Quantifying and Valuing the Ecosystem Services of Pastoral Soils under a Dairy Use

A thesis presented in partial fulfilment of the requirements for the degree

of

Doctor of Philosophy

in

Ecological Economics

at

Massey University, Palmerston North, New Zealand

Estelle Jeanne DOMINATI

2011

Abstract

The full range of ecosystem services provided by soils are rarely recognised or understood, nor is the link between soil natural capital and these services. Understanding these concepts is more important than ever to meet the food and fibre demands of a growing global population, while ensuring the sustainability of the finite resource that is soil. The objective of the thesis was to develop a framework to describe the natural capital and ecosystem services of pastoral soils, and to apply it to quantify and value soil ecosystem services under a dairy use in New Zealand.

A new conceptual framework was developed from current scientific understanding of land classification, soil formation, soil processes and ecosystem services concepts. The framework links soil formation, maintenance and degradation processes to soil natural capital stocks, and provides a basis for exploring the influence of drivers like climate and land use on soil natural capital stocks and the flow of ecosystem services. The soil services identified included provision of food, support to human infrastructure and animals, flood mitigation, filtering of nutrients and contaminants, detoxification and recycling of wastes, carbon storage and greenhouse gases regulation, and pest and disease populations regulation. Based on the conceptual framework, new methodology was developed to quantify and model each provisioning and regulating service from soils. Proxies based on soil properties and a process-based model were used to explore the impacts of soil type (Horotiu silt loam and Te Kowhai silt loam) and dairy management practices on soil properties and processes behind each service at the farm scale. Neoclassical economic valuation techniques were then used to value soil ecosystem services for the case study examples.

Under a dairy operation, the total value of soil ecosystem services was $15,777/ha/yr for a Horotiu silt loam. Regulating services ($11,445/ha/yr) had a greater value than provisioning services ($4,322/ha/yr). The ecosystem services from a Te Kowhai silt loam were less valuable, $11,687/ha/yr. The difference in value between soils reflects differences in their physical structure and associated hydraulic properties, the natural capital stocks behind many services. Valuing some services (e.g. filtering of P) was challenging since some services cannot be substituted by artificial inputs or manufactured capital.

This new approach provides for the first time land managers and policy makers with the ability to compare the total utility of soils, not just their productivity and versatility for different land uses. It also provides a powerful practical tool for evaluating the environmental impact of farm management practices, resource management options and policy alternatives at the regional and national levels, by enabling direct linkages between the economy and the environment. This study allows the value of soil to be benchmarked against commonly used indicators of economic performance such as GDP at the national level and net profits at the farm scale. The case study examples showed that the value of 'un-priced' soil ecosystem services to be significantly higher than net profit of dairy farms.

Acknowledgements

Firstly, I would like to dedicate this work to my dad, Jacques Raymond Dominati (1947-2005), the first and real doctor of the family. He passed away before I even started my Ph.D., but I know he would have enjoyed this work very much, passionate by gardening and the outdoors as he was. I miss him every day.

A great thank you to each and every one of all the many people who have supported me throughout my Ph.D. First, I would like to thank my partner, Mike Cook, for his tremendous patience, kindness, love, and endless support. He has kept me sane through the whole process, and was always available to listen to my "PhD ranting". I owe him more than I can tell.

I am particularly grateful to my supervisor, Dr. Alec Mackay, for the countless hours spent with me, his constant encouragements, and his much appreciated and contagious enthusiasm and energy. Twice, he gave me the opportunity to come to New Zealand and work on captivating projects, and for that I will be forever indebted to him.

A very special thank you to Dr. Steve Green, for his patience with "frog girl", the endless hours of modelling, and his great contribution to this work. It wouldn't have been possible without him. I also want to thank Professor Murray Patterson for his supervision, advice and great insight.

To my dear family, thank you for always being there for me, even from the other side of the world. Thank you to my mum, Sylviane, for her love, and always wanting the best for us. To my little brother, Emmanuel, who managed to become a doctor before me. He's always been the crafty one. To my aunt Maryse and my uncle Max for their support. And to all the rest of you, dearest to my heart.

I am indebted to a number of people who have been a great help, passing on knowledge, especially Dr. Brent Clothier, Dr. Kevin Tate, Dr. Surinder Saggar, and many unnamed others at Landcare Research, Plant and Food Research and AgResearch. It's been great to work with you all these past four years.

I would also like to thank my wonderful friends, in New Zealand and in France, especially Julie Piazza and Elsje Van der Merwe: thank you for all the girly chats and the much needed laughter. But also Aurelie, Liat, Amelie, Anne-Gaelle, Elodie, Emilie, Cyriele, Natalie, "the girls" back home, and all the others, you know who you are!

Last but not least, I would like to thank AgResearch for their financial support, as a member of the sustainable land use research initiative (SLURI), a formal collaboration between three Crown Research Institutes.

Table of Contents

List of Figures

List of Tables

Acronyms and Symbols

Acronyms:

ASC: Anion storage capacity

AWC: Available water capacity

BCA: Benefit cost analysis

BD: Bulk density

CEC: Cation exchange capacity

DFE: Dairy farm effluents

DM: Dry matter

DOC: Dissolved organic carbon

DOM: Dissolved organic matter

DON: Dissolved organic nitrogen

DOP: Dissolved organic phosphorus

EDCs: Endocrine-disrupting chemicals

ES: Ecosystem services

FC: Field capacity

GHG: Greenhouse gas

GHGs: Greenhouse gases

GT: Grazing time

HM: Heavy metals

HR: Horotiu silt loam

IPCC: Intergovernmental Panel on Climate Change

K sat: Hydraulic conductivity

MAF: Ministry for agriculture and forestry in New Zealand

MEA: Millennium ecosystem assessment

MfE: Ministry for the environment in New Zealand

Mp: Macroporosity

MS: Milk solids

NC: Natural capital

NZ: New Zealand

OM: Organic matter

PL: Plastic limit

RF: Rainfall

RO: Runoff

Sat: Saturation

SP: Stress point

SPASMO: Soil plant atmosphere model

SR: Stocking rate

SWC: Soil water content

TDF: Typical dairy farm

TEV: Total economic value

TK : Te Kowhai silt loam

WFPS: Water-filled pore space

WP: Wilting point

WTA: Willingness to accept compensation

WTP: Willingness to pay

Symbols:

Ammonia: NH_3

Ammonium: NH_4^+

Boron: B

Calcium: Ca

Carbon dioxide: CO_2

Carbon: C

Chloride: Cl

Cobalt: Co

Copper: Cu

Hydrogen: H

Iron: Fe

Magnesium: Mg

Manganese: Mn

Methane: CH_4

Molybdenum: Mo

Nitrate: NO_3^-

Nitrogen: N

Nitrous oxide: N_2O

Oxygen: O_2

Phosphorus: P

Potassium: K

Silicon: Si

Sodium: Na

Sulphur: S

Zinc: Zn

Chapter One

Introduction

1.1 New Zealand - A land based economy:

New Zealand's continued wealth generation is more than ever highly dependent on its soils. Almost half of New Zealand's land is farmed commercially – of that 80% of the total land area is pasture and arable cropping land, about 14% is exotic forest, and less than 4% is orchard or market garden (Fig 1.1). The remainder is under indigenous forest and shrubland (33%), tussock grassland (14%), urban area (1%), alpine zone, low-lying wetland or coastal sand (Gillingham, 2009).

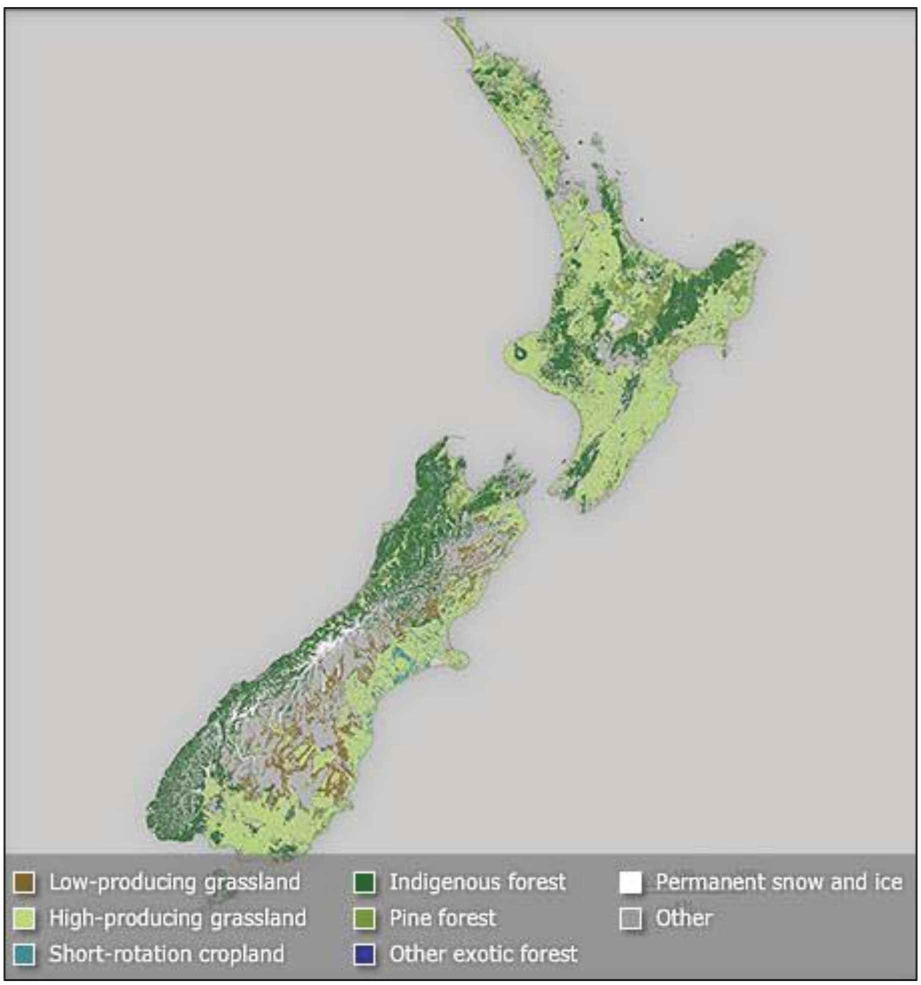

Figure 1-1: North island land uses (Gillingham, 2009).

For the last 100 years agriculture and forestry have been New Zealand's largest sectors of the economy, with farming being the main export earner. In 2005, 63% of export earnings were

from agriculture (25%, by value, from the dairy industry, 24% from meat, 4% from wool, 10% from timber and wood products) (Pawson, 2010).
About 90% of what New Zealand farmers produce is exported (FF, 2010):

- 1.78 million tonnes of dairy products, representing 21.8% of the world trade of dairy products,
- Nearly 600,000 tonnes of sheep meat, representing 55% of the world trade of sheep meat, and 75% of world lamb exports,
- 634,000 tonnes of beef and veal, representing only 1.1% of the world trade,
- 140,000 tonnes of wool.

For the year ended March 2008, the New Zealand agricultural sector generated $19 billion in gross revenue. Agriculture made a direct contribution to Gross Domestic Product (GDP) of over $8.2 billion (5% of total GDP, excluding downstream processing (Statistics New Zealand). Including downstream processing, agriculture is estimated to contribute over 15% of total GDP. Together with its support and processing components, the agriculture industry regularly contributes almost a quarter of New Zealand's GDP (FF, 2010).
Another driver of New Zealand economy is tourism, which also depends on natural environments including soils. International tourism took off in the 1960s. By the early 2000's, tourism's annual contribution to the economy was $18.6 billion (9% of GDP then). In the year ending March 2009, there were 2.4 million international tourists visiting New Zealand.

Soils and landscapes and the services they provide are therefore an essential factor in the economic well-being of New Zealand, as they are for the economy of most nations (Daily, 1997). Franklin D. Roosevelt said in 1937 "the Nation that destroys its soil destroys itself". Surprisingly, little attention is given to analysis of current or future land uses, or the greater value of soils to the economy, despite our ongoing dependency on this finite resource.

1.2 Soils - An undervalued and threatened resource:

Differences in the productive capacity and versatility of soils are well known and understood. Soil science has been very effective in classifying soils based on their capability (Lynn et al., 2009) and versatility (Webb and Wilson, 1995), but struggles to place a monetary value on these and other attributes. The current valuation of the land is driven primarily by its productive capacity and little else, putting aside location in relation to distance from urban centres and iconic coastal and lake side locations. The other roles soils play, which include the provision of support for human infrastructure and animals, flood mitigation, the filtering of nutrients and contaminants, the recycling of wastes, the regulation of GHGs emissions, carbon storage and the regulation of pest and disease populations are not valued. They are often not even recognised.

Some scientists noticed early that soils have roles beyond production. For example, Daily (1997) and Wall et al. (2004) described in detail the services soils provide to human society, from being a substrate for plant growth, to buffering floods or recycling wastes. Daily (1997) noted that soils are a very valuable asset that "takes hundreds to hundreds of thousands of years to build and very few to be wasted away" (Daily, 1997, p. 113).

The challenges faced by modern societies of population growth and associated growing food demand intensify the pressures on natural resources, including soils, and the wider environment. This is raising increasing questions about the sustainability of agricultural systems, and our ability to mitigate the impacts of ongoing production gains on the environment. In New Zealand, over the past 10 years, the agricultural sector multi-factor productivity has grown at a rate of 1.8% per year, double the rate for the economy as a whole (FF, 2010). This growth strategy is based on the assumption that natural resources, such as land and water, are inexhaustible resources, which they are not, raising the question of the viability of the current economic model (Munda et al., 1994).

Of the soils in New Zealand, very few are without limitations. Only 5% of the land mass is classed as having 'elite and versatile soils', and these are increasingly under pressure for urban development (Rutledge et al., 2010). The expansion of urban areas over elite and versatile soils is of concern in New Zealand. The Ministry of Agriculture and Forestry estimated that urban areas are expanding at about 5% a year – approximately 40,000ha each year (MAF, 2010). In comparison, New Zealand's $3 billion horticultural exports come from just 70,000ha (MAF, 2010; Rowarth, 2010).

To meet the growing demand for more food, a wide range of very effective and successful technologies have been developed to overcome soil and climate limitations. These technologies effectively address inherent weaknesses in our soils. To date, land development has focussed on technologies that remove limitations with additional inputs (fertilisers, irrigation or drainage). As a consequence of this approach, intense agricultural activities are having more and more impacts on the environment. Erosion and increased nutrient concentrations in water are New Zealand examples of this impact (Rowarth, 2010). Such externalities are not taken into account when determining land value, and are certainly not incorporated into on-farm production costs or food prices. Given land is a finite resource, consideration of the overall value of our land resource, including all the services provided by our soils, and especially by elite and versatile soils, would be warranted. This would better inform land managers and policy makers in progressing towards more sustainable and environmentally friendly land development and management practices.

In a world more and more concerned by the sustainability of industries and our living environment, it is very important for a country like New Zealand whose economy depends on its soils, to look more closely at this resource to ensure it is being preserved and used efficiently to sustain the country and its place on the world markets. It therefore becomes crucial that the roles soils play are understood and accounted for by policymakers and land managers to ensure the soil resource is there for future generations.

1.3 Tools to achieve sustainability:

In the quest for sustainable land management, a number of disciplines need to be brought together. Soil science has been successful in describing the soil resource and understanding the differences in the productive capacity and versatility of soils, but has come up short in identifying, quantifying and valuing all the services provided by soils. This is where Ecological Economics may offer some utility.

The concepts of natural capital and ecosystem services come from the discipline of Ecological Economics. There has been a growing interest in these concepts since the late 1960s. Ecological Economics is a recently developed field, which sees the economy as a subsystem of a larger finite global ecosystem (Martinez-Alier, 2001). Ecological economists question the sustainability of the economy, because of its environmental impact and its material and energy requirements (Martinez-Alier, 2001). The main focus of Ecological Economics is to develop physical indicators and indexes of sustainability. Different techniques have been developed by ecological economists, including assigning monetary values to ecosystem services and correcting macroeconomic accounting to include the environment.

Natural capital refers to the extension of the economic idea of manufactured capital to include environmental goods and services and has been defined has the “stocks of natural assets (e.g. soils, forests, water bodies) that yield a flow of valuable ecosystem goods or services into the future” (Costanza and Daly, 1992, p. 38). The concept of ecosystem services gained real momentum in 1997 thanks to Costanza et al. (1997). In 2005, the Millennium Ecosystem Assessment introduced ecosystem services to the general public as “the benefits people obtain from ecosystems” (MEA, 2005). Most of the studies on ecosystem services focus on above ground ecosystems and the contribution of soils and more generally below ground ecosystems to the provision of services is afforded little consideration, not only by economists but also by ecologists and soil scientists. The difficulty of bridging science disciplines and communicating effectively due to differences in language and terminology may be one of the reasons why Soil

Scientists have not engaged more with the ecosystem services approach yet (Robinson et al., 2012).

To convince financial institutes, economists and others in society of the value of soils, the current market approach to valuing land needs to be extended to include all the services provided by a soil. Approaches that have the capacity to quantify and value all the ecosystem services coming from soil natural capital stocks, would provide an advancement on the current method, and for the first time, a commentary on the total costs and benefits of land use. Currently, the full range of ecosystem services from soils are rarely recognised and generally not well understood, nor are the links between soil natural capital and ecosystem services.

1.4 A new framework for soil natural capital and ecosystem services:

The millennium ecosystem assessment (MEA, 2005) was very successful in informing people of the different roles of ecosystems and how much human societies depend on them. Since then, governments and policy-making bodies have begun adopting the idea of an ecosystems approach in resource management to incorporate life supporting values into decision making (e.g., Department for Environment, Food and Rural Affairs, 2007) (Robinson et al., 2012). Sadly, however, soils were treated as a black box. The MEA framework while advancing our thinking contains a number of limitations if used for the quantification and valuation of ecosystem services from soils. These include:

- The terms used (benefit, good, function, service, process) are ambiguous, which makes the framework difficult to apply to a specific ecosystem (Boyd and Banzhaf, 2007),
- There are no links allowing the application of the MEA framework to soils. What are supporting, provisioning and regulating services from soils?
- What is Natural Capital? How does it fit with soil science concepts?
- How do drivers like land use or climate influence natural capital stocks and ecosystem services?

Knowledge available in Soil Science, including soil processes, pedogenesis and pedology could be used within a framework based on the MEA framework to bring life to the black-box and create links between land use, soils and ecosystem services, strengthening ecosystem based resource management. A new framework that captures these elements would inform land management by enabling links to be made between land resources inventories, land use and outcomes at different scales, thus conveying the importance and value of soils to decision makers.

1.5 Research aims and objectives:

1.5.1 Overall aim:

1- A conceptual framework for identifying and classifying the ecosystem services provided by a soil can be developed, from our current understanding of soil formation and processes, land classification, and ecosystem service concepts, that links soil formation, maintenance and degradation processes to the soil natural capital stocks defined by soil properties, and provides a basis for exploring the influence of drivers (anthropogenic and natural) on natural capital stocks and the flow of ecosystem services.
2- A methodology can be developed to quantify and value each of the soil services identified and described in the framework under a dairy farm operation.

1.5.2 Specific Objectives:

Specific objectives for this research project include:

1. Develop an understanding of soil natural capital and how it links to the provision of ecosystem services from below ground ecosystems since such knowledge is lacking from existing ecosystem services framework.
2. Investigate the properties and processes behind each soil ecosystem service, including where these services come from and what influences their provision.
3. Define a methodology to quantify soil ecosystem services at the farm scale.
4. Identify techniques appropriate for valuing soil services and develop a methodology to value them at the farm scale.
5. Examine the impacts of soil types and farm management on the provision of soil services, from a dairy farm for the Waikato region.
6. Value soil ecosystem services in the context of a dairy farm operation.

1.5.3 Methodological Approach:

The methodology used in this thesis consists of five integrated components:

1- A critical review and synthesis of the relevant literature relating to the quantification and valuation of ecosystem services from soil. This includes: (i) natural capital and ecosystem services concepts developed for above ground ecosystems and their applications to soils, (ii) soil science knowledge and soil functioning behind soil natural capital and the provision of ecosystem services, and (iii) theory around the economic valuation of ecosystem services and the different techniques available. The purpose of the literature review and syntheses is to provide insight from a theoretical perspective of Ecological Economics and Soil Science for the construction of a soil natural capital and ecosystem services framework, and the quantification and valuation of soil ecosystem services.

2- Develop a conceptual framework for exploring the links between soil natural capital, ecosystem services and human needs, using Soil Science concepts, including pedogenesis, pedology, soil properties and soil processes, since current ecosystem services framework do not take soils into account. A particular focus of this thesis is to link Ecological Economics concepts of natural capital and ecosystem services to Soil Science to enable utilisation of the extensive scientific knowledge on soil processes to inform the provision of ecosystem services from soils.

3- Develop tools to inform the provision of soil services and quantify them. Existing Soil Science enables us to identify proxies and calculate indicators based on soil properties to measure soil services. These indicators of the provision of soil services then need to be linked to the dynamics of soil properties depending on land use, farm management and climate.

4- Add extra-functionality to an existing soil-plant-atmosphere process-based model (SPASMO). In order to look at the impact of management practices on the dynamics of the provision of services from soils, the impact of practices on soil properties and processes needs to be identified, described and incorporated into a dynamic model.

5- Economic valuation of soil ecosystem services. The valuation of soil ecosystem services needs to be closely linked to the measures of each service, in order to investigate changes in the value of the services. The quantification and valuation of soil ecosystem services then need to be implemented at the farm scale, through different scenarios, to explore the impacts of soil types and management practices on the provision and value of soil ecosystem services.

1.6 Thesis organisation and outline:

This thesis is divided into three distinct yet related parts. The interrelationships between these different parts of the thesis and the constituent chapters are described by Figure 1.2.

Part One, Conceptual and Quantification Framework, addresses the following questions: What is behind soil natural capital and ecosystem services? How can we measure them?
Chapter Two reviews the general literature on ecosystem services, as well as the literature on soil ecosystem services. A framework is developed drawing on scientific understanding of soil formation, functioning and classification systems and current thinking on ecosystem services,

to identify and classify soil natural capital, show where climate and land use impact, and how this all links through to ecosystem services. The details of this framework and the quantification of each service are developed further, for each provisioning and regulating soil ecosystem service, in chapters Three and Four.

Chapter Three discusses cultural services briefly, and then describes the properties, processes and drivers influencing the provisioning services from soils. For each soil provisioning service, critical soil properties linked to the service are identified and indicators are proposed to quantify and model the provision of these services.

Chapter Four describes the properties, processes and drivers influencing the regulating services from soils. For each soil regulating service, critical soil properties linked to the service are identified and indicators are proposed to quantify and model the provision of these services.

Part Two, Methodology, builds on Part One. It describes and develops tools and critically reviews methods used to quantify and value soil ecosystem services.

Chapter Five presents the context of the study, as well as the process-based model (SPASMO) used to quantify the dynamic provision of soil services. This chapter details the additional functions added to the model to capture and describe the impact of specific management practices on the soil properties and processes behind specific soil services.

Chapter Six critically reviews neoclassical economic valuation methods as well as the emergence of alternative approaches in Ecological Economics. Different valuation methods are examined against a number of criteria specific to the valuation of soil services.

Part Three, Empirical Results for Dairy Farm Soils, uses the methods developed in Parts One and Two to quantify and value each soil ecosystem service at the farm scale, for a dairy farm in the Waikato, using different scenarios.

Chapter Seven details the quantification and valuation of each of the ecosystem services provided by a Horotiu silt loam soil, under a typical Waikato dairy farm.

Chapter Eight examines the impact of soil type on the provision of soil services through two scenarios.

Chapter Nine examines the impact of management practices, namely stocking rate, fertilisation and the use of a standoff pad, on the provision of soil services through twelve scenarios.

Chapter Ten identifies the key contributions of the thesis, and identifies areas for further research and development.

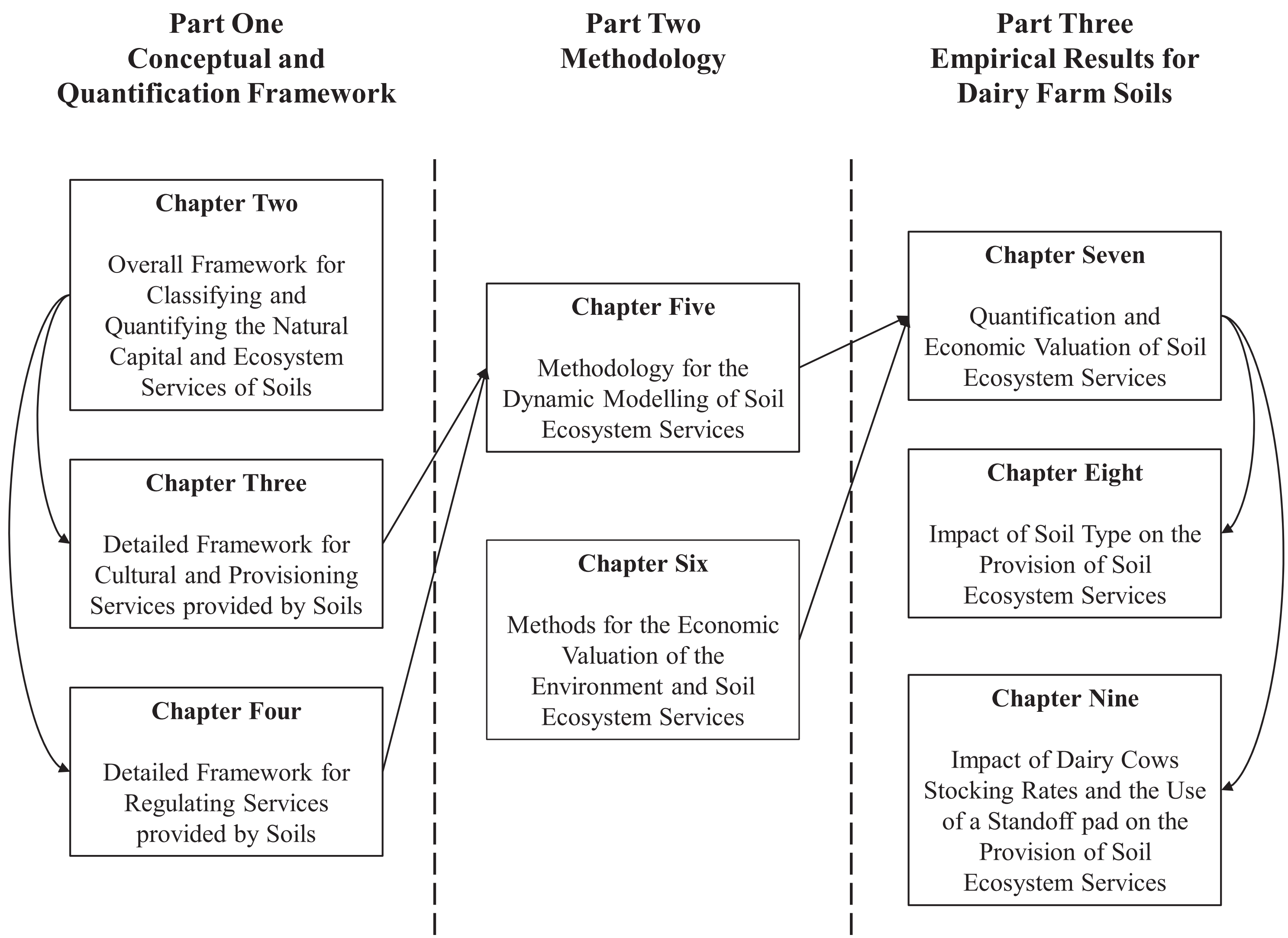

Figure 1-2: Relationships between thesis chapters.

PART ONE

CONCEPTUAL AND QUANTIFICATION FRAMEWORK

Chapter Two

Overall Framework for Classifying and Quantifying the Natural Capital and Ecosystem Services of Soils

In this chapter, a framework for identifying and classifying the natural capital and ecosystem services of soils is developed, that draws on our scientific understanding of pedogenesis, soil processes and soils and land classification systems, and current ecosystem services concepts to classify and quantify soil natural capital and ecosystem services.
The focus of this chapter and the PhD thesis is on the ecosystem services from soils. Existing ecosystem services frameworks focus mainly on the above ground component of ecosystems. A future step would be to bring the belowground and aboveground frameworks together to provide a more holistic view of the functioning of our environment.

2.1 Context and terminology:

Since the late 1960s there has been a growing interest in the analysis of the services provided by ecosystems (Westman, 1977) and the need to include them in decision-making processes in order to achieve sustainable development. Several studies have provided frameworks for the description and valuation of ecosystem services (Costanza et al., 1997; de Groot et al., 2002; MEA, 2005) but all too often soils, the basic substrate for many ecosystems and human activities, have been considered a black-box within these frameworks, because their focus is on what happens above ground. Many authors (Balmford et al., 2002; Daily et al., 1997a; Kroeger and Casey, 2007; Swinton et al., 2006; Swinton et al., 2007; Turner and Daily, 2008) agree that our ability to understand soil natural capital and the ecosystem services it provides is incomplete, despite a good understanding of pedogenesis and soil processes. Because soils are an important determinant of the economic status of nations (Daily et al., 1997a), it is essential to include them in ecosystem service frameworks that inform decision making and environmental policies.

One of the difficulties in constructing a coherent "natural capital - ecosystem services" framework for soils is the confusion created by the use of terminologies borrowed from at least three disciplines: Ecology, Economics and Soil Science. Many of the terms used have multiple definitions. In particular, there is considerable confusion between the terms process, function and service. For sake of clarity, in this thesis, we define each of the terms used. We recognise that these terms may be used differently in other disciplines or field.

Natural capital refers to the extension of the economic idea of manufactured capital to include environmental stocks. Natural capital, like all other forms of capital, is a stock as opposed to a flow. Natural capital consists of "stocks of natural assets (e.g. soils, forests, water bodies) that yield a flow of valuable ecosystem goods or services into the future" (Costanza and Daly, 1992, p. 38). Soils are considered here as natural capital and provide services such as flood mitigation (Fig.2.1).

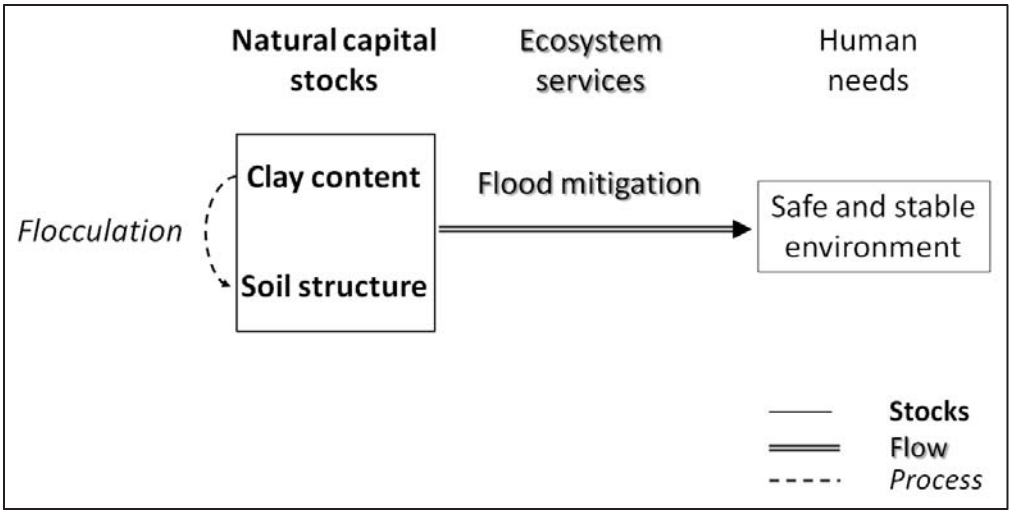

Figure 2-1: Illustration of the use of the key terms employed in this thesis

To describe soils, pedologists use different concepts like soil components and soil properties. A *soil component* is defined here as a biogeochemical species (e.g. nitrate NO_3^-) or an aggregation of biogeochemical species (e.g. clays, Fig. 2.1) that make up soils. Soils consist of four major categories of soil components: solids mineral and organic, liquids, and gases. Soil properties are the physical (e.g. porosity, texture), chemical (e.g. pH, readily available phosphate), and biological (e.g. microbial biomass) characteristics of a soil. Soil properties are often measurable quantities that allow soil scientists to place soils on relative scales. For example, clays (Fig. 2.1) are soil components which play an important role in the formation of soil structure. Clay content is a property quantifying the amount of clay in a soil.

Authors (Costanza and Daly, 1992; Daly and Farley, 2003; de Groot et al., 2002; Ekins et al., 2003a) agree that natural capital yields ecosystem services but the nature of these ecosystem services is still debated in the literature (Costanza, 2008; Fisher and Turner, 2008; Wallace, 2007). Controversy revolves around the definitions of the terms *function* and *process* used to define ecosystem services and the boundaries between them. In Ecology, the traditional definition of an *ecosystem function* was the role the ecosystem plays in the environment, but in recent years, the term ecosystem function has been used as a synonym for *ecosystem process* (Wallace, 2007), as in soil science. In this chapter, the term "process" is used rather than "function" and is defined as the transformation of input into outputs. Some processes are chemical (e.g. oxidation), some physical (e.g. diffusion), others are biological (e.g. denitrification). All processes must only involve the transformation of energy and mass, to qualitatively different forms, with both mass and energy quantities being conserved (e.g. $Mass_{in} = Mass_{out}$; $Energy_{in} = Energy_{out}$). For example (Fig. 2.1), flocculation is a process leading to the formation of soil structure. At the molecular level, water molecules and cations link negatively charged clays together. When the soil dries out the clays are brought together into more stable aggregates. In the Soil Science literature the terms "property" and "attribute" are synonyms. Soil properties are often measurable quantities that allow soil scientists to place soils on relative scales. Soils differ in their properties and in response to a use. That's how soil scientists compare soils.

The existing literature on ecosystem services tends to focus exclusively on the ecosystem services rather than holistically linking these services to the natural capital base from which they arise. To avoid this, ecosystem services are defined here as the beneficial flows arising from natural capital stocks and fulfilling human needs. We argue that ecosystem services are not processes but flows (amount per unit time), as opposed to stocks (amount). For example (Fig. 2.1), soil structure presents pores able to store water. The provision of the ecosystem service flood mitigation depends on the amount of water a soil can store (stock) and also the timing of the availability of the storage volume regarding a rainfall event.

When considering the term ecosystem service, it can be argued that, to some extent the adjective 'ecosystem' is a misnomer, as ecosystem services can occur at higher levels of ecological organisation/scale than an 'ecosystem', e.g. greenhouse gases regulation comes from different ecosystems (soils, forests) and impacts at the biosphere level. The noun 'service' is also arguably a misnomer as it includes 'goods' such as food, wood and fibre products, as well as actual services like flood mitigation or aesthetics.

Keeping in mind these concepts, this chapter undertakes to assess the importance of soils as natural capital and provider of ecosystem services. First, existing ecosystem services and soil services frameworks are critically reviewed. Then, a new framework is developed which introduces soils as natural capital, illustrates natural capital formation, maintenance and degradation and the natural and anthropogenic drivers impacting on these processes. Finally, the chapter describes the ecosystem services provided by soils, and outlines how soil ecosystem services fulfil human needs.

2.2 Existing classification schemes for ecosystem services:

Before presenting our new framework, the strengths and limitations of general ecosystem services frameworks found in the literature, as well as agro-ecosystem services frameworks which include soils are examined.

2.2.1 General ecosystem services frameworks:

With heightening awareness of the importance of ecosystem services, over the last two decades general typologies and classification systems have emerged (Table 2.1). De Groot's classification system (1992), one of the first, defined ecosystem functions as "the capacity of natural processes and components to provide goods and services that satisfy human needs, directly or indirectly" and grouped these functions into four primary categories (Table 2.1):

- Regulation functions to regulate essential ecological processes and life support systems and the maintenance of ecosystem health,
- Habitat functions to provide refuge and reproduction habitat to wild plants and animals,
- Production functions for processes creating living biomass used for human consumption (food, raw materials, energy resources, genetic material),
- Information functions to provide opportunities for reflection, spiritual enrichment, cognitive development, recreation and aesthetic experience.

Costanza et al. (1997) detailed seventeen goods and services, including most of de Groot's (1992) functions. Noël and O'Connor (1998) classified "the specific roles or services provided by natural systems that support economic activity and human welfare" into five categories - "the five S's" (Table 2.1) - including, *Source* of biological resources, food, raw materials and energy in various forms; *Sink*, or place of controlled and uncontrolled disposal of waste products and energy of all sorts; *Scenery*, covering all forms of scientific, aesthetic, recreational, symbolic and informational interest; *Site* of economic activity, including land uses and occupation of space for transportation; and *life Support* for human and non-human living communities: the capacity to sustain ecosystem health.

Daily (1999) also produced an "ecosystem services framework" including five services (Table 2.1):

- Production of goods: Food, pharmaceuticals, durable materials, energy, industrial products, genetic resources,
- Regeneration processes: Cycling and filtration processes, translocation processes,
- Stabilizing processes: Regulation of hydrological cycle, stabilization of climate, coastal and river channel stability,

- Life-fulfilling functions: Aesthetic beauty, cultural, intellectual, and spiritual inspiration,
- Preservation of options: Maintenance of the ecological components and systems needed for future.

A common thread through all these classification systems is the recognition of the diversity of roles played by ecosystems (Table 2.1). The concepts proposed in different classifications tally with each other (Table 2.1). For instance de Groot's (1992) production functions correspond to what Noël and O'Connor (1998) called the "source" role of ecosystems, and Noël and O'Connor (1998) 'Sink' function is similar to Daily's (1999) 'stabilising processes'.

More recently, de Groot et al. (2002) identified 23 functions in the four primary categories established in earlier work (de Groot, 1992) and detailed the corresponding processes and services, noting that "ecosystem processes and services do not always show a one-to-one correspondence" (de Groot et al., 2002, p. 397). To the four categories, they later introduced a fifth, a carrier function (Table 2.1) and specified that the "regulation functions provide the necessary pre-conditions for all other functions" (de Groot, 2006, p. 177). As part of the CRiTiNC project, Douguet and O'Connor (2003) and Ekins et al. (2003b) used a similar classification (Table 2.1) to that of Noël and O'Connor (1998) to argue that the principles of environmental sustainability must be based on the maintenance of the important life-support "functions of nature" that form the basis on which the "functions for people" are fundamentally dependent.

Table 2-1: Ecosystem roles mentioned by different classification systems.

Authors	Ecosystem roles[1]					
	Life Support	Production	Regulation	Habitat provision	Physical support	Information and Culture
De Groot (1992, 2002)	Regulation functions	Production functions	Regulation functions	Habitat functions	NC	Information functions
Noël & O'Connor (1998)	Life Support	Source	Sink	NC	Site	Scenery
Daily (1999)	Regeneration processes	Production of goods	Stabilising processes	NC	NC	Life filling functions, Preservation of options
Ekins (2003)	Life Support	Source	Sink	NC	NC	Human health and welfare
MEA (2005)	Supporting services	Provisioning services	Regulating services	NC	NC	Cultural services
De Groot (2006)	Regulation functions	Production functions	Regulation functions	Habitat functions	Carrier functions	Information functions

[1] Roles as described by the original authors. NC: not considered.

The novel idea that de Groot et al. (2002), Douguet and O'Connor (2003) and Ekins et al. (2003b) advanced is that some ecosystem functions – or processes as we call them in this study – support others. Ecosystem processes insure ecosystems health and functioning, whereas ecosystem services are flows coming from these ecosystems. The Millennium Ecosystem Assessment (MEA, 2005) took up this idea in a "framework of ecosystem services" (Table 2.1). It assessed the consequences of ecosystem change for human well-being, defining ecosystem services as "the benefits people obtain from ecosystems" (MEA, 2005, p. 40). The MEA framework classified ecosystem services in four categories: Provisioning services are the products obtained from ecosystems; Regulating services are the benefits obtained from the regulation of ecosystem processes; Cultural services are the nonmaterial benefits people obtain from ecosystems through spiritual enrichment, cognitive development, reflection, recreation, and aesthetic experiences; and Supporting services are those that are necessary for the production of all other ecosystem services. Their impacts on people are often indirect or occur over a very long time (MEA, 2005). The first three categories of services directly affect people, whereas the supporting services are there to maintain the other services. It is interesting to point out that the MEA's four categories are close to the categories of functions of de Groot (1992) with the difference that de Groot's "regulation functions" seem to include both of the MEA's "supporting and regulating services" (Table 2.1). The approach set out in the MEA has been adopted and used widely (Barrios, 2007; Lavelle et al., 2006; Sandhu et al., 2008; Swinton et al., 2007; Zhang et al., 2007) as a conceptual framework.

Some roles of ecosystems are mentioned unanimously by the authors cited above (Table 2.1), including:

- The production (or source) role – the capacity of an ecosystem to produce resources of interest for humans,
- The regulation role – the capacity of ecosystems to auto-regulate themselves, absorb human emissions, recycle them, and remain stable,
- The information role – the capacity of ecosystems to inspire people and produce non-material goods.

However, as has been reported by a number of authors (Boyd and Banzhaf, 2007; Costanza, 2008; Fisher and Turner, 2008; Wallace, 2007), some common challenges are still found with existing frameworks.

First, not all existing frameworks recognise that some processes sustain others. Of the classification systems covered (Table 2.1), only Ekins et al. (2003b), de Groot (2006), and the MEA (2005) acknowledge that some processes "support" other processes. Failure to make the distinction can lead to double-counting in the measurement and valuation of ecosystem services. Once it has been recognised that some processes support others, the challenge is to identify precisely the ecosystem services provided and make the distinction between these services and the processes directly supporting them (Boyd and Banzhaf, 2007; Fisher and Turner, 2008).

Second, the definitions and use of terms to describe ecosystem services vary across the published classification systems. The ecosystem services literature often refers to groups of processes such as, for instance, "nutrient cycling" (MEA, 2005) as a service. It has been argued (Balmford et al., 2011; Boyd and Banzhaf, 2007; Fisher and Turner, 2008; Wallace, 2007) that doing so mixes up the "means of production", the processes, with the actual services. Photosynthesis, for example, is an essential process for plant growth and should not be confused with the ecosystem service it supports, which is the provision of food and fibre.

Third, as different authors (Boyd and Banzhaf, 2007; Wallace, 2007) have pointed out, in valuing ecosystems it may be more helpful to focus on ecosystem components, and use them as proxies for services, rather than on processes, because science gives us much more information on the structure and composition of ecosystems than on the processes involved in their functioning. For these reasons, a number of authors (Balmford et al., 2011; Fisher et al., 2009) have tried redefining ecosystem services and disaggregating them into sets which differ in their proximity to human well-being. Fisher et al. (2009, p. 645) argued that "the functions or processes become services if there are humans that benefit from them. Without human beneficiaries they are not services". They proposed to make the difference between "intermediate services, final services and benefits". Similarly, Balmford et al. (2011) proposed the distinction between "core ecosystem processes, beneficial ecosystem processes and ecosystem benefits". Both these frameworks recognise that some processes support others, that processes and ecosystem services are distinct, and that care needs to be taken when using ecosystem services frameworks to avoid double-counting.

General ecosystem service frameworks (Table 2.1) do little justice to the roles of soils in the provision of ecosystem services and as a consequence fail to recognise the large differences that exist between soils in their ability to provide services. For instance, the MEA mentions "soil formation" as a supporting service and recognises that "many provisioning services depend on soil fertility" (MEA, 2005, p. 40). It also mentions the role of soils in the provision of regulating services like erosion regulation, water purification and waste treatment, but does not explicitly identify the part played by soils in the provision of these services and more generally in the provision of services from above ground ecosystems. Moreover, general ecosystem services frameworks do not detail the relationships between soil properties, soil processes and soil services. For these reasons, ecosystem services frameworks that accord more importance to soils and their different roles needed to be examined more precisely.

2.2.2 Soil ecosystem services frameworks:

Many agree (Daily, 1997; Dale and Polasky, 2007; de Groot et al., 2003; Haygarth and Ritz, 2009; Straton, 2006) that a better characterisation of ecosystem services supplied by soils is overdue. Daily (Daily et al., 1997a, p.128) indicated that "research is needed to better characterise the ecosystem services supplied by soils", along with a better understanding of the "interrelationships of different services supplied by soils and other systems". While a few authors (Daily et al., 1997a; Haygarth and Ritz, 2009; Wall et al., 2004; Weber, 2007) have proposed soil specific frameworks for ecosystem services, others (Barrios, 2007; Lavelle et al., 2006; Porter et al., 2009; Sandhu et al., 2008; Swinton et al., 2007; Zhang et al., 2007), mainly working on wider agro-ecosystems, have detailed services provided by soils (Table 2.2). These studies enable us to start identifying where, and in which way, soils affect the provision of ecosystem services. A comparison of Daily (1997), Wall et al. (2004) and Weber (2007) specific soil services classifications, with those that have based their classification on the MEA (Barrios, 2007; Haygarth and Ritz, 2009; Lavelle et al., 2006; Sandhu et al., 2008; Swinton et al., 2007; Zhang et al., 2007) (Table 2.2) highlights the following roles of soils in the provision of services:

- Fertility role: soil nutrient cycles ensure fertility renewal and the delivery of nutrients to plants, therefore contributing to plant growth,
- Filter and reservoir role: soils fix and store solutes passing through and therefore purify water. They also store water for plants to use and take part in flood mitigation,
- Structural role: soils provide physical support to plants, animals and human infrastructure,
- Climate regulation role: soils take part in climate regulation through carbon sequestration and greenhouse gases (N_2O and CH_4) emissions regulation,
- Biodiversity conservation role: soils are a reservoir of biodiversity. They provide habitat for thousands of species regulating for instance pest control or the disposal of wastes,
- Resource role: soils can be a source of materials like peat and clay.

Table 2-2: Soil ecosystem services and agro-ecosystem services classifications and the concordances between them.

Reference	Type of framework	Services attributable to soils									
		Nutrient					Water			Structure	
		Provision to plants	Contribution to plant production	Renewal through soil formation	Cycles regulation	Movements	Provision to plants	Filtering	Flood control	Support provision	Erosion control
Daily (1997)	Soil specific	Retention and delivery of nutrients to plants	NC	Renewal of soil fertility	Regulation of major element cycles	NC	NC	NC	Buffering and moderation of the hydrological cycle	Physical support of plants	NC
Wall et al. (2004)	Soil specific	Retention and delivery of nutrients to plants	Contribution to plant production for food, fuel and fibber	Generation and renewal of soil and sediment structure and soil fertility	Regulation of major biogeochemical cycles	Translocation of nutrients, particles and gases	NC	Provision of clean drinking water	Mitigation of floods and droughts	Contribution to landscape heterogeneity and stability	Erosion control
Lavelle et al. (2006)	Soil specific	NC	Enhancement of primary production	Soil formation	Nutrient cycling	NC	Water supply	NC	Flood control	NC	Erosion control
Barrios (2007)	Soil specific	Nutrient uptake	NC	NC	Nutrient cycling	NC	Water flow	NC	Storage of water	NC	Regulation of soil erosion
Weber (2007)	Soil specific	NC	Production function	NC	Reactor function	NC	NC	Filter function	Buffer function	Carrier function	NC
Swinton et al. (2007)	Agro-ecosystems	NC	Food, fibre	NC	NC	NC	NC	Water pollution	NC	NC	NC
Zhang (2007)	Agro-ecosystems	NC	Food, fibre production	Soil formation	Nutrient cycling	NC	Water provision	Water purification	NC	NC	Soil retention
Sandhu et al. (2008)	Agro-ecosystems	Soil fertility	Food	Soil formation	Mineralisation of plant nutrients	NC	Hydrological flows	NC	NC	NC	NC
Porter et al. (2009)	Agro-ecosystems	NC	Food production	Soil formation	N regulation	NC	Hydrological flow	NC	NC	NC	NC

NC: not considered

Table 2-2: (continued)

Reference	Services attributable to soils							General agro-ecosystems' services			
	Climate regulation			Biodiversity			Resources	Pollination	Culture		
	General	Carbon sequestration	Greenhouse gas production	General habitat	Populations regulation	Recycling actions			General	Recreation	Aesthetics
Daily (1997)	NC	NC	NC	NC	NC	Disposal of wastes and dead OM	NC	NC	NC	NC	NC
Wall et al. (2004)	Modification of anthropogenically driven global change	NC	Regulation of atmospheric trace gases	Vital component of habitats important for recreation and natural history	Control of potential pests and pathogens	Bioremediation of wastes and pollutants	NC	NC	NC	NC	NC
Lavelle et al. (2006)	Climate regulation	NC	NC	NC	Regulation of animal and plant populations	NC	NC	NC	NC	NC	NC
Barrios (2007)	NC	Carbon sequestration	NC	NC	Biological control of pests and diseases	NC	NC	NC	NC	NC	NC
Weber (2007)	Climate regulating function		NC	Habitat function	NC	NC	Resource function	NC	Cultural and historical function	NC	NC
Swinton et al. (2007)	NC	Carbon sequestration	NC	Biodiversity conservation	NC	Odours Health risks	NC	NC	NC	Recreation	Aesthetics
Zhang (2007)	Climate regulation	NC	NC	Genetic diversity	Pest control	NC	NC	Pollination	NC	NC	Aesthetic landscapes
Sandhu et al. (2008)	NC	Carbon accumulation	NC	NC	Biological control of pests	NC	Raw materials	Pollination	NC	NC	Aesthetics
Porter et al. (2009)	NC	Carbon accumulation	NC	NC	Biological control of pests	NC	Raw material production	Pollination	NC	NC	Aesthetics

NC: not considered.

To progress the recent advances made in soil specific ecosystem service frameworks, several remaining limitations need to be addressed. Extending the existing frameworks to show the links between soil natural capital stocks and ecosystem services to provide a more holistic approach would be one of the major challenges (Robinson et al., 2009). Like general ecosystem service frameworks, existing soil ecosystem service frameworks fail to recognise that some processes support other processes which lead to confusion in the wording of the services. For instance, Wall et al. (2004) mention as services the "retention and delivery of nutrients to plants" and the "contribution to plant production for food" (Table 2.2). The first one is a group of processes, whereas the second one is the service. Moreover, existing frameworks tend to ignore a great deal of scientific knowledge that has been acquired about soils and do not acknowledge the complexity of soil functioning. When applying the existing frameworks for valuation, some authors tend to use a one-to-one correspondence between processes and services without acknowledging the complexity of soil processes. Sandhu et al. (2008) and Porter et al. (2009) used similar methodologies for the valuation of ecosystem services from agro-ecosystems, including some soil ecosystem services. For each one of the services they valued, they identified one soil process underlying the service (e.g. soil formation), using one indicator to measure that process (e.g. the population of earthworms). The economic valuation was then based on that single indicator. While the methodology used in both studies helps to illustrate the links between soil processes and properties and the provision of services from soils, limiting each service to one indicator fails to recognise that each soil service is the product of multiple properties and processes. Nevertheless, Porter et al. (2009) did consider a more sophisticated function when dealing with nitrogen regulation, showing that it is necessary to acknowledge that soils are very complex ecosystems. Services are underpinned by more than one process or property and the use of process-based models that capture the scientific knowledge available is required to fully comprehend them (Robinson et al., 2009). Dale and Polasky (2007) argued about general ecosystem service frameworks that "a thorough understanding of how ecological systems function" is needed and that "ideally, it would be useful to have the ability to accurately measure the flow of ecosystem services from agro-ecosystems at several scales of resolution" (Dale and Polasky, 2007, p. 287). Existing frameworks also pay little attention to those factors over which managers of soils have control and therefore have had limited utility as tools to explore the impacts of land uses and practices on the provision of soil ecosystem services.

The limitations of existing frameworks mentioned above highlight the need for a better framework (Haygarth and Ritz, 2009). In the following section, we present a framework for the provision of ecosystem services by soil that addresses some of these limitations.

2.3 Proposed framework for soil natural capital and ecosystem services

The conceptual framework for classifying, quantifying and modelling soil natural capital and ecosystem services (Fig. 2.2) provides a broader and more holistic approach than previous attempts to identify soil ecosystem services by linking soil ecosystem services to soil natural capital. It shows how external drivers impact on processes that underpin soil natural capital and ecosystem services, and how soil ecosystem services contribute to human well-being. This new framework builds on existing frameworks for ecosystem services including the MEA framework (MEA, 2005) and makes a number of original contributions. The framework consists of five main interconnected components: (1) soil natural capital, characterised by standard soil properties well known to soil scientists; (2) the processes behind soil natural capital formation, maintenance and degradation; (3) drivers (anthropogenic and natural) of soil processes; (4) provisioning, regulating and cultural ecosystem services; and (5) human needs fulfilled by soil ecosystem services.

2.3.1 Soil natural capital:

Soil natural capital is defined here as a stock of natural assets yielding a flow of either natural resources or ecosystem services (Costanza and Daly, 1992). Since the flow of services from ecosystems requires that they function as whole systems, the structure, composition and diversity of the ecosystem are important components of natural capital (Robinson et al., 2009). By incorporating the idea of soils as natural capital into the conceptual framework, we provide a more complete picture, as well as infuse soil science knowledge into the discussion. Doing so creates the opportunity to value the natural capital of soils and also to track the changes in these values for a given human use. The natural capital of soils can be characterised by soil properties (Robinson et al., 2009).

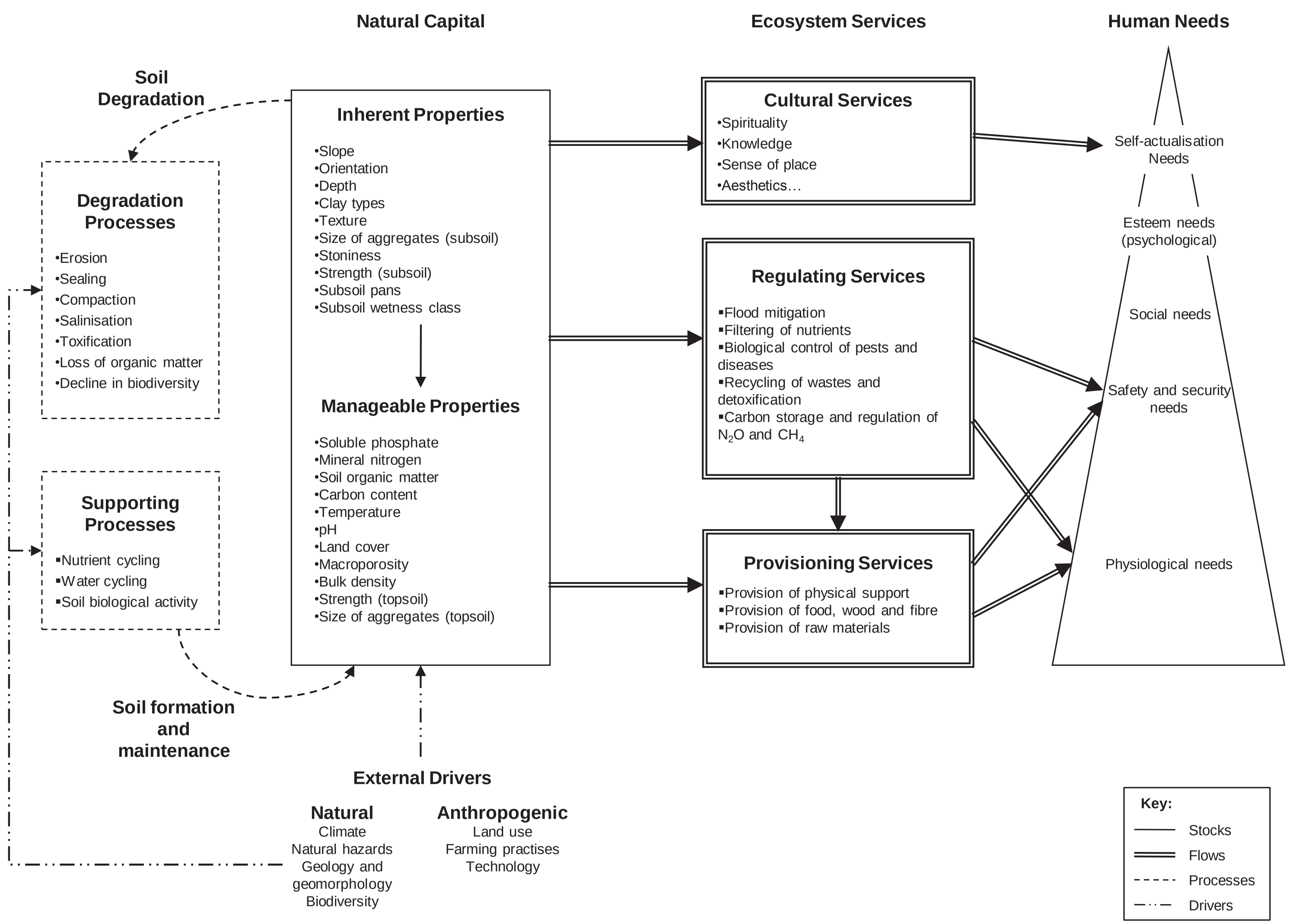

Figure 2-2: Framework for the provision of ecosystem services from soil natural capital.

The idea of soil properties is central to soil science and it is the way in which soil scientists and agronomists describe and characterise soils. As measurable quantities, soil properties enable soil scientists to compare soils on different criteria. The concept of soil properties can be traced back to the 1840s when scientists studied the chemical properties of soils: first, soil weak-acid properties and the capacity to absorb and exchange cations (Way, 1850) and anions, and later the colloidal properties of soil clays and their mineralogy (Schloesing, 1874). In parallel, soil physics was developed as a discipline about soil moisture and water physics, based on the work of Darcy (1803–1858) but also the principles and determination of the grain-size distribution in soils (i.e. clay, silt and sand fractions) that influences both physical and chemical properties. Understanding mechanical properties of soils came later (beginning of the 20th century) with rheology (study of deformation and flow of matter) informing us of the behaviour of soils under stress (Yaalon, 1997).

A soil property can refer to any soil component that can be measured and used to compare or assess soils. For instance, when soils contain stones, the properties related to stones can be size, percentage of stones in soil volume or percentage of stones in soil mass. Soil properties are routinely evaluated in terms of three broad dimensions – physical, chemical or biological. For example, texture is a physical soil property representing the relative proportion of sand, silt and clay in the soil. Texture is a determinant factor of aggregate size and soil structure and is also an indicator of other soil properties like water storage capacity and hydraulic conductivity. Cation exchange capacity (CEC) is a chemical property. It is a quantitative measure of the soil's ability to hold cations, and indicates the quantity of negative charges present per unit mass of soil. CEC is influenced by the amount of organic matter (OM), the types and amounts of clays, and pH (Fig. 2.3). Microbial biomass and its activity are biological soil properties. It refers to the size and diversity of microbial populations associated with organic matter decomposition and nutrient transformations.

Soil properties are interrelated with each other and with soil components (Fig. 2.3). For example, physical properties influence soil moisture content and water movement, which then influence soil chemical and biological properties. In return, soil chemical and biological processes and properties influence physical properties by the production of precipitates and colloids, for example. Properties influence the intensity at which the processes occur and are, at the same time, products of these processes. It is very important when quantifying and valuing the natural capital stocks of a soil that double-counting of soil properties does not occur and there is a clear understanding of the influence soil properties have on soil processes and how they collectively contribute to ecosystem services (Palm et al., 2007).

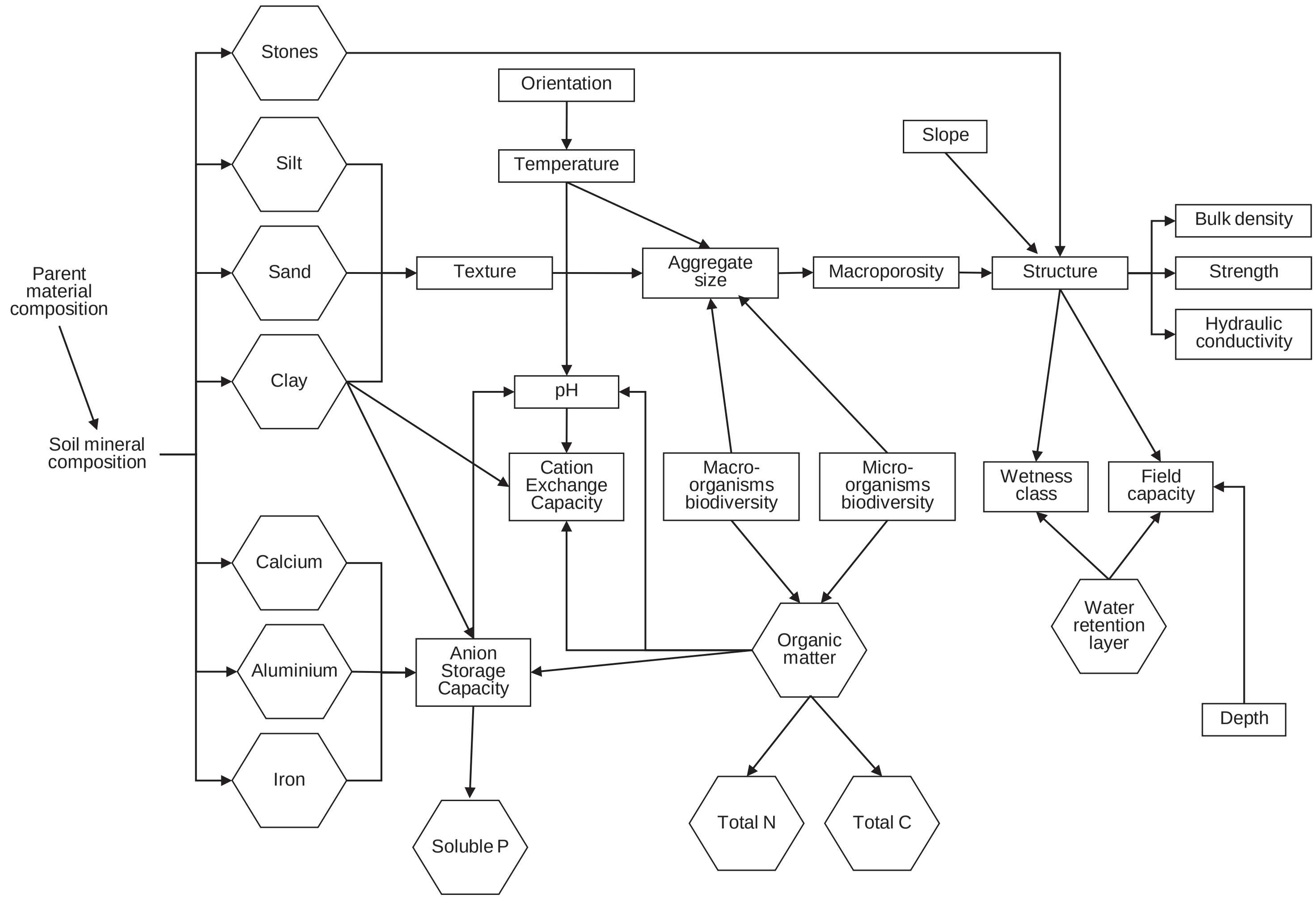

Figure 2-3: Simplified relationships between some soil components (hexagons) and properties (rectangles).

Most of the modern soil classifications are based on the properties of horizons within the soil. Soil classification provides a framework that facilitates communication and understanding amongst pedologists, when there is a prior agreement on concepts. They also make information more accessible to non-specialists. The properties chosen to build up classification schemes are those that can be observable or measured in the field or measured in the laboratory. Those linked directly to use are of particular interest. In the past, climate parameters were utilised in the classification of soils. The World Reference Base for Soil Resources (WRB) (FAO, 1998) is the international standard taxonomic soil classification system endorsed by the International Union of Soil Sciences (IUSS), replacing the previous Food and Agriculture Organisation (FAO) soil classification. The WRB is inspired by modern soil classification concepts, including the United States Department of Agriculture (USDA) soil taxonomy (USDA, 1975), the legend for the FAO Soil Map of the World, the French Référentiel Pédologique, and Russian concepts. The WRB classification is based mainly on soil morphology as an expression of soil formation conditions. Soil classifications and associated properties alone cannot be used for compiling an inventory of soil natural capital stocks and their value. Human use (land use) or purpose must be added to soil classifications before a value can be assigned to the natural capital stocks by quantifying the ecosystem services they provide. For example, a deep stony soil will be suited for grape growing, average for sunflower cropping, and unsuitable for arable cropping because these different crops require different optimal water and drainage conditions. Land use is therefore a very important component of the relationship between soil natural capital stocks, ecosystem services and human welfare (Haygarth and Ritz, 2009). Notwithstanding the difficulties and intricacies of applying soil classification schema to a natural capital and ecosystem services framework, the existence of soil classification systems does provide a rigorous way of considering soil stocks, on which ecological economists and others concerned with managing soil ecosystem services can draw on as a basis for recognising differences between soils.

When describing soil natural capital stocks and the sustainable productive capability of soils, it is useful to make the distinction between inherent soil properties derived from soil formation conditions and those properties that respond to active management (Fig. 2.2). Lynn et al. (2009, p. 86), make the distinction between "permanent, removable and modifiable limitations". Robinson et al. (2009, p. 1906) made a similar distinction between "inherent and dynamic properties". In this chapter, we make the distinction between inherent and manageable soil properties (Fig. 2.2). Inherent soil properties typically include slope, depth, cation exchange capacity, and clay types. They cannot readily be changed without significant modification of the soil, its environment, or without involving prohibitive costs. Manageable soil properties typically include soluble phosphate, mineral nitrogen, organic matter contents and macroporosity (Fig. 2.2). In an ecosystem services management framework, although recognising and taking account of inherent soil properties, the manageable properties assume more practical importance as they provide the opportunity for agronomists, farmers and other stakeholders to optimise the provision of ecosystem services from soils. Knowing what type of properties are involved in the processes and the services they support is therefore essential. For this reason, in putting forward the conceptual framework of soil natural capital and ecosystem services, we put major emphasis on recognising and distinguishing the differences between inherent and manageable soil properties within soil natural capital stocks. The ability to track changes in the inherent properties of soils provides a tool for both industry and policy to separate the effects of short-term management practices from the long-term changes in our soil resources.

A distinction also needs to be made between soil natural capital and added capital, with the latter associated with technologies employed to lift the productive capacity of soils (e.g. irrigation to overcome limited water holding capacity). For this reason, variations in the soil natural capital can lead to very marked differences in land use and farming systems and associated environmental footprint (Mackay, 2008).

2.3.2 Soil natural capital formation, maintenance and degradation:

Soil natural capital, like any type of capital (manufactured, social, human), is formed, maintained and degraded over time. The following section details the processes involved in these phenomena.

2.3.2.1 Soil natural capital formation and maintenance: Supporting processes:

Soils are complex dynamic systems consisting of soil components (abiotic and biotic) interconnected by biological, physical and chemical processes. Soil processes support soil formation, which is the development of soil properties and soil natural capital stocks. Soil processes also form the core of soil functioning and allow the establishment of equilibria and the maintenance of natural capital stocks (Fig. 2.2). What we call here “supporting processes” (Fig. 2.2) are, strictly speaking, categories of processes driving soil natural capital formation and soil functioning. We chose this denomination to relate to the Millennium Ecosystem Assessment framework (MEA, 2005) but we depart from the MEA by talking about supporting processes rather than services. The definitions of the terms given in this chapter allow us to make that distinction since these processes do not directly affect human well-being. Supporting services are similar to the “core ecosystem processes” of Balmford et al. (2011).

The following supporting processes are included in the conceptual framework (Fig. 2.2):

- Nutrient cycling, which refers to the processes by which a chemical element moves through both the biotic and abiotic compartments of soils. Nutrient cycles are a way to conceptualise the transformations of elements in a soil. The transformation, or cycling, of nutrients into different forms in soils is what maintain equilibria between forms, e.g. soil solution concentrations of nitrate drive many processes such as plant uptake, exchange reactions with clay surfaces or microbial immobilisation.

- Water cycling, which refers to the physical processes enabling water to enter soils, be stored and released. Soil moisture is the driver of many chemical and biological processes and is therefore essential in soil development and functioning. The continuous movements of water through soils carrying nutrients disturb chemical equilibria, and thereby drive transformations.
- Soil biological activity: soils provide habitat to a great diversity of species, enabling them to function and develop. In return, the activity and diversity of soil biota are essential to soil structure, nutrient cycling, and detoxification. Biological processes include predation, excretion and primary production among others.

These processes are at the core of soil formation (pedogenesis), building up the physical, biological and chemical stocks of soils. Pedogenesis is the combined effect of physical, chemical, biological, and anthropogenic processes on soil parent material. Soils are formed from the rock materials that make up the earth's crust. Soils can be formed from the underlying bedrock, from material moved relatively small distances (e.g. down slope) or even considerable distances from where the bedrock was originally exposed to the environment. The formation of a soil in these mineral deposits is a complex process. It may take centuries for a developing soil to acquire distinct profile characteristics. Minerals derived from weathered rocks undergo chemical weathering creating secondary minerals and other compounds that vary in water solubility. These constituents are translocated through the soil profile by water and biota. In addition to chemical weathering, physical weathering also takes place. It refers to the disintegration of mineral matter into increasingly smaller fragments or particles. Pedogenic processes, driven by nutrients and water cycles and biological activity, include the accumulation of organic matter, leaching, the accumulation of soluble salts, calcium carbonate and colloids, nutrient redistribution, gleying, and the deposition and loss of materials by erosion, and are very important in soil development and defining soil properties.

Five factors control soil development and natural capital formation: parent material, climate, vegetation, topography, and time (Jenny, 1941). The mineralogy of the parent material influences weathering products and the mineral composition of the soil. Rainfall influences the intensity of weathering and the leaching of weathering products, while temperature will change the speed of chemical and biological reactions. Some indirect climatic effects are through biomass production and rates of organic material decomposition. Species of flora and fauna have a significant effect on the type of soil formed but in time the distribution of flora and fauna depends on climate, topography, and parent material. Landscape relief affects soil formation in different ways, including soil depth, modification of local climate, and available water.

Thus, we saw how, with time, supporting processes gradually build up and create soil properties and ensure the maintenance of the dynamic equilibria underpinning soil natural capital. However, soil natural capital is also degraded over time.

2.3.2.2 Soil natural capital degradation: Degradation processes

The MEA (2005) brought to attention the degradation and loss of ecosystems, but there has been very little recognition of degradation processes in the soil ecosystem services literature (Palm et al., 2007). However, the idea of ecosystem "dis-services" has begun to emerge (Swinton et al., 2007). The notion of dis-service refers to an adverse change in a stock or in a process leading to a loss of ecosystem services. There is a real need to consider the degradation of soil natural capital, and the degradation of natural capital stocks in general, and to identify and quantify the processes behind this degradation because losing natural capital stocks means losing ecosystem services. By limiting soil natural capital degradation, we can act on ecosystem services provision.

Soils can be qualitatively (e.g. salinisation) and quantitatively (e.g. erosion) degraded over time (Palm et al., 2007). Again, this is analogous and conceptually the same as the degradation (or depreciation) of manufactured capital used in national economic accounts and macro-economics. There are a number of types of soil degradation processes: physical, chemical and biological (Palm et al., 2007). Physical degradation processes refer to the structural breakdown of the soil through aggregate disruption. This results in the loss of pore function, which leads to a reduction in surface infiltration, increased water run-off and decreased drainage, in time leading to a decrease in oxygen availability to plants and biota. Physical degradation processes include (Fig. 2.2) (Palm et al., 2007):

- Erosion: the loss of soil material. Soil particles from disrupted soil aggregates or even soil horizons are removed from site by gravity, water, ice or wind. Erosion causes the loss of soil profile, which impacts on soil depth and therefore on the levels of stocks of nutrients and organic matter, for example.
- Sealing and crusting: the formation of a structural seal at the soil surface that crusts once dry. The impact of raindrops causes physical disintegration of surface aggregates. The physico-chemical dispersion of clay particles into pores results in decreased porosity and infiltration. Surface sealing and crusting also prevent seedling emergence.
- Compaction: loss of soil structure leading to lower infiltration, decreased drainage and increased surface run-off. It also reduces the movement of soil gases (O_2, CO_2).

Farming practices including high cow stocking rates or tillage destroy soil aggregates and can lead to the formation of a compacted layer at depth.

- Chemical degradation refers to the processes leading to soil chemical imbalances. Main chemical degradation processes include (Fig. 2.2):
- Salinisation: the accumulation of salts like sodium or magnesium chloride. It lowers the water potential, making water harder to take up by plants. Salt crystals can also destroy roots and breakdown soil aggregates.
- Loss of nutrients by leaching and run-off. It decreases the levels of macronutrients on exchange sites (clays, OM) and in soil solution.
- Acidification: it occurs when cations are excessively leached from soils, when mineralisation is too intense because of soil structure perturbation.
- Toxification: the excessive build-up of some elements (e.g. aluminium, iron) and heavy metals (e.g. mercury, chromium, lead). It can be caused by excessive weathering or industrial activities. In New Zealand, cadmium and fluor can accumulate as by-products of P application.

Biological degradation processes can also degrade the natural capital of soils. The artificial disruption of soil structure (tillage, cattle treading) can lead to excessive activity of the soil biota due to oxygenation and therefore excessive mineralisation of organic matter leading to the loss of structure and nutrients. All the processes mentioned above add to, maintain or degrade soil natural capital. One needs to acknowledge that they can be influenced by a number of drivers, natural and anthropogenic.

2.3.3 External drivers:

Soil processes are influenced by many drivers more or less external to the system where the processes take place. These drivers can come from natural origins or be anthropogenic, influencing soil processes in different ways, including the nature and speed of the processes. The drivers impacting on the inputs to, or outputs of, a system will influence the type of reactions taking place. By influencing soil processes, external drivers will therefore also impact on the levels and nature of soil natural capital stocks (Fig. 2.2). Natural drivers influencing soil processes and natural capital stocks include climate, natural hazards, geology and geomorphology, and biodiversity (Fig. 2.2). Climate has a very significant impact on soil processes and therefore on the provision of ecosystem services from soils. The characteristics of local climate (rainfall intensity, temperature, sunshine) influence supporting processes, degradation processes and biodiversity by driving soil moisture and temperature. Anthropogenically driven climate change therefore impacts on both soil natural capital stocks and ecosystem services. Natural hazards, like earthquakes or volcanic eruptions for example, can change a soil environment (e.g. bury it or compromise the integrity of soil structure at different scales), thereby modifying supporting and degradation processes like water cycling or erosion. The geological origin of the parent material determines the initial minerals in soils that will drive soil development and properties. Geological history, as well as the climate of the area, determines the morphology of landscapes, therefore the undergoing intensity of degradation and supporting processes. Biodiversity is the agent of biological reactions; therefore the type and variety of species present in an area will influence the type and intensity of the biological processes.

Anthropogenic drivers, such as land use, farming practices and technologies, also influence soil processes (Fig. 2.2). The type of land use (e.g. cropping, livestock) determines the type of disturbance (e.g. tillage, treading, use of agrochemicals) as well as inputs (e.g. excrements, synthetic fertilisers) applied to the soil. Farming practices determine the level of intensity of the disturbances (e.g. organic versus conventional cropping) and the amount of inputs to the soil (e.g. quantity and timing of fertilisation). The evolution of technology provides humans with more tools to manage soil processes and the impacts of the pressures applied to the environment, for example, the use of nitrification inhibitors can reduce nitrate leaching losses and nitrous oxide emissions from soils. Soil scientists have been studying the impacts of many of these drivers on soil processes and properties for many years and some areas like the impacts of farming practices and climate on soil properties, are therefore well understood and documented.

We saw that soil natural capital stocks can be characterised by soil properties, that the formation, maintenance and degradation of these stocks are determined by soil processes and that soil processes can be influenced by external drivers. By showing how soil properties and processes link to soil natural capital, the large body of knowledge on soil processes from the soil science literature can be included into the framework for the provision of ecosystem services from soils. In the following section, we detail soil ecosystem services.

2.3.4 Provisioning, regulating and cultural ecosystem services from soils

Ecosystem services are defined here as the beneficial flows arising from natural capital stocks and fulfilling human needs. Soils take part in the provision of a number of ecosystem services that we identified by talking with soil scientists and compiling the literature (Table 2.2). We chose to classify these soil services according to the MEA (2005) model, so the reader can relate to more general ecosystem service frameworks (Haygarth and Ritz, 2009). Soils provide three types of services: provisioning, regulating and cultural services. Provisioning services are defined as “the products obtained from ecosystems” (MEA, 2005, p. 40). Soils specifically provide a number of products useful for humans:

- The provision of food, wood and fibre: Humans use a great variety of plants for a diversity of purposes (food, building, energy, fibre, medicines). By enabling plants to grow, soils provide a service to humans. Soils physically support plants and also supply them with nutrients and water. The natural capital stocks insuring the provision of the service are embodied by soil structure, water holding capacity and nutrients fertility.
- The provision of physical support: soils form the surface of the earth and represent the physical base on which animals, humans and infrastructures stand. Even an otherwise unproductive soil may provide physical support to human infrastructure (e.g. stretches of the Trans-Australia Railway across the Nullarbor Desert**)**. Soils also provide support to animal species that benefit humans (e.g. livestock). The strength, intactness and resilience of soil structure represent the natural capital stocks behind this service.
- The provision of raw materials: soils can be source of raw materials like, for example, peat for fuel and clay for potting. These materials stocks are the source of the service. However, renewability of these stocks is questionable (de Groot et al., 2002).

Soils also provide regulating services which enable humans to live in a stable, healthy and resilient environment. The regulation that these services provide comes from soil processes and their effect on the establishment of equilibria between natural capital stocks. Soil regulating services included in our framework are (Fig. 2.2):

- Flood mitigation: soils have the capacity to store and retain quantities of water and therefore can mitigate and lessen the impacts of extreme climatic events and limit flooding. Soil structure and more precisely macroporosity, as well as processes like infiltration and drainage will impact on this service.
- Filtering of nutrients: if the solutes present in soil (e.g. nitrates, phosphates) are leached, they can become a contaminant in aquatic ecosystems (e.g. eutrophication) and a threat to human health (e.g. nitrate in drinking water). Soils have the ability to absorb and retain solutes, therefore avoiding their release into water. Natural capital stocks of clays and OM, as well as processes like adsorption and precipitation regulate this service and therefore drive the quality of run-off and drainage waters and wider water bodies such as ground water, lakes and rivers.
- Biological control of pests and diseases: by providing habitat to beneficial species, soils can support plant growth (rhizobium, mycorrhizae) and control the proliferation of pests (crops, animals or humans pests) and harmful disease vectors (e.g. viruses,

bacteria). Soil conditions (e.g. moisture, temperature) determine the quality of the soil habitat and thereby select the type of organisms present. This service depends on soil properties and the biological processes driving inter- and intra-specific interactions (symbiosis, competition).

- Recycling of wastes and detoxification: soils can self-detoxify and recycle wastes. Soil biota degrades and decomposes dead organic matter into more simple forms that organisms can reuse. Soils can also absorb (physically) or destroy chemical compounds that can be harmful to humans, or organisms useful to humans. This service depends on biological processes like mineralisation and immobilisation and therefore is also related to the natural capital stocks of nutrients available for soil biota or for chemical reactions.
- Carbon storage and regulation of N_2O and CH_4 emissions: soils play an important role in regulating many atmospheric constituents, therefore impacting on air quality. Perhaps most important is the ability of soils to store carbon as stable organic matter which is a benefit when talking about off-setting greenhouse gases emissions. This service is mainly based on OM stocks and the processes driving them but also on soils conditions (e.g. moisture and temperature) which regulate soil biota activity and thereby the production of greenhouse gases like nitrous oxide (N_2O) and methane (CH_4).

Soil provisioning and regulating services arise at very different scales ranging from microns (habit for micro-organisms) to landscape (flood mitigation) to the globe (air quality).

Notably, none of the previous studies (Barrios, 2007; Daily et al., 1997a; Lavelle et al., 2006; Wall et al., 2004; Weber, 2007) on soil ecosystem services cover or identify “cultural services” (Table 2.2), apart from Haygarth and Ritz (2009). This is a curious omission as soils alone, as part of landscapes that support vegetation, have across many cultures been a source of aesthetic experiences, spiritual enrichment, and recreation. Many deities and religious beliefs refer specifically to the earth and its sacredness and soils also have various cultural uses across the globe from being a place to bury the dead, a material to build houses or a place to store and cook food (Māori hāngi). The point here is not to detail all the cultural services provided by soils but to acknowledge that these services, even if almost always forgotten, are of tremendous consequence.

We have examined services provided by soils and acknowledge that they can be of a different nature, but to complete our framework, in the following section we need to look at human needs and how ecosystem services fulfil them.

2.3.5 Human needs fulfilled by soil ecosystem services:

Ecosystem services exist because they meet a human need. This is the very essence of the anthropocentric concept of ecosystem services. However, few studies in the ecosystem services literature go as far as specifying how and what human needs are potentially or actually fulfilled by ecosystem services. One very notable exception is the Millennium Ecosystem Assessment (2005), which, although not explicitly acknowledging it, shows how ecosystem services contribute to human well-being by using a framework that resembles Maslow's "Hierarchy of needs" (1943). Maslow's (1943) classic study of the so-called "Hierarchy of needs" is the foundation study in this domain. This hierarchy has five levels: the first four levels are deficiency needs: physiological needs, safety and security needs, social (love and belonging) needs, and esteem (psychological) needs; the last level is self-actualisation needs. Deficiency needs must be met first, the individual prioritises them; the higher needs can be considered only when the lower needs are met. Maslow's framework has been widely criticised (Wahba and Bridwell, 1976) on a number of grounds. Probably the most persistent critique is that Maslow's framework is based on a hierarchal structure for which there is a lack of strong evidence. For example, a starving artist may be self-actualised while his/her physiological needs (e.g. food) may be inadequately fulfilled. In this context, Chilean economist Manfred Max-Neef's "matrix of needs" (1992) is perhaps a better reflection of reality. In this framework many needs are complementary and different needs can be fulfilled simultaneously. Max-Neef classifies fundamental "axiological categories" – subsistence, protection, affection, understanding, participation, idleness, creation, identity, and freedom – that are split into four "existential categories" (being, having, doing and interacting), thereby forming a matrix of needs. Ecological economist Herman Daly somewhat bravely presents an even broader contextualisation of human needs, in terms of his "end-means" spectrum (Daly and Farley, 2003). This spectrum links ultimate ends (final cause and "God") to intermediate ends (health, safety, comfort) to ultimate means (material cause, low entropy matter energy). However, whatever philosophical construct of human needs is selected, it is inevitably a poor representation of the complexity, subtlety or ever-changing nature of human needs.

Even though Maslow's hierarchy of needs (1943) is an overly simplistic picture, it's easy to comprehend and thereby enables us to point out that ecosystem services relate to human needs on two different levels. First, at the physical level, provisioning services provide goods useful for the fulfilment of some physiological needs: food, fibre for clothing, sources of energy, and support for infrastructures (Fig. 2.2). Regulating services also fulfil some physiological needs like clean air and clean water by regulating greenhouse gases emissions and filtering water. Moreover, provisioning and regulating services also fulfil safety and security needs by ensuring the stability of human habitat through soil structure stability, flood mitigation, the control of pests and the recycling of wastes (Fig. 2.2). Second, at the non-physical level, ecosystems provide aesthetics, spiritual and cultural benefits through cultural services, thereby fulfilling self-actualisation needs. Again, the fulfilment relationships between services and human needs are not a one-to-one correspondence.

As shown in Fig. 2.2, it should also be noted that some needs in Maslow's hierarchy (1943) (social and esteem needs) cannot be fulfilled by ecosystem services. This is because these needs are only based on our own self-perception of emotionally-based relationships with other human beings (or even animals).

2.4 Conclusion:

This chapter uses Soil Science and the current ecosystem services concepts to develop a framework for classifying and quantifying the natural capital and ecosystem services of soils. The framework shows how soil natural capital stocks can be characterised by soil properties and how the provision of ecosystem services from soils is linked to both manageable and inherent soil properties. We argue supporting processes ensure the formation and maintenance of soil natural capital and that degradation processes drive natural capital depletion. The framework also shows for the first time that both natural and anthropogenic drivers impact on natural capital stocks and soil processes. Such information now enables land managers to link changes of these drivers, e.g. land use, to outcomes and changes in the provision of ecosystem services. Including this scientific knowledge in the framework opens the soil black box and creates the opportunity to value the ecosystem services provided by soils and also to track the changes in these values for a given human use. It also allows, for the first time, the inclusion of differences between soils into broader ecosystem service frameworks.

Throughout the entire thesis, the focus is on soil as the studied ecosystem. The boundary of the analysis is the soil profile, from the soil's surface to the parent material. However, soils are not separable from the vegetation growing on them, which will influence flows of matter and energy, and impact on soil processes and properties. A dairy grazed pasture based system is considered in this study but the principles applied and methodology developed to investigate the provision of services from soils will be applicable to any combination of soil and vegetation type.

In the following chapters, the framework presented is implemented to quantify and value ecosystem services from soils at the farm level. The framework concepts are used to incorporate the vast scientific modern-day understanding of soil processes and taxonomy (Chapters Three and Four) into a process-based model to link the soil biophysical processes and properties at the origin of the provision of each soil ecosystem service to a biophysical measure of each service (Chapters Five and Seven). Doing so enables us to show how soil natural capital, farming practices and soil management impact on the provision of ecosystem services. The quantification of soil services is then paired with an economic valuation of soil services (Chapters Six and Seven) to provide a very powerful management tool for land managers and policy makers to better understand the provision of ecosystem services from soils and weigh more carefully soil natural capital and soil services values in rural development processes.

An edited version of this chapter was published in the journal Ecological Economics in 2010 (Dominati et al., 2010a). The publication triggered a commentary by Robinson and Lebron (2010), which we responded to in the form of another commentary (Dominati et al., 2010b).

Chapter Three

Detailed Framework for Cultural and Provisioning Services provided by Soils

This chapter builds on the framework developed in Chapter Two. The major focus of the chapter is on the provisioning services provided by soils but, it first comments on the cultural services provided by soils. New conceptual thinking for the quantification of each service is developed and presented. The properties, processes and drivers influencing the provisioning services from soils under a dairy grazed system are described and parameters suitable for capturing their dynamics and quantifying them explored and documented. The examples used to describe degradation processes and external drivers impacting on the provisioning services provided by soils are specific to a dairy grazed system. The same steps would be taken in an analysis of the impacts of any other land use.

3.1 Cultural services:

Numerous authors (Daily, 1999; de Groot, 1992; de Groot, 2006; de Groot et al., 2002; Ekins et al., 2003a; MEA, 2005) highlight that ecosystems fulfil both physical and non-physical human-needs. The non-material benefits people obtain from ecosystems are referred by the Millennium Ecosystem Assessment (2005) as "cultural services". They include spiritual enrichment, cognitive development, reflection, recreation, and aesthetic experiences. Soils, as part of landscapes and support to vegetation, provide to many cultures a source of aesthetic experiences, spiritual enrichment, and recreation. The earth and its sacredness are referred to by many deities and religious beliefs. Soils have a diversity of cultural uses across the globe, from being a place to bury the deceased, a material from which to build houses and a place to store and/or cook food (e.g. Maori hangi).

Cultural services provided by the soils of a dairy grazed system could be the aesthetics associated with the farm landscape, the opportunity for on-farm recreation, the spiritual and religious values associated with the farm location and particular soil types, the educational and social opportunities of the farming system, through to the cultural heritage value of the farm site or farming practices.

Cultural services cannot be measured in biophysical terms in the same way as provisioning and regulating services, although the valuation of provisioning and regulating services can utilise some of the tools and methodologies developed for the valuation of cultural services. To fully inform the provision of ecosystem services from soils, cultural services should be considered and included in the valuation scheme. In this thesis, quantification and valuation does not extend to cultural services, but is limited to the provisioning and regulating services, as a first assessment of the framework described in Chapter Two.

In the following section, the properties, processes and drivers influencing the delivery of soil provisioning services from soils under a dairy farm operation are described. The information produced is then used in combination with current knowledge of soil processes and utilisation of process-based models (Chapter Five) to investigate the relationship between soil properties and processes in the provision of soil services. This forms the basis of the operational model for quantification and valuation of these services (Chapter Seven).

The rest of the chapter focuses on the provisioning services provided by soils.

3.2 The provision of food, wood and fibre:

Humans use a great variety of plants for a diversity of purposes. For the last 3 million years man has relied on plants as a source of food. Ten thousand years ago, man discovered how to cultivate crops and domesticate animals. Nowadays, plants are used for food, directly or as forage for farmed animals, as a source of wood for building and energy resources, fibre for clothing, medicines, and ornaments. The availability of a diversity of plants thereby fulfils a variety of human needs from basic physiological needs like food, to higher needs like aesthetics. Soils are the substrate at the base of any cropping or grazing system. By enabling plants to grow, soils provide a service to humans.

To grow, plants require a number of elements provided by soils and the atmosphere. Soils provide plants with nutrients, water and support and also provide a habitat for organisms favourable to plants. The range and quantity of nutrients available in soil solution, the quantity of water available, the capacity of the soil structure and soil biota all strongly affect plant growth. Soil structure and fertility are the natural capital stocks on which plant growth is based and thereby the basis for the provision of food, wood and fibre.

In dairy grazed systems, the provision of food is embodied by legume based pasture growth consumed *in situ* by grazing animals. Pasture yield depends on climate, soil natural capital and a range of farming practices. The consumption of pasture by lactating dairy cows, drives the production of foods like milk and meat, but also other materials from grazing animals like leather (cattle), wool (sheep) or velvet (deer).In dairy farm operations, soil natural capital enables pasture growth, which supports milk production[1]. Efficient milk production is also

[1] It should be noted that milk is not only used as food. Considerable amounts of milk are converted to non-food output such as casein. These outputs can be used in non-food applications like adhesives,

dependent on healthy and productive animals. The provision of all required nutrients from soils through pastures is therefore essential to ensure optimum animal development, health and productivity.

Dairy farms can also include plant species beyond forage, such as wood lots like pine forests, native bush or shelter belts. The uses of these plant species is not considered in this study, but could easily be added to any further analyses.

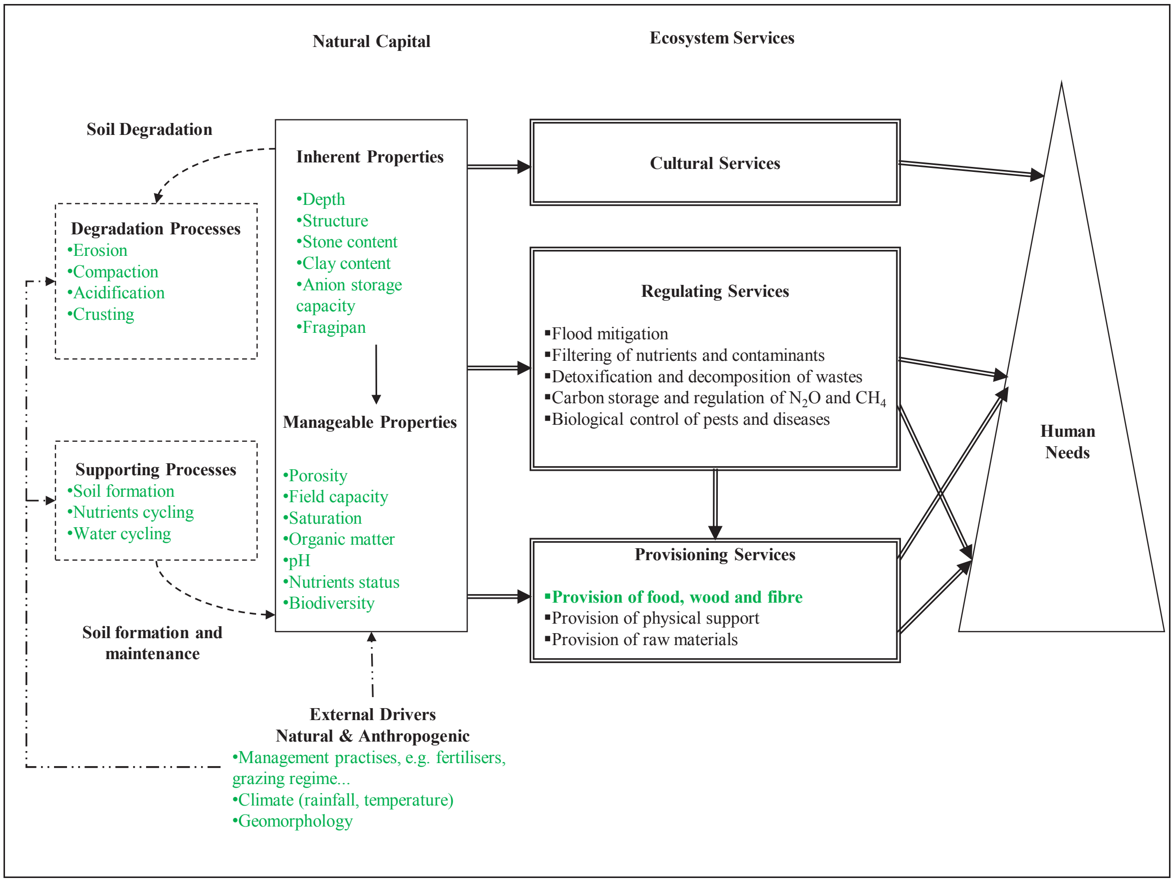

Figure 3-1: Detail of the conceptual framework applied to the provision of food, wood and fibre.

In the following section, the soil properties and processes sustaining pasture growth and animal production that underpin the provision of food from a dairy grazed system are examined (Fig. 3.1). This includes an investigation of the soil properties and supporting processes behind pasture growth and animal health (Fig. 3.1) and then, the drivers and degradation processes impacting on soil natural capital and associated service (Fig. 3.1). Finally, the methodology chosen to quantify the provision of food is presented.

paper coating, leather finishing, pharmaceuticals and synthetic fibres. Therefore, milk products have a considerable added value to the economy.

3.2.1 Soil properties and supporting processes involved in the provision of food:

The soil natural capital stocks, namely soil properties, contributing to pasture growth, as well as the supporting processes regulating these stocks are highlighted in Fig. 3.1. These are shown in greater detail in Fig. 3.2, which highlights the complexity of the interactions between the soil properties behind the provision of food.

To establish the part played by natural capital stocks in the provision of the service, the key properties and supporting processes directly impacting on plant growth and animal health are identified and described. This forms the basis for the quantification of soil services. It is acknowledged from existing Soil Science that soil structure, available water capacity (AWC) and soil fertility are the natural capital stocks supporting pasture growth and pasture quality. This section details the properties and supporting processes behind these natural capital stocks, including how these stocks are formed, maintained, impact on pasture growth and quality, and through grazing animals, on the provision of food.

3.2.1.1 Role of soil structure: the provision of support to plants.

Soil structure is a key natural capital stock supporting plant growth (Fig. 3.1). A soil with a well developed structure enables plant roots to penetrate easily and air and water to flow freely. It allows roots to respire and obtain oxygen and stands physical loading. Soil structure and especially pore size distribution and conductivity, influence the supply of gases, water and nutrients to plant roots thus regulating plant growth. A lack of physical structure, namely a lack of natural capital, can in some circumstances have a very marked negative effect on plant growth, irrespective of chemical fertility (Pande et al., 2000).

The provision of support to plants by soils is embodied by the condition and attributes of soil structure. Soil structure refers to the size, shape and degree of development of aggregation of soil particles, as well as the assemblage of aggregates (McLaren and Cameron, 1990). Soils acquire structure over very long periods of time through pedogenesis. Some aspects of a soil structure are inherent (e.g. depth, texture) and others are manageable (e.g. topsoil macroporosity) (Fig. 3.1). The supporting processes (Fig. 3.1) behind the development of soil structure are pedogenic processes, which include chemical and physical weathering, which slowly form and shape soil aggregates. The shape, size and packing arrangement of soil aggregates define soil pore space. The pore system is essential for drainage, aeration, root growth and habitat for soil biota. The stability of structural aggregates will determine if the pore system stays open or collapses under a physical load. Soil porosity (Table 3.1) is defined as the ratio of pores volume to total soil volume (%).

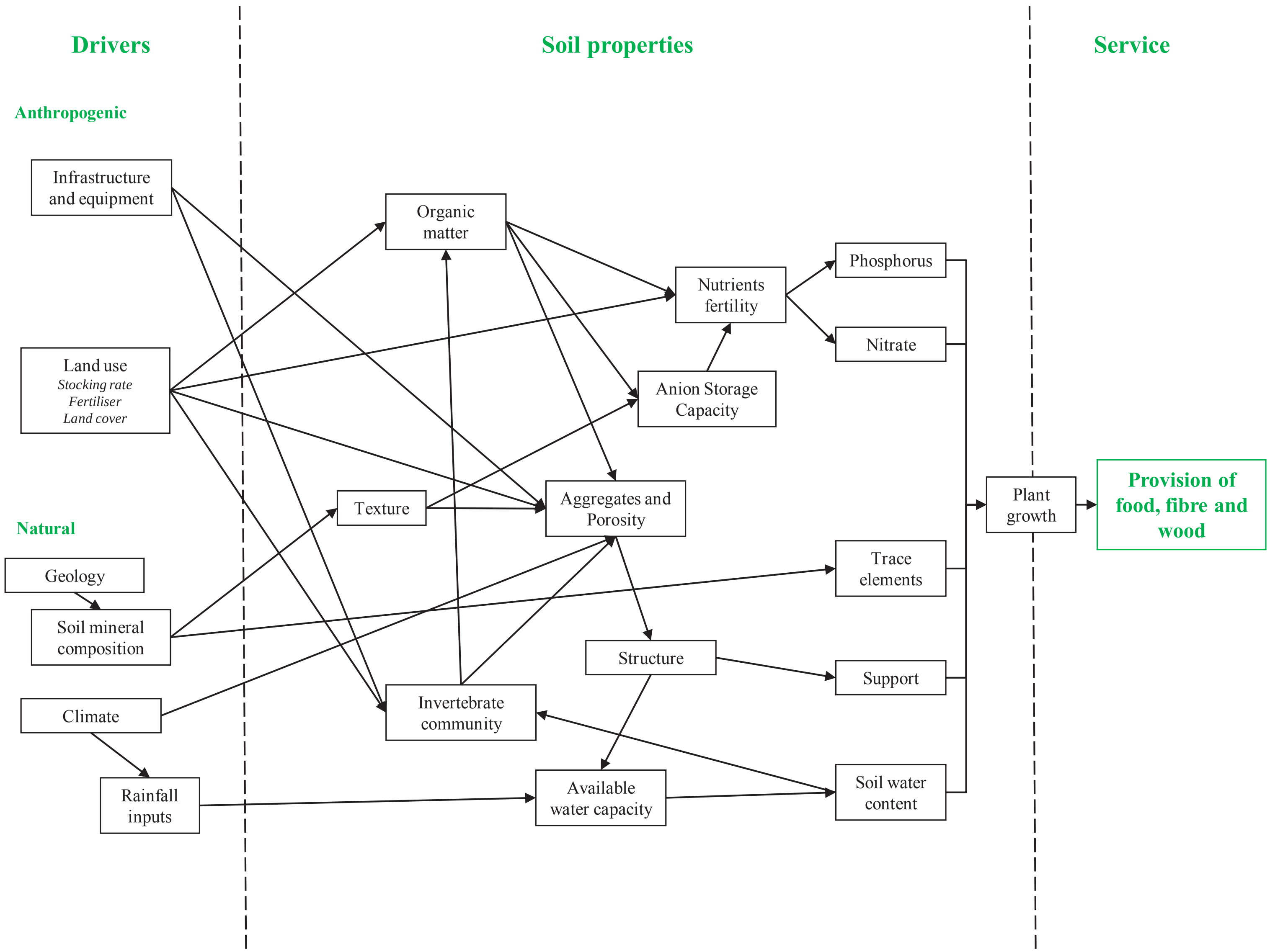

Figure 3-2: Drivers and soil properties influencing the provision of food, wood and fibre.

Table 3-1: Classification of pores according to size and function (from McLaren and Cameron, 1990).

Pore diameter (µm)	Pore description	
> 300	Air pores	Macropores
300 – 30	Transmission pores	
30 – 0.2	Water storage pores	Micropores
< 0.2	Residual pores	

Soil texture (silt, sand and clay fractions) plays an essential role in structure formation since the silt, sand and clay contents influence both the size and stability of aggregates. Formation of soil structure occurs at a range of scales. At the molecular level, water molecules and cations link negatively-charged soil colloids (clay and OM) together. When the soil dries out the colloids are brought together which enables their flocculation – a first step in structure formation. At the microscopic level, sand and silt particles coated with clay or OM are linked into micro-aggregates, making up soil microporosity (Table 3.1). At the macroscopic scale, micro-aggregates are bound together by fungal hyphae (Lavelle et al., 2006), plant roots and other stabilising agents into macro-aggregates, making up macroporosity (Table 3.1).

The supporting processes contributing to soil structure development include (McLaren and Cameron, 1990, p. 66) (Fig. 3.1):

- Accumulation, depletion and cycling of organic matter: OM levels are critical for structure formation and stability. As described previously, OM flocculation and the OM break-down carried out by soil biota are essential processes for structure development;
- Root growth: The pressure applied on soil aggregates by growing roots as well as plant water uptake help to bind soil aggregates. Moreover, root exudates act as glues and stabilise soil aggregates;
- Soil fauna activity: Micro-organisms produce organic glues that help stabilise soil aggregates (Barrios, 2007; Lavelle et al., 2006). Earthworms mix OM with soil, initiating aggregate building, and create pores while burrowing. They also ingest soil, thereby creating very stable earthworm casts;
- Wetting and drying cycles, as well as freezing and thawing cycles: These cycles break down clods producing finer aggregates, but also assist soil aggregation.

Features of a soil with a well developed structure are stable aggregates, uniform pore size distribution, well developed pore continuity and abundance of macropores. Several studies

(Betteridge et al., 2003; Drewry et al., 2008) covering a range of soils indicate that optimum soil macroporosity for maximum pasture and crop yield in the soil top 10 cm is between 10-20%; this ensures plant root penetration and optimum air and water flows.

A number of interconnected soil properties contribute to soil structure (Fig. 3.2). A change in macroporosity can lead to a reduction in pasture growth especially if macroporosity falls under a critical level (Drewry, 2006; Drewry et al., 2008). Macroporosity < 10% often indicates limiting conditions for soil aeration when the soil is wet. Drewry et al. (2008; 2004) showed that when soils are wet in spring, macroporosity at 5-10 cm is a useful indicator to predict pasture yield, with pasture yield increasing 1-5% per unit increase in macroporosity. Alone, a low macroporosity indicates a greater risk of aeration limiting plant growth when a soil is wet (i.e. pores are filling with water). Topsoil macroporosity is a manageable soil property (Fig. 3.1) which farmers can use, through management practices, to improve pasture growth and pore function.

3.2.1.2 Role of available water capacity: the provision of water to plants.

Soil water, or more precisely the soil's capacity to store rainfall and irrigation water as plant available water, is an important natural capital stock for plant growth (Fig. 3.1). Plants need water from soils in order to grow and transpire. Soil solution is the vehicle by which plants acquire nutrients from soils. The total amount of water a soil can store and provide is crucial for plant development, as is the ability of a soil to remove excess water by drainage. Soil structure and more precisely the pore volume and size distribution determine the amount of water the soil can store and move. The amount of water available to plants also depends on the volume of soil explored by plant roots.

Terms used to describe natural capital stocks as they relate to the provision of water, include:

- Saturation: A soil is at saturation before drainage occurs through macropores when water (rainfall or irrigation) has replaced all the air in the topsoil (Hillel, 1980). The saturation point is the amount of water in the soil "when all the pores are full of water and no air remains in the soil" (McLaren and Cameron, 1990, p. 82). For most soils, this is a temporary state as drainage occurs rapidly (in situations where the presence of a water table is not a factor that influences drainage). To a degree, the saturation capacity of a soil is a manageable soil property (Fig. 3.1) influenced strongly by macropore volume (section 3.2.1.1).
- Field capacity (FC): A soil is at field capacity when the water in the macropores has drained, after water application (Hillel, 1980). Field capacity can be described as the state of the soil "after rapid drainage has effectively ceased" (McLaren and Cameron,

1990, p. 83). Field capacity is reached quickly in free draining soils, but drainage time can take up to several days, as drainage speed can be very slow depending on soil texture and structure. FC is often defined by the soil moisture content at a potential between -10 kPa (0.1 bar) and -33 kPa (0.33 bar) depending on the authors (Hillel, 1980). FC depends on soil micropore volume (Table 3.1), an inherent soil property (Fig. 3.1) that cannot be modified.

- Permanent wilting point (PWP): The soil dries below field capacity as plants extract water and water evaporates from the soil surface (Hillel, 1980). Water remaining in the soil becomes more and more difficult to obtain by plants, because it is held in smaller and smaller pores, requiring greater energy for extraction (Table 3.1). The PWP is described as "the amount of water in the soil at which plants are permanently wilted" (McLaren and Cameron, 1990, p. 83) as they can no longer extract water. It corresponds to a water potential of -1500 kPa (15 bars) and varies with soil properties. PWP is an inherent soil property (Fig. 3.1).
- Available water capacity (AWC): This can be defined as "the amount of water which a soil can store for plant growth" (McLaren and Cameron, 1990, p. 83). It is the amount of water held at field capacity minus the amount held at the permanent wilting point (AWC% = FC% - PWP%). Total rooting depth is often used to calculate total available water. All the water within the AWC is not equally available to plants. Water becomes more difficult for plants to absorb the closer the soil water content at the rooting depth moves toward the PWP. Some scientists talk about 'stress point' (SP) when water is still available to the plant but its extraction becomes more difficult, slowing plant growth (Scotter et al., 1979). AWC is an important soil property for plant growth that differs greatly between soil types and that farmers cannot manage.

Another important aspect of the provision of water to plants is the soil's behaviour in case of an excess, or a deficit of water, as it impacts directly on pasture growth, but also indirectly through farm management practices. The volume available for the storage of water depends on soil structure and especially the size range of soil pores and their volume (Table 3.1). Therefore, the properties and processes involved in the development of soil structure mentioned in the previous section (3.2.1.1) are equally important when it comes to soil water storage. Soil texture, structure and biota all influence soil water content (Fig. 3.2). In addition to soil structure, there are a number of other properties that also influence soil moisture:

- Depth of soil: Soil depth defines the overall volume of soil available for water storage. The total amount of water stored in a soil profile, equals the sum of available water stored in each layer. The effective depth of profile for plant uptake actually depends on the depth of rooting of the plant rather than on the soil depth itself.

- Soil profile layering: Distinct textural boundaries affect hydraulic conductivity across soil horizons. Small differences in textures influence the movements of water and the water storage dynamics of a soil. It also influences rooting depth and density.
- Impermeable layers: The presence of impermeable layers like fragipans or iron pan slows down or all together stops water movement in the profile. This renders the volume of soil under the impermeable layer inaccessible for roots and therefore of no utility in water storage.
- Stone content: The volume taken up by stones has no capacity to store water but the presence of stones will assist with drainage by creating preferential flows depending on soil texture.
- Salt content: A high concentration of salts in the soil solution lowers the water potential, making water harder to take up by plants.

The first three properties mentioned above are inherent whereas the last two are manageable (Fig.3.1). The stone and salt content of top soils can be managed, while at large expense to the farmer.

Water circulation in soils is influenced by a number of supporting processes (Fig. 3.1). The speed (rate) at which these processes occur depends primarily on the soil pore structure, volume and connectivity. The processes involved in the water cycle include:

- Infiltration: It is the process by which water enters the soil surface. The infiltration rate affects recharge, and surface runoff and associated sediment and nutrient loss in overland flows (Hillel, 1980; Miyazaki, 2005). Infiltration rates are highly influenced by aggregate stability (structure) and the presence of swelling clays (texture), as well as the condition of the soil surface. Damage to soil surface aggregates from rain drops can lead to sealing and crusting. Soil surface can also exhibit hydrophobicity often referred to as water repellency (Aslam et al., 2009). At saturation, infiltration rates are very low, leading to greater amounts of runoff, increasing the risk of erosion and nutrient losses.
- Preferential flow and drainage: In field soils, cracks, earthworm burrows, root channels, stones and macropores influence water redistribution and can be the vehicle for preferential flows. Water movements in soil are driven by gravity and matric potential gradients which allow pores to fill (Hillel, 1980). If the amount of water applied causes saturation, drainage will occur, emptying the macropores and leaving the soil at field capacity. Water movement is dependent on the structure and texture of each profile horizon: these properties will influence water movement and regulate soil moisture status. Drainage water transports solutes, this is called leaching and it can

result in a reduction of soil fertility and contribute to contamination of surface and ground water. The extent of leaching depends on the volume of drainage and on the bio-chemical and physical processes regulating nutrient concentrations in the soil solution (Miyazaki, 2005). The drainage class of a soil depends on its texture, structure and macroporosity (Table 3.1). Therefore, managing macroporosity impacts on drainage (Fig. 3.1).

- Evaporation: Water evaporates from the soil surface. The evaporation rate depends on climatic factors such as solar radiation, temperature, humidity, wind speed, and on the characteristics of the soil surface. The dry surface layers have a "self-mulching" effect and reduce the loss of water from lower in the profile (Hillel, 1980; Miyazaki, 2005).
- Plant uptake: To satisfy their transpiration demand, plants need water. Plants absorb water by osmotic absorption or mass flow absorption, depending on the rate of transpiration (Miyazaki, 2005). When a soil dries through evapotranspiration, the soil water potential decreases and the gradient between the soil and the plant becomes insufficient. The rate of absorption by plants decreases and eventually the plant starts to wilt.

Water cycle processes mentioned above drive soil water availibility and plant growth, and also take part in supporting the formation and maintenance of soils' natural capital stocks (Fig. 3.1 and 3.2).

3.2.1.3 Role of soil fertility: the provision of nutrients to plants:

Soil fertility, or nutrient status, is another important natural capital stock influencing plant growth (Fig. 3.1). Plants require a considerable number of different chemical elements (between 16 and 20) to grow successfully. These are called nutrients. The elements consumed in the greatest quantities are the non-mineral nutrients, carbon (C), hydrogen (H), and oxygen (O), taken up by plants in the form of water (H_2O) and carbon dioxide (CO_2). Most of the other nutrients needed by plants are mineral-nutrients taken up by plant roots from soil solution. Large differences exist in mineral-nutrient status between soil types. Such differences come from differences in parent material, rates of weathering, leaching, land-uses and management practices. Therefore, part of a soil nutrient status is inherent, from parent material, and part is manageable from inputs (added capital) (Fig. 3.1).

Mineral-nutrients are divided into two groups: macronutrients and micronutrients. Macronutrients are the elements essential for plant growth which are needed in large quantities and include: nitrogen (N), phosphorus (P), potassium (K), and sulphur (S). These macronutrients are usually low in recent or undeveloped soil and thus limit plant growth. Some authors also include calcium (Ca), magnesium (Mg) and sodium (Na) to the list of macronutrients. These three nutrients are usually present in sufficient quantities in soils not to limit plant growth. Micronutrients, also called trace-elements, are elements essential for plant growth, but needed in only very small quantities (ppm). They include iron (Fe), manganese (Mn), copper (Cu), cobalt (Co), zinc (Zn), selenium (Se), molybdenum (Mo), boron (B), chlorine (Cl) and silicon (Si).

Since man started to cultivate land and grow plants the ability of soils to provide nutrients and to sustain sufficient fertility has been the subject of continued studies. The key dimension in nutrient supply is the range of nutrients required, the amount of each nutrient and the availability of each nutrient at different stages of plant growth, to prevent nutrient deficiency and sustain optimum growth, without causing toxicity.

In dairy grazed systems, the provision of food is embodied by pasture growth and quality since pasture is consumed *in situ* by grazing animals. Pasture quantity and quality determine animal growth, health and milk production. Like plants, animals need a range of nutrients to be healthy. The provision of nutrients from soils to animals is supplied through pasture; therefore soil nutrient status is also critical for animal health. Trace-element deficiencies in soils, which include selenium (Se), cobalt (Co) and copper (Cu) (Table 3.2), lead to deficiencies in livestock, which can pose problems like depressed conception rate, and reduced growth rate or nervous disorders (Ellison, 2002; Grace, 1994).

In New Zealand, some soils are deficient in some trace-elements. These are linked to a number of metabolic diseases in livestock (Table 3.2). Some New Zealand soils are naturally deficient in Co and/or Se and/or Cu (Ellison, 2002). Figure 3.3 presents a map of Co deficient soil in the North Island. Se is deficient in about 30% of soils in New Zealand (<0.5ppm) (Fleming, 2003). Livestock grazing pasture grown on these soils may be deficient in one or more of these trace elements, and therefore susceptible to ill-thrift and diseases like ‘white muscle’ disease (Table 3.2). Moreover, the availability of Ca, Mg and I, when limited by different processes, can also lead to a number of metabolic diseases in livestock (e.g. milk fever, grass staggers) (Table 3.2).

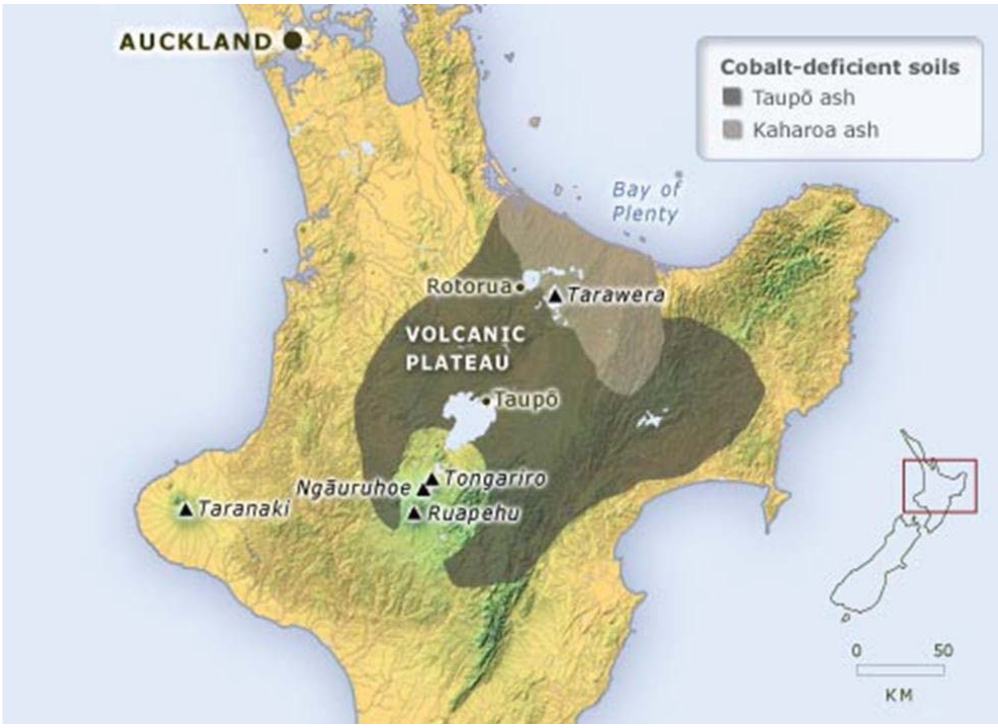

Figure 3-3: North Island cobalt deficient soils (Tonkin, 2010).

Table 3-2: Some dairy livestock pest and disease and their relation to soils (DairyNZ, 2010; Fleming, 2003; Grace, 1994).

Metabolic disease	Damages	Link to soil
Milk fever: Calcium deficiency	Loss of muscle function	Soil availability of Ca impact on grass quality and thereby on animal intake
Ketosis: Hypoglycemia	Low energy, lower production	Soil nutrient status and soil water content impact on grass quality
Grass staggers: Magnesium deficiency	Hyper excitability, convulsions and even death	Soil availability of Mg impact on grass quality and thereby on animal intake
Selenium deficiency	White-muscle' disease (weakness or failure of skeletal or cardiac muscle), ill-thrift, depress conception rate, calf survival, and growth rate, stiffness	Soil availability of Se impact on grass quality and thereby on animal intake. Deficient soil orders are Pumice, Peat, Podzols, Gley and Sands
Cobalt deficiency	Ill-thrift, depression of growth rates.	Soil availability of Co impact on grass quality and thereby on animal intake. Deficient soil types are pumice, soil from granite parent material, sands, and leached soils.
Copper deficiency	Nervous disorder, depress fertility and calf survival, bone and join abnormality	Soil availability of Cu impact on grass quality and thereby on animal intake. Deficient soil orders are Peat, Podzols, Sands and Melanic.
Iodine deficiency	Reduction in the levels of hormones produced by the thyroid (goitre), weakness at birth, hair defects, infertility	Soil availability of I impact on grass quality and thereby on animal intake.
Cu * Mo * S interaction	Increasing the intake of Molybdenum (Mo) in the presence of sulphur (S) can dramatically reduce the absorption of Cu, leading to Cu deficiency.	Pasture Mo concentrations can increase several folds during the late winter and early spring.

A number of soil properties (Fig. 3.2) directly influence the quantity of readily available nutrients, including:

- Parent material: It will determine the types and amounts of nutrients originally present in the soil. Trace-element deficiencies are high ly correlated to soil parent material. Soils with granite parent material are generally low in Co and Se (Grace, 1994). Soils formed on sands are low in Cu, whereas soils from basaltic origin are high in Co (Grace, 1994),
- Degree of soil development: An old weathered, leached soil will have less readily available nutrients than a soil in the early stages of weathering,
- Soil retention capacity: The retention capacity of a soil comes from different origins, mineral and organic. Soil clays and OM form an exchange surface on which charged nutrients can get sorbed and therefore act as a buffer to regulate soil nutrient content. Soils scientists use cation exchange capacity (CEC) and anion storage capacity (ASC) to measure a soil retention capacity. CEC is a quantitative measure of the soil's ability to hold exchangeable cations. It indicates the quantity of negative charge present per unit mass of soil, that is the quantity of sites able to hold cations (McLaren and Cameron, 1990). ASC estimates the soil's capacity to sorb anions. ASC used to be referred to as P retention capacity because it is the soil's ability to retain phosphate ($H_2PO_4^-$, HPO_4^{2-}) that is tested,
- Soil pH influences the availability of nutrients by driving ion exchange reactions with charged soil particles,
- Soil salinity: The excessive levels of salts (sodium, calcium or magnesium chlorides or sulphates) in soils leads to alkaline pH which impacts on nutrient availability,
- Soil biota: The diversity of the invertebrate and microbial community and their trophic structure drive nutrient cycling, and the availability for plants of nutrients, regulated by the organic fraction.

Stocks of macronutrients and micronutrients in soils can be found in different forms:

- Soluble forms in soil solution and readily available to plants,
- Labile forms readily able to move into solution (weakly adsorbed inorganic forms, soluble precipitates and easily mineralised organic forms),
- Non-labile forms insoluble and unavailable to plants (soil biomass, undecayed plant and animal residues, stable OM (humus), forms strongly absorbed and/or occluded by hydrous oxides, forms fixed on silicate minerals and insoluble precipitates).

These stocks of macronutrients and micronutrients can be quantified (ASC, CEC) but their availability to plants is highly dynamic and depends on a number of supporting processes (Fig.3.1). Nutrient cycling insures the formation and maintenance of that natural capital (Fig.3.1). Nutrients can take different forms (e.g. mineral, organic) more or less available to plants. Soil processes drive the transformations between these forms and regulate soil solution concentrations and nutrient availability (McLaren and Cameron, 1990). They include:

- Inputs to soils: The amount and type of organic and chemical inputs to soils like plant litter, animals faeces or fertilisers, influence the amount and type of nutrients present in soils, and available for nutrient cycling and plant growth.
- Plant uptake: When plants take up nutrients from the soil solution, it changes soil solution concentration and equilibriums, causing cation and anion exchange reactions. Nutrients that are transported by the transpiration stream can accumulate at the root surface.
- Symbiotic fixation: Some plants (legumes) have a symbiotic relationship with bacteria (rhizobium). The plants roots have nodules containing the bacteria (rhizobium) that can fix atmospheric di-nitrogen (N_2) and convert it to plant available N. The legume supplies the bacteria with carbohydrate in exchange for N.
- Leaching: Water draining through the soil can leach solutes, which results in a reduction of soil fertility and the contamination of surface and ground water.
- Gaseous losses: Some nutrients (e.g. nitrogen, sulphur) can be transformed into gaseous forms by biological or chemical reaction, and then escape the soil by diffusion.
- Inorganic adsorption: Some nutrients are held tightly on the charged surface of clays and OM, rendering them more or less unavailable for plants. Availability depends on conditions like soil water content and solution concentrations.
- Dissolution / precipitation: Nutrients can sometimes precipitate depending on the diversity of elements present and soil concentrations. The dissolution of the precipitates will depend on soil conditions.
- Mineralisation / immobilisation: Organisms living in soils (e.g. bacteria, fungi, macrofauna) remove soil nutrients to function. These nutrients are returned to the soil pool when the organisms die (Barrios, 2007). The OM entering the soil is degraded by micro-organisms (mineralisation) who transform it into more labile forms. These forms can be further mineralised into simpler, readily useable forms of nutrients or transformed – immobilised – into non-labile OM and vice and versa (Barrios, 2007; Lavelle et al., 2006).

Soil biota is a highly dynamic natural capital stock (Fig. 3.1) whose activity has enormous consequences on soil fertility and nutrient cycling (Fig. 3.1 and 3.2) and thereby on plant growth and the provision of other services.

Soil biota, by recycling dead OM (wastes or plant litter), is a main agent in nutrient cycling. A wide diversity of organisms live in the soil, ranging in size from large animals (rabbits) to microscopically small ones (bacteria) (Fig. 3.4). The species and number of animals vary greatly between soils. Micro-organisms and earthworms make the bulk of the soil fauna biomass.

Each organism has a different role in nutrient cycling and plays a different role in the decomposition of OM and wastes. The amount and quality of inputs to the soil impact on the type and abundance of trophic groups and therefore on decomposition pathways and the efficiency of decomposition and nutrient cycling.

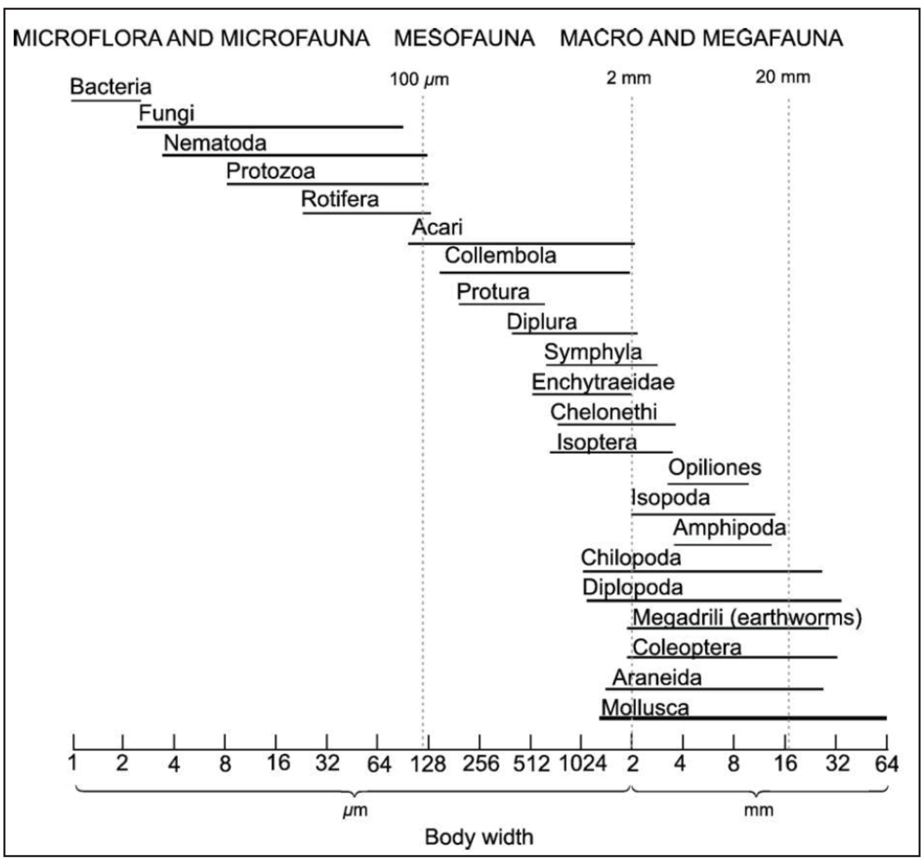

Figure 3-4: Body width of soil fauna (from Swift et al., 1979).

Macrofauna species (body diameter >2 mm) like earthworms constitute an important group for nutrient cycling. They require reasonably moist conditions, satisfactory aeration, and depend on a constant supply of OM and calcium. They play an important role in the initial incorporation and mixing of surface applied material including dead plant roots, plant litter and animal dung, which they digest or mix, thereby starting the recycling of nutrients. Their burrowing activity has important effects on the physical properties of the soil. It promotes aeration and drainage (Schon et al., 2010c). They are often referred to as ‘ecosystem engineers’ (Fig. 3.5).

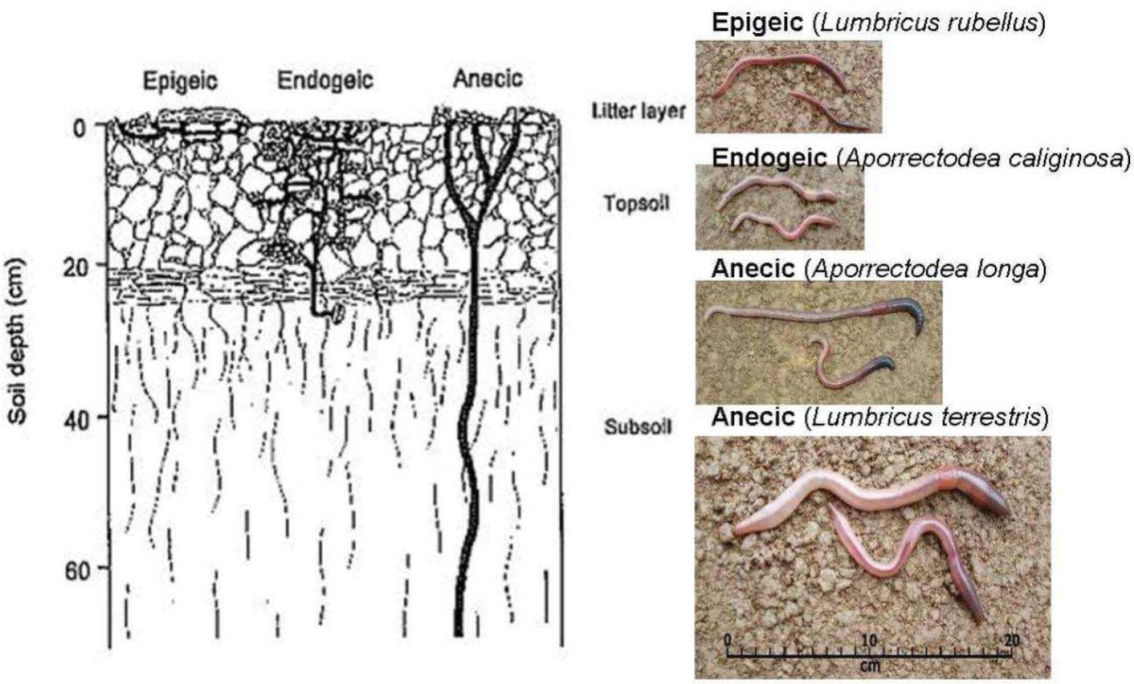

Figure 3-5: Earthworm functional groups (from Fraser and Boag, 1998).

There are three earthworm functional groups (Fig. 3.5). Epigeic earthworms feed on plant litter and dung on the soil surface and do not form permanent burrows. Endogeic earthworms inhabit the mineral soil horizons and ingest soil, feeding on the humified organic material within. They form semi-permanent burrows in the topsoil which have few openings to the soil surface, as they don’t feed on the surface. Anecic earthworms draw plant litter and dung from the soil surface into their burrows and feed on it underground. Their burrows are deep and permanent or semi-permanent (Bardgett and Cook, 1998; Schon, 2010). Epigeic and anecic earthworms are particularly useful in organic matter incorporation. Endogeic and anecic earthworms are important for soil structure and porosity. Earthworm casts have an extremely stable structure, contain an intimate mixture of organic and mineral matter, and are extremely rich in soluble nutrients that can return to soil solution and availability to plants (Syers et al., 1979).

The soil meso fauna (body diameter 0.1–2 mm), is dominated by Acari and Collembola. The soil contains a considerable number of these animals (Schon, 2010). They are involved in litter incorporation and the breakdown of organic material and are especially important in soils with low earthworm numbers. Some species are extremely efficient for soil mixing. A high proportion of soil Acari are represented by longer-lived general detritivores Oribatida, and shorter-lived predatory Mesostigmata. Collembola are shorter-lived general detritivores (Schon, 2010).

The soil micro fauna (body diameter <100 µm) represents a considerable proportion of the soil biomass (McLaren and Cameron, 1990). It is dominated by nematodes which are also extremely important in OM recycling. Nematodes are represented in all trophic groups within the decomposer food-web (Yeates and Pattison, 2006). They feed on plant, microbial and animal remains. Many are parasitic on larger animals like earthworms and insects, some are crop pests (Yeates, 1999). Bacterial-feeding and fungal-feeding nematodes are very important for nutrient cycling, while predatory and omnivorous nematodes, in turn, regulate the populations of other micro fauna (Yeates and Pattison, 2006).

In addition to nematodes, soil micro fauna taking part in nutrient cycling includes:

- Bacteria: They are single-celled organisms. They are the most numerous of all the organisms in soil. They exist in colonies. They are aquatic organisms and live in the film of moisture surrounding soil particles. Most species of soil bacteria are heterotrophic, obtaining both energy and C from organic material. They release CO_2 and nutrients through the breakdown of plant and animal residues. Their population is higher in topsoil. A small number of species are autotrophic: they obtain their C from CO_2 and their energy from the oxidation of various mineral constituents.
- Fungi: They represent the largest part of the total microbial biomass. They grow as filaments or hyphae, are heterotrophic and one of the most important agents in OM decay and nutrient cycling. They are extremely efficient for the decomposition of very resistant compounds (cellulose, lignin). Their hyphae (filaments) play an important role in soil aggregate building and stabilisation. Some species can live in a permanent structural association with living roots known as mycorrhizae, where the association is beneficial for both the plant and fungus.
- Actinomycetes: These organisms are related to both fungi and bacteria. They play an important role in the decomposition of plants and animal residues. Some species are even more effective than fungi. They are aerobic organisms and can survive in soil of low moisture content.
- Algae: These organisms contain chlorophyll and are photosynthetic. They are early colonisers of developing soils. Since they require light to function, they are situated in the very topsoil. Some blue-green algae are able to fix atmospheric N.
- Protozoa: They are unicellular aquatic organisms which feed on other soil micro-organisms.

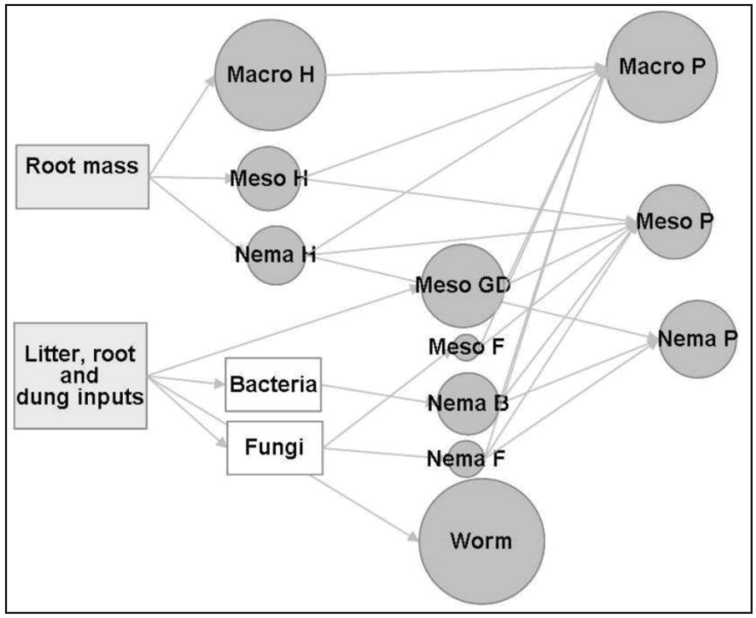

Figure 3-6: Food-web of selected soil faunal groups. Macrofauna (Macro), mesofauna (Meso), nematode (Nema), herbivores (H), earthworms (Worm), general detritivores (GD), bacterial-feeders (B), fungal-feeders (F), predators (P) (Schon, 2010).

Macrofauna, mesofauna and nematode herbivores feed on plant material (Fig.3.6). Earthworms, general detritivores, bacterial-feeders and fungal-feeders feed on detrital inputs and associated micro flora. These perform the initial steps of the recycling of nutrients from decaying and dead OM and dung. These organisms are also very important in waste decomposition. Macrofauna, mesofauna and nematodes are in turn consumed by predators of each group (Schon, 2010).

The quantity and quality of food resources entering the soil food-web, that is the amount and type of dead OM and wastes, change the trophic structure of the invertebrate community (Bardgett, 2005). The quality of food resources increases the relative dominance of bacterial-mediated decomposition over fungal-mediated decomposition in the soil food-web (Schon, 2010). The bacterial-decomposition pathway is associated with faster nutrient cycling in the short-term, but may also be associated with greater nutrient leaching from the soil (Bardgett, 2005). To achieve a maximum decomposition of OM and efficient nutrient cycling, a diversity of organisms is needed.

Soil biodiversity is a central and critical component of soil natural capital as it drives and supports many soil processes and impacts on numerous soil properties (Fig. 3.1 and 3.2). Soil biodiversity is to some extent manageable through the artificial introduction or removal of some species (e.g. earthworms), the quality of litter inputs to the soil, and the degree of physical disturbance (Fig. 3.1). However, the extremely complex interactions between different species are still not well understood. There is an added complexity in New Zealand as many of the Macrofauna found in our pasture soils are accidental introductions. Their distribution is therefore patchy (Springett, 1992).

The natural capital stocks embodied by soil structure, AWC, nutrient fertility and soil biota are at the core of plant growth enabling soils to provide food to humans through plants and farm animals. These natural capital stocks are sensitive to a number of degradation processes (Fig. 3.1). Degradation of natural capital stocks could result in reduced plant growth and decreased service provision.
Likewise, a number of external drivers can impact on the properties and supporting processes behind soil natural capital (Fig. 3.1) and also affect plant growth. These degradation processes and drivers are examined in the next section.

3.2.2 Degradation processes and drivers impacting on the provision of food:

There are a number of degradation processes and external drivers that can affect the soil natural capital behind the provision of food, including soil structure, available water capacity, nutrient fertility and soil biota, and supporting processes (e.g. nutrient cycling and water cycling), thereby impacting on plant growth (Fig. 3.1). In this section, these degradation processes and external drivers are identified and discussed.

3.2.2.1 Degradation processes affecting the provision of food:

Since the components of soil natural capital (soil properties) are interrelated, the processes degrading one property have repercussions on many other properties and thereby on the provision of services (Fig. 3.2). For example, the processes that degrade soil structure also affect available water capacity, soil biota and nutrient fertility by impacting on aeration, water movement, habitat, roots growth and so forth. Similarly, processes that deplete soils of nutrients impact on soil biota and structure and thereby on plant growth. The degradation processes impacting on natural capital stocks and supporting processes underpinning plant growth are:

- Erosion: The less stable soil structure is, the more the soil will be prone to loss of intactness and erode. If soil aggregates are strongly bound together it is more difficult for water or wind to tear off soil particles. The removal of topsoil material by erosion means loss of natural capital including loss of structure, loss of OM, loss of nutrients available for plants, loss of soil volume for water storage, loss of biodiversity and so forth. Erosion can take massive proportions up to the complete removal of the soil profile.

- Compaction: It affects soil structure and refers to a loss and collapse of pores. Compaction is described as “the compression of an unsaturated soil body resulting in a reduction of the fractional air volume” (Drewry et al., 2008; Hillel, 1980). Compaction decreases soil porosity, particularly the volume of the large inter-aggregate pores (macropores). Compaction is due to the application of pressure at the soil surface resulting in the collapse of soil aggregates and the closure of soil macropores. Compaction occurs more easily when soil is moist (Houlbrooke et al., 2009). A reduction in porosity will impact on seedling emergence, root penetration, air and water movement and nutrient availability (Fig. 3.2), impacting on many biochemical processes, soil biota and plant growth. Soil compaction has a strong effect on soil fauna (Schon et al., 2010d) directly because of, e.g. the destruction of a number of animals by livestock treading, and indirectly by the loss of habitable pore space (Fig.3.2).
- Pugging also affects soil structure: It is the deformation of topsoil. Soil pugging involves the deformation and remoulding of soil (Houlbrooke et al., 2009). In grazing systems, pugging occurs when the animals hooves penetrate the topsoil deeply and deform the soil (Houlbrooke et al., 2009). When the soil is pugged, pasture plants can also be directly damaged, buried or uprooted (Betteridge et al., 2003). Immediately beneath the depth where the hoof penetration stopped, compaction can occur. Depending on the intensity of the treading, pugging damage to the soil structure can take a long time to recover. Pugging, like compaction, affects soil porosity and decreases air and water movement and nutrient availability (Mackay et al., 2010) and thereby plant growth (Houlbrooke et al., 2009).
- Sealing and crusting – the formation of an impermeable layer at the soil surface makes water infiltration more difficult and thereby impacts the soil water content. If the soil water cannot be recharged, plants wilt.
- Hydrophobicity often referred to as water repellency (Aslam et al., 2009) slows or prevents water movement from the soil surface into the soil, depending on macroporosity (Robinson et al., 2010).
- Loss of OM: OM levels are essential in the development of soil structure (Barrios, 2007). An excessive loss of OM will lead to a more fragile soil structure and a higher sensitivity to degradation. OM is also a source of nutrients for plants.
- Loss of biota: The abundance and diversity of soil invertebrates play a critical role in nutrient cycling and in sustaining pore structure (McLaren and Cameron, 1990; Schon

et al., 2008), therefore, a loss of biota will lead to a decreased nutrient turn-over and lower nutrient availability for plants.

- Leaching: When water drains through soil, it carries solutes and nutrients, removing them from the soil profile. The resulting reduction in soil fertility decreases plant growth because of nutrient shortages.
- Chemical processes like salinisation or acidification can make soil water harder to take up by plants and also change soil solution equilibriums. The accumulation of chemicals in soils (heavy metals, salt) can lead to toxic levels for plants. Soil chemical equilibriums also drive the availability for plants of trace-elements. For example, high soil pH (basic) makes Cu and Co unavailable for plants. Similarly, high Mn, Ni or Fe content within the soil decreases Co availability (Fleming, 2003).

The occurrence and intensity of some of these degradation processes can be modified by management. The following section presents the drivers, natural and anthropogenic, affecting supporting and degradation processes and thereby soil natural capital.

3.2.2.2 External drivers affecting plant growth

A number of external drivers affect soil structure, AWC, nutrient fertility, and biota, and thereby plant and animal growth and the provision of food, by acting on the frequency and intensity of supporting and degradation processes (Fig. 3.1). Figure 3.2 shows where some of these drivers impact on soil properties.

Natural drivers like geology and climate influence the soil natural capital stocks supporting plant growth (Fig. 3.1). The mineral composition of a soil parent material and the proportions of silt, sand and clay (soil texture) constitute the basic elements for structure formation and impact on available water capacity (AWC). The amounts of inorganic soil colloids (clays and oxides and hydrous oxides of iron, aluminium and manganese) are particularly important because they act as binding agents to form soil aggregates and stabilise soil structure. The mineral composition of the soil parent material determines the quantities of mineral-nutrients in the soil, and especially the quantities of trace-elements. This determines underlying soil nutrient fertility.

Climate drives wetting and drying cycles, as well as freezing and thawing cycles which impact on structure and AWC. Rainfall and temperature drive soil water content and soil temperature respectively. This impacts on soil faunal activity and a wide range of biochemical processes that influence aggregate stability through to soil nutrient status.

Anthropogenic drivers like land use and farming practices also influence soil structure, AWC, nutrients fertility, and the invertebrate community directly and indirectly (Fig. 3.2). For example, tillage is known to impact on soil structure from reducing infiltration to creating a barrier preventing soil roots to penetrate. Soil structure is also sensitive to compaction and deformation (degradation processes) due to tractor wheels and to the treading action of livestock. The sensitivity to livestock treading is highly related to soil moisture and is maximised during the wet months of the year (May to October). A well drained soil will have limited susceptibility to treading damage because it will be wet for shorter periods. To limit treading damage, farmers can consider different management options: they can monitor the type and number of animals per hectare, drain wet soils to remove excess water, consider off-pasture standing areas to avoid treading when the soil is wet, feed-pads to allow soil structure recovery, or even grazing the animals off the farm. These practices limit the effects of treading damage (degradation process) and allow time for supporting processes to take place and soil to recover. Soil structure has been shown to recover from compaction faster in summer and autumn than winter due to drying and cracking processes (Drewry, 2006).

Farming practices can also influence soil nutrient status directly through fertiliser application (N, P, lime), and indirectly through the choice of plant species grown, the removal of plant material (crops, grazing), and the number and type of animals, which drives dung and urine inputs to soil. Inputs of nutrients and OM to soils impact on soil biota and supporting processes like nutrient cycling (Fig. 3.1). The main nutrients applied by dairy farmers in New Zealand are nitrogen (N), phosphorus (P), potassium (K), sulphur (S) and magnesium (Mg) to correct nutrient deficiencies. Most N inputs to the soil come through biological N fixation by legumes. Many New Zealand soils have a low natural labile P status, i.e. P that is readily available to fast-growing plants such as legumes and grasses (Parfitt et al., 2008a). This is also the case for S. Farmers have dramatically raised the P and S status of soils with P and S fertilisers, which in turn, enables clover to flourish and raises the nitrogen (N) status of soils, to the point where grasses become more competitive. A legume based pasture system is a "self-regulating system where clover and grasses compete for resources such as light and water" (Parfitt et al., 2008a, p. 37) so there is an upper limit on production. Additions of N fertilisers allow yields to be lifted further. Topdressings of pastures with Se, Co or Cu are also common practice in New Zealand. It is an effective method of providing additional trace-elements to animals. The frequency of applications depends on soil deficiency as well as the element applied (Grace, 1994). Topdressing with Se can maintain adequate pasture levels for up to 24 months, whereas topdressing of Co can be very efficient on pumice, but only elevate pasture Co levels for 6-12 weeks (Fleming, 2003). Specific technologies can also be employed to directly control some soil processes, e.g. nitrification inhibitors are used to prevent the transformation of ammonium

(NH_4^+) into nitrate (NO_3^-) to lower the risk of nitrate leaching and nitrous oxide (N_2O) emissions.

Production technologies and farming practices have been very successful in providing plants and animals with nutrients and water, overcoming soil limitations and reducing differences between soil natural capital stocks. Indeed, where resources become scarce, it is important to have a good understanding of the production levels that can be sustained by the soil natural capital and what part of the production comes from added capital. By quantifying the soil natural capital and ecosystem services, the value of these services, which differs significantly between soils, can be determined.

3.2.3 Quantifying plant growth and the provision of food from soil natural capital stocks:

3.2.3.1 Previous attempts to quantify the provision of food from soils:

Only a few studies have attempted to investigate ecosystem services provision from soils (Table 2.2, Chapter Two) and among these only two (Porter et al., 2009; Sandhu et al., 2008) have attempted to use soil properties to inform the provision and valuation of soil services.
Here, the processes and properties behind plant growth and the provision of food are investigated and modelled to separate the production due to natural capital from the part that is attributed to added capital (e.g. fertilisers). Porter et al. (2009) and Sandhu et al. (2008) valued the services directly using production levels. There are two issues with their methodology. First, there is a problem of double accounting. Both Porter et al. (2009) and Sandhu et al. (2008) in their studies of ecosystem services from agro-ecosystems talk about food production as a service. They consider crop yield as an indicator for the service, but in Porter et al. (2009) 'N regulation', 'soil formation' and 'hydrological flows', and in Sandhu et al. (2008) 'soil formation', 'hydrological flows', 'nitrogen fixation' and 'soil fertility' stand as separate ecosystem services and each of these are valued in addition to the food provision service. It is argued here that these 'services' are not directly fulfilling human needs and, as such, are only supporting processes contributing to a service (Fig. 3.1). These processes, termed services by Porter et al. (2009) and Sandhu et al. (2008) are critical for supporting plant growth, but are not services as such. As mentioned above, plant growth and therefore the provision of food depends on soil structure and fertility, but it is argued here that putting a value on the processes contributing to plant growth, in addition to putting a value on plant yield, is double accounting, with respect to the provisioning services (Chapter 4). Second, Porter et al. (2009) and Sandhu et al. (2008) do not differentiate the contribution to production of soil natural capital from the contribution of added capital. The failure to unmask the differences between these two

contributions raises the risk of overestimating the value of the service. The methodology employed by Porter et al. (2009) and Sandhu et al. (2008) does recognise that services come from natural capital stocks, that is soil properties, but does not, in our opinion, do justice to the relationships between soil properties, processes and services. The study undertaken here represents a major shift from previous attempts to model and inform the provision of food from soils. It avoids double accounting, and importantly allows an examination of the influence of a change in soil properties on the provision of services.

Because of the interrelationship between soil structure, AWC and nutrient fertility, as they influence plant growth, their contributions to the provision of food can only be explained in a dynamic environment. Process-based models that link dynamically soil properties and processes to the dynamics of plant growth provide a very useful tool for examining in detail the influence of these three properties on the provision of food.

3.2.3.2 Parameters chosen to quantify plant growth and the provision of food:

In this thesis, measures of soil natural capital stocks form the basis of the quantification of soil services. This new methodology is applied to the quantification of each service. To model plant growth and the provision of food, natural capital stocks and soil processes need to be dynamically linked to pasture growth (Fig. 3.1). The parameters chosen to inform each of the natural capital stocks behind pasture growth are soil structure, AWC and nutrient fertility[2].

<u>Macroporosity: a parameter to inform soil structure and the provision of support to plants:</u>

Macroporosity was chosen to examine the influence of soil structure on providing support to plants. Drewry et al. (2004) showed that macroporosity at 0-5 and 5-10 cm was a useful indicator for predicting spring and summer/autumn pasture yields. They suggested a linear response between macroporosity and spring pasture yield, with a 1-5% increase in yield for every 1% increase in macroporosity. Other studies (Betteridge et al., 2003; Drewry et al., 2008; Houlbrooke et al., 2009) have reported macroporosity as a sensitive indicator of soil physical health linked to plant growth, with links to air and water transmission and plant root exploration (Fig. 3.2). This is the basis for using macroporosity as an indicator of soil structural condition, and the provision of support to plants. Macroporosity can be measured in the laboratory in cores, or assessed visually in the field with the Visual Assessment System method (VAS) of Shepherd (2000).

---

[2] As mentioned before, these three natural capital stocks were chosen to inform plant growth as they are linked directly to many of the supporting processes underpinning this service. Their dynamics have been well studied and they are known to be directly linked to pasture growth.

When examining soil structure and the provision of support to plants, the impacts of degradation processes and drivers on the dynamics of soil macroporosity and pasture yield need to be considered. Intensive dairy grazing, when associated with wet soil conditions can lead to severe deterioration of soil physical state, and consequently decrease pasture production (Houlbrooke et al., 2009). Spring (September to November) has been identified as a critical period for soil damage on New Zealand dairy farms, because of high soil moisture coupled with intensive grazing practices (Houlbrooke et al., 2009). Soils differ in sensitivity to physical damage depending on their physical properties. A well drained soil with high macroporosity drains quickly, reducing the period it is vulnerable to damage by stock treading. The level of damage depends on different factors: soil water content at the time of grazing, animal live weight, stocking rates and the duration of grazing. Soil water content impacts strongly on the type of damage endured by the soil. Plastic limit (PL) is the gravimetric water content at which a soil changes from being friable to plastic under pressure (Hillel, 1980). For some soils the field capacity (FC) is close to the PL (Drewry et al., 2008). The risk of compaction is greater at soil water contents below the PL. Above the PL and around the liquid limit, soils deform rather than compact (Betteridge et al., 1999; Drewry et al., 2008). Treading close to FC and above should be avoided as the soil is likely to be compacted or deformed (Drewry et al., 2008) which leads to a lower macroporosity and reduced pasture growth. To inform soil physical support to plants, the dynamics of macroporosity needs to be linked to the intensity of treading (degradation process), soil structure recovery (supporting process) and pasture yield (Fig. 3.1).

The PL differs between soil types. In the absence of data on the PL of soils and the changes that occur in this soil property under treading pressure (Betteridge et al., 1999), FC was chosen as a parameter to indicate the point above which the soil becomes vulnerable to treading damage. Moreover, pasture growth is reduced by compaction and even more by pugging, but it is difficult to separate the direct effects of treading and plant damage from the indirect effect of changes in soil physical support on plant growth (Drewry et al., 2008). To quantify and model soil physical support to plants, soil water content at the time of grazing (below or above FC), stocking rate and the time animals spend on the pasture need to be linked to assess the impact of treading on macroporosity and pasture growth (Chapter Five).

<u>Soil N and P: parameters to inform soil fertility and the provision of nutrients to plants:</u>

To inform the provision of nutrients to plants, the study focuses on nitrogen (N) and phosphorus (P). These two macronutrients are the ones that severely limit plant growth when deficient. For this reason, they are the most applied by New Zealand farmers. The approach could be extended to include other nutrients also managed by farmers such as S and K.

To model the provision of nutrients to plants, the influence of different drivers on nutrient cycles (supporting process) needs to be taken into account, including climate, land use and management practices, inputs to the soil (fertilisers, animal dung and urine and plant litter), and changes in soil structure and soil moisture (Fig. 3.2). Making the distinction between N and P coming from soil natural capital and N and P coming from fertiliser inputs which is "added capital" is an added challenge, as farmers apply these nutrients to compensate for the lack of soil mineral-nutrients or natural capital. To quantify the service, the natural capital and added capital need to be separated. The distinction also needs to be made between N inputs as fertilisers and N coming from legume fixation, with the latter driven by P fertiliser inputs.

The methodology chosen to inform the N and P status of soils and availability to plants is described below. New Zealand soils have received P additions for up to 100 years (Parfitt et al., 2008a). Applied P drives legume growth. Legumes fix gaseous N, which in turns increases grass growth, as the N is released from the legume. The value of the P fertiliser inputs is to a large degree a product of the improved N status of the soil, derived from the increased N input from legume growth, and associated N_2 fixation, as a consequence of the increase in P availability in soil.

The inherent stocks of P come from the soil parent material. During soil development (hundreds of years) some P is lost by leaching and erosion, but annual losses are extremely small. P is stored in soil mainly as inorganic forms like precipitates or it can be adsorbed on the surface of different minerals or included in the matrix of soil components. Organic-P is stable and available to plants through mineralisation, but these rates are very slow in a low P status soil. Well-developed soils have low levels of plant-available P because, with time, labile-P transfers to non-labile forms. Soil solution P concentration depends on the labile P pool, which consists of easily mineralised organic P, P weakly adsorbed to clays and soluble precipitates (Ryden and Syers, 1977). To quantify the soil natural capital stocks of P, the native P or background native stocks of labile inorganic P need to be considered. In New Zealand, the Olsen P test is used to assess the amount of labile inorganic P, "plant available P" in the soil, which is the soil P fraction that is in, or replenishes, soil solution P. Organic P forms are quite stable, so their contribution to soil solution P is minor, limited to P in the microbial fraction. The Olsen P test involves extraction of P with 0.5M sodium bicarbonate solution at pH 8.5. In New Zealand, a modified Olsen-P test, based on extracted volume rather than weight of soil (dried and sieved soil and Olsen P=μg P cm^{-3}), is used (Edmeades et al., 2006). The Olsen P test can be used to determine the amount of labile inorganic P in a soil that hasn't been fertilised for a long period. This provides an indication of the inherent natural capital stocks of P. Parfitt et al. (2009) carried out a study to quantitatively estimate N and P cycles under sheep-grazed pastures at two stages of N saturation. One of the sites had not received P fertilisers since 1980, and had an Olsen P value of 8 (Parfitt et al., 2009). This study gives an indication of the inherent P status of a soil, namely P from natural capital stocks. The inherent Olsen P value of a soil depends on the soil type. For example, an Allophanic soil rich in allophanes will have a higher inherent Olsen P than a Gley soil. In New Zealand, the relationship between Olsen P and relative pasture yield has been established for all major soil orders, based on a large number of field sites (Morton and Roberts, 2001). These relationships (Fig. 3.7) enable us to determine what relative yield would be attained with an Olsen P close to the native soil P stock (Edmeades et al., 2006).

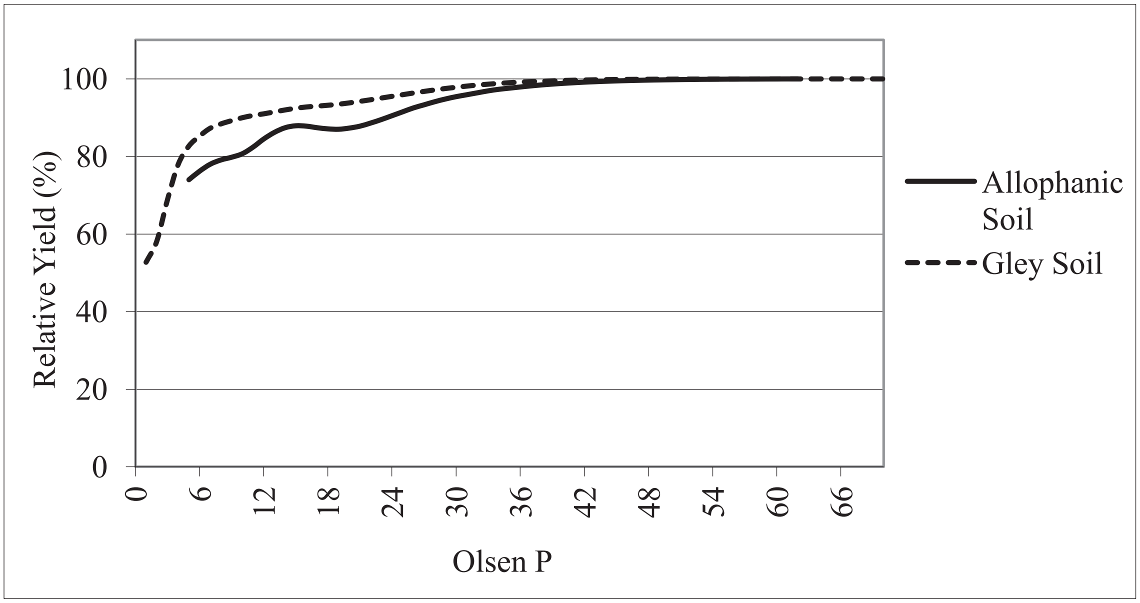

Figure 3-7: Relative pasture yield as a function of Olsen P for two different soil orders (Morton and Roberts, 2001).

It is more complicated to assess the inherent N status of a soil, because soils store very little labile N. N is stored in soils mainly in OM (Sparling et al., 2006). The mineralisation of OM releases N (NH_4^+ and NO_3^-) in soil solution for plants to use. In pasture soils, the P status of the soil drives legume growth. Legumes fix N, which in turn is an input into the soil organic N fraction. Most of the labile N in soils, which is taken up by pasture plants, comes from the impacts of P fertilisers on legume N fixation. Parfitt et al. (2009) found that the net N mineralised level of a sedimentary soil under sheep-grazed pastures that had not been fertilised since 1980, were 50 times lower than the same soil with P fertiliser inputs at a level where P was not limiting growth, indicating N supply from that legume-based pasture was at steady state under that management regime. Such studies give an indication of the inherent N status of a soil. Therefore, to quantify production levels sustainable by soil natural capital stocks, inherent Olsen P values need to be considered since they can support some clover growth, and thereby an inherent N status associated with a sustainable pasture yield.

To inform the risk of livestock metabolic diseases due to trace-element deficiencies, the soil trace-element status also needs to be assessed, from parent material, soil type and location. Soil trace-element status is a proxy for pasture quality

Number of days with available water: a parameter to inform the provision of water to plants

To quantify the provision of water to plants, the number of days per year when pasture growth is not restricted by soil water supply should be calculated. Pasture growth can be restricted by a lack or excess of water. When soil water content goes below the "stress point" (SP), water left in the soil becomes more and more difficult to absorb. If the soil water content decreases further reaching the permanent wilting point (PWP), plants cannot take up water and wilt. When soil water content is above field capacity (FC), aeration and the transmission of gases decrease and plant roots lack O_2 as CO_2 increases, which affects plant growth. The proxy chosen for the provision of water to plants is the number of days per year when water is not restricting plant growth, that is when SP<SWC<FC. This parameter is a measure of soil natural capital (Fig. 3.1) and is influenced by the amount of rainfall received per year (wet or dry years), soil texture and structure differences (AWC). For all practical purposes, farmers monitor soil sensitivity to damage based on SWC. Houlbrooke et al. (2009) recommended the monitoring of soil moisture directly as an appropriate tool for accurately predicting critical soil conditions for making decisions on the exclusion of animals or restricted grazing, when SWC is too high.

A measure of the service can then be obtained by linking macroporosity, the number of days with plant available water and inherent Olsen P to a sustainable pasture yield. The sustainable yield from natural capital stocks is a measure of the provision of food from soils (pasture quantity).

To inform pasture growth, the soil parameters mentioned above must be linked dynamically. The parameters chosen can be measured over a period of time or modelled to reveal their dynamics against degradation processes and drivers (Fig. 3.1). Process- based soil models can enable us to follow closely the dynamics of the parameters chosen under the influence of different degradation processes and drivers and allow an examination of the impacts of these processes and drivers on the provision of the service. To inform the provision of food from pastoral soils, a process-based model is used to examine the influence of the dynamics of soil structure and AWC on pasture yield (Chapter Five and Seven).

3.3 Provision of support for human infrastructure and animals:

Soils form part of the surface of the earth and represent the physical base on which plants grow and animals, humans and infrastructures stand. The previous section demonstrated that the provision of food, fibre and wood depends on plant growth, that soil structure is the natural capital behind the physical support to plants, and that the processes behind soil structure formation are the supporting processes behind the provision of support to plants (Fig. 3.1).

Soils also provide support to humans, infrastructure and animals. The provision of support to humans is an ecosystem service that directly fulfils human needs (e.g. human habitat). Humans need stable soils as a substrate to support transport, leisure, houses and infrastructure. A sterile soil is not able to grow plants but can still provide physical support for human infrastructure and activities. For example, the soils of the Giza plateau support the Egyptian pyramids.

Soils also provide physical support to animals, including species that directly benefit humans like livestock. In dairy grazed systems, pasture is consumed *in situ*, making the grazing regime and animal production dependent on soil structure and the stability of soil aggregates. The pressure on this service increases as stocking rates and liveweight loading is increased. When the bearing strength of a soil is exceeded due to wetness, animal's hooves penetrate the surface, the soil deforms and animal movement is restricted. Animal foraging declines and the animal uses more energy to move; plants are buried and feed wasted. This can affect animal's health and the level of milk production. At the extreme, the animal gets trapped and dies. Excessive soil moisture also leads to soft hooves, increasing vulnerability to damage, and leading to footrot and lameness. The provision of support to animals is an ecosystem service, because humans use animals for many different purposes to fulfil human needs (cattle for food, horses for traction or recreation, pets for aesthetics and company). New Zealand animals graze perennial pasture *in situ* year round so the provision of support is critical year round.

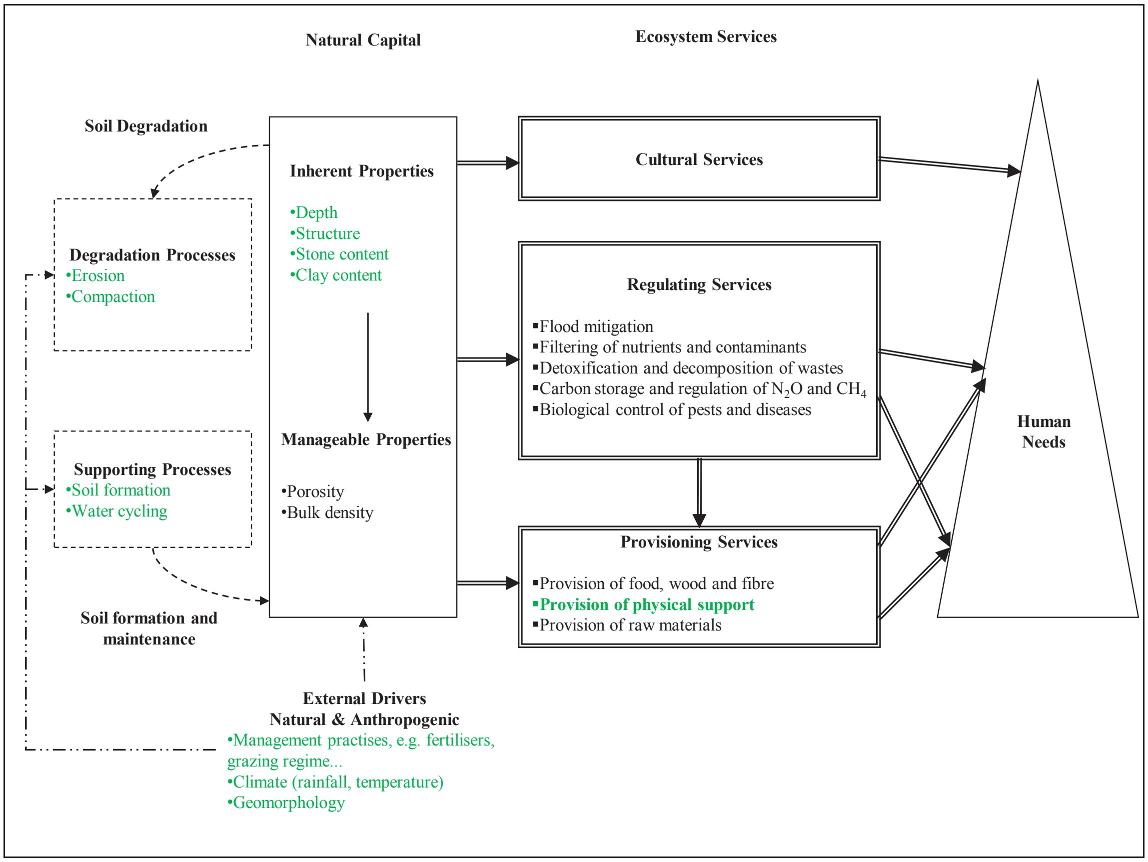

Figure 3-8: Detail of the conceptual framework applied to the provision of support.

In the following section, the properties and supporting processes involved in the provision of support to human infrastructure and animals are detailed. Then the drivers and degradation processes impacting on soil natural capital stocks and the provision of support are examined.

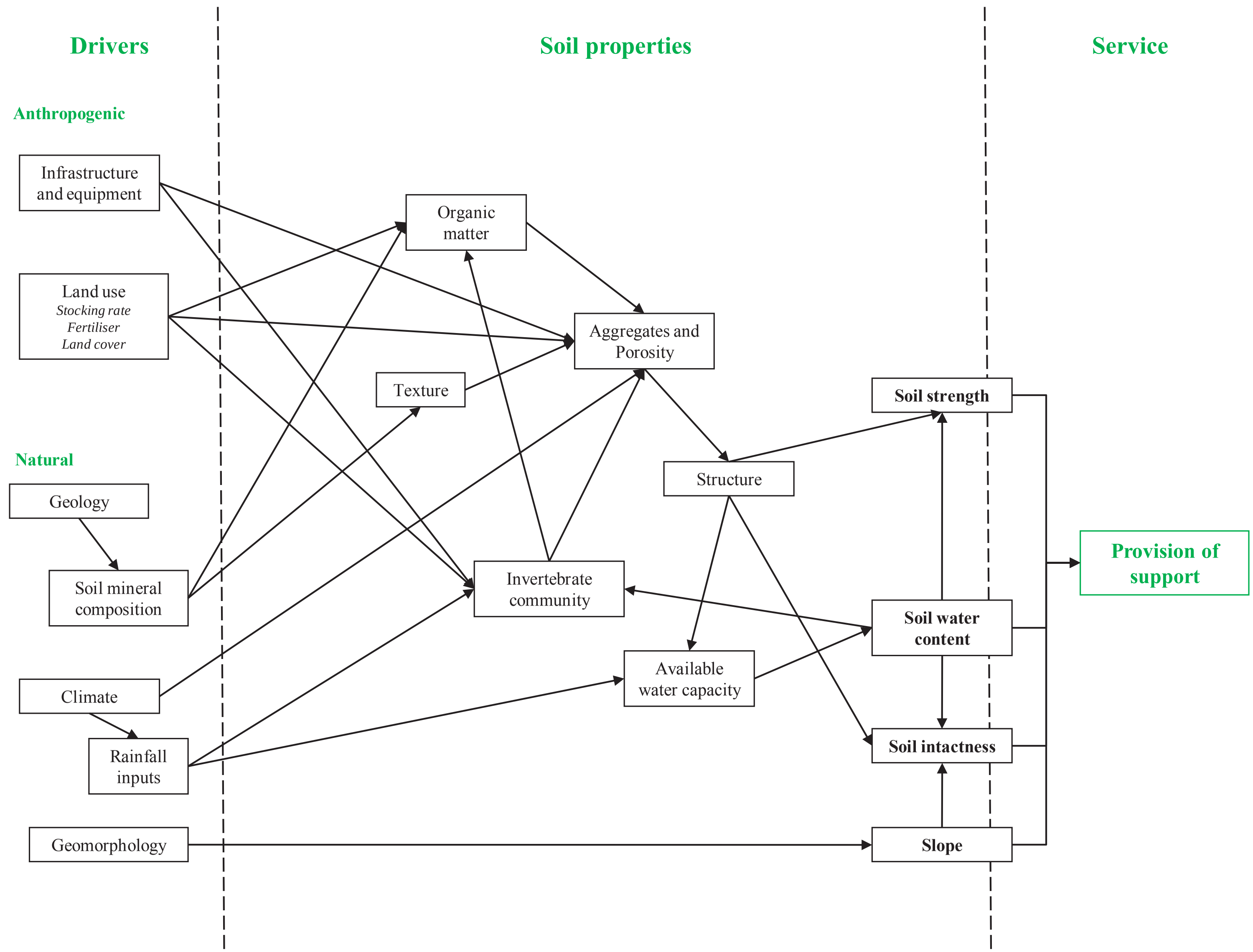

Figure 3-9: Drivers and soil properties influencing the provision of support for human infrastructure and animals

3.3.1 Soil properties and supporting processes involved in the provision of support to human infrastructure and animals:

The support of humans, their infrastructures and animals is dependent on soil natural capital, namely soil structure and especially soil strength (Fig. 3.8).

Soil strength is defined as the "ability of soil to resist a force without shearing" (Hewitt and Shepherd, 1997). When a force is applied to a soil, if the stress exceeds the strength of the material, the soil fails by fracture or plastic flow. At that point, the soil does not recover its original size when the stress is removed (Marshall, 1996). The shear strength of a soil is a combination of its cohesive strength (bonding between particles) and its internal friction (the friction between particle surfaces when they slide over each other). Wet clay soils have a good cohesion and low internal friction, whereas dry sand has no cohesion, and high internal friction when compacted. Soil types differ in their shear strength. Soil texture is one determinant, as clay content influences the soil's cohesive strength, and silt and sand contents impact on internal friction (Fig. 3.9). OM plays a role in soil structure formation and the bonding of soil aggregates, both of which influence soil strength (Fig. 3.9). Soil strength tends to increase with increasing bulk density (BD) and decreasing water content (Marshall, 1996). Bulk density is defined as the ratio of the mass of dry soil to the total volume of soil. It is normally expressed in grams per cubic centimetre (g/cm^3). BD takes into account the pore space of the soil so it gives an indication of the level of compaction. When a pressure is applied on soil, the BD reached depends on the soil particle size distribution and water content (Hillel, 1980; Marshall, 1996). Soil strength is always considered by engineers before construction. Marshall (1996, p. 242) declared "for road construction, an optimum water content is sought at which soil can best be compacted to obtain the required density and strength required for this purpose. This is determined by packing a sample of the soil in a cylinder under a set number of standardised blows from a hammer in the method of Proctor (Proctor compaction test, 1933)"; this is conducted at a number of water contents. A curve (Fig. 3.10) is constructed to obtain the Atterberg limits, which include the optimum water content for compaction corresponding to the maximum BD (on a dry basis) (Marshall, 1996).

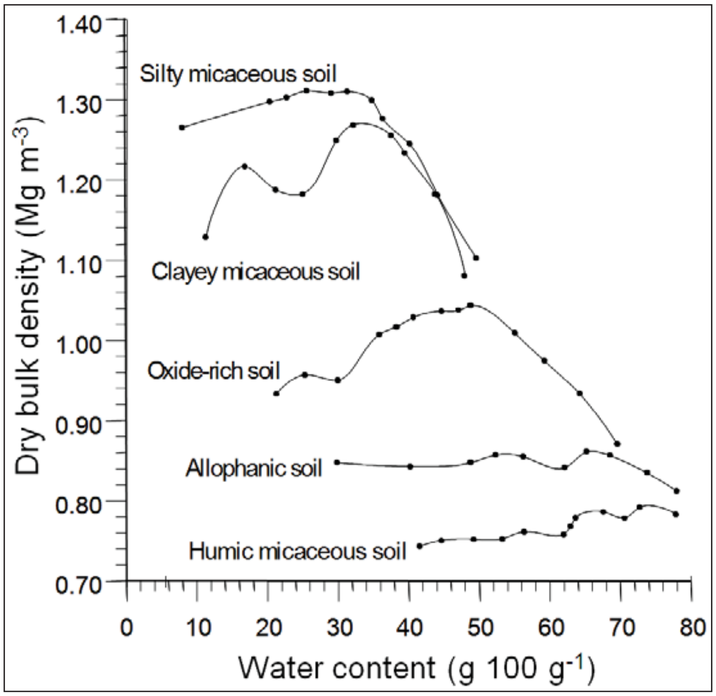

Figure 3-10: Relationship between soil water content and bulk density for different soils (from Graham Sheperd, pers.com.)

The 'Atterberg limits' (Hillel, 1980, p. 348-349) are widely used to determine the appropriate SWC for maximum bearing capacity (Fig. 3.11).

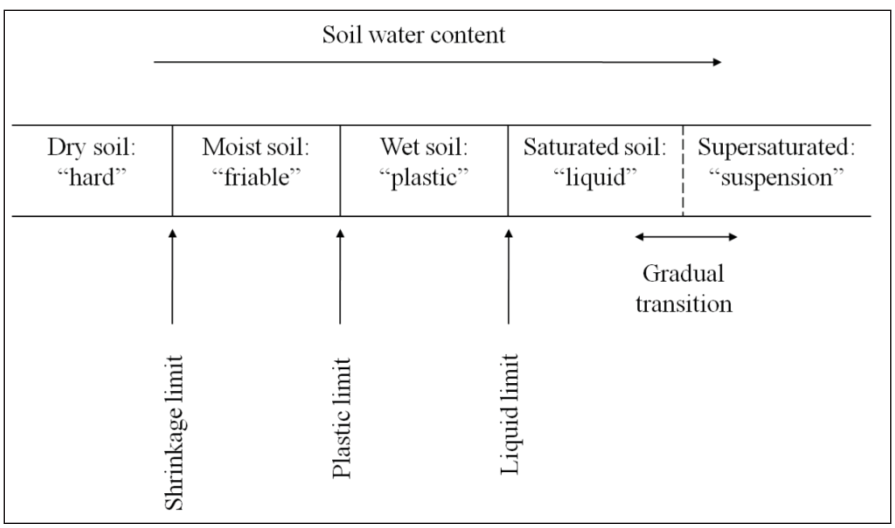

Figure 3-11: The Atterberg consistency limits (schematic) (from Hillel, 1980, p. 349)

Soil strength can be measured with various methods (Fig. 3.12). Field measurement of soil strength can be made with different instruments like a vane-shear tester or a penetrometer (Marshall, 1996).

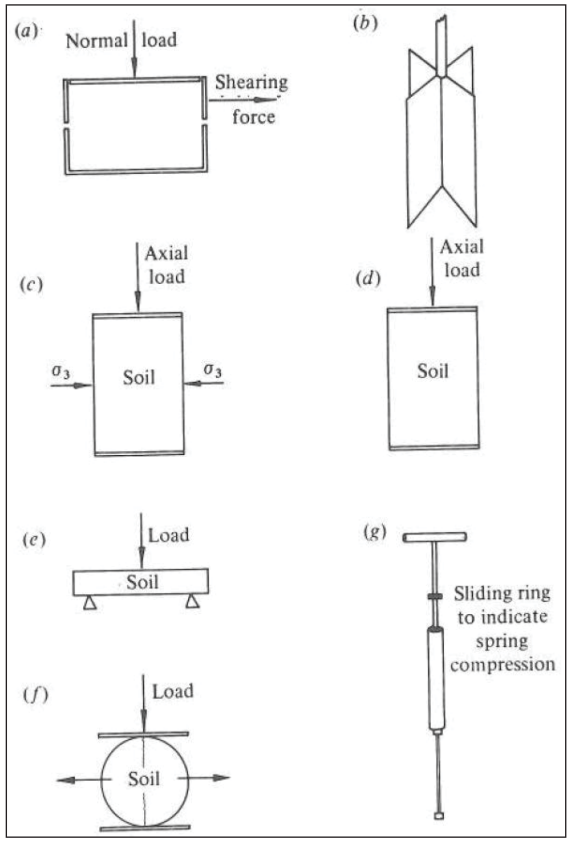

Figure 3-12: Methods of measuring soil strength: (a) direct shear; (b) vane; (c) triaxial compression; (d) unconfined compression; (e) rupture; (f) indirect measurement of tensile strength; (g) penetrometer, (Marshall, 1996, p. 233).

The provision of support to animals depends on soil structure (natural capital) and more specifically on soil sensitivity to degradation processes associated with livestock treading including compaction and deformation (Fig. 3.8). If a soil can't sustain its integrity under cattle treading, its ability to provide support is compromised, placing not only the provision of that service at risk, but also the provision of a number of other services (Fig. 3.8). The provision of support is highly related to soil moisture (Fig. 3.9), and is at its weakest during periods of prolonged wetness, frequently the wet winter and spring months. A soil with low macroporosity, that is low natural capital, becomes saturated when it rains, before a soil with a good structure. The net effect is that a soil with low macroporosity is at risk of compaction or deformation for longer periods of the year (Betteridge et al., 1999; Drewry et al., 2008). Well drained soils will be able to support animals for longer periods than poorly drained soils, before becoming exposed to treading damage risks. Frequent pugging can impact on animal health and well-being due to difficulties with foraging, including ingestion of soil, walking and health issues associated with softness of hooves, footrot and lameness, all of which also impact on milk production.

The supporting processes involved in the development and stability of soil structure mentioned previously (section 3.2.1.1) are the same processes behind soil strength (Fig. 3.9) and thereby the provision of physical support to humans and their infrastructure and animals. Therefore, the reader is referred to section 3.2.1.1 for details about the supporting processes behind the provision of support (Fig. 3.8).

3.3.2 Degradation processes and drivers impacting on the provision of support:

Soil structure and strength are sensitive to a number of degradation processes and external drivers (Fig. 3.8) which can modify these natural capital stocks and thereby have repercussions on the provision of support. Natural drivers like climate, geology and degradation processes like erosion influence natural capital stock and supporting processes like soil structure formation, and thereby impact on soil strength. Physical support is important at different scales. At the farm scale, soil's capacity to support animals at the paddock scale depends on the BD and compaction of the upper horizon. Support of buildings and farm tracks depend more on the strength of the deeper horizons and the subsoil. At the landscape level, geomorphology (slope, orientation) and soil's sensitivity to landslides also impacts on the provision of support. The cohesion within soil horizons, as well as between horizons, and between soil and bedrock influences the possible movement of soils at the landscape scale. Climate, by driving soil water content, impacts on soil strength and stability. Wind and water impact on the type of degradation process, namely erosion type, applied to the soil (Fig. 3.8). Different types of erosion have different impacts on the provision of support by soils. Surface erosion involves the movement of a thin layer of particles across the ground by water, wind or gravity (Lynn et al., 2009) and impacts on the structure of the soil surface. Mass movement erosion includes a wide range of erosion types like soil slip, debris flows, debris avalanche or earth flow, where material moves down slope, as a more or less coherent mass, under the influence of gravity (Lynn et al., 2009). At the landscape scale, mass movement erosion can be caused by water infiltration and soil saturation. Fluvial erosion involves the removal of material by channelised running water (Lynn et al., 2009). It starts at the soil surface, but if water keeps running, it can dig the whole of soil depth (river beds).

Anthropogenic drivers like land use and farming practices also influence soil structure and soil strength. Some farming practices compensate for the lack of support from soils that is a lack of service (Fig. 3.8). Farmers have a range of management options to deal with limited supporting processes (drainage) generally associated with wet soils. Farmers can control the type and number of animals per hectare and the duration of the grazing period to limit the damage to soils with limited drainage. Stand-off pads or feed-pads are effectively substitutes for a soil's lack of support service (Betteridge et al., 2003). Standing the animals off the soil also allows macroporosity recovery, carried out by supporting processes (Fig. 3.8). Artificially draining the soil also reduces the risk of treading damage by reducing the length of the wet period.

3.3.3 Quantifying the provision of support for human infrastructure and animals:

To our knowledge, no one has previously attempted to quantify the provision of support from soils.

To quantify and model the provision of support to human infrastructure, soil strength, the natural capital behind the service, needs to be considered. For building purposes, compacted soils that are very stable and won't sink and deform or erode, when under a building or a road, have the most value. For example, soils with low bulk density require compaction before building. Bulk density and macroporosity are indicators of soil compaction. For building purposes, the first 10 cm of soil are usually removed, therefore to inform the provision of support for human infrastructure, the BD below 10 cm was chosen as a proxy to measure the service. The service in itself is defined as the difference between a minimum BD and the actual BD of the chosen soil. This measure represents the already existing compaction, that is the existing support.

To inform the provision of support to animals, the interactions between soil texture, structure and moisture need to be considered. The less sensitive to treading damage a soil is, the better support it provides to animals, the easier for humans it is to use this soil for farming animals. Winter and spring time (May to October) has been identified as a critical period for soil damage on New Zealand dairy farms because of high soil moisture contents (Houlbrooke et al., 2009). Houlbrooke et al. (2009) used a hand-pushed cone penetrometer as a decision support tool to identify soil conditions under which grazing by animals would produce treading damage. They found the penetrometer wasn't the most efficient at predicting critical soil conditions. They recommended the monitoring of soil moisture directly, as a more appropriate tool for accurately predicting critical conditions. The maximum risk of deformation occurs when soil water content is around and above the plastic limit (Betteridge et al., 1999; Drewry et al., 2008). At these water contents, animal hooves penetrate the soil surface, deforming (pugging) the topsoil. To avoid soil deformation, production losses and health issues, farmers are increasingly taking animals off pastures before significant damage occurs. Therefore, to illustrate and quantify the provision of support to animals, the number of days per year when the soil can support animals can be calculated. To calculate this parameter, the dynamics of soil water content between May and October can be followed and the days when the SWC< (FC+Sat)/2 identified. The moisture (FC+Sat)/2 was chosen because FC is usually well documented whereas data on PL is often difficult to access. Moreover, saturation levels are linked to the dynamics of soil macroporosity. A well drained soil will be above this limit less often than a poorly drained one. Process-based models can describe SWC dynamics, enabling us to calculate the value of this parameter. The measure of the service was then defined as the difference between the number of days when SWC could be above (FC+Sat)/2 (184 days between May and October) and the modelled or calculated wet periods for the chosen soil.

3.4 Provision of raw materials:

The provision of raw materials from ecosystems has been mentioned by a number of authors. Costanza et al. (1997), de Groot et al. (2002), and the Millennium ecosystem assessment (2005) identified raw materials provided by ecosystems as renewable biotic resources (wood, strong fibres, biochemicals or biodynamic compounds like latex, gums, oils, waxes, tannins, dyes, hormones, etc.) and energy resources (fuel wood, organic matter, animal power and biochemicals). The raw materials in question are constituents of ecosystems, namely natural capital stocks (Fig. 3.13) directly of interest for humans and therefore removed, harvested and mined.

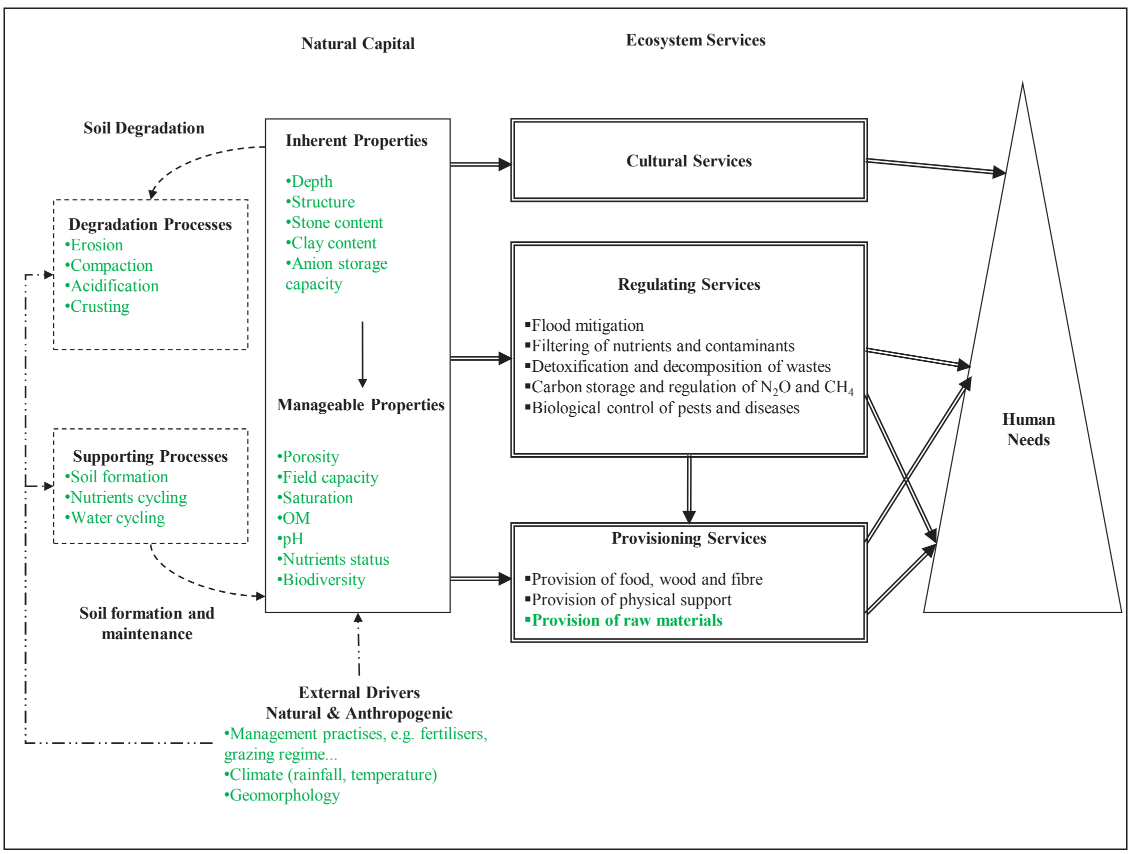

Figure 3-13: Detail of the conceptual framework applied to the provision of raw materials.

De Groot et al. (2002) specified that the provision of abiotic resources like minerals and fossil fuels cannot be considered as ecosystem services because these resources "are usually non-renewable and/or cannot be attributed to specific ecosystems". Consequently, in examining soils' capacity to provide raw materials, the distinction needs to be made between renewable and non-renewable resources. In this study, the discussion is limited to raw materials found within the soil profile, not in the bedrock like fossil fuels, minerals or gases. Materials in soils, like peat or clays, are generally considered non-renewable so their provision shouldn't be considered as an ecosystem service (de Groot et al., 2002). An argument, however, could be mounted to suggest peat is renewable, as a product of plants and the result of OM accumulation (Fig. 3.13).

Peats or organic soils develop in sites that support dense vegetation cover and that are more or less permanently waterlogged. The process dominating peat formation is the accumulation under anaerobic conditions of an organic horizon. The thickness of a peat soil and its rate of formation depend on vegetation growth which can be very slow in low temperature areas like, e.g. the Taiga in Siberia. As organic soils, peats are a large reservoir of sequestered carbon. Borren et al. (2004) studied the role of Siberian peatlands as a sink for atmospheric CO_2 and found average peat accumulation rate varied from 0.35 ± 0.03 to 1.13 ± 0.02 mm/yr. The long-term apparent rate of carbon accumulation value of bogs and fens varied from 19.0 ± 1.1 to 69.0 ± 4.4 gC/m^2/yr. In Northern Ireland, peatlands are considered a scarce, endangered ecosystem even though they still occupy 12% of the country (Cruickshank et al., 1995). However, cutting fuel peat is still practised since it can reduce household fuel costs, and some peat fuel is sold to gain income. Cruickshank et al. (1995) who studied peat extraction in Northern Ireland, mentioned that Irish peat bogs do not exceed a depth of 2 meters.

At the farm level, knowing that average consumption of peat per household is around 10 m^3 every year, and that peat accumulation rate is around half a millimetre per year (Borren et al., 2004), for a household to *sustainably* consume peat for fuel, the peat would need to be extracted from a 2 ha bog (10 m^3 / 0.5 mm = 20,000 m^2 = 2 ha). If the extraction rates of peat are greater than the accumulation rates, extraction of peat would see a decline in the soil natural capital and may therefore be considered a non-renewable resource in an annual set of accounts.

The same argument goes for clays. At the farm level, these materials are often not present or not exploited, but even if they were, based on the above analysis, their use wouldn't be sustainable, and they would only provide a very small annual benefit for the farmer. For this reason, in this study, the provision of raw materials from soils is not included, but it is acknowledge that it could make a contribution in some situations, for example, at a different scale like the region or country.

3.5 Summary of the quantification of soil provisioning services:

The conceptual thinking and information presented in this chapter is used to quantify and measure the provision of provisioning services from soils. A summary of the soil natural capital stocks behind provisioning services, and the parameters based on these stocks chosen for the quantification of the provision of the services are presented in Table 3.3. In the next chapter, Chapter Four, the same exercise is undertaken for the quantification of the regulating services provided by soils. Then, Chapter Five shows how an existing process-based soil model was modified to include the necessary relationships between supporting processes and soil properties, to enable the calculation of the parameters chosen here and to follow their dynamics.

Table 3-3: Summary of soil natural capital stocks behind provisioning services and parameters chosen for the quantification.

Natural capital stocks	Provision of food, wood and fibre			Provision of physical support	Provision of raw material
	Support	Water	Nutrient		
Inherent Properties					
Depth		X			
Structure	X	X		X	
Texture	X	X	X	X	
Soil strength				X	
Stone content		X		X	X
Clay content			X	X	X
Fragipan	X	X			
Drainage class of subsoil		X		X	
Inherent mineral contents			X		X
Manageable Properties					
Biodiversity			X		
Organic matter		X	X		X
Dissolved organic matter					
Anion storage capacity			X		
Cation exchange capacity			X		
pH			X		
Porosity	X	X		X	
Bulk density	X			X	
Nutrient status			X		
Saturation levels			X		
Temperature			X		
Soil water content		X	X	X	
Field capacity	X	X			
Saturation capacity		X			
Available water capacity		X			
Plastic limit	X			X	
Drainage class of topsoil		X		X	

Table 3.3: Continued.

Natural capital stocks	**Provision of food, wood and fibre**			**Provision of physical support**	**Provision of raw materials**
	Support	**Water**	**Nutrients**		
Parameters chosen	Macroporosity proxy for soil structural health	Available water capacity	Trace-elements status	Provision of support for human infrastructure: Bulk density	Not included in this study
	Field capacity proxy for soil sensitivity to treading damages		Sustainable yield (from native Olsen P)	Provision of support to animals: SWC<(FC+Sat/2)* (number of days/year when the soil can support animals)	

* SWC: soil water content; FC: field capacity; Sat: saturation.

Chapter Four

Detailed Framework for Regulating Services provided by Soils

This chapter builds on the framework developed in Chapter Two, and follows on from Chapter Three. It describes the soil properties, processes and drivers influencing the regulating services from soils, under a dairy grazed system. These services include flood mitigation, the filtering of nutrients and contaminants, detoxification and recycling of wastes, carbon storage and greenhouse gas regulation and the biological regulation of pest and disease populations. New concepts for the quantification of each service are developed and presented. The parameters chosen to quantify and explore the dynamics of these services are examined.

4.1 Flood mitigation

The ability of soils to store water provides a service to humans, buffering excessive rainfall, and in doing so, reducing flood risk. The buffering of rainfall by soils is an ecosystem service because human well-being benefits directly from being able to live in a safe, dry and practicable environment.

Rainfall water infiltrates the soil and is stored. When soil water content (SWC) reaches FC the soil profile starts to drain. If rain keeps falling, SWC increases to saturation. Once the soil is saturated, and rainfall exceeds drainage rates, water can no longer infiltrate and is lost as surface-runoff in overland flow. Accumulated runoff water contributes to peak flow in streams and rivers. The amount of water that can be stored in soil before saturation is reached (e.g. saturation capacity) depends on both inherent and manageable soil properties. It provides a service by reducing the amount of land at risk from flooding and the need for manmade flood-protection structures. Flood mitigation doesn’t remove the risk of flooding, but rather reduces its likelihood.

Surface-runoff is also one of the determinants of soil erosion. Decreased runoff also means lower risk of soil erosion and transport of materials (sediments, nutrients) off-site. The ability of soils to absorb and store significant amounts of rain and to drain quickly before runoff starts reduces peak flow, by decreasing runoff intensity and introducing a delay before the flood peak.

In the following section, the soil properties and processes behind flood mitigation are examined (Fig 4.1), including an investigation of the properties and supporting processes behind the service and of the drivers and degradation processes impacting on soil natural capital stocks. The methodology developed to quantify and model flood mitigation is also presented.

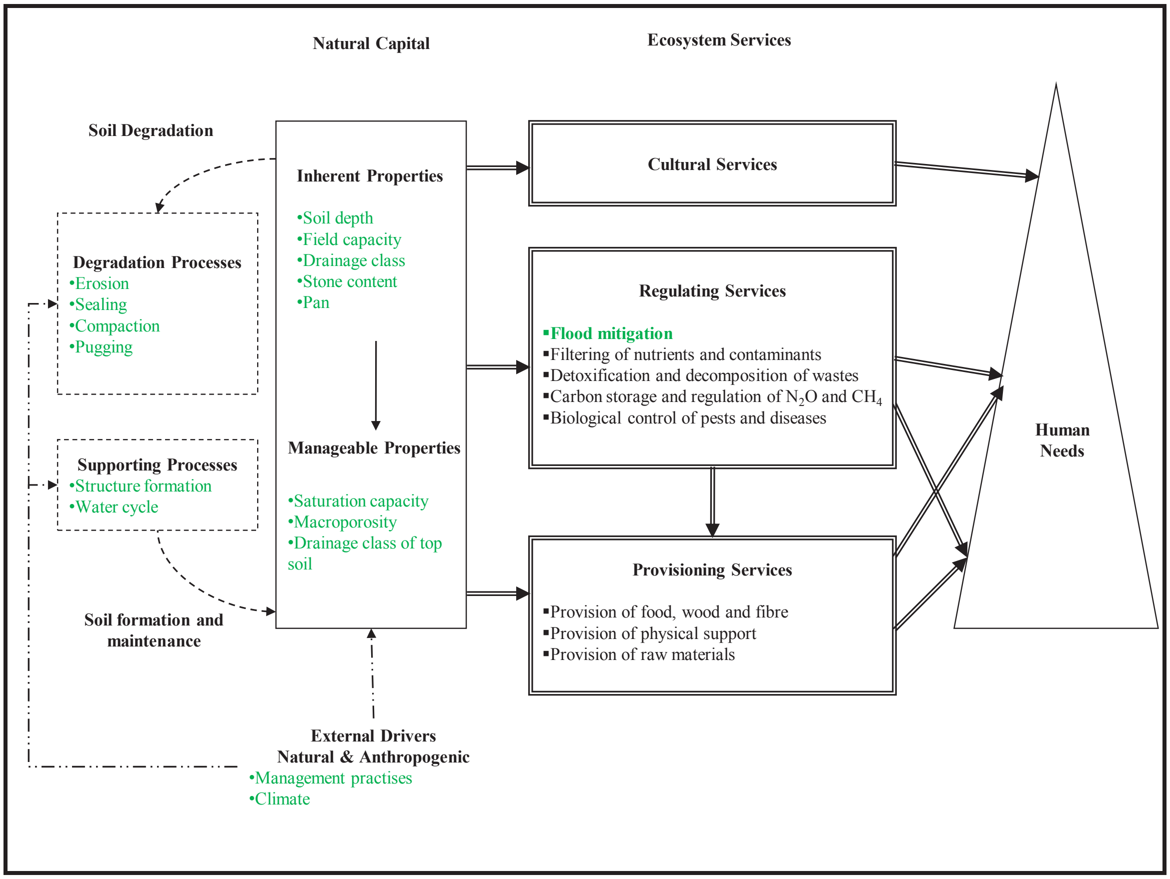

Figure 4-1: Detail of the conceptual framework applied to flood mitigation.

4.1.1 Soil properties and supporting processes contributing to the provision of flood mitigation:

The flood mitigation potential of a soil depends on how much water the soil can absorb and store before runoff starts, as well as the drainage class of the soil: the faster a soil drains, the shorter the saturation period; this regulates surface runoff.

Soil saturation capacity (SC) and drainage class depend on soil structure, an inherent property, and porosity, a manageable property (Fig 4.1). The properties and processes influencing flood mitigation are the same as those affecting soil structure, detailed in Chapter Three. Figure 4.2 summarises how soil properties influence each other and which ones impact on the natural capital stocks behind flood mitigation. The amount of water a soil can store before it starts draining depends on the volume of pores >30μm (Mp) (Table 4.1). The volume of pores >30μm (Table 4.1) determine the volume of water a soil can store above FC to saturation, before runoff starts. Soils with good structure have a high macroporosity (i.e. abundant pores >30µm) and therefore can store a greater volume of water before becoming saturated, whereas soils with low macroporosity store less water above FC. The depth of the soil profile, an inherent property (Fig 4.1), impacts on the total volume of water stored. The deeper the soil profile, the greater the volume available for storage. Other inherent properties of the soil profile (Fig 4.1 and 4.2) can also have a big impact on water storage. The presence of a pan, or impermeable layer, within the profile can impede or prevent the infiltration of water lower in the profile (e.g. drainage). A pan also isolates the volume of soil under the impermeable layer, and reduces the potential storage volume available. The stone content of the soil affects water storage, because the volume taken up by stones cannot be used for water storage. The depth of the water table can also limit water storage. When the water table is shallow, the volume of soil already under water and is no longer available for water storage (Fig 4.1 and 4.2).

Table 4-1: Soil pores function in relation to their size (adapted from Marshall, 1996, p.208)

Pore size	Water relation
1 mm to 10 mm	These pores transmit water freely but only if soil is saturated
30 µm to < 1 mm	These pores transmit water during infiltration. They are drained at field capacity
200 nm to < 30 µm	These pores retain water available to plants and soil fauna
1nm to < 200 nm	These pores are within clay complexes and change size as the soil water content changes (swelling, shrinking)

The supporting processes influencing the development of soil structure and macroporosity have been discussed in detail earlier (Chapter Three) and include wetting and drying cycles, root growth, soil fauna activity (especially earthworms) and the cycling of OM. Flood mitigation also depends on drainage and runoff, processes of the soil water cycle (Fig 4.1). Drainage rate determines how fast water leaves macropores and thereby determines how fast a soil reaches saturation and when water starts running off.

Runoff depends on the surface infiltration rate (Fig. 4.2), therefore properties and processes influencing infiltration rate will impact on flood mitigation. Infiltration rate affects soil water recharge and depends on surface aggregate stability and pore size distribution. Slope also influences infiltration and runoff (Fig. 4.2); when rainfall is exceeding infiltration rate, water will flow down slope. In the case of heavy rain on hill country, water starts running-off before the soil is saturated. Runoff water ends up in streams, rivers and lakes increasing flood risk. The presence of an impermeable layer at depth (Fig. 4.2) can also increase runoff by blocking drainage. If the water can't drain through the profile, saturation is reached faster and therefore runoff starts earlier and lasts longer. The processes decreasing SWC like plant uptake and evapotranspiration also delay saturation and free some water storage volume.

The natural capital stocks behind flood mitigation, embodied by soil structure and Mp, are sensitive to a number of degradation processes (Fig. 4.1). Similarly a number of external drivers can impact on these properties and supporting processes (Fig. 4.1). The net result is a decreased provision of the flood mitigation service. These degradation processes and drivers are examined in the next section.

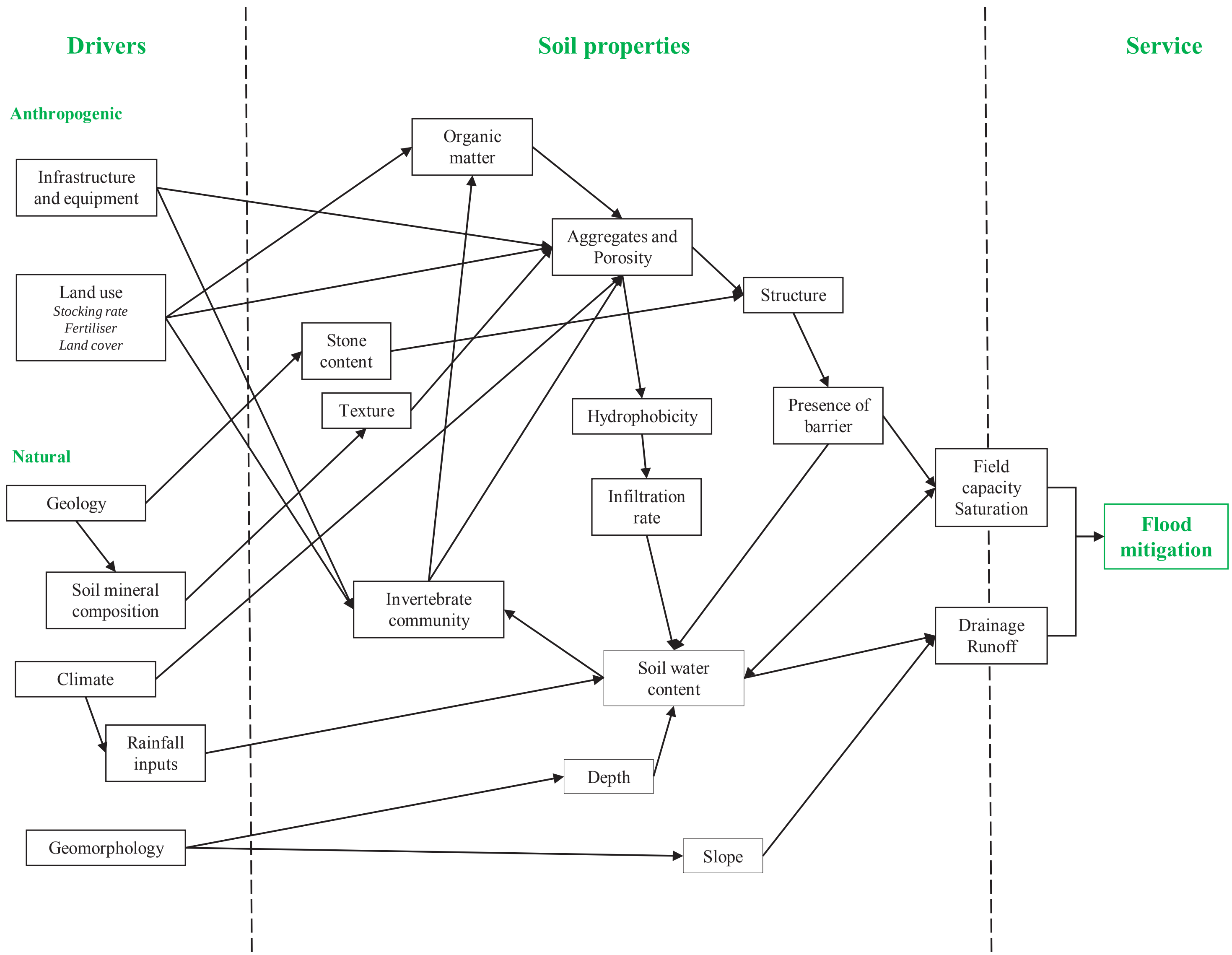

Figure 4-2: Drivers and soil properties influencing flood mitigation.

4.1.2 Degradation processes and drivers influencing flood mitigation:

Flood mitigation is driven by the following natural capital stocks (Fig. 4.1): soil structure and more specifically as structure influences FC and saturation capacity, soil depth, stone content, and landscape position. The degradation processes and external drivers influencing these properties will have an impact on flood mitigation (Fig. 4.1).

The degradation processes affecting soil structure have already been described in detail (Chapter Three, section 3.2.2.1). Erosion affects soil structure by removing soil volume and thereby also impacts on the amount of water the soil can store. Soil scientists know that the greater the slope gradient, the greater the erosive power. If the velocity of the surface runoff water is doubled then its erosive power is increased four-fold (McLaren and Cameron, 1990, p. 137). Soil erosion in itself increases runoff, which in turn makes erosion greater. Compaction reduces the volume of pores available for water storage. Animal pugging reduces porosity as well as infiltration rates. Surface sealing also slows infiltration rates. Rain drops damage aggregates at the soil surface, forming a seal that in drying creates a crust. This blocks access to the surface pores and decreases infiltration rates (Kladivko et al., 1986). Other processes that reduce infiltration include soil saturation and hydrophobicity (Aslam et al., 2009). Both prevent water penetration (Fig. 4.2). If water cannot penetrate the soil, it will pond at the surface and evaporate or if the slope is sufficient, run off down slope.

Natural drivers have an obvious influence on flood mitigation. Climate determines the amount of rainfall a soil receives. The timing of the rainfall event and its intensity are as important in determining floods as soil natural capital stocks. Soils have the capacity to buffer flood peaks to some extent. Even a well structured soil with high saturation capacity, good drainage and high infiltration rate would be saturated and prone to runoff if it received 5,000 to 6,000 mm of water per year in big storm events (e.g. Fiordland, New Zealand). Geomorphology and the position of the soil in a landscape will also greatly influence runoff. Slope gradient will affect infiltration and, together with slope length, will impact on erosion.

Anthropogenic drivers like land use and farming practices affect flood mitigation mainly through their impacts on soil structure. Farming practices like tillage or animal treading can reduce soil macroporosity and in particular the structure of the soil surface, leading to compaction and pugging (Fig. 4.1). These practices decrease both surface infiltration rate and the volume of pores available for water storage, impacting on both drainage and runoff (Brauman et al., 2007). The integrity of the land cover will also influence infiltration rate. A bare soil is more prone to surface degradation of aggregate by rain drops, resulting in loss of soil structure and pore function, than a soil under a permanent vegetation cover like pasture.

4.1.3 Quantification of flood mitigation:

4.1.3.1 Previous attempts to quantify flood mitigation:

A number of authors (Barrios, 2007; Brauman et al., 2007; Daily et al., 1997b; Lavelle et al., 2006; Wall et al., 2004; Weber, 2007) mentioned flood mitigation as a service provided by soils. Porter et al. (2009) and Sandhu et al. (2008), in their studies of ecosystem services from agro-ecosystems, talked about 'hydrological flows' as an ecosystem service that they valued. In both instances, 'hydrological flows' only referred to the water supplied by soils to plants not as it might influence flood mitigation.

On the other hand, Ming et al. (2007) modelled, mapped and valued the flood mitigation provided by wetland soils in China. They quantified the flood mitigation service of a wetland soil by subtracting the quantity of water in the soil at FC from the quantity of water at saturation. Parameters used included saturation water content, FC water content, bulk density, soil depth, the area of flood mitigation and water density. The spatial distribution of soil bulk density was overlapped with saturation water content and FC water content and the water quantity of flood mitigation was then calculated in m^3/ha. Ming et al. (2007) used the replacement cost valuation method to value flood mitigation by wetlands. They estimated the investment needed in the construction of reservoirs to replace wetlands was $5700/ha/yr, if the flood mitigation service provided by the wetland soil was lost.

4.1.3.2 Parameters chosen to quantify flood mitigation:

To inform the flood mitigation capacity of soils, soil natural capital stocks, including the amount of water a soil can store before runoff starts, need to be quantified. Processes also need to be considered, including drainage and runoff (Fig. 4.1). The amount of water in runoff over a number of years also needs to be considered to accommodate for the variation in rainfall amount and intensity between years in order to understand the extent and limits of the flood mitigation capacity of a soil to the community. Flood protection schemes are designed to prevent flooding in 1 in 50, 1 in 100 and 1 in 200 years, based on long-term climate data and the value of the resources at risk from flooding. Drainage water passing through the soil profile generally takes more time, suggesting that drainage water participates less in the building of the peak flow, and flooding events. Therefore to model the flood mitigation capacity of soils, the amount of water drained was not taken into account.

In this thesis, measures of soil natural capital stocks form the basis of the quantification of soil services. This new methodology is applied to the quantification of each service. If the soil surface was impermeable, all rainfall could potentially runoff. Therefore, the service was defined as the difference between rainfall and actual runoff, which is the amount of water that doesn't run off due to soil water storage capacity or the amount of water the soil buffers. Process-based soil-plant-atmosphere models enable us to be able to follow the daily dynamics of rainfall, soil water content and runoff, and measure the service through the years.

4.2 Filtering of nutrients and contaminants:

Soils receive rainfall and are the substrate through which water passes before entering water bodies like rivers, lakes, ground water and oceans. Soils act as filtering agents. In dairy grazed systems, a number of materials are applied to pastures and soils like animal dung and urine, dairy farm effluents (DFE), fertilisers and pesticides. These materials contain a number of constituents entering the soil, including nutrients, organic matter, pathogens, endocrine-disrupting chemicals (EDCs) and heavy metals. Soils can sorb and retain these nutrients and contaminants and avoid their release to free water, controlling water quality. If the solutes present in soil (e.g. nutrients like N and P, but also pathogens) are leached, they can become a contaminant in aquatic ecosystems causing eutrophication, and reducing biodiversity. These also represent potential threats to animal and human health (e.g. nitrate, endocrine-disruptors or pathogens in drinking water).

The soil's ability to filter nutrients and contaminants is directly linked to the quality of the receiving fresh water bodies. This ecosystem service fulfils a physiological human need for drinkable water, but also higher needs for recreation (swimming, fishing...) (Fig. 4.3).

In the following section, the properties (soil natural capital stocks) and supporting processes involved in the provision of this service are investigated.

4.2.1 Soil properties and supporting processes contributing to the filtering of nutrients and contaminants:

The filtering capacity of a soil refers to its ability to retain nutrients and contaminants (e.g. pathogens, endocrine-disrupting chemicals (EDCs), pesticides...) by weakly to strongly bonding them to the surface of soil, and thereby preventing their release into water passing through the soil profile.

A soil nutrient retention capacity is an inherent property (Fig.4.3) and has a number of dimensions. First, soil properties determine the number and type of sites capable of retaining nutrients. Second, nutrients and contaminants can take different forms, stabilities and solubilities, all of which influence the probability of them being retained or released to soil solution. Third, soil processes drive the transformations between different nutrients and contaminant forms, including the rates of sorption and desorption from sorption sites.

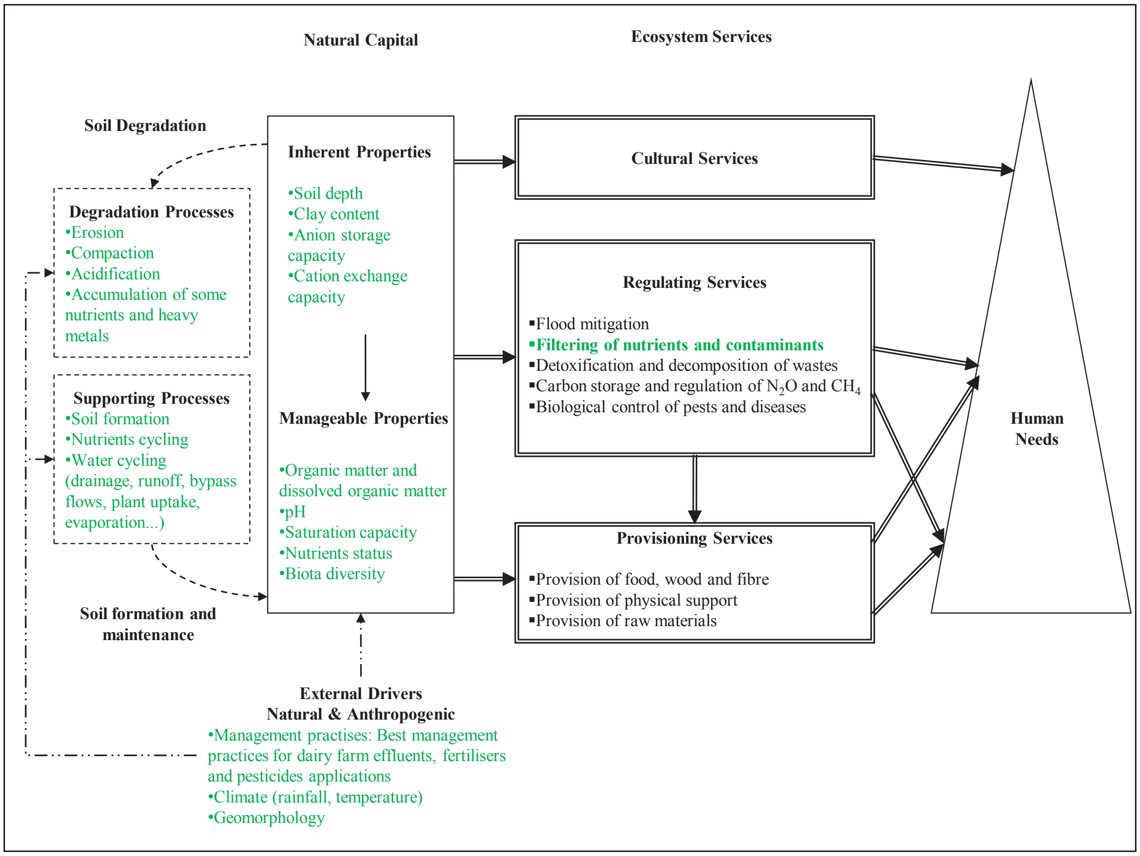

Figure 4-3: Detail of the conceptual framework applied to the filtering of nutrients and contaminants.

Differences in the nutrient retention capacity of a soil are a product of the mineralogy and organic matter content of the soil. To refer to soil nutrient retention capacity, soil scientists talk about cation exchange capacity (CEC) for cations, and anion storage capacity (ASC) for anions (Fig. 4.3 and 4.4). CEC is a quantitative measure of the soil's ability to hold exchangeable cations. It indicates the quantity of negative charge per unit mass of soil, that is the quantity of sites being able to attract cations (McLaren and Cameron, 1990). ASC estimates the soil's capacity to sorb anions. ASC used to be referred to as P retention. Soil clays and OM form exchange surfaces on which charged nutrients, contaminants (pesticides, EDCs) and even negatively charged microbes (McLeod et al., 2008) can get sorbed, effectively removing them from drainage water.

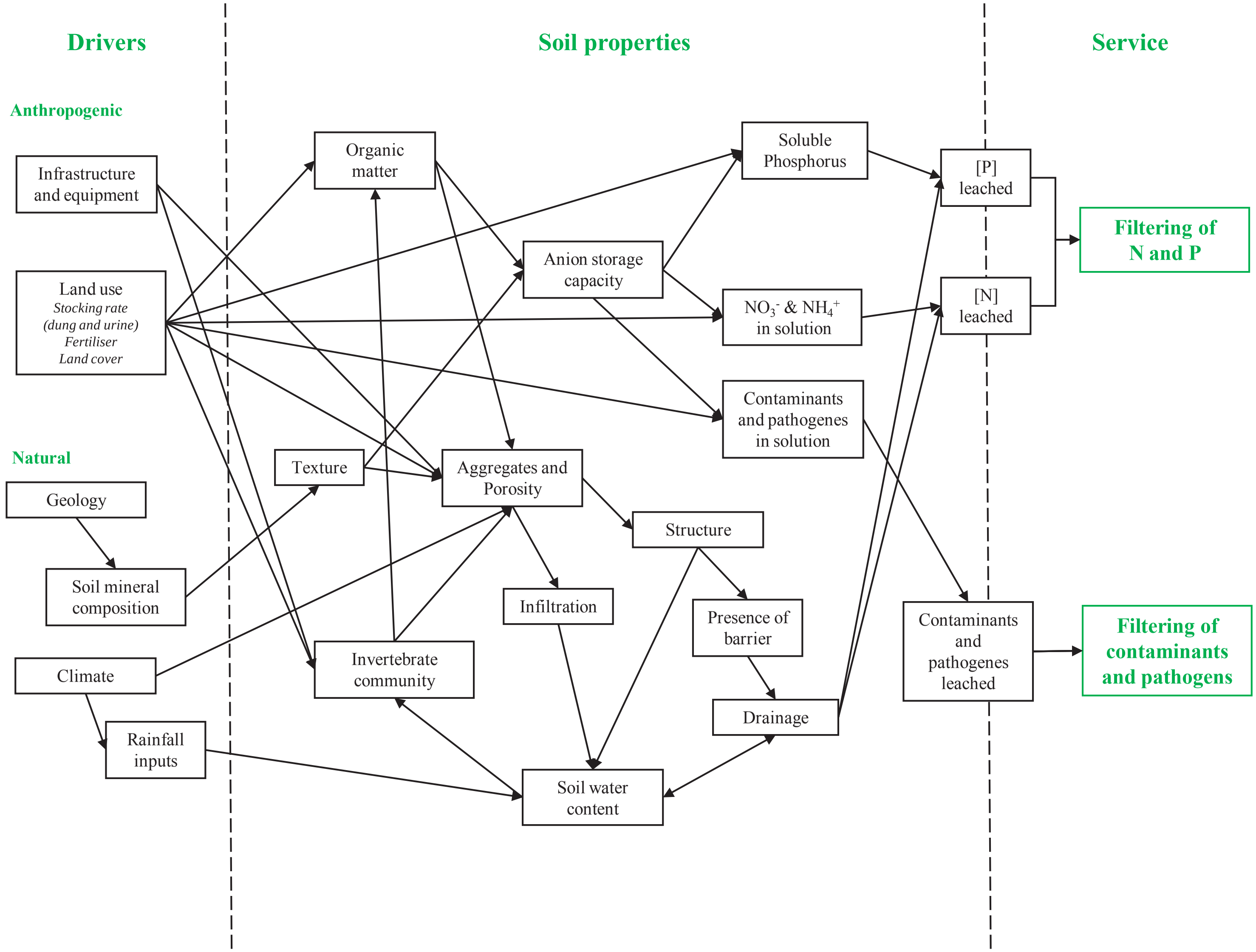

Figure 4-4: Drivers and soil properties influencing the filtering of nutrients

The properties (i.e. natural capital stocks) influencing the number and type of exchange sites include (Hedley and McLaughlin, 2005; Stevenson, 1999) (Fig. 4.4):

- *The nature and quantity of clay minerals*, an inherent property (Fig. 4.3) which depends on the soil parent material. The surface of clays carries charges, negative and positive, which attract ions and holds them on their surface (adsorption). Some clay minerals are also able to expand and contract, trapping nutrients inside the mineral (absorption or occlusion). Primary minerals like micas, found in young soils, don't have the same ability to expand and contract as secondary minerals - more altered and hydrated (e.g. clay minerals like kaolinite or aluminosilicates (allophane)) (McLaren and Cameron, 1990).
- *Organic matter content*: the OM content of soils is manageable (Fig. 4.3). OM, like clays, has an overall negatively charged surface which attracts nutrients. OM can also hold elements by absorption or occlusion. Dissolved organic matter (DOM) has a strong affinity for organic contaminants (e.g. pesticides like DDT) (Aislabie et al., 1997) and can improve their movement through soils and release in water. The DOM content of a soil will therefore impact on its sorption efficiency.
- *pH*: soil pH is manageable (Fig. 4.3) and driven by the relative concentrations of H^+ and OH^- ions. The addition or removal of these ions from functional groups on OM and mineral surfaces will change the charge of the surfaces, influencing the sorption or release of other ions.
- *Soil depth* is an inherent property (Fig. 4.3): the deeper the soil, the more exchange sites for removing nutrients from the soil solution.
- *Soil nutrient status*: The nature and quantity of ions present in the soil will influence the type of reactions taking place.
- *Levels of saturation*: the saturation level of a soil nutrient retention capacity will determine future retention and thereby nutrients quantities in leaching waters.

Contaminants like pesticides or endocrine disrupting chemicals can be found intact or in various breakdown residues after biodegradation by soil biota (Aislabie et al., 1997).

Similarly, nutrients can be found in soils in different forms (McLaren and Cameron, 1990):

- Soluble – free – inorganic and organic compounds in soil solution,
- Labile –weakly sorbed– forms readily able to move into solution:
    - weakly adsorbed inorganic forms
    - soluble precipitates

  - easily mineralised organic forms
- Non-labile –strongly sorbed– forms insoluble, with a very low availability for plants:
  - sparingly soluble organic forms: soil biomass, undecayed plant and animal residues, stable soil organic matter (humus)
  - strongly absorbed and/or occluded by hydrous oxides
  - held by silicate minerals
  - sparingly soluble precipitates.

Nutrient cycling processes (Fig. 4.3) drive transformations between the different forms of nutrients and contaminants that can be found in a soil. Nutrients, and contaminants, are sorbed, or in precipitates, when they are not in soil solution. The concentrations of nutrients in their soluble form are generally in rapid equilibrium with the labile fraction, whereas reactions between labile and non-labile fractions are much slower.

Soil processes transform nutrients from soluble to labile or non-labile forms and vice and versa. These processes affect the degree of saturation of the soils' exchange sites and the soil nutrient retention capacity (McLaren and Cameron, 1990). These processes are:

- *Ion exchange*: Ions are attracted and accumulate on charged surfaces of soil colloids (clays and OM). The solid phase of most soils generally carries a net negative charge. These ions are not held irreversibly, but remain in equilibrium with nutrients in the soil solution. The charged surfaces of soil colloids can act as either a sink or a source of nutrients, depending on the net flow of nutrients.
- *Adsorption*: certain elements can chemically react (ligand exchange reaction) with functional groups on the surface of several types of clay minerals and/or organic matter. They get adsorbed as particles or as coatings. Adsorption is reversible. Adsorbed elements are still labile, but adsorption can be followed by occlusion.
- *Occlusion*: adsorbed elements can become non-labile if they are occluded within the matrix of a soil component. Occlusion can occur by diffusive penetration – the element slowly penetrates the structure of a soil mineral – or incorporation – the elements gets trapped on a soil mineral by developing coatings of hydrous oxides.
- *Precipitation*: when soils have been accumulating ions and dry out because of high evapotranspiration rates, soil solution concentrations increase and ions precipitate. The solubility of the precipitate formed depends on its nature (e.g. tri-calcium P ($Ca_3(PO_4)_2$) is insoluble whereas mono-calcium P ($Ca(H_2PO_4)_2$) is soluble).

Other supporting processes (Fig. 4.3) drive soil solution concentrations by removing or adding nutrients to the soluble pool. This impacts on transformations between nutrient forms and thereby affects the degree of saturation of soil nutrient retention capacity (Fig. 4.4). The supporting processes driving soil solution concentrations are (McLaren and Cameron, 1990):

- *Water evaporation from soil surface*: As the soil surface dries, water deeper in the profile rises by capillarity, changing soil solution concentrations.
- *Plant uptake*: plant uptake of nutrients, contaminants and water from soil solution changes soil solution concentration, which in turn modifies chemical equilibriums and impacts on transformation processes.
- *Drainage:* drainage removes nutrients and contaminants from the soil profile changing soil solution concentrations. The nature of soil flows (matrix or preferential flows) determines how fast nutrients are removed (Houlbrooke and Monaghan, 2009; McLeod et al., 2008).
- *Runoff:* runoff removes nutrients and organic compounds from the soil surface preventing them entering into the soil nutrients pool.
- *Mineralisation:* micro-organisms decompose organic compounds like plants and animal residues and release nutrients in mineral forms to the soil solution. Some organic compounds are resistant to degradation, becoming part of the soil humus complex. Micro-organisms also biodegrade contaminants, changing their form and concentration in solution.
- *Immobilisation:* micro-organisms utilise mineral forms in soil solution, incorporating them into cellular material forming their biomass.

Nutrients behave differently depending on their structure. For example, the phosphate ion ($H_2PO_4^-$) is a small molecule, in comparison to the nitrate (NO_3^-) anion, and is specifically sorbed and tightly held by soil minerals. In solution, it moves by diffusion down a solution concentration gradient, not in the transpiration stream. The nitrate ion is non-specifically sorbed and weakly held by soil minerals. In solution, it moves in mass flows (Barber, 1995). The ammonium ion (NH_4^+) is also an important source of N for plants held more tightly than nitrate. Whereas most of the P in a soil is in an inorganic form, most N is found in the organic fraction, with mineralisation and immobilisation processes dominating solution concentration, and hence plant availability. In comparison, it is the inorganic chemistry of the soil that determines P in solution.

Some degradation processes and external drivers impact on the properties and supporting processes regulating the filtering of nutrients and contaminants (Fig. 4.3) and thereby affect the provision of the service. These degradation processes and external drivers are investigated in the following section.

4.2.2 Degradation processes and drivers influencing the filtering of nutrients and contaminants:

The ability of soils to filter nutrients and contaminants and ensure water quality mainly depends on the following natural capital stocks: the amount and types of clay minerals in the soil, organic matter content and soil nutrient status (Fig. 4.3). Any degradation process and external driver impacting on these properties has the potential to impact on the provision of the service.

Degradation processes impacting on organic matter levels and soil nutrient status (manageable properties) will impact on the provision of the service (Fig. 4.3 and 4.4). Erosion, by removing soil particles, minerals and OM, will decrease the quantity of sorption sites and therefore decrease soil nutrient retention capacity.

Soil compaction decreases soil porosity by packing soil aggregates together, reducing the percentage of the soil matrix available for exchange, and causing preferential flows.

Soil acidification caused by excessive leaching of anions or cations, influences ion exchanges with clays and OM and thereby the amount of nutrients and contaminants held. For example, in grazing systems, urine patches show accumulation of nitrates which when leached lead to acidification.

The accumulation of some nutrients or heavy metals can influence the quality (preferential use) or quantity (luxury consumption) of nutrients taken up by plants, or stop it all together (phytotoxicity) (Wang et al., 2004).

The loss of soil invertebrates from treading, dry conditions or competition between species, as well as changes in soil properties that influence micro and mesofauna, will impact on mineralisation and OM levels, but also immobilisation rates and other biological processes.

Natural drivers like geology, vegetation, biodiversity and climate, also influence the filtering capacity of soils (Fig. 4.3 and 4.4). The diversity of minerals present in a soil is inherent (Fig.4.3) and depends on the parent material. The degree of weathering of the soil and the degree of development of the soil structure impact on exchange capacity, but also processes like drainage. The mineral composition of soil changes through time with weathering, as does the soil's ability to retain nutrients. Secondary clay minerals or aluminosilicates (allophanes) are more altered and hydrated therefore react more with soluble nutrients than primary minerals (micas) found in recent soils whose structure is more stable (McLaren and Cameron, 1990). Vegetation type can have a major influence on soil development (podzols) (Edwards et al., 1994b). The genetic and functional diversity of soil biota influence all biological reactions and thereby impacts on soil nutrient status and the biodegradation of pathogens and contaminants, and therefore their availability for release in water. Finally climate, including temperature and rainfall (Fig.4.3 and 4.4), influences soil water content and the intensity of biological and chemical reactions. Rainfall also influences the amount of water draining through the soil, the amount of nutrients and contaminants lost and the degree of saturation of the soil nutrient retention capacity.

Anthropogenic drivers like land use and management practices also influence soil filtering capacity at different levels (Fig. 4.3). Land use impacts on nutrient status directly through fertiliser use and plant species introduction. It also directly influences the diversity of plants and animals living on and in the soil (Fig. 4.4). This affects the amount and nature of inputs to the soil, thereby driving the replenishment of soil solution, but also soil organic matter levels and soil exchange capacity. For example, in pasture systems grazing animals often show camp behaviour. When they are not eating, cows gather in shaded areas, next to a water source, or on a flat dry part of the paddock. These areas can accumulate nutrients. The disproportioned deposition of dung and urine on these areas effectively makes the sites potential point-sources for leaching and runoff.

Management practices impact directly on the quantity and type of nutrients and contaminants added to the soil via fertilisers, dairy farm effluents (DFE) or pesticides. Fertilisers are used to sustain pasture production by keeping nutrients available for plants in the optimum range. Once in soils, nutrients enter the soil solution, are sorbed by clays, precipitated, leached, incorporated into OM by micro-organisms, or taken up by plants. Fertilisers, if used incorrectly, can contribute to environmental problems through leaching (eutrophication) or the accumulation of nutrients and heavy metals in soils. DFE are utilised for their nutrient content, but inappropriate application rates or timing of application, in relation to the SWC, can lead to surface runoff, leaching and groundwater contamination by nutrients and bacteria, nutrient imbalances in soils, animal health problems or water logging of soils (Hawke and Summers, 2006). Scheduling effluent irrigations, grazing events and applications of fertilisers and pesticides, based on soil physical properties and SWC is important to prevent the direct discharge of nutrients and contaminants into surface or groundwater due to direct runoff or drainage (Houlbrooke et al., 2004). Moreover, technology provides tools to prevent some processes. For example, nitrification inhibitors are used to slow down the transformation of ammonium (NH_4^+) into nitrate (NO_3^-) to prevent nitrate leaching and losses of N as nitrous oxide (De Klein and Eckard, 2008).

4.2.3 Quantifying the filtering of nutrients and contaminants:

4.2.3.1 Previous attempts to quantify the filtering of nutrients and contaminants:

A number of authors (Swinton et al., 2007; Wall et al., 2004; Weber, 2007; Zhang et al., 2007) have mentioned the ability of soil to filter nutrients as an ecosystem service, but to the knowledge of the author, no one has tried to model the provision of this service. Process-based soil models (Green et al., 2006) exist that inform the release of nutrients in water and the sorption and retention of contaminants, but to our knowledge modelling hasn't been used before to explore the provision of this soil service.

The ability of a soil to filter water has been long recognised and even managed at the watershed scale. For example, New York City has one of the few sources of natural, unfiltered water in the US, the Catskill/Delaware watershed. The natural filtering abilities of the wetlands soils and waterways of New York's ecosystems were being threatened by development, runoff from agricultural lands and impervious surfaces, and discharges from wastewater treatment plants at a time when the city faced the potential major investment in a new treatment facility. Between 1997 and 2007, New York City chose to implement a comprehensive watershed protection program to preserve and restore natural filtration services, as a more cost effective means of maintaining water quality than water treatment. Watershed management measures included land acquisition and comprehensive planning, water quality monitoring and disease surveillance, and upgrading existing wastewater treatment plants.

4.2.3.2 Parameters chosen to quantify the filtering of nutrients and contaminants:

To inform soil filtering capacity, the focus is on two contrasting nutrients, nitrogen (N) and phosphorus (P), both posing a potential threat to New Zealand surface and ground waters. These two nutrients are also very important for pasture growth and hence actively managed by farmers. Poor management of these two nutrients can lead to elevated losses to runoff and drainage waters, causing potential environmental problems. Since the quantities and chemistry of N and P are different, their regulation in soil solution is also very distinct.

In grazing systems the loss of N is due primarily to leaching of nitrate (NO_3^-), originating from urine patches, down through the soil to below the roots. The amount of N deposited on a urine patch can reach the equivalent of 200-1000 kgN/ha (Hoogendoorn et al., 2010). Some N can also be lost as NH_4^+ in leachate. NO_3^- is a weekly sorbed anion and, as a consequence is not held tightly on soil surfaces and is easily leached. This occurs mainly during the period of the year when net drainage occurs (usually May to September). The amount of NO_3^- leaching losses from a grazed pasture depends on the number of animal urine patches. Therefore, when animal numbers and production increase, so do NO_3^- leaching losses (Ledgard et al., 1999).

Phosphorus is a specifically sorbed anion tightly held by the soil. P loss occurs largely via surface runoff, unless the soil demonstrates preferential flow (e.g. cracking clays) or has very low sorption capacity (e.g. podzols) (Edwards et al., 1994b). P is lost in two forms, soil-bound P and dissolved-P, with the former often the dominant (60-90%) mechanism in less intensively farmed hill catchments (Parfitt et al., 2009). In comparison with N losses, the quantities of P lost are smaller and a significant proportion of the P lost on an annual basis can occur during single-storm events (Parfitt et al., 2009). N losses can be of agronomic significance, whereas P losses are generally not.

To describe soil nutrient retention capacity and its level of saturation, soil scientists have tests available to determine ASC and CEC as well as soil saturation. High ASC soils that are saturated can't sorb anymore nutrients so act as low ASC soils. High ASC soils non-saturated are known to adsorb more added P fertilisers, requiring farmers to apply higher amounts of P fertilisers to pasture to sustain a given solution P concentration and level of yield.

To quantify the filtering of N (NO_3^- and NH_4^+), a measure of the service was defined as the difference between a maximum loss (MaxNloss) specific to a soil type, depending on soil nutrient status, N inputs, management and production intensity, and the actual N loss (N leaching) (Fig.4.5). This quantification method is very innovative. This measure represents the amount of N the soil doesn't lose, that is the amount filtered. The maximum loss value is the amount of N that could potentially leach, but does not due to the soil filtering, or nutrient retention capacity. It depends on the soil absorption capacity, but also the amount of nutrients entering the soil and the amount of nutrients being used by plants, as well as the soil's drainage class. For the same nutrient status, a well-drained soil usually loses more nutrients than a poorly drained soil, as more water drains through it.

A number of different approaches were considered to quantify the Max N leaching loss:

1- Consider the amount of N inputs to the soil (fertilisers, N fixation by clover and dung and urine) as the maximum quantity that could be leached: This was not realistic since plants can use N very quickly, some N is lost by denitrification and some stored in soil as organic-N. This approach would tend to over-estimate the maximum potential N loss.

2- Model N losses on a free draining soil, to generate high leaching losses: The N cycle on an extremely free draining soil is very different from the one used in this study. For example, a free draining soil usually has a very low plant available water capacity, which means it grows less grass. This changes all the N dynamics of the system.

3- Use N leaching data over a number of years to define the top of the range as the maximum possible N loss for the soil: Considering the max N losses from a dataset as the potential maximum loss from the soil wasn't deemed satisfactory either because N losses are the result of a range of soil processes including N uptake by plants, mineralisation and denitrification. Only comparing N leaching data for different years would not have been successful in isolating the part of N leaching due to soil nutrient retention capacity.

4- A fourth option considered and subsequently used was to isolate leaching losses due to the soil nutrient retention capacity from inevitable losses from plant turnover and mineralisation. To do so, potential maximum N loss could be determined by modelling N losses for a soil with extremely low ASC close to zero, with a process-based model. Process-based soil models that link soil nutrient retention capacity, saturation levels and plant growth to the dynamics of nutrients in soil solution, drainage and runoff, offer an approach for exploring and quantifying this soil service.

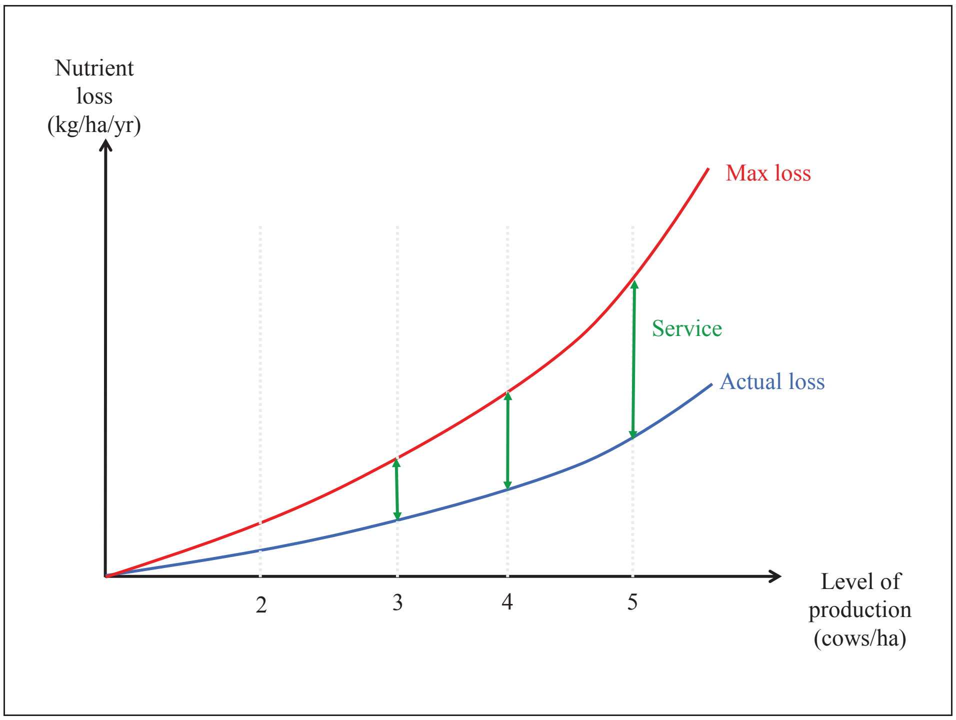

Figure 4-5: Quantification of the filtering of nutrients.

Similarly, to quantify the amount of P retained by the soil, the difference between the maximum amounts of P the soil could lose and the actual P losses (mainly runoff) was chosen as a measure of the service (Fig.4.5). The potential maximum P loss would depend on soil anion storage capacity, rainfall, SWC and runoff intensity.

The same methodology could be applied to quantify the filtering of pathogens and contaminants from DFE, pesticides and fertilisers. The quantities of pathogens and contaminants filtered by the soil could be determined by calculating the potential maximum amount of contaminants the soil could lose and subtracting from that value, the actual quantities of contaminants lost.

Detoxification and the recycling of waste are examined in the next section.

4.3 Detoxification and recycling of wastes:

A number of materials are applied to New Zealand soils each year. This includes wastes, like farm animal dung and urine, effluents from dairy sheds, standoff-pad effluents, piggery or poultry farm effluents, sludge from effluents ponds and composts. To that list can be added fertilisers and pesticides. These materials contain two types of threats:

- Compounds (organic or chemical) potentially harmful to the environment, and directly to animal and human health, including organic contaminants from pesticides, heavy metals and endocrine-disrupting chemicals (EDCs).
- Living organisms (pathogens) like viruses, bacteria, or parasites harmful to animals and humans.

A range of processes enable soils to detoxify, decompose and recycle wastes. These processes (biodegradation) release breakdown components in the form of nutrients reusable by plants and soil fauna or stable non-toxic compounds. Soils are also able to physically deactivate harmful compounds by sorbing them (detoxification). Detoxification and waste decomposition constitute an ecosystem service, linked directly to human health and the fulfilment of human need for a safe habitat.

The distinction needs to be made between the decomposition and recycling of plant litter and dead soil fauna, which is the supporting process (nutrient cycling) behind plant growth and the provision of food, wood and fibre examined in Chapter Three (section 3.2.1.3) and the service of detoxification and recycling of wastes. The supporting processes involved in detoxification and the recycling of wastes are similar to nutrient cycling processes, but because they are targeted at pathogens and contaminants potentially harmful to humans and the environment, they constitute a service.

In New Zealand dairy farms, animals graze perennial pasture *in situ* year round depositing dung and urine directly on soil surface. Moreover, dairy livestock spending time in yards, stand-off pads, and the milking shed also produce nutrient-rich dairy farm effluents (DFE), which consists of livestock excreta diluted with wash down water. Traditionally, DFE has been treated in standard two-pond systems and then discharged into a receiving fresh water stream (Houlbrooke et al., 2004). However, since the Resource Management Act (1991), most regional councils now prefer dairy farms to land treat their DFE, that is apply them to land, to allow the water and nutrients they contain to be utilised by pasture plants and improve soil fertility (Hawke and Summers, 2006; Houlbrooke et al., 2008; Wang et al., 2004). Since 1975 (Fig. 4.6) the number of dairy cows in New Zealand has doubled (LIC and DairyNZ, 2009) and since 1993 the proportion of Waikato farmers who irrigate DFE onto pasture rose from 35% to nearly 70% in 1997 to effectively 100% in 2004 (Hawke and Summers, 2006). Therefore, there are increasing amounts of wastes deposited on New Zealand pastures, and with them OM, pathogens, heavy metals and EDCs.

Fertilisers and pesticides (insecticides and herbicides) are also applied to dairy pastures and contain harmful compounds, including chemical residues and heavy metals.

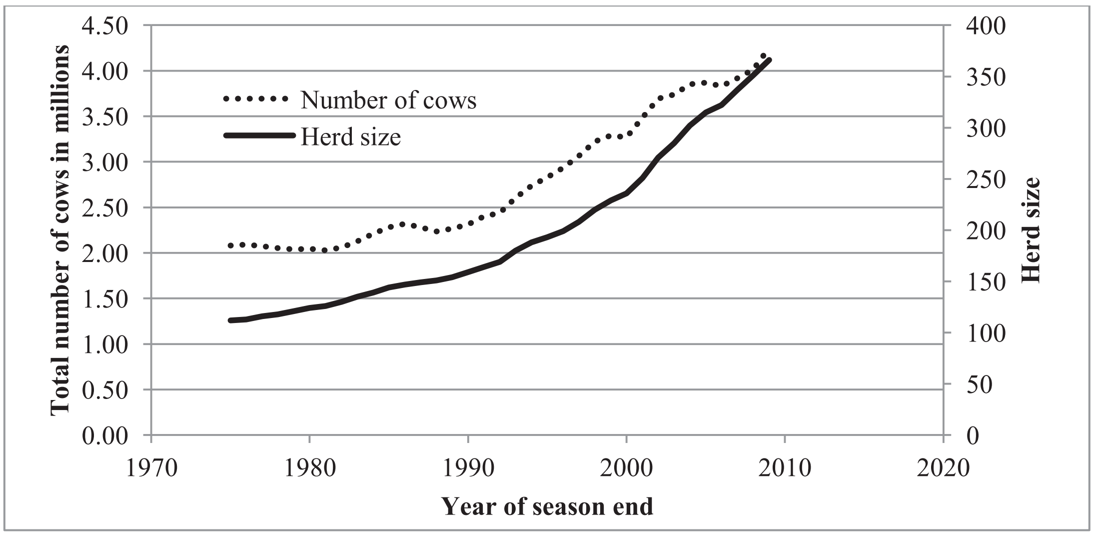

Figure 4-6: Total number of cows in New Zealand and herd size since 1975 (from LIC and DairyNZ, 2009).

The deposition of DFE on pastoral soils is known to change soil properties, or modify natural capital stocks (Fig.4.7). DFE is very rich in nutrients like N and C but its composition can vary greatly (Ghani et al., 2005; Saggar et al., 2004b). The impacts of irrigation with DFE of pastoral soils have been well studied in New Zealand but the results of these studies are soil type dependent (Hawke and Summers, 2006). The impact of irrigation with DFE on soil total C has been of particular interest since it is a potential technique to increase C sequestration. Some researchers have shown that effluent irrigation can increase total soil C (Hawke and Summers, 2006). Others have shown effluent irrigation to decrease total soil C (Sparling et al., 2001); or result in no change (Degens et al., 2000; Sparling et al., 2001). For example, Sparling et al. (2001) reported the effects of long-term application of dairy factory effluent to pastures on Horotiu (Allophanic Soil) and Te Kowhai (Gley Soil) soils. After 22 years of effluent application every 2 weeks, there was no effect on C content on the Te Kowhai soil, but an apparent decline in C content of the Horotiu soil. Further investigations by Degens et al. (2000) showed that the decline occurred only in the surface soil and that, lower in the profile, there had been a compensating accumulation of C, and no changes lower than 50 cm. Long-term effluent application seemed to have speeded the movement of C down the profile. A number of authors (Hawke and Summers, 2006; Sparling et al., 2001) also reported changes in soil structure mainly due to the increase in soil OM. Sparling et al.(2001) reported an increase in unsaturated hydraulic conductivity of both Horotiu and Te Kowhai soils and a decrease in bulk density of the Horotiu soils after long term effluent irrigation. However, effluent irrigation can also result in plugging of pores, changes in the pore size distribution of the topsoil and aggregate collapse (Hawke and Summers, 2006).

Land treatment with DFE also has effects on the amount of nutrients stored in soils like N and P. Regular application of DFE can saturate the soil capacity to retain these nutrients. Even if the results of different studies are mitigated (Hawke and Summers, 2006) most researchers have shown that effluent irrigation increases total soil N (Degens et al., 2000; Houlbrooke et al., 2004). Since the majority of N in DFE is organic, and so slowly available, the application of effluents would increase the availability of soil N over the long term (Hawke and Summers, 2006).

There are a number of adverse effects associated with dairy cow wastes (fresh or as effluents), fertilisers and pesticides if handled inappropriately. These adverse effects include (Hawke and Summers, 2006; Houlbrooke et al., 2004; Wang et al., 2004):

- Nitrate and phosphorus loss to ground water and waterways (section 4.2),
- Odour and gaseous emission and notably N_2O emissions (section 4.4),
- Heavy metal accumulation in soil (this section),
- Enhanced organic contaminant (pesticide) mobility in soil (section 4.2),
- Nutrient imbalances inducing nutritional disorder of animals (this section),
- Pathogen movements (section 4.2) and survival (section 4.5) and related health risks,
- Endocrine-disrupting chemicals (EDCs) contamination of soils (this section) and waterways (section 4.2).

The ability of soils to filter nutrients (de Klein and Ledgard, 2001; Ghani et al., 2005; Parfitt et al., 2008b), pathogens and contaminants and thereby prevent their release in waterways is an ecosystem service in itself and is treated in the previous section (section 4.2). The regulation of emissions of greenhouse gases (GHGs) from soils due to urine and effluent applications to land (Saggar et al., 2007a; Saggar et al., 2004b; Saggar et al., 2004c) is another ecosystem service and is examined later in this chapter (section 4.4). The biological regulation of pest and disease populations and pathogen survival in soils is also an ecosystem service and is examined later in this chapter in section 4.5.

This section focuses on the ability of soils to deactivate non-organic contaminants, and biologically degrade organic wastes, by investigating the properties and supporting processes that impact on toxic compound deactivation and waste degradation and recycling (Fig.4.7). Also examined are the degradation processes associated with land application of DFE, fertilisers and pesticides, including nutrients imbalances and heavy metal and EDCs contamination, as well as the external drivers impacting on the service (Fig.4.7).

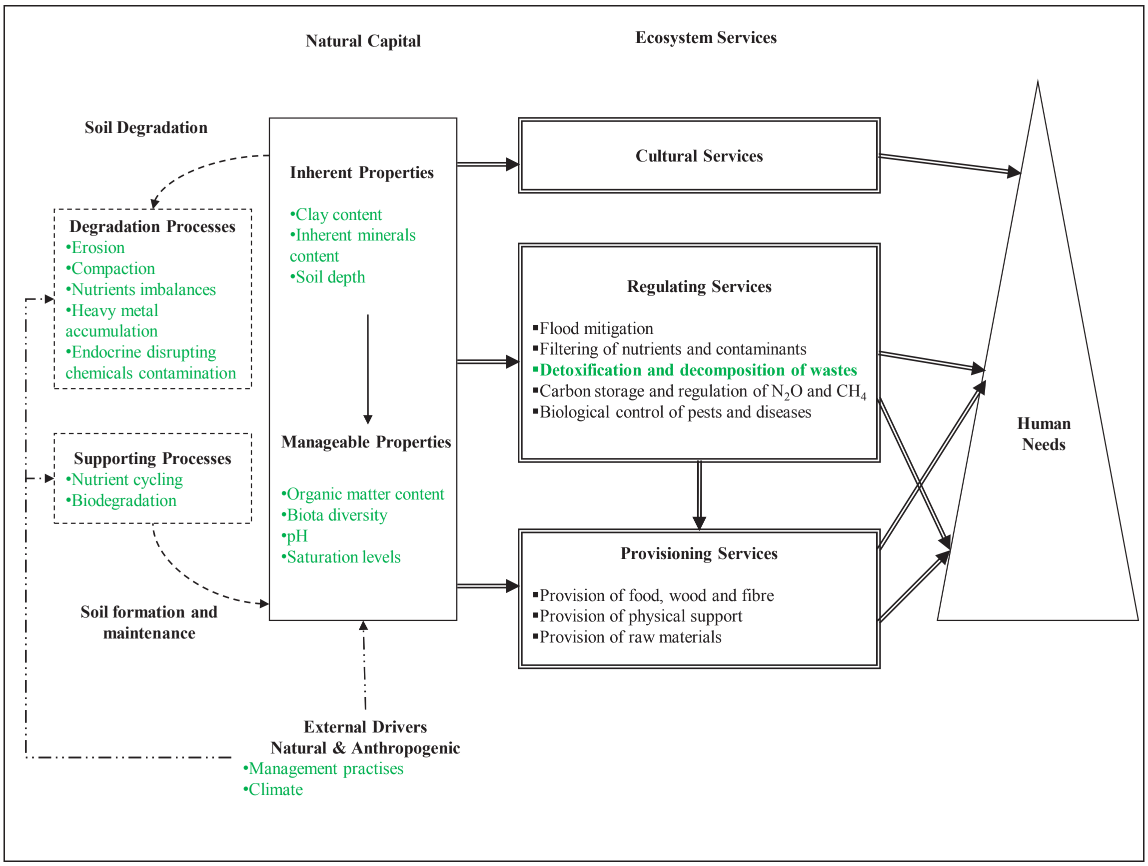

Figure 4-7: Detail of the conceptual framework applied to detoxification and the recycling of wastes.

4.3.1 Soil properties and supporting processes contributing to detoxification and the recycling of wastes:

There are two main supporting processes involved in detoxification and the recycling of wastes: the sorption of compounds on soil particles and biological degradation (Fig.4.7).

First, soils are able to adsorb compounds on clays and OM surfaces. Section 4.2.1 examined in detail the properties (natural capital stocks) behind soil nutrient retention capacity and the supporting processes driving nutrient movements in soils. The sorption of contaminants (pathogens, organic or chemical) depends on the same properties and processes. Soil properties like pH, soil depth and soil saturation levels impact on the capacity of soils to retain contaminants and prevent their release in waterways (section 4.2.1) (Fig.4.7 and 4.8). The sorption of contaminants on clays and OM also plays a role in their deactivation, and thereby on soil detoxification. It also enables soil biota to access and degrade them. Negatively charged microbes (McLeod et al., 2008), organic chemicals, heavy metals or EDCs can all be sorbed and retained on soil particles. DDT, an insecticide that was used against grass grubs in New Zealand, has been demonstrated to have a high affinity for soil organic matter (Aislabie et al., 1997). Once bound to soil, DDT residues are detoxified and lose their activity. Moreover, soil microbes are known to play an important role in the binding of pesticide residues to soil organic matter (Aislabie et al., 1997). However, the accumulation of toxic compounds in soils (some nutrients, heavy metals, EDCs, chemicals) can be a problem in some areas and is considered a degradation process (Fig.4.8).

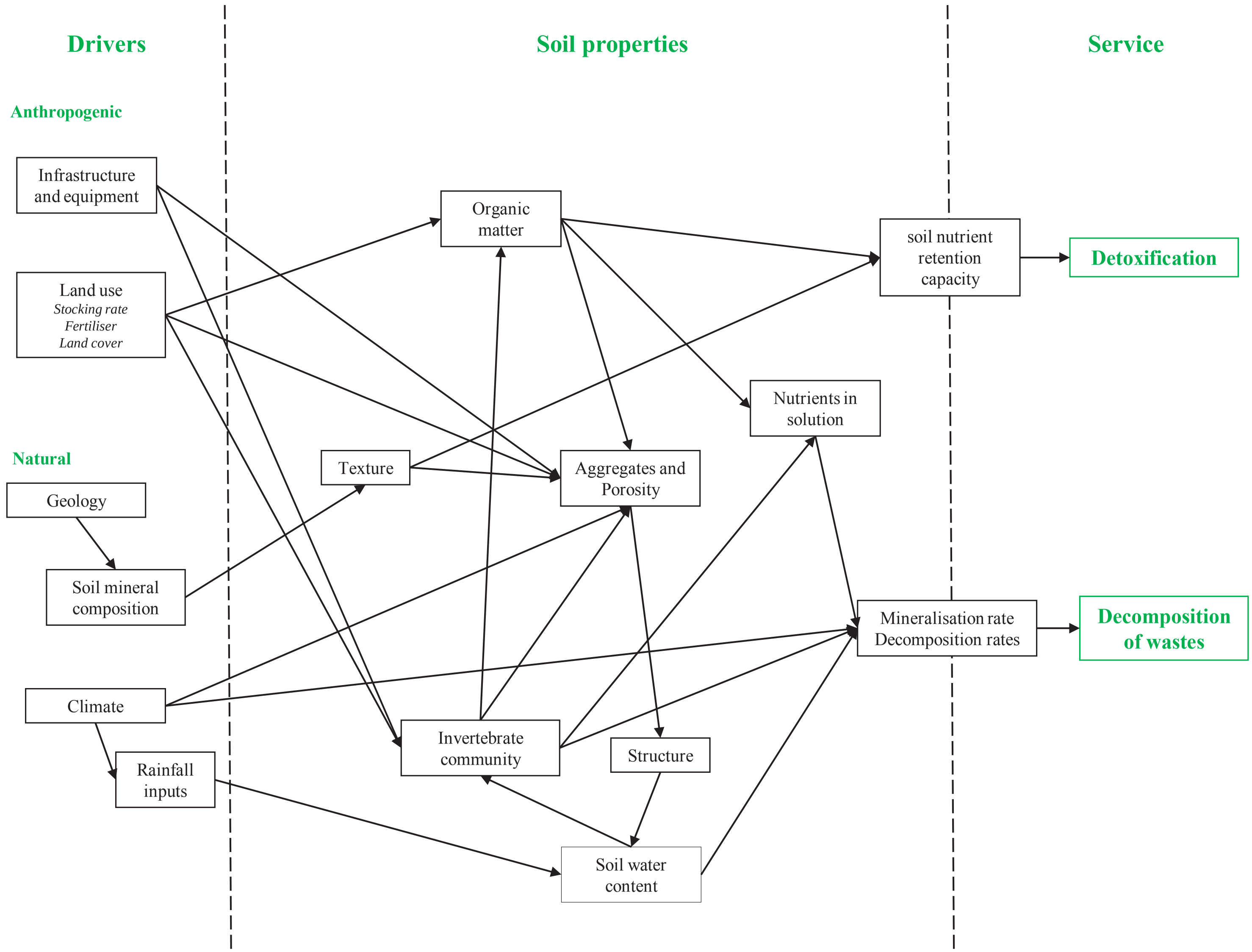

Figure 4-8: Drivers and soil properties influencing detoxification and the recycling of wastes.

Secondly, soils are able to decompose organic and chemical materials. The recycling of wastes (similarly to the recycling of plant and animal residues) goes through several stages (McLaren and Cameron, 1990). First, materials deposited on the soil surface have to be incorporated into the soil. This can occur by transport in drainage water, by treading, and importantly by the activity of macro-fauna species like e.g. earthworms (Epigeic and Anecic) or detritus-feeding Collembola (Bardgett and Cook, 1998; Schon, 2010). Heterotrophic organisms present in the soil use the residues as a source of food, effectively degrading and in many instances detoxifying toxic compounds. When these organisms die, their bodies enter the food pool. The biological activity breaks down the organic compounds present in the residues, releasing other compounds and CO_2. Some of the compounds present in wastes and some of the bio-products formed during the decomposition process are resistant to further oxidation and are involved in the formation of humus. The biodegradation of chemical contaminants can take different forms. The biodegradation of the insecticide DDT by bacteria and fungi involves co-metabolism which means that the microbes are growing at the expense of a growth substrate (alternative C source) and are able to transform DDT without deriving any nutrient or energy for growth from the process (Aislabie et al., 1997). Some fungi and bacteria are resistant to heavy metals thanks to mechanisms of tolerance and detoxification including the production of chelating agents that bind metals and reduce their toxicity or, the enzymatic attack of the compounds (Kavamura and Esposito, 2010).

The biodegradation of contaminants is controlled by the availability of some nutrients in the soil, which is a manageable property (Fig. 4.7 and 4.8). Aislabie et al. (1997) suggested that the availability of extra N and C can enhance co-metabolic metabolism of DDT. The C:N ratio of the material added to the soil impacts on the speed of the decomposition process. Soil biomass has a C:N ratio of between 9:1 and 4:1. Most of the material entering the soil has a C:N ratio >30:1 (plants range from 20:1 to 100:1, cow dung is around 30:1). Therefore, wastes entering the soil have too little N for micro-organisms to convert all the C. To decompose materials with wide C:N ratios, micro-organisms need N, which they may take up in mineral form (NO_3^-) present in the soil. This process is called immobilisation (McLaren and Cameron, 1990). The critical value above which N immobilisation occurs is usually 20:1. A decrease in the C:N ratio makes greater quantities of the mineralisable forms of soil N available for plant uptake, (Hawke and Summers, 2006). Soils rich in available mineral N usually show faster decomposition rates.

After the addition of a fresh food source, the populations of organisms increase to a maximum, which coincides with the breakdown of the more easily decomposed organic compounds (sugars like lactose in DFE, or simple proteins) and the maximum release of CO_2. Authors (Degens et al., 2000; Ghani et al., 2005; Sparling et al., 2001) have reported increases in microbial biomass due to irrigation with DFE and the associated inputs of available C (particularly lactose) (Degens et al., 2000). The increase in the microbial biomass pool is linked to enhanced immobilisation of nutrients, mainly N and S, which are greater at higher temperature (Ghani et al., 2005). Immobilisation due to the decomposition of wastes has effects on the availability of N for plant growth (Ghani et al., 2005).

After the simple compounds have disappeared, the number of organisms decline and only those that are capable of decomposing complex compounds like cellulose and lignin remain. Decomposition rates are slower and only material indistinguishable from humus remains.

The bio-products from the decomposition of wastes get added to the soil OM pool but soil OM levels do not increase indefinitely: humus is also slowly decomposed and C released as CO_2 (McLaren and Cameron, 1990). During soil formation, although there is a continual turnover between formation and decomposition of humus, soil organic matter ultimately reaches a steady state under a given set of management practices.

Different degradation pathways exist for aerobic and anaerobic conditions (Fig. 4.8). Wet soils support a different micro-flora than well-drained soils. In wet soils, the decay of organic materials differs qualitatively and quantitatively from that of aerobic soils. In anaerobic conditions, the decomposition of wastes is relatively slow, as the activity of a lot of micro-organisms is limited.

Environmental conditions and soil properties (natural capital stocks), including soil nutrient status, temperature and water content (Fig.4.7) are critical in determining the efficiency of biodegradation of wastes by micro-organisms.

Detailed in the next section are a number of degradation processes and external drivers that impact on natural capital stocks and supporting processes behind the detoxification and recycling of waste (Fig.4.7 and 4.8).

4.3.2 Degradation processes and drivers influencing the detoxification and recycling of wastes:

Degradation processes that impact on soil natural capital stocks and supporting processes affect the efficiency of detoxification and the recycling of wastes. They include:

- Compaction: by reducing habitable pore space for macro-fauna, soil aeration, slowing drainage, and increasing the potential period soils are anaerobic, compaction affects the numbers and types of organisms present, and therefore the efficiency of the decomposition.

- Erosion and leaching remove nutrients used by soil biota to efficiently decompose wastes. Erosion also removes soil material, decreasing the amount of surface available to sorb nutrients and contaminants.
- Chemical processes like salinisation, acidification or the accumulation of heavy metals or some nutrients, change soil chemical equilibriums, thereby impacting on the amounts of nutrients and contaminants in solution and the activity of soil fauna. Some organisms can't live, or are less efficient, under acidic conditions, resulting in slower decomposition of organic matter.
- The land application of DFE, fertiliser and pesticides, if poorly managed, can have adverse effects including nutrient imbalances, heavy metal accumulation, and EDCs contamination (Hawke and Summers, 2006; Wang et al., 2004). The application of effluents is currently based only on N loading, with no attention given to the concentration of the effluent in other elements like K, Mg, heavy metals or EDCs. These effects are processes degrading soil natural capital stocks and the soil's ability to provide ecosystem services (Fig 4.7). Heavy metal accumulation can lead to phytotoxicity for some crops, reducing the number of land use options available for contaminated soils (Mackay, 2008).

Nutrient imbalances:

DFE generally contains high concentrations of K, but relatively low concentrations of Mg and Ca. The excessive supply of K to soils can result in excessive K uptake and decreased Ca and Mg uptake by pasture, leading to increased K, and decreased Ca and Mg intake by animals (Bolan et al., 2004; Wang et al., 2004) inducing animal health problems. Ca and Mg deficiencies increase risks of "milk fever" (hypocalcaemia) and "grass staggers" (hypomagnesaemia) respectively in livestock leading to decreased milk production. High levels of soil K also disperse clays and can degrade soil structure (Wang et al., 2004), an important natural capital stock (Fig 4.7).

Heavy metal accumulation:

Research has focused on accumulation of N and P due to land application of DFE (Degens et al., 2000; Ghani et al., 2005; Hawke and Summers, 2006; Houlbrooke et al., 2008) but DFE can also contain metals such as copper (Cu) and zinc (Zn) (Wang et al., 2004) derived from the animal diet (supplements to treat Cu deficiencies), medicines from disease prevention (e.g. Zn to treat facial eczema, Cu to treat lameness), or growth promoters (Bolan et al., 2003). Cu and Zn are strongly bound to OM in effluents, therefore application of DFE is likely to result in the accumulation to toxic levels of these metals in the topsoil. Bolan et al. (2003, p.230) showed that land application of DFE based on N loading of 150 kg N/ha, is likely to add up to 31.5 kg Cu/ha as effluent, and 73.7 kg Cu/ha as manure sludge.

Land application of fertilisers and pesticides (insecticides and herbicides) are also responsible for the accumulation of heavy metals in soils. Longhurst et al. (2004) showed that total soil cadmium (Cd) (of 398 New Zealand soils) was highly correlated ($P<0.001$) to total soil P, suggesting Cd enrichment in pastoral soils was related to P fertiliser applications. Plant uptake and soil ingestion by livestock are entry point of Cd into the food chain resulting in animal and human health problems (Longhurst et al., 2004).

Endocrine-disrupting chemicals contamination:

DFE and some pesticides (e.g. DDT residues) are potential sources of endocrine-disrupting chemicals (EDCs) as they contain natural hormones like oestrogens. The release of EDCs in the environment can induce reproductive disorders in wildlife even at very low concentrations (Wang et al., 2004). Oestrogens can be sorbed by soils and degraded by various micro-organisms, reducing the risk of contamination of water. These processes are still under active investigation.

Organic contaminants:

Pesticides (insecticides and herbicides) contain toxic organic compounds that can get sorbed on soil clays and OM and accumulate. Dissolved organic matter (DOM) from DFE has a strong affinity with these compounds and can lead to their release from soil particle surfaces, improving their movement through soils, and release in water. The association of DDT with the soluble humic fractions of the soil can result in an increase in the solubility and hence mobility of DDT (Aislabie et al., 1997), therefore the amount of DOM present in soil will impact on the movements of organic contaminants.

There are also a number of natural drivers influencing the detoxification and recycling of wastes (Fig. 4.7 and 4.8). Climate, by driving soil temperature and water content (Fig. 4.7), influences the intensity of soil fauna activity and therefore, the rate of biodegradation of wastes. Schon et al. (2010d) looked at the impacts on soil invertebrates of changes in the physical environment and feed availability in intensive pastoral systems. They studied a well structured loamy Andosol soil in two seasons (autumn and winter sampling). They showed that prolonged soil water content deficit had suppressed decomposition, causing an accumulation of dung material, providing more potential food resources for detritus-feeding Collembola, which play an important role in the incorporation of litter (and carbon) into the soil. As soil water content increased in winter, Collembola abundance decreased and the bacterial-pathway could quickly utilise this dung and litter, reducing food resources for Collembola. Local variations in topography modify the microclimate of the soil and thereby influence soil fauna habitat and the efficiency of waste degradation. The nature of soil parent material (Fig. 4.8) determines the types of mineral present in the soil like clays, thereby determining soil nutrient retention capacity and the capacity of soils to sorb and detoxify contaminants, as well as nutrients and minerals available to soil fauna for waste degradation.

Anthropogenic drivers like land use and farming practices also impact on detoxification and waste decomposition. Several farming practices (Fig. 4.7) influence decomposition rates through their influence on the type and amount of wastes entering the soil (irrigation with DFE, grazing regime, pasture harvest) and thereby, the feed availability for soil biota. Farming practices thereby influence the level of biological activity available for the decomposition of wastes, as well as soil nutrient status. The deposition of dung or DFE on pastures returns nutrients to the soil in easily metabolisable forms, favouring the faster bacterial-decomposition pathway over fungal-feeding pathway (Schon et al., 2010d). Hence in theory, soil having received effluent before should degrade wastes faster than a soil where a different fauna is established. The quantity (depth applied) and timing (regarding SWC) of DFE application or loading is critical and will determine if the nutrients and contaminants contained in the effluent are drained or leached or if they reside long enough in soils to be decomposed (Houlbrooke et al., 2004). Increased aeration (oxidation) due to e.g. tillage means increased microbial respiration and decomposition rate. Repeated wetting and drying, through cultivation or irrigation, increase decomposition rates by maintaining ideal soil water content and aeration conditions.

Excessive use of fertilisers and application above the plant requirements can lead to nutrient imbalances, the accumulation of heavy metals to toxic levels and the saturation of sorption sites, making soils unable to detoxify. Fertiliser and lime use, by increasing soil concentrations of available nutrients can enhance decomposition rates, by enabling micro-organisms to decompose wastes with high C:N ratios more efficiently. For example, different methods of bioremediation are used to rehabilitate contaminated soils, e.g. soils that have suffered oil spills. Bioremediation is a technique that uses living organisms in order to degrade or transform contaminants into their less toxic forms. It is based on the existence of microorganisms with the capacity to enzymatically attack the compounds (Kavamura and Esposito, 2010). Nutrients can be applied to the soil to increase the activity of bacteria naturally present in the soil, that actively consume oil-derived toxic compounds, transforming them into CO_2 (Kavamura and Esposito, 2010).

4.3.3 Quantifying detoxification and the recycling of wastes:

4.3.3.1 Previous attempts to quantify detoxification and the recycling of wastes:

A number of authors have mentioned soils ability to recycle wastes. Costanza et al. (1997) talk about "waste treatment". The MEA (2005) mention a regulating service "water purification and waste treatment". De Groot (2006) talks about regulation functions like "nutrient regulation" and "waste treatment". Other authors (Daily, 1997; Swinton et al., 2007; Wall et al., 2004) mention "disposal of wastes" or "nutrient cycling and mineralisation" (Barrios, 2007; Lavelle et al., 2006; Porter et al., 2009; Sandhu et al., 2008; Weber, 2007; Zhang et al., 2007). Costanza et al. (1997) valued waste treatment of different ecosystems by using techniques based on the 'willingness-to-pay' of individuals for the service. Sandhu et al. (2008) assessed the rate of mineralisation of plant nutrients using bait-lamina probes during field experiments. They used as indicators total organic matter content of soil, total nitrogen and the ratio of organic matter to nitrogen (20:1). Porter et al. (2009) assessed mineralization of organic matter provided by soil microorganisms and invertebrates by using data obtained from field experiments. They used total amount of N in soil, soil bulk density, soil volume and mineralization percentage (%, obtained from bait-lamina probes). These methods though recognising the link between soil properties and the provision of the service, do not make the distinction between plant litter decomposition, a supporting process (nutrient cycling) behind the provision of food, wood and fibre, and detoxification and waste decomposition, a service in its own right. These methods use some soil properties (OM content, total N content) to model mineralisation, but do not include the dynamics of soil water content.

To the knowledge of the author, no one has specifically modelled detoxification and the recycling of wastes as part of an ecosystem services framework before.

4.3.3.2 Parameters chosen to quantify detoxification and the recycling of wastes:

Ideally, to inform the detoxification service, pathogens and contaminant loads need to be considered, as well as the soils potential to retain and degrade them. The risk of bypass flows, and its effect on the efficiency of detoxification should also be included in any analysis.

To quantify the recycling of wastes in the context of a dairy grazed system, soil condition, including SWC, aeration (macroporosity), macrofauna populations and soil nutrient status (nitrates concentration), the key soil properties driving the efficiency of microbial activity, need to be linked to the amount and timing of dung deposition on pasture. A measure of the service can be defined as the difference between the total amount of dung deposited on pastures and the amount of dung deposited under restricting conditions for dung decomposition. This measure would represent the amount of dung deposited on pastures in ideal conditions for waste decomposition that is the amount of dung which is theoretically efficiently recycled. This method of quantification would use soil properties as a basis, and focus on the service actually provided by the soil.

Process-based soil models could be used to follow soil conditions daily and identify soil conditions when grazing events occur.

Carbon storage and greenhouse gases (GHGs) regulation are examined in the next section.

4.4 Carbon storage and greenhouse gases regulation:

Soils can store carbon (C) which has become very interesting for signatory countries of the Kyoto Protocol. For instance, New Zealand has the option at a future date to include soil C in its GHGs inventory. Globally, there is more organic C stored in soils than the total amount in living land plants and the atmosphere (Table 4.2). C flows to and from soil are as important, because measuring them enables us to understand if a soil is a net sink or source of C.

Table 4-2: Estimated major stores of C on the Earth (Pidwirny, 2010).

Sink	Amount in Billions of Metric Tons
Marine Sediments and Sedimentary Rocks	66,000,000 to 100,000,000
Ocean	38,000 to 40,000
Fossil Fuel Deposits	4000
Soil Organic Matter	1500 to 1600
Atmosphere	578 (as of 1700) - 766 (as of 1999)
Terrestrial Plants	540 to 610

Soils also contain a diversity of gases coming from diffusion from the atmosphere (oxygen O_2, carbon dioxide CO_2) or being produce within the soil by biological (methane CH_4, nitrous oxide N_2O, CO_2) or chemical reactions. The storage of C by soils is one of the processes behind the regulation of greenhouse gases (GHGs) emissions from soils to the atmosphere.

C storage and GHGs emissions are soil processes (Fig. 4.9) but the fact that soils regulate and buffer these processes constitutes an ecosystem service, since GHGs emissions impact on air quality and are potentially harmful to humans through global warming. Their regulation fulfils humans need for a safe environment.

To inform the provision of this service, the properties and supporting processes (Fig. 4.9) behind C storage and the regulation of emissions of CH_4 and N_2O are examined.

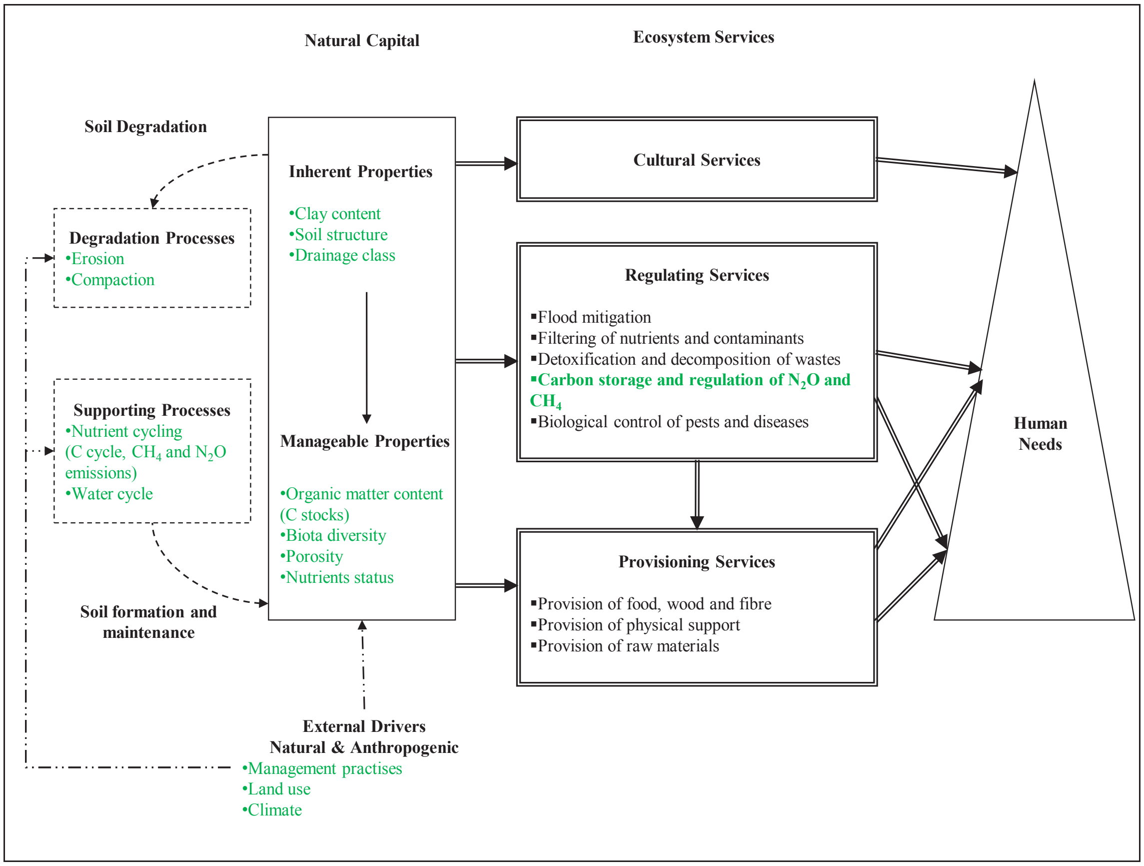

Figure 4-9: Detail of the conceptual framework applied to carbon storage and greenhouse gases regulation.

4.4.1 Soil properties and supporting processes contributing to carbon storage and greenhouse gases regulation:

The soil–atmosphere exchanges of CH_4, N_2O and CO_2 depend on complex interactions between soil properties, biota, climate, and agricultural practices (Fig. 4.9) (Saggar et al., 2008). This section details the properties and supporting processes (Fig. 4.9) behind the overall regulation of GHGs by soils. The supporting processes considered here are C storage, the net flows of C and the net flows of CH_4 and N_2O emissions (Fig. 4.9).

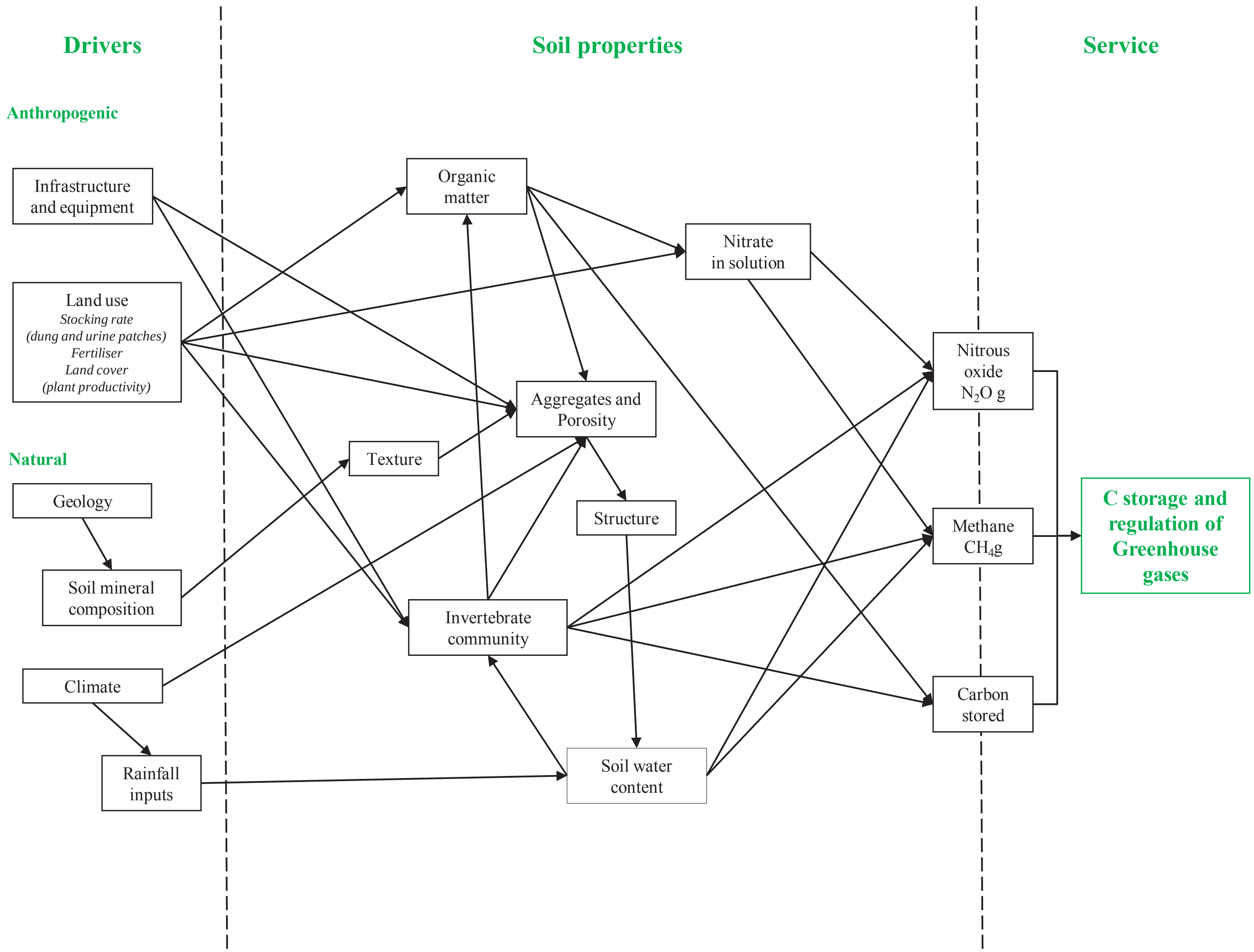

Figure 4-10: Drivers and soil properties influencing carbon storage and greenhouse gases regulation.

4.4.1.1 Carbon stocks and carbon flows:

Carbon stocks: the property behind carbon storage

Soils are particularly important, as they are the largest 'reservoir' of C in the terrestrial biosphere (Scott et al., 2002). Soils store C mainly as organic matter (OM). In fact, soil OM is about 60% C. The C content of a soil has an inherent component (Fig. 4.9) which depends on soil type and is determined by parent material, climate, the age of the soil, and vegetation (Fig. 4.11). It also has a manageable component (Fig. 4.9) that changes with land use and management practices.

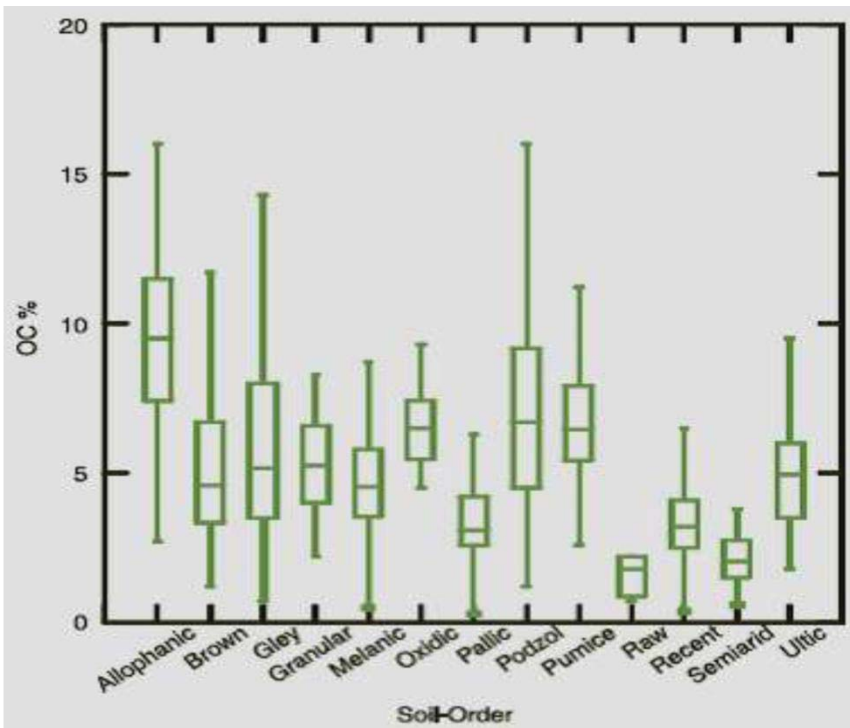

Figure 4-11: Organic carbon as percentage of soil mass in different New Zealand Soil Orders (from the New Zealand Soils Database).

Land use as well as management practices (Fig. 4.9) have an great impact on soil C stocks (Parson et al., 2009). The complex interactions between these drivers and C stocks are currently the focus of considerable research to understand more precisely the drivers and timeframe of soil C changes (Parson et al., 2009).

In New Zealand, legume based pasture grazed in situ is the dominant land use (Table 4.3). Pasture soils contain by far the largest amount of soil C (Tate et al., 2005b).

New Zealand, as a signatory to the United Nations Framework Convention for Climate Change and the Kyoto Protocol, has developed a national system of C inventory and a policy to reduce net GHGs emissions (MfE, 2005; MfE, 2009a; MfE, 2009b). The potential of soils to sequester atmospheric CO_2 has been widely talked about especially for countries owning large areas of arable land and having the option to switch from conventional tillage practices to no till practices or perennial plants such as pastures. However, in New Zealand, the potential for increasing storage of C in soils to the extent needed to offset the shortfall under the Kyoto protocol is very limited (Tate et al., 2005a). There are a number of reasons for this. First, in New Zealand, less than 1% of managed land is under arable crops, and more than half managed land is under a grassland system (Table 4.3).

Table 4-3: Areal extent of major land-cover types in New Zealand in 2000 (from Tate et al., 2005b)

Land cover	**Area** (Mha)	**Area** (% of total land)
Grassland	**13.61**	**50.9**
Forest land		29.9
Indigenous	6.25	
Exotic/Planted	1.73	
Cropland		1.2
Arable and grain	0.21	
Horticulture	0.09	
Shrub land	2.65	9.9
Others	2.20	8.2
Total land	26.70	100

Pastures are already highly effective in storing C, because of their high productivity and prolific root systems (Tate et al., 2005a). In the last 15 years, a few studies (Scott et al., 2002; Tate et al., 1997; Tate et al., 2003; Tate et al., 2005b; Trotter et al., 2004) have assessed New Zealand soils C stocks and flows to assist the country to achieve its CO_2 emissions reduction target under the Kyoto Protocol. Tate et al. (2005b) developed an IPCC-based Carbon Monitoring System (CMS) to monitor New Zealand soil organic C stocks and flows. They concluded that most of New Zealand's soil C is stored in 14 Mha of pasture land (1480±60 Mt soil C to 0.3 m depth), mainly improved pastures (Table 4.4).

Table 4-4: Soil (0-0.3 m) C stocks (1990) for different land uses (from Tate et al., 2005b)

Land use	**Area** (Mha)	**Soil C** (Mt)	**Soil C** (t/ha)
Grazing land	**14.0**	**1480 ± 58**	**105.7**
Exotic forest	1.3	77 ± 23	59.2
Natural (shrub) vegetation	2.7	244 ± 18	90.4
Cropland	0.3	26 ± 3	86.7

A major issue with the determination of C stocks (natural capital stocks) under pastures is that changes in pasture management alter several properties and processes of the C cycle at the same time (Fig. 4.9). Therefore it is difficult to predict in any one situation the effects on soil C stocks of a change in management. Because of the complexity of the C cycle, very different combinations of fertility and grazing intensity may give rise to similar stocks and flows of C (Parson et al., 2009). Nevertheless, some studies show clearly the impact of soil type and management on C stocks. For example, Schon et al. (2010a) showed that total C stocks vary on two soil types, in the 0-75 mm layer, under three managements in the Waikato, New Zealand (Table 4.5).

Table 4-5: Soil properties in dairy-grazed pasture under three stocking rates on two soil types, Waikato, New Zealand (from Schon et al., 2010a).

Soil Type	**Horotiu silt loam**			**Te Kowhai silt loam**		
Stocking rate (cows/ha)	2.3	3	3.8	2.3	3	3.8
Total C (%) [1]	6.2	5.1	4.9	8.1	7.4	5
C:N ratio [1]	10.6	10.5	10.7	11.9	10.8	10.8

[1] measured at 0-75 mm depth.

Schipper et al. (2010) re-sampled 83 soil profiles in New Zealand to investigate whether changes in soil C stocks were related to land use. Over an average of 27 years, soils (0-30 cm) of lowland dairy pastures lost on average 0.73±0.16 tC/ha/yr. They observed no significant change in soil C in lowland pasture grazed by "dry stock" (e.g. sheep, beef), or in grazed tussock grasslands (Table 4.6). Grazed hill country soils (0–30 cm) appeared to be gaining 0.52±0.18 tC/ha/yr. C:N ratios also declined significantly. Their results reported to 60 and 90 cm show that the pattern of losses and gains extend beyond the IPCC accounting depth of 30 cm (Schipper et al., 2010).

Table 4-6: Change in total C of grazed land for different land categories (tC/ha/yr) for 0-30 cm depth. Standard error of the mean in parenthesis (from Schipper et al., 2010).

Land form	**Land use**	**Number of samples**	**Average**
Lowland	Dry stock	27	-0.14 (0.15)
Lowland	Dairy	29	-0.73 (0.16)***
North Island hill	Dry stock	15	0.52 (0.18)*
South Island tussock	Dry stock	12	0.00 (0.13)

*,**,*** Significantly different from 0 at $P < 0.05$, $P < 0.01$ and $P < 0.005$ respectively.

The stability of soil C stocks highly depends on net flows of C to the soil, which are examined in the next section.

Processes behind Carbon storage:

The amount of C stored in soil (natural capital stock) (Fig. 4.9) is the result of many processes, C flows, making up the C cycle (Fig. 4.12). Plants fix CO_2 through photosynthesis (primary production). They also emit CO_2 through respiration. Soil autotrophic respiration specifically refers to root respiration. The net fixation of C by plants (CO_2 fixed - CO_2 emitted) is called "net primary production" (NPP) and is usually expressed as a rate in tC/ha/yr. In dairy grazed systems, some of the plant material is eaten by grazing animals and the plant C is then returned to the soil as dead OM through dung excretion. Similarly, when plants die, the dead OM enters the soil and is degraded by soil fauna. Micro-organisms which degrade OM emit CO_2 in the process: this is called soil heterotrophic respiration. Soil C stocks are mainly organic C, but soils also contain mineral C in solution (HCO^{3-}, CO_3^{2-}), adsorbed on clays (Fig. 4.12). Soils can lose C by leaching of dissolved organic C (Ghani et al., 2010), but the main processes responsible for C loss are either OM degradation or erosion. During transport of eroded soil to the sea, OM is oxidised and CO_2 is lost (Dymond and Baisden, 2010).

When investigating soil C stock, it's very important to consider net flows of C from soils, because they determine C stock stability.

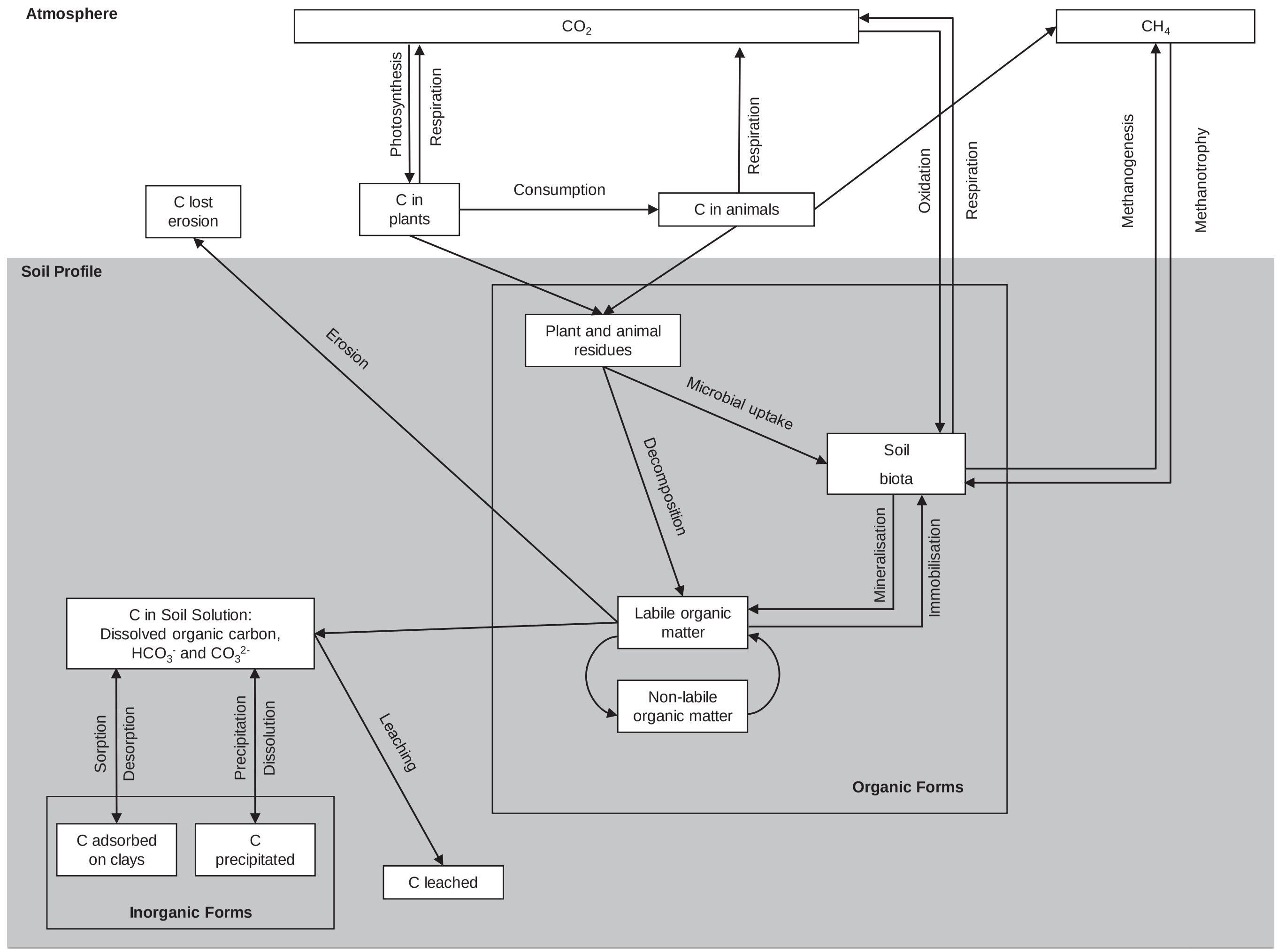

Figure 4-12: Soil carbon cycle (McLaren and Cameron, 1990).

The properties (Fig. 4.9) linked to the C cycle and thereby impacting on C flows and C storage are:

- Soil conditions influencing net primary production, and hence the potential size of the C pool.
- Porosity and aeration: they determine the activity of soil biota responsible for mineralisation, immobilisation and soil respiration (Fig. 4.12). Porosity also impacts on drainage and thereby SWC and DOC leaching.
- Soil biota: the type of organism present impact on all the biological processes of the C cycle.
- Clay content: clays can sorb C and stabilise OM.
- Actual C stocks: some ecosystems are already at equilibrium and cannot store more C (pastures).
- N status: N is required to decompose OM; therefore the amount of N available will impact on mineralisation.

Carbon flows at the New Zealand scale:

Net C flows and the C storage potential of different land-uses has been well studied in New Zealand regarding the Kyoto protocol and the New Zealand emission trading scheme. The most important factor is the rate of carbon accumulation. Because of their prolific root systems, pastures are highly effective in storing C (Tate et al., 2005a). Until recently soil C levels in uneroded pastures in New Zealand were believed to be nationally at, or near, steady state (Saggar et al., 2001) (Fig. 4.12). The accumulation or loss of organic C in pastures depends on the balance between C inputs to soil from surface litter and roots, and C losses like decomposition of soil humus and soil respiration. This balance is influenced by climate, soil type, landscape, land-use and management practices (Fig. 4.9 and 4.10). When the C input rate comes close to the rate of decomposition, soil C concentration stabilises.

Trotter et al. (2004) realised a multi-scale analysis of the New Zealand carbon budget. They estimated C gains and C losses from different land uses at the national scale. The net terrestrial C balance or net ecosystem production (NEP) is the difference between the C fixed for a given land-use and the C emitted. To calculate NEP, Trotter et al. (2004) first calculated net primary production (NPP), the amount of C fixed by plants for each land cover class. NPP is based on climate and land cover, including both above- and belowground components.

Then they adjusted values of NPP to account for 2 components of net terrestrial C loss:

- Soil autotrophic respiration (RA) which is the C emitted by the growth and maintenance respiration of roots.

- Soil heterotrophic respiration (RH) which is the C emitted by oxidation of organic matter by micro-organism. It is the difference between measured values of total soil respiration and soil autotrophic respiration for each land cover class.

The definition of NPP they used already accounted for losses by autotrophic respiration, therefore calculation of NEP thus requires heterotrophic respiration (RH) only to be subtracted from NPP: NPP-RH=NEP. Table 4.7 presents national NEP values. It shows that improved grasslands lose around 0.9 tC/ha/yr, whereas exotic forests accumulate around 3.08 tC/ha/yr. The values of soil respiration reported here have been calculated from a dataset unaffected by soil water content limitations.

Table 4-7: National values of net ecosystem production by land cover (from Trotter et al., 2004).

Land use		Area (Mha)	NPP	RT	RA	RH	NEP
Improved grassland	Tg CO_2-C/y	6.67	59	99	34	65	-6
	t CO_2-C /ha/yr	1	8.85	14.84	5.10	9.75	-0.90
Exotic forest	Tg CO_2-C/y	1.62	16	14	3	11	5
	t CO_2-C /ha/yr	1	9.88	8.64	1.85	6.79	3.09

Note: NPP: net primary production, RT: Total soil respiration, RA: soil autotrophic respiration, RH: soil heterotrophic respiration, NEP: net ecosystem production; RT=RA+RH and NPP+RA-RT=NEP. 1t = 10^6g; 1Tg = 10^6t = 10^{12}g.

These numbers should be handled with caution, because they result from the subtraction of 2 very big numbers, C gains and C losses. The RT value of 14.84 t CO_2-C /ha/yr corresponds to a rate of soil respiration of 40.7 kg CO_2-C/ha/day. This data is comparable with other New Zealand studies. Aslam *et al.* (2000) measured field-CO_2 emissions from a Ohakea silt loam of 55 kg CO_2-C/ha/day. Similarly, soil surface respiration in a grazed pasture measured by Brown et al. (2009) was 53.15 kg CO_2-C/ha/day. The soil respiration value modelled by Trotter et al. (2004) therefore seems a bit low but already results in a negative net ecosystem production - that is a loss- for pastures.

Some recent evidence reinforces the same theory. Schipper et al.(2007) showed losses of soil C from soil profiles under pasture during the past 20 years. Significant losses averaged 106 gC/m^2/yr (1.06 tC/ha/yr). One of their sites showed extreme C loss of 21 kgC/m^2/yr. Omitting this site resulted in an annual C loss of 80 gC/m^2/yr (0.8 tC/ha/yr). C losses were not confined to top soils only but observed through the top meter of the profile (Schipper et al., 2007). More recently Schipper et al. (2010) showed losses of soil C from dairy grazed pastures to be 0.73(±0.16) tC/ha/yr. These results are in accordance with the Trotter et al. (2004) study which showed that improved grasslands loose around 0.9 tC/ha/yr (Table 4.7). Large losses of soil C are concerning because they are likely to contribute to an increase in atmospheric CO_2. But as Schipper et al. (2007) argue, it is unknown whether these losses reflect the shift to a new equilibrium of C or whether they are ongoing. C losses could be explained by changes in grazed organic matter inputs by livestock, changes in pasture species composition, the amount of litter and litter quality, or the rates of incorporation.

Other possible causes for C losses are listed below (Fig. 4.9) (Schipper et al., 2007):

- Enhanced C leaching from urine patches,
- Intensified N cycling increases N losses, which limits C storage since N is required to decompose high C:N ratio materials,
- Leaching of dissolved organic matter is increasing,
- Pasture harvest index has increased, carrying more C and N off site.
- Plant, faunal and microbial diversity and functional groups responsible for litter removal and incorporation into the soils profile are decreasing,
- Macroporosity has declined, affecting the C cycle,
- Climate change has enhanced soil respiration more than C inputs,
- Erosion has increased.

Even though losses of C by erosion are a significant issue in New Zealand, little account is taken of it in the New Zealand C budget. Studies estimated soil C loss to the sea from erosion at around 3±1 Mt /yr (Trotter et al., 2004). Scott et al. (2006) estimated that New Zealand's rivers export 4 ± 1 t C /km^2/yr of dissolved organic carbon (DOC) and 10 ± 3 t C /km^2/yr of particulate organic carbon (POC) which is 2 and 6 times the global average. Dymond & Baisden (2010) used an erosion model to calculate losses of POC to the sea. They reported that the North Island of New Zealand is estimated to export 1.9 (-0.5/+1.0) Mt POC/yr and the South Island 2.9 (–0.7/+1.5) Mt POC /yr. Although the soil C transported to the ocean represents a loss from terrestrial ecosystems, it is not a loss that impacts directly on the atmosphere, which means that assuming exported C is buried at sea with an efficiency of 80% gives New Zealand a net C sink of 3.1 (–2.0/+2.5) Mt C/yr (Dymond and Baisden, 2010). To assume a loss of 20% of eroded soil C during fluvial transport seems sensible because river lengths in New Zealand are short, and flow rates are high by international standards, limiting the opportunity for oxidation losses during sediment transport (Trotter et al., 2004).

Soils recovering from erosion present the potential to store more C. Pastures are most efficient at storing soil C, but when recovering from erosion, storing rates are less than or equal to 1 tC/ha/yr (Tate et al., 2005a). Dymond & Baisden (2010) calculated that North Island and South

Island soils sequester approximately 1.25 (-0.3/+0.6) MtC/yr and 2.9 (-0.7/+1.5) MtC/yr, respectively, from the atmosphere through recovery from erosion.

Carbon flows at the pasture scale:

In pasture soils, soil C can be changed substantially by management practices (Parson et al., 2009; Schon et al., 2010a), an external driver (Fig. 4.9). Adding fertiliser increases total plant growth and thereby alters the amount of C flowing to soil and also the proportion of C partitioned to roots. It also changes the quality (e.g. the C:N ratio) of all the material cycling in the system (Parson et al., 2009). Increases in livestock density and grazing regimes alter plant and soil biota species diversity (Parson et al., 2009; Schon et al., 2010e), as well as pasture composition, notably the presence of legumes. Thereby pastoral management has a large effect on inputs to the soil (plant litter, both above and below-ground, dung inputs from grazing animals and living plant roots) (Schon et al., 2010c), and on soil processes regulating C cycling. Parsons and Chapman (2000) described (Fig. 4.13) how grazed pasture management alters the C budget (annual total per ha) above ground, over a grazing season, for both a low and higher fertility case. The magnitude of C flows a management practice creates is indicated by drawing a vertical line over the graphs. Of the C fixed by photosynthesis nearly half is respired by shoots and returns as CO_2 to the atmosphere. Half of what remains returns to the soil as tissue turnover. It is the size of this turnover that creates the potential for the substantial amount of C sequestered in grassland soils (Parsons and Chapman, 2000). Increasing the intensity of utilisation by grazing a greater proportion of what is grown, and so maintaining a lower average pasture leaf area index, reduces all the fluxes of C. Thus increasing stocking rate would decrease the flow of C to soil, and so reduce the potential for C sequestration (Parsons and Chapman, 2000). The assumption made in the model of Parsons et al. (2009) is that all other variables that influence the C cycle remain unchanged.

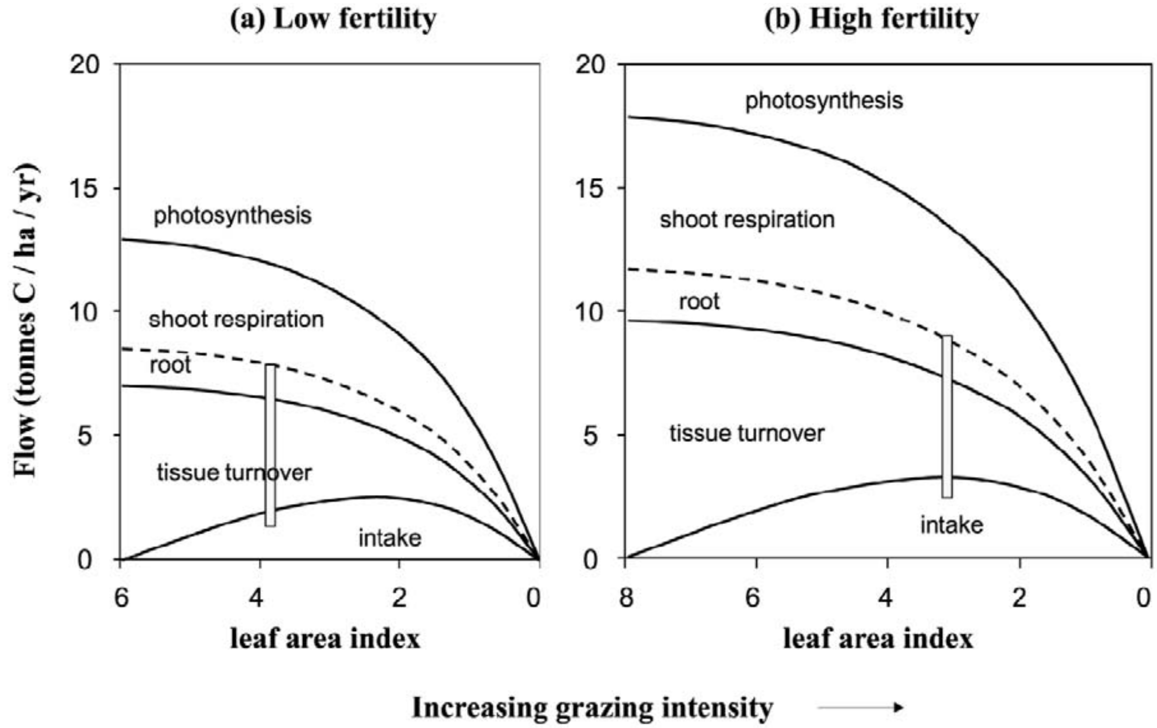

Figure 4-13: The major flows of C (tonnes C/ha/year) through plants and animals in grazed pastures in relation to the intensity of grazing, as defined by the leaf area index sustained under (a) low fertility and (b) high fertility conditions. Vertical bars show examples of the potential total flow of C to soil (from Parsons and Chapman, 2000).

Similarly, Saggar & Hedley (2001) studied the seasonal changes in assimilation and partitioning of photo-assimilated C in the plant–root–soil components of a temperate pasture. They found that losses by respiration were high (66–70%) during the summer, autumn and winter season, and low (37–39%) during the spring and late-spring season. Overall, at this high fertility dairy pasture site, 18,220 kgC/ha (55.5% of total C assimilated 32,850 kgC/ha) was respired, 6,490 kg (19.8%) remained above-ground in the shoot, and 6,820 kg (21%) was translocated to roots, and 1,320 kg (4%) to soil (Saggar and Hedley, 2001). This study showed that soil respiration presents important seasonal changes. It also indicated that more than half of the C fixed by pastures (55.5%) is lost as CO_2 respired.

Soil fauna and carbon cycle:

Soil fauna is a key agent of the C cycle, especially for the recycling of OM. In order to sustain economic profitability and high production levels, legume-based dairy pasture systems in New Zealand are being intensified by the import of supplementary feed onto the milking platform and the addition of N fertilisers which increase pasture production. Supplementary feeding and faster-growing pastures mean that more dung and more plant detritus are entering the soil food-web, increasing nutrient inputs to the soil-plant system (Schon et al., 2010d). Increased pasture growth can support a greater abundance of soil invertebrates and stimulates populations of plant- and bacterial-feeding fauna, with higher root mass and a dominance of higher quality pasture species. However, intensive management practices like high stocking rates also influence soil fauna habitat (e.g. reduce macroporosity), thereby selecting species and reducing biodiversity. Soil food-webs of more intensive systems seem to be dominated by smaller, short-lived organisms and by bacterial-decomposition pathways, faster to return nutrients to the soil pool than fungal-feeding pathway (Schon et al., 2010a; Schon et al., 2010d) leading to increased rate of nutrient cycling and faster C cycles. However, in higher input pasture systems, despite having potentially more organic matter available for incorporation into the soil profile, the lower abundance of litter incorporating fauna these systems present (earthworms, Collembola and Oribatida) may collectively contribute to a decline in soil C (Schon et al., 2010a) because litter left on the surface may be quickly oxidised and lost as carbon dioxide (CO_2).

The section above showed how C stocks (property) and flows (processes) (Fig. 4.9) are linked and influence C storage and how they relate to CO_2 emissions. The next section examines the properties and supporting processes (Fig. 4.9) behind the regulation by soils of the emissions of other GHGs (N_2O, CH_4).

4.4.1.2 Greenhouse gases regulation from soil:

New Zealand is unique in having a GHGs emissions inventory not dominated by CO_2 like other developed countries, but by methane (CH_4) and nitrous oxide (N_2O). Together, these two GHGs represented 52.1% of New Zealand's total emissions, in 2007, on a CO_2-equivalent basis (MfE, 2009b, p.V). Their dominance in New Zealand GHGs inventory results from two facts: first the New Zealand economy is strongly based on agriculture, and second, New Zealand has relatively low levels of heavy industry and vehicular CO_2 emissions per unit land area (Saggar et al., 2008). In New Zealand, CH_4 and N_2O mainly come from agricultural activities: CH_4 from enteric fermentation of farm animals and N_2O from soils (Table 4.8).

Table 4-8: CH_4 and N_2O emissions from agriculture, from New Zealand GHGs inventory (MfE, 2009b).

	2007 Emissions in Gg CO_2-e	% of total emissions	% of emissions from agriculture
CH_4 from enteric fermentation	23,326.40	30.9	64.0
N_2O from soils	12,298.10	16.3	33.8

Note: Gg = 10^9g; CO_2-e = CO_2 equivalent

In this study, the focus is on the processes of CH_4 emission and consumption by soils. CH_4 emissions from enteric fermentation are not considered. Globally, according to IPCC (Intergovernmental Panel on Climate Change) estimates, natural and cultivated submerged soils (landfills not included) contribute about 55 % of the CH_4 emitted into the atmosphere, while emerged soils are responsible for 6 % of the CH_4 consumption. Soils are therefore a major contributor in the global CH_4 cycle (Le Mer and Roger, 2001). At the paddock scale, methane flows are small, but at the scale of a country or the globe, they can be very significant (Saggar et al., 2008).

N_2O emissions from soils are another process considered here (Fig. 4.9). The emissions of N_2O from soils have been very much studied, because the global warming potential of N_2O is 310 times the one of CO_2. Methane's global warming potential is 21 times the one of CO_2 (MfE, 2009b). The regulation by soils of N_2O emissions is a service (Fig. 4.9). All soils emit N_2O, but some soils emit less than others thanks to a series of supporting processes regulating NO_3^- levels and therefore N_2O emissions. It is these supporting processes (Fig. 4.9) that are investigated further in the next section.

<u>Methane (CH_4) regulation:</u>

Even if methane (CH_4) has a short residence time in the atmosphere (about 10 years), its ability to absorb infrared radiation makes it a GHG 21 times more efficient than CO_2, on a 100 year basis (Le Mer and Roger, 2001; MfE, 2009b). Agriculture is the main anthropogenic source of CH_4. The atmospheric concentration of CH_4 has more than doubled in the past 200 years, mainly as a result of increased CH_4 emissions from anthropogenic sources, such as fossil fuel exploration, rice production, large-scale animal husbandry of ruminants, biomass burning and landfills (Tate et al., 2007). Decreased CH_4 uptake by soils because of land-use changes, may also have contributed to increasing atmospheric CH_4 concentrations.

There are two biological supporting processes (Fig. 4.9) regulating atmospheric CH_4 concentrations: methanogenesis and methanotrophy (Fig. 4.12). CH_4 is produced by the anaerobic digestion of organic matter in anoxic environments (submerged soils) by methanogenic bacteria. CH_4 is also eliminated in soils by microbial oxidation (methanotrophy)

(Le Mer and Roger, 2001). CH_4 oxidation by aerobic soils (methanotrophy) represents a globally significant sink. Uptake of CH_4 by aerobic soils removes a significant amount from the atmosphere (10–44 Tg/y), and accounts for up to 10% of the global CH_4 sink (Tate et al., 2007).

In New Zealand, the profile of GHGs emissions differs from most other countries. In 2007, CH_4 emissions from enteric fermentation were 64% (23,326.4 Gg CO_2-e) of agricultural emissions and 30.9% of New Zealand's total emissions (MfE, 2009b) (Table 4.8). For that reason CH_4 oxidation by aerobic soils (methanotrophy) has been studied a lot in New Zealand in the past 10 years as a possible tool to offset CH_4 emissions from enteric fermentation.

Methane emissions from soils: Methanogenesis

Under anaerobic conditions, soils are a source of CH_4. Methanogenic bacteria degrade organic wastes by anaerobic fermentation. Methanogenic fermentation of organic materials occurs under strictly anaerobic and low oxydo-reduction potential (Eh<-200mV) conditions, where sulphate and nitrate concentrations are low: $C_6H_{12}O_6 \rightarrow 3\ CO_2 + 3\ CH_4$ (Saggar et al., 2004b). The main properties determining the extent of CH_4 production are the amount of degradable organic matter available (Saggar et al., 2004b) and the type micro-organisms present (Fig. 4.9). Most soils, such as forest, pastures and cultivated soils, emit CH_4 only when they are waterlogged. Soils that are often submerged or water-saturated and where a significant methanogenic activity develops at intervals are generally also most efficient in methanotrophy (Le Mer and Roger, 2001).

A soil is a CH_4 source when the balance between production by methanogenic bacteria and consumption by methanotrophic bacteria is positive, leading to net CH_4 emission. When the balance is negative, the soil is a CH_4 sink (Le Mer and Roger, 2001). Temperate and tropical oxic soils that are continuously emerged above water and exposed to atmospheric concentrations of CH_4 are CH_4 sinks. They usually exhibit low levels of atmospheric CH_4 oxidation but, because of the large areas they cover, they are estimated to consume about 10 % of the atmospheric CH_4. Among upland soils, forest soils are probably the most efficient CH_4 sink (Le Mer and Roger, 2001).

Methane consumption by aerobic soils: Methanotrophy

Two forms of CH_4 oxidation are recognised in soils. 'High affinity oxidation' occurs at CH_4 concentrations close to that of the atmosphere (< 12 ppm), in soils without high NH_4^+ concentrations (Le Mer and Roger, 2001). 'Low affinity oxidation' occurs at CH_4 concentrations higher than 40 ppm. "It is performed by bacteria called methanotrophs and is considered as methanotrophic activity sensu stricto" (Le Mer and Roger, 2001, p.28). Methanotrophs use CH_4 as a C and energy source. Their activity is mainly limited by oxygen availability (Le Mer and Roger, 2001). Therefore, a well drained soil has a higher methanotrophic potential than a waterlogged soil. Temperate soils that are continuously emerged are CH_4 sinks (Le Mer and Roger, 2001; Tate et al., 2007). In New Zealand, half (13.6 M ha) the usable land is grassland used for livestock farming, and forests cover about 30% of the land (8.0 M ha) (Table 4.3) which means that more than 80% of New Zealand land is potentially acting as a CH_4 sink (Tate et al., 2003).

Nitrous oxide (N_2O) regulation:

The production of N_2O by soils is a major concern for New Zealand. In 2007, the agricultural sector contributed 48.2% of New Zealand's total GHGs emissions. N_2O emissions from agricultural soils were 33.8% of agricultural emissions and 16.3% of total emissions (MfE, 2009b) (Table 4.8).

N deposited in the form of animal urine and dung, and N applied as fertilisers are the principal sources of N_2O production in New Zealand (Saggar et al., 2008). N is present in soils in 3 major forms:

- Organic compounds associated with plant material, soil organisms and soil humus (94-98%) unavailable for plants;
- Ammonium N held by clay minerals (1-6%);
- Mineral N forms (Ammonium NH_4^+, Nitrite NO_2^- and Nitrate NO_3^-) available to plants (1-2%).

The processes regulating N_2O emissions (Fig. 4.9) involve mineral N forms. Gaseous N losses are the product of denitrification. There are two sorts of denitrification processes: biological denitrification carried out by bacteria called nitrobacteria, producing N_2O, and chemical denitrification producing N_2.

Biological denitrification is possible in any anaerobic conditions like waterlogged soils or poorly drained soils, or, at smaller scales in zones imperfectly drained or at the centre of soil aggregates. Anaerobic bacteria called nitrobacteria, can instead of oxygen, use nitrate (NO_3^-) as an electron acceptor for the oxidation of available C (e.g. organic matter). Aerobic oxidation of carbohydrates produces CO_2 and H_2O, but anaerobic oxidation of carbohydrates produces also N_2.

The reduction of NO_3- leads to a series of nitrogen oxides to di-nitrogen (N_2):

NO_3^- (nitrate)→ NO_2^- (nitrite)→ NO (nitric oxide)→ N_2O (nitrous oxide)→ N_2 (di-nitrogen).

In pasture soils, nitrous oxide (N_2O) is the GHGs produced and released in the greatest quantity. Usually, it escapes as gas before being reduced to di-nitrogen (N_2). More than half the N_2O emissions for New Zealand come from uneven deposition of excretal N in grazed pastoral soils, while another 30% come from indirect emissions from leached and volatilized excretal-N (de Klein et al., 2003). It is challenging to measure N_2O fluxes accurately from grazed pastures, because of the high levels of spatial variability, which constitute one of the largest sources of uncertainty (Saggar et al., 2007a). Nitric oxide (NO) is usually produced in acidic conditions, but in generally small amounts. Even if the soil is not anaerobic, when ammonium (NH_4^+) fertilisers are applied, gaseous N losses can occur by chemical denitrification. The high amounts of ammonium (NH_4^+) reduce the activity of nitrobacteria and thereby decrease nitrification which leads to the accumulation of nitrite (NO_2^-) in the soil. Nitrite (NO_2^-), or nitrous acid (HNO_2), then reacts with organic matter, or ammonia (NH_3), or urea (the reaction with urea also produce CO_2) to produce di-nitrogen (N_2). These reactions don't involve microbial activity. Chemo-denitrification is enhanced by low or neutral pH.

A number of degradation processes and drivers impact on the natural capital stocks and supporting processes behind C storage and GHGs regulation (Fig. 4.9). They are investigated in the section below.

4.4.2 Degradation processes and drivers influencing carbon storage and greenhouse gases regulation:

C storage and the regulation of CH_4 and N_2O emissions are driven by interactions between soil properties, soil micro-organisms, climate and agricultural practices (Saggar et al., 2008).

4.4.2.1 Degradation processes:

Like C storage, CH_4 and N_2O emissions depend on micro-organism population and activity. They are sensitive to soil environmental conditions like soil water content and soil nutrient status, and are therefore sensitive to disturbance by management. Therefore all processes impacting on soil water content and soil nutrient status impact on the provision of the GHGs regulation service.

Compaction, by affecting drainage and soil water content, affects CH_4 oxidation and N_2O emissions. Soil water content and fertility are known as key controls of CH_4 oxidation in soils (Tate et al., 2007). Soil methanotrophic activity is related to soil water content, because soil submersion reduces the size of the oxidised zones and O_2 availability. Methanotrophy is more significant where gas diffusion is easy (Le Mer and Roger, 2001) which means that soil texture and aeration are critical. Different studies showed that CH_4 oxidation generally decreases as water-filled pore space (WFPS) increases (Saggar et al., 2007b; Tate et al., 2007). This seems particularly true above the soil field capacity (FC). At lower water contents (<FC),

methanotrophy seems to depend upon soil fertility (Le Mer and Roger, 2001; Saggar et al., 2007b; Tate et al., 2007).

Biological denitrification is an anaerobic process; therefore a well-drained soil which does not easily water-log, won't produce as much N_2O as a poorly-drained soil. Moreover, soil respiration is also influenced by soil water content. CO_2 losses by soils are higher in warm and dry conditions (Saggar and Hedley, 2001).

Erosion, by removing soil material and especially OM will impact on C stocks but also biological processes using OM as a source of energy like biological denitrification or methanogenesis.

4.4.2.2 External drivers:

A number of natural drivers impact on C storage and the regulation of CH_4 and N_2O emissions from soils. Geology and the conditions of formation of the soil determine soil texture and clay mineralogy which influence soil C storage (Fig 4.10). Depending on the soil type and the clay minerals present, strong interactions between clay minerals and organic matter can stabilise organic matter and may protect it from losses associated with degradation, erosion and land-use change (Scott et al., 2002).

Climate influences soil water content dynamics, which, with temperature (Fig 4.10), drive biological processes, impacting on soil organic C turnover (Scott et al., 2002) but also methanogenesis, methanotrophy and denitrification, and thereby GHGs regulation.

Anthropogenic drivers like land use and farming practices also impact strongly on C storage and the regulation of CH_4 and N_2O emissions from soils which are very sensitive to management. Land use determines the plant and animal species interacting with the soil and therefore affects the type and amount of organic inputs to the soil (Fig 4.10). In pastures, C inputs are plant litter (both above and below-ground), dung inputs from grazing animals, living plant roots (Parson et al., 2009; Schon et al., 2010c) and dairy farm effluents application. Below-ground inputs are readily decomposable, leading to high microbial activity in the rhizosphere (Scott et al., 1997). In exotic forests for example, detrital material contains more components with higher lignin content which are less readily decomposable than inputs in pasture ecosystems. These differences impact on the type and amount of C stored in soils as well as the nature and quantity of nutrients in solution. In return, soil nutrient status drives biological processes like methanogenesis, methanotrophy and denitrification.

Consequently, land use change has been shown to impact on C storage and N_2O and CH_4 emissions. For example, in New Zealand soils, CH_4 uptake rates have been found to vary markedly with land use "with inhibition of CH_4 oxidation being attributed to disturbance effects on methanotroph populations and activity" (Tate et al., 2007, p.1438). CH_4 uptake rates are comparatively high (averaging 7.9 kg CH_4–C /ha/yr) for an evergreen native beech (Nothofagus) forest soil, intermediate (3.2–10.5 kg CH_4–C /ha/yr) for soils under Pinus radiata (pine), and lowest (<1kg CH_4–C /ha/yr) for pasture and cropping soils (Tate et al., 2007).

Several farming practices impact on C storage and the regulation of CH_4 and N_2O emissions (Fig 4.10). Increased aeration of the soil due to e.g. tillage means increased biological activity and therefore increased aerobic processes like mineralisation and methanotrophy. Repeated wetting and drying through cultivation or irrigation can increase microbial activity, increasing mineralisation and C turnover. Cultivation or animal treading, by modifying soil structure, impact on soil porosity and therefore soil aeration and water content, but also affect the exposure of previously inaccessible organic matter. Compaction impacts on soil fauna habitat and thereby selects biota species. Schon et al. (2010d) showed how earthworm species substitute under increased treading pressure from surface species to deep burrowers, reducing the incorporation of plant litter.

The use of fertilisers, or lime, impacts on soil nutrient status and plant production, and thereby on soil fauna and all biological processes. Liming is known to increase the activity of earthworms and other fauna (Springett and Syers, 1984). Ammonium (NH_4^+) is known to inhibit methanotrophy through soil acidification (Le Mer and Roger, 2001). When applied to soil, urea rapidly hydrolyses to NH_3 and NH_4^+. Micro-organisms then transform NH_4^+ into NO_3^- (nitrification), releasing at the same time H^+ ions which acidify the soil. Methanotrophs are more tolerant to pH variations than methanogens, but they are, however sensitive to the acidification of the environment. Soils rich in available P seem also more prone to methanotrophy (Le Mer and Roger, 2001). Earlier New Zealand studies (Saggar et al., 1997) have shown that increased fertiliser use increases pasture production, translocates more C to roots, but also enhances decomposition of soil organic C, and the rate of C loss (Tate et al., 2005a). Moreover, techniques like the use of nitrification inhibitors are used to prevent the transformation of ammonium (NH_4^+) into nitrate (NO_3^-) to prevent nitrate leaching and nitrous oxide (N_2O) emissions.

4.4.3 Quantifying carbon storage and greenhouse gases regulation:

4.4.3.1 Previous attempts to quantify carbon storage and greenhouse gases regulation:

A few ecosystem services studies mention C storage (Barrios, 2007; Porter et al., 2009; Sandhu et al., 2008; Swinton et al., 2007) and GHGs regulation (Lavelle et al., 2006; Wall et al., 2004; Weber, 2007; Zhang et al., 2007). Only two (Porter et al., 2009; Sandhu et al., 2008) attempted to model C storage. For example, Porter et al. (2009) and Sandhu et al. (2008) both mentioned C accumulation as an ecosystem service. They estimated the amount of plant and root residues from crop yields and supposed that 40% was C accumulated. This methodology, while recognising that natural capital stocks of C are the source of ecosystem services, fails to apprehend the complexity of the soil processes behind that service. These authors (Porter et al., 2009; Sandhu et al., 2008) do not consider existing C stocks and also fail to consider net C flows. Sandhu et al. (2008) mentioned C accumulation in soils as a service being considered as "an alternative to offset the emissions of carbon dioxide in the atmosphere by industry and other human activities", but did not consider the regulation of GHGs emissions by soils.

4.4.3.2 Parameters chosen to quantify carbon storage and greenhouse gases regulation:

To inform the regulation of CH_4 and N_2O emissions and C storage from soils, each of the processes involved need to be considered.

To quantify C storage in soils, process-based models can be used to calculate the net flows of C from soils to determine if the soil is a net source or sink of C. Data from the literature can be then compared to model outputs.

Scott et al. (2002) designed a soil C monitoring system for New Zealand stratified by soil type, climate, and land use. Soils were placed in six IPCC (Intergovernmental Panel on Climate Change) soil categories to reduce the number of cells in the system. The soils considered in this study fall into the High Clay Activity (Gley Soil, Te Kowhai silt loam) and Volcanic (Allophanic Soil, Horotiu silt loam) categories. They showed that soil type was clearly the most important single factor and soil C differences between soil types were highly significant. Table 4.9 shows soil C values (0-0.3 m) in tC/ha for improved pastures for comparable soil / climate categories to the one used in this study (Scott et al., 2002).

Table 4-9: Improved pastures soil C values (0-0.3 m) (tC/ha) (from Scott et al., 2002).

Soil	Climate	IPCC (tC/ha)	Scott et al. 2002 (tC/ha)
High Clay Activity	Dry temperate	50	75
High Clay Activity	Moist temperate	80	**94**
Volcanic	all	70	**134**

These data show that New Zealand soils have much higher soil C levels than similar soils within the IPCC approach (Table 4.9) and are consistent with other data which gave average values (105.7 t C ha^{-1}) for New Zealand pastures (Tate et al., 2005b).

To take into account net flows of C, autotrophic respiration, heterotrophic respiration and C lost by leaching and erosion need to be subtracted from C fixed by plants (NPP) (Fig. 4.12). The literature (Schipper et al., 2007; Trotter et al., 2004) gives us data of C losses from pasture (Table 4.10), not including C losses by erosion or leaching.

Table 4-10: Carbon losses rates from pastures.

References	Loss from pastures
Trotter et al. (2004)	-0.9 t C/ha/yr
Schipper et al.(2007)	-0.8 t C/ha/yr

To quantify C storage, the net flows of C need to be considered. If they are negative C losses should be considered as a degradation process, and the impacts of C loss on natural capital stocks and the provision of soil services should be investigated.

To quantify the flows of CH_4 from soils, it is important to remember that grazed pastures are net CH_4 sinks, unless the soil is wet (SWC>FC) (Saggar et al., 2007b; Tate et al., 2007). Soil water content is a key control of CH_4 oxidation in soils, therefore to quantify net CH_4 oxidation, the dynamics of soil water need to be linked to the dynamics of CH_4 oxidation. Tate et al. (2007) reported the difference in daily methane uptake flux of soils with different macroporosity and drainage status. To take into account the difference between soils, they considered the influence of water-filled pore space (WFPS) (Tate et al., 2007). Saggar et al. (2007b) found that in a poorly drained pastoral soil (Tokomaru silt loam) mean daily CH_4 flux for the summer months (-2.22 ± 0.63 gCH_4–C/ha/day) was two to three times the consumption in wet winter months (-0.68 ± 0.17 gCH_4–C/ha/day), indicating a strong seasonal pattern of soil CH_4-sink capacity. Moreover, Saggar et al. (2003a; 2004a) found a well-drained soil had a higher winter CH_4 uptake than a poorly drained soil (Saggar et al., 2008). Poor winter drainage in a heavy-textured soil often results in anaerobic conditions preventing methanotrophy. Differences between soils with contrasting natural capital is most likely to show in winter when soil is wet (SWC>FC) and aeration limited.

Table 4.11 presents data including the seasonal pattern of soil CH_4-sink capacity and the different winter CH_4 uptake between a well-drained soil and a poorly drained soil (Saggar et al., 2008).

Table 4-11: Seasonal methane uptake (sink) from two soil types (modified from Saggar et al., 2008)

	Well drained soil gCH_4-C/ha/day	Poorly drained soil gCH_4-C/ha/day
spring	1.3	1.3
summer	2	2
autumn	1.8	1.8
winter	*1.3*	*0.3*
Total in kg/ha/yr	0.57	0.48

Annual methane uptake of around 0.6 CH_4-Ckg/ha for a well drained soil and 0.5 CH_4-C kg/ha for a poorly drained soil, are coherent with other studies concluding that annual methane uptake for a dairy grazed pasture is around 0.5-0.6 CH_4-Ckg/ha/yr (Saggar et al., 2008).

Since soil water content and N are the key controls of CH_4 oxidation in soils, to quantify CH_4 oxidation from soils, SWC can be used as a proxy to build a methane oxidation function with data from the literature (Saggar et al., 2007b; Saggar et al., 2008). SWC is considered as an indicator to reflect reduced CH_4 oxidation with increased water filled pore space for pastoral soils based on New Zealand results (Saggar et al., 2007b; Saggar et al., 2008; Tate et al., 2007).

The measure of the service, CH_4 regulation, can then be defined as the difference between the minimum CH_4 oxidation (that is none) and the actual CH_4 oxidation for each year, under a given management. This measure represents the amount of CH_4 actually consumed by the soil.

The dynamics of N_2O emissions involve a number of properties and processes which make it very complex. N_2O emissions are dependent on soil water content and soil solution concentration of NO_3^-. To model N_2O emissions, the dynamics of soil water content and nitrate concentrations need to be linked. The IPCC methodology to calculate N_2O emissions from soils is globally recognised. It uses N inputs to the soil and emission factors to calculate direct and indirect N_2O emissions estimations. Some emission factors have been recalculated to better fit New Zealand conditions (de Klein et al., 2003). However this methodology is unable to account for the distinction between soil types and different moisture status, which makes it a useful, but non-precise tool. Complex models have been developed to predict N_2O emissions from soils. In New Zealand, the NZ-DNDC (Giltrap et al., 2008; Saggar et al., 2007a; Saggar et al., 2007b; Saggar et al., 2004c) has proven a reasonable fit with field data. The model includes soil water content and nitrate concentrations as parameters, along with emission factors, to predict N_2O emissions.

Here, the IPCC methodology is used to quantify N_2O emissions, with specific emission factors to take into account the impacts of soil water content on N_2O emissions. Process-based soil models can enable us to follow SWC and thereby estimate more precisely N_2O emissions from soils. The measure of the service, N_2O regulation, can then be defined as the difference between the maximum N_2O emissions for a given management in case the soil was water-logged, and the actual N_2O emissions calculated each year. This measure represents the N_2O that wasn't emitted from the soil, which is the N_2O regulated.

The regulation of pest and disease populations is examined in the next section.

4.5 Biological regulation of pest and disease populations:

This section discusses plant and animal pests and diseases. The influence of soils on pest plants (weeds) is not considered here but there is recognition of the interaction between soil natural capital and pest plant invasions (Popay et al., 2010).

Soils are the home of a great diversity of invertebrate species ranging from unicellular organisms (1µm) like bacteria to macro fauna (several cm) like earthworms. Of the great diversity of species inhabiting soils, many are beneficial to humans and support ecosystem services, but some are parasites, cause disease, or are pests for plants and animals useful to humans. Human soil-borne parasites include ascaris, toxoplasmosis and different types of worms. Fungal and bacterial soil-borne diseases include anthrax, botulism, meningitis or tetanus.

In pastoral systems, farmers often have to deal with a number of pests and diseases affecting pasture plants (e.g. grass grubs, porina, clover root weevil, root feeding nematodes) (Table 4.12 and 4.13) and livestock (e.g. parasites like nematodes or fungi like facial eczema) (Table 4.14), that impact on production and animal survival.

Livestock manure and dairy farm effluents (DFE) are potential sources of many diseases including pathogens like E-coli, Campylobacter or Salmonella (Wang et al., 2004). Moriarty et al. (2008) sampled freshly deposited bovine faeces from four New Zealand dairy farms over a year and enumerated them for E. coli, Enterococci, Campylobacter, Giardia, Cryptosporidium, Salmonella and STEC. The overall median bacterial counts (/g wet weight) were E. coli = 5.9×10^6 ; Enterococci = 1.3×10^4 and campylobacter = 3.9×10^5 (Moriarty et al., 2008). Numbers varied markedly between faecal samples, but they concluded the fresh bovine faeces are a significant source of E. coli, Enterococci and Campylobacter on New Zealand pastures. Similarly, land application of DFE, if managed poorly, has the potential to spread harmful organisms onto soils and pastures.

A number of species inhabiting soils are economically important pasture pests in New Zealand (Table 4.12 and 4.13). Species range from micro-fauna (bacteria, clover nematodes), to macro-fauna (moths, weevils, beetles, crickets). Some of these pests and diseases live in soils, or have part of their reproduction cycle (e.g. larvae) in soil. Therefore the potential exists to reduce their impact on pasture plants through regulation by soil properties and processes. The main pasture pests are listed in Table 4.13 along with details on the damage they do, and the soil properties they're sensitive to.

Table 4-12: Pasture pests and diseases and their relation to soils.

Pests	**Part of the cycle in soil**
Porina	Caterpillars
Tasmanian grass grub	Larvae (Grubs)
Grass grub	Larvae (Grubs)
Clover root weevil	Larvae (Grubs)
Black field cricket	Adult lives in cracks in the soil during the day. Eggs are laid in damp soil.
Slugs	Adult in moist soil.
Argentine stem weevil	Eggs and larvae
Black beetle	Larvae
Blue green Lucerne aphid	None
Clover flea / springtails	None
Clover nematode	Adult
White fringed weevil	Eggs and larvae
Diseases	
Rust	None
Pepper spot	None
Grass - leaf spot, blights & blotches	None
Clover - leaf spot, blights & blotches	None
Clover viruses	None
Damping off diseases	None

Table 4-13: Main pasture pests of New Zealand (DairyNZ, 2010; Fleming, 2003; PGGwrightson, 2010).

Pests	Latin name	Forms	Host	Damages and Occurrence	Relevant soil property
Porina	*Wiseana*	The adult is a moth. The larva is a caterpillar.	Ryegrass, White clover, Lucerne	Larvae (caterpillars) feed on aerial parts of plants, eating off the foliage at ground level. The damage caused by caterpillars is apparent in autumn, throughout winter.	Caterpillars live in the soil. They are vulnerable to water logging and dry conditions and trampling by livestock.
Grass grub	*Costelytra zealandica*	The adult is a brown beetle. The larva is a grub.	White clover, Rye-grass	Larvae (grub) graze plant roots during autumn and winter, killing plants and leaving bare patches.	Grubs live in soil. Young grubs are sensitive to dry conditions water-logging and trampling.
Clover root weevil	*Sitona lepidus*	The adult is a brown weevil. The larva is a creamy white grub.	White clover	Adults live on the soil surface, and feed on clover leaves. They are most abundant in spring and autumn. Larvae feed exclusively on clover roots, and associated nodules, reducing nitrogen fixation, and clover production. They are present throughout the year.	Eggs (on pasture) and larvae (in soils) are sensitive to heat and dry conditions.
Clover nematode	*Heterodera sp.* (clover cyst nematodes) *Meloidogyne sp.* (root knot nematodes)	Microscopic worms	White clover	Nematodes invade clover roots and cause damage at the growing tip which allows root diseases to occur. Damage may result in dwarfing, discolouration, wilting, and plant death. Nematodes are present all year round.	Nematodes live in water films therefore they are sensitive to dry conditions.

Similarly, dairy livestock can be affected by a number of pests and diseases (Table 4.14) impacting on animal health and, thereby on milk production. These organisms are influenced by soil properties, with soil processes impacting on the level of infection of a number of animal diseases.

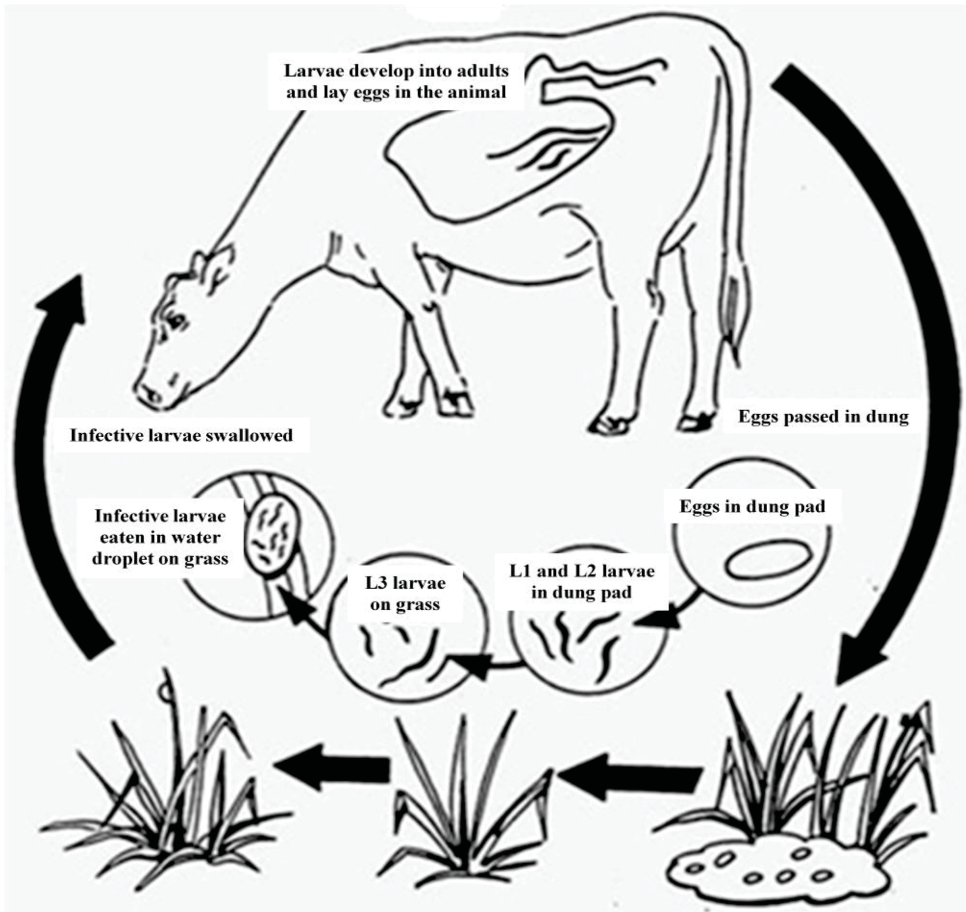

Figure 4-14: Typical life cycle of most internal parasites (Fleming, 2003).

Nematodes can be internal parasites of dairy cows. Adults nematodes live in the animal's gut and produce eggs, which are passed out with dung onto pasture. Over the next few days to several weeks, depending on moisture and temperature, the eggs hatch and develop through three larval stages (L1, L2 and L3) (April to September) (Fig.4.14). L1 and L2 stages are sensitive to climatic conditions. In comparison the effective L3 larvae are resistant to climate extremes. The L3 larvae migrate up moist grass blades and are consumed by grazing animals (Fig.4.14). Internal parasites can have a major impact on young animals (lamb, calves). Internal parasite infestations are very costly, especially for the sheep meat industry. However, adult dairy cows usually develop resilience and are not affected much.

Some animal diseases like footrot are supported by moist soil conditions, since prolonged wet periods soften the animal hooves. Facial eczema depends on the abundance of plant litter at the soil surface, so it is influenced by the rate at which litter is incorporated into the soil (Table 4.14).

Table 4-14: Some dairy livestock pests and diseases and their relation to soils (DairyNZ, 2010; Fleming, 2003).

	Symptoms	Life cycle	Link to soil
Internal parasites			
Parasitic nematodes	Diarrhea, reduction in live weight gain and wool production, depression of appetite, changes in mineral skeleton, changes in protein metabolism, gastrointestinal abnormalities	Adults in livestock, eggs in dung, larvae (L1 and L2) in dung, on grass and in soil (L3). The animal get infected by swallowing larvae (L3)	Dry and very hot conditions kill eggs. Wet condition favors larvae transport to soil. Soil biology activity reduces effective L3 numbers.
Tapeworms	Predisposition to fly-strike, enterotoxaemia	Adults in animal intestine, eggs in dung. Orbatid mites eat the eggs. Livestock eats the mite with grass.	Eggs can be transported to soil.
External parasites: Lice, keds, ticks	Damage to skin and more, infections	All the reproduction cycle takes place on the animal	Soil can act as a reservoir of eggs
Metabolic diseases			
Facial Eczema (fungus: Pithomyces chartarum)	FE is a disease which causes lowered milk production and sometimes death from liver damage. The damaged liver cannot rid the body of wastes and a breakdown product of chlorophyll builds up in the body causing sensitivity to sunlight, which in turn causes inflammation of the skin.	Fungal spores produced by the fungus *Pithomyces chartarum* growing on pasture produce a toxin which when ingested by livestock damage the liver and bile ducts.	The fungus grows on soft litter at the base of the pasture. The rate of incorporation of litter and soil fauna influences the risk of infection.
Ryegrass staggers	Muscle tremors, loss of coordination, death. Reduce growth rate and hormonal production.	Caused by a fungal mycotoxin, from an endophyte (fungus within the ryegrass)	Soil nutrient status can drive toxin concentration in ryegrass.
Infectious disease:			
Footrot	Bacterial infection of the soft tissue between and above the hoof	NA	Get more common when soil is wet and hoof soft
Other Infectious disease: Mastitis, Woody tongue, Lump jaw, Johne's Disease, Tuberculosis...	Bacterial infections causing losses of condition and drops in production	NA	NA

The regulation of pest and disease populations by soils has several dimensions. First, soils provide the habitat for many species, beneficial or harmful. The provision of habitat is a supporting process (Fig.4.15) behind the service. Second, the soil properties characterising that habitat, that is soil natural capital stocks (Fig.4.15) influence the diversity of soil fauna, the dynamics between different populations (e.g. symbiosis, competition, predation) and thereby the natural levels of biocontrol provided by the soil. The characteristics of the habitat provided drive the implantation and development of some species and the disappearance of others. Third, soil properties (e.g. nutrients and trace-elements status) also impact on litter and pasture quality (nutrient content) and thereby on livestock and human health (Ellison, 2002; Lambert et al., 2004).

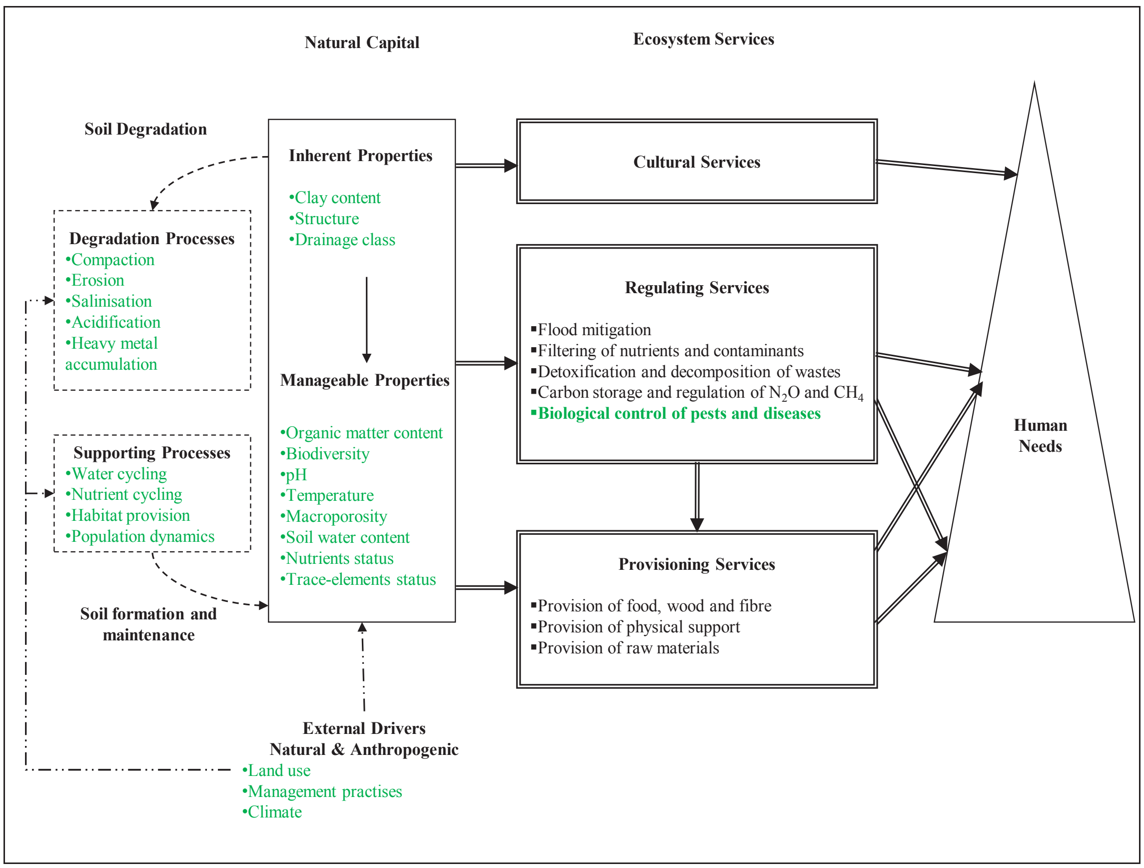

Figure 4-15: Detail of the conceptual framework applied to the regulation of pest and disease populations.

By providing a habitat which is favourable to beneficial species (e.g. earthworms) and detrimental for unfavourable species (e.g. pests larvae), soils are able to regulate the development of populations of unfavourable species like pests and diseases. In this study, the provision of habitat is not considered as an ecosystem service in itself, because it is not directly useful to humans, but as an attribute supporting soil's natural biocontrol and the regulation of the populations of pests and diseases, the real service.

The role played by soils in the regulation of pests and diseases is an ecosystem service because it relates directly to human health but also to the health of plants and animals used by humans for different purposes.

In the following section, the properties and supporting processes (Fig.4.15) behind the regulation of pest and disease populations by soils are examined.

4.5.1 Soil properties and supporting processes contributing to the regulation of pest and disease populations:

Soil properties can impact on the regulation of pests and diseases either directly or through the quality of the habitat provided and the selection of soil biodiversity. The provision of habitat is an essential supporting process (Fig.4.15) driving the population composition and dynamics behind the regulation of pests and diseases. The habitat provided to soil fauna can be characterised by different soil properties which constitute the natural capital stocks behind the service (Fig 4.15).

4.5.1.1 Regulation of population dynamics of soil biota:

Population dynamics are the main processes (Fig. 4.15) behind the regulation of pest and diseases. They are driven by habitat properties and especially macroporosity, soil water content and food resources (Fig. 4.17) which are manageable properties (Fig. 4.15) as well as the composition of the existing fauna. Therefore processes like nutrient and water cycles driving these properties are also critical for the regulation of pest and disease populations.

Different species of soil fauna have very different food requirements and are interrelated in complex food-webs with plants and animals wastes. Macrofauna, mesofauna and nematode herbivores feed on plant material, whereas earthworms, general detritivores, bacterial-feeders and fungal-feeders feed on detrital inputs and associated micro flora (Schon, 2010). The macrofauna, mesofauna and nematodes are in turn consumed by predators of each group (Schon, 2010). Therefore the quality and amount of organic inputs to soils, as well as the OM already present in the soil and soil nutrient status, will directly impact on soil populations and the competition between them.

In temperate pastoral systems, earthworms (macro-fauna), microarthropods (meso-fauna) and nematodes (micro-fauna) are the most abundant in terms of both number and biomass.

Earthworms form the greatest biomass (70±80% of the total) of the soil fauna (Bardgett and Cook, 1998). Numerically, microarthropods (Collembola, Acari and Protura) are the most abundant non-aquatic group in soils. Most microarthropods in grassland are feeding on soil micro-organisms (fungi, bacteria, actinomycetes and algae) and/or dead organic matter and plant litter. Nematode populations are dominated by plant-feeders and bacterial feeders (Bardgett and Cook, 1998).

In general, intensive management of grassland, with large inputs of inorganic fertilisers and high livestock stocking rates impact negatively on the diversity, but not necessarily the density or biomass of soil fauna (Bardgett and Cook, 1998). High pressure management systems characterised by "fast" nutrient cycles and dominated by labile substrates and bacterial decomposition pathways tend to favour opportunistic, bacterial-feeding fauna. On the other hand, low-pressure systems characterised by "slow" nutrient cycles dominated by more resistant OM and fungal decomposition pathways, tend to show a more heterogeneous habitat and contain a more diverse fauna with persistent species, in general, fungal-feeders (Bardgett and Cook, 1998; Schon et al., 2010a).

For example, it has been suggested (Bardgett and Cook, 1998) that a "hump-backed" relationship (Fig. 4.16) between soil fauna species diversity and abundance, and disturbance may occur (Grime, 2001). Bardgett and Cook argued that a stable, uniform environment with abundant resources (e.g. unmanaged grassland) favours dominance of particularly competitive species, forcing competitive exclusion. Moderate stress (e.g. organically managed, low-input system) may reduce the likelihood of competitive exclusion and allow other organisms to proliferate. At the other extreme, severe stress (e.g. intensive agriculture) clearly leads to a reduction in soil faunal diversity (Bardgett and Cook, 1998, p. 274).

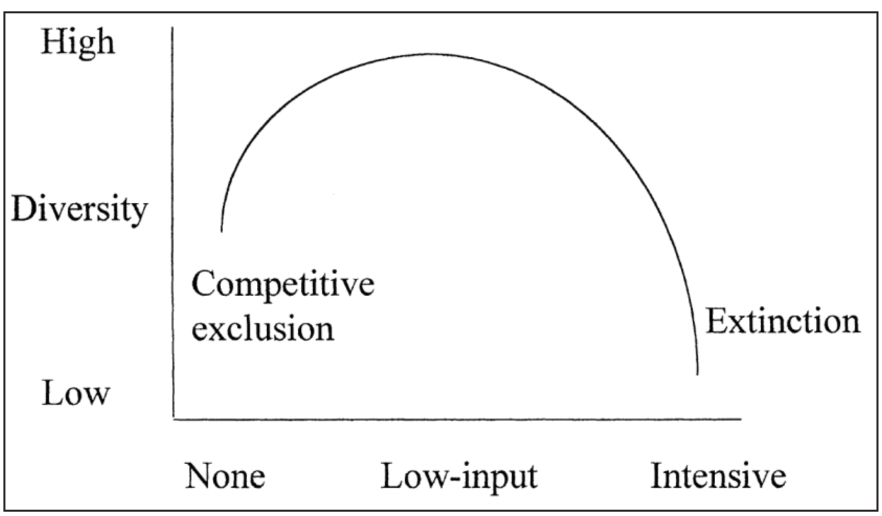

Figure 4-16: Hypothetical model of the effects of management intensity on the diversity of soil fauna in agricultural grasslands (from Grime, 2001).

Also, Schon et al. (2010d) showed the importance of initial diversity of functional groups in providing resilience to increasing external pressures. They showed that anecic earthworms (deep burrowers) can substitute litter-incorporating epigeic earthworms (surface burrowers) vulnerable to treading in intensively managed pastoral systems by taking on the incorporation of litter, as well as being important soil engineers.

4.5.1.2 Regulation of animal pests and diseases:

DFE are a source of pathogens. Pathogen survival in soils presents a threat to animal and human health since pathogen movement and discharge to waterways can lead to water contamination. Pathogen survival in soils depends on resident biota populations (natural biocontrol) and also on a number of soil properties (Donnison and Ross, 2009) (Fig. 4.17). Soil water content is a major factor determining bacterial survival with greater survival associated with moist soils (Wang et al., 2004). Low temperatures and neutral pH are also known to favour survival (Donnison and Ross, 2009). Clay and OM matter content impact on the sorption of negatively charged microbes (McLeod et al., 2008) and thereby influence their retention in soils or release in water. The more time pathogens spend in soil, the greater the chance that they are consumed by the local soil biota. Muirhead (2009) showed that mean concentration of E-Coli in the soil under dung pads remained high for up to 6 months after the animals were removed. Moreover, soil structure and the type of flows within the profile (preferential or matrix flows) determine the time spent by pathogens in soils and the numbers of individuals transferred to waterways (McLeod et al., 2008). McLeod et al. (2008) argued that “soils with a drainage impediment or those with well developed soil structure have a high potential for microbial bypass flows, whereas soils with less developed, porous soil structure (tephra, recent soils) have a low potential for microbial bypass flows” (McLeod et al., 2008).

Soil nutrient status can impact directly on livestock diseases. For example, nitrogen concentrations are suspected to increase toxin concentration in ryegrass, thereby favouring ryegrass staggers (Lambert et al., 2004) (Table 4.14). Trace-element deficiencies in soil can lead to metabolic diseases in livestock. Animal intake of Molybdenum (Mo) from soil and pasture, in the presence of sulphur (S), can dramatically reduce the absorption of Cu, leading to Cu deficiency in the animal.

Soil water content is an important property of habitat (Fig 4.17) which can select the species living in soil. Some organisms like micro-organisms (bacteria, fungi) or nematodes live in the thin film of moisture around soil particles. Some bacteria species are fragile and can be killed by small changes in the soil environment, whereas other species are able to withstand severe heat, cold or drying. Some can lie dormant for decades waiting for favourable conditions.

Earthworms absorb oxygen and release carbon dioxide directly through their moist cuticle. They can live for some time in water-logged soils if the oxygen supply is adequate but they suffocate if the oxygen content is too low.

Soils can act as a significant reservoir of infective larvae of parasitic nematodes (Leathwick et al., 2010; Stromberg, 1997; Waghorn et al., 2002). Stromberg (1997) argued that in a wet climate, the larvae are washed down into the soil. If the environment is dry, movement onto surrounding herbage is difficult, thus forcing movement or migration into the soil beneath the dung pad. Nematode larvae could find refuge in the moist soil beneath dung pads, before migrating back onto the surrounding herbage, when conditions get better (Stromberg, 1997). He also observed that soil type may have a major effect on the ability of larvae to migrate (Stromberg, 1997). Leathwick et al. (2010) showed that the numbers of larvae of *Teladorsagia circumcincta* and *Trichostrongylus colubriformis* (two parasitic nematodes) recovered from soils exceeded those recovered from herbage and faeces. However, the spatial dynamics of nematode larvae for ingestion by grazing livestock is still poorly understood (Leathwick et al., 2010). In a recent study, Leathwick (pers. com.) found, in a comparison between L3 numbers recovered from the field and from larval cultures in the laboratory, that the potential for parasite development is far greater than that which occurs in the field, which emphasises the role that climatic and soil factors play in limiting parasite infestation. Even under the most favourable conditions in the field, yield of L3 was still less than 25% of potential growth from laboratory faecal cultures. This work indicates that soil properties and processes play a significant role in regulating the number of eggs that hatch and the numbers of L1 and L2 larvae that survive through to the effective L3 stage, and the survival of L3 larvae.

Soil fauna has been shown to play an important role in the spatial distribution of larvae of different parasites, as well as in their survival. Gronvold (1979) showed that earthworms play a role in the transmission of infective *Ostertagia ostertagi* larvae (parasite nematode) from cow pats to the surrounding soil. Earthworms eat infected cow dung; they ingest larvae, which pass through their intestinal canal to the surrounding soil surface alive (Gronvold, 1979). Waghorn et al. (2002) showed that the numbers of infective larvae were reduced by the action of earthworms. Moreover they suggested that parasite eggs and/or larvae are damaged in some way following their ingestion by earthworms. On the other hand, artificial dung burial seemed to favour larvae development and migration and protect larvae against consumption by earthworms (Waghorn et al., 2002).

4.5.1.3 Regulation of plant pests:

Soil fauna species vary greatly in size and therefore need different habitable pore space. Most of the species present in soils live in macropores >30 µm. Therefore, soil macroporosity is a good indicator of soil biological health. A number of pasture pests like Porina, Grass grub and Clover root weevil (Table 4.13) have big larvae that develop in soil macropores, therefore soils with low macroporosity are less likely to offer these pests a favourable environment to develop. Moreover, the degradation processes decreasing macroporosity impact negatively on these pests.

The moisture useful for plants, soil invertebrates and micro-organisms is stored in soils in pores ranging in size from 30 µm to 0.2 µm (Table 4.1), but unlike plants which are available to grow roots deeper in the profile to reach more water, most of the fauna of the topsoil can tolerate dry conditions only to a limited extent, and hence require more water in the bigger soil pores (>10 µm). Strong seasonal moisture fluctuations impact firstly on meso- and macro-fauna living in bigger pores, whereas micro-fauna like nematodes, which survive in water films, are generally less readily affected by moisture stress (Schon et al., 2010b). The seasonality of soil water content therefore determines the type of species present in soils and their abundance, and thereby regulates the development of pests like porina, grass grub and clover root weevil, which all have a larval stage (caterpillars or grubs) that is sensitive to dry and wet conditions (Table 4.13).

New Zealand pastures can contain a limited diversity of plant species (generally only ryegrass and white clover) and are therefore an ideal food source for insects (pests) who feed specifically on the leaves (porina, crickets, slugs) or roots (grass grub, clover root weevil) of these plants. The presence of soil animals, such as nematodes or collembola, directly affects the biomass and activity of the microbial community (fungi and bacteria) since they are feeding on them, or indirectly by breaking down organic matter and altering nutrient availability (Bardgett and Cook, 1998).

Bacterivore and fungivore species are very important to regulate diseases, while natural predators have been used a lot to biologically control pest populations. For example, the fungus *Metarhizium anisopliaea* and NPV virus can kill porina caterpillars. Grass grub larvae are sensitive to a number of diseases (bacteria), and parasitic wasps have been used to regulate clover root weevil populations.

Jackson (1990) reported initially low numbers of grass grub (*Costelytra zealandica*) in young pastures, that commonly rise to a peak 4-6 years from sowing, before declining. He also noted that grass grub numbers in older pastures fluctuate but rarely reach the same levels as the early peak. Natural biological control agents such as birds, invertebrate predators, toxin-producing endophytes, parasites and diseases are the key factors causing mortality in grass grub populations and limiting damage in older pastures (Jackson et al., 2002).

A similar phenomenon was observed with Porina (*Wiseana*) by Kalmakoff, et al. (1993). He reported a dramatic rise in the porina population in the first year after sowing a new pasture, followed by a decrease in population in year 2 and 3. Kalmakoff, et al. (1993) noted that viruses increased as larval density decreased. Porina larval density of a new pasture in the first year was four times the density of an old pasture where viruses and parasites were well established.

Maximising the benefit of biocontrol is now used by some farmers as a bio protection strategy for preventative action against pasture pests (Jackson et al., 2002).

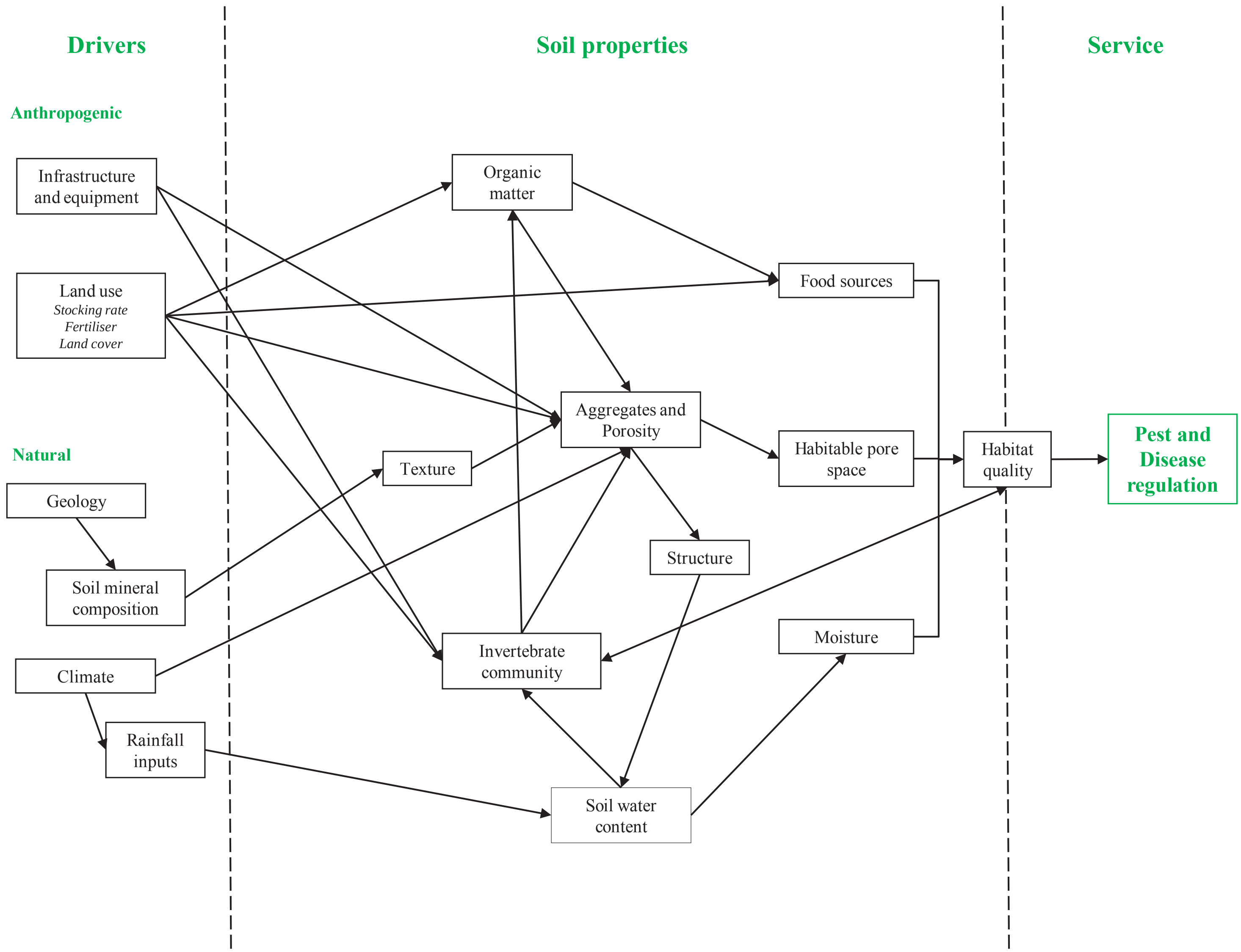

Figure 4-17: Drivers and soil properties influencing the regulation of pest and disease populations.

In the next section, the degradation processes and external drivers impacting on the regulation of pest and disease populations are investigated.

4.5.2 Degradation processes and drivers influencing the regulation of pests and diseases:

There are a number of degradation processes impacting on habitat quality and thereby on the regulation of pest and disease populations.

Soil compaction is known to have a strong effect on soil fauna (Schon et al., 2010d) directly because of the destruction of a number of animals from practices like tillage or livestock treading, and indirectly from the loss of habitable pore space (Fig 4.17). Cultivation techniques, like preparing the soil to sow a new pasture, reduce pathogen levels. Stock management (high stocking rate) is used by some farmers to mitigate infestations of pasture pests like crickets, porina and grass grubs. Schon et al. (2010d) explored the influence of dairy cow stocking rates (3, 4 and 5 cows/ha) on the abundance and diversity of soil invertebrates at two depths (0–7.5 and 7.5–15 cm) on a well structured loamy Andosol soil, in two seasons (autumn and winter). They found little change in soil structure as stocking rates increased but macro-fauna predators were absent at high stocking rates. The most sensitive invertebrates to stock treading seem to be those that don’t have the ability to create or maintain their own habitable pores and don’t have sufficient mobility to escape physical disturbance (i.e. Oribatida and large nematodes) which includes the larvae of a number of pasture pests like porina, grass grub and clover root weevil (Table 4.13) (Schon, 2010; Schon et al., 2010d). Compaction also affects drainage impacting on soil water content and aeration.

Erosion, by removing soil material also removes some soil fauna, as well as OM and nutrients, a source of food and energy which can impact on population dynamics and equilibriums between services. Chemical processes like salinisation or acidification change soils chemical equilibriums and soil conditions which can lead to the disappearance of some species and the development of others.

Natural drivers like climate influence pest and disease regulation by driving the selection of species present in soils (Fig. 4.15 and 4.17). Rainfall drives soil water content ,and thereby determines, throughout the year, the seasonality of the development of specific species (Schon et al., 2010b). Temperature also selects soil fauna, especially micro fauna like nematodes, fungi, bacteria, and impacts on biological processes. Micro-organisms like bacteria have a range of temperature where they develop the fastest and are the most efficient. The larvae of pasture and livestock pests are particularly sensitive to temperature and SWC variations (Table 4.15 and 4.17).

Anthropogenic drivers like land use and management practices also influence greatly the regulation of pest and disease populations (Fig. 4.15 and 4.17). Land use determines the nature and quantity of inputs to the soil (plant litter, animal wastes). Forests and pasture usually accumulate OM because leaves and roots decay, whereas in cropping systems most of the above ground biomass is removed, which restricts food available for soil fauna. In livestock farming, the nature of the OM returned to the soil is different because it includes animal dung and urine which are quickly utilised by bacterial pathways and general detritivores (Bardgett, 2005; Schon, 2010). Monocultures like crops or pastures are especially sensitive to pests because they have low plant biodiversity levels and provide an abundant source of food for specific species. A number of farming practices impact on the type and diversity of soil fauna, and thereby on pest and disease populations. It is difficult to separate from one another the effects of individual management inputs to grassland on soil fauna (Bardgett and Cook, 1998; Schon, 2010). Farming practices like grazing regimes and livestock treading impact on soil porosity, and thereby mostly on species unable to create their own pore space and bigger species like macro and meso-predators (Schon et al., 2010d). Studies (Bardgett and Cook, 1998) show that the numbers of nematodes, in particular bacterial-feeders and plant-feeders, are higher in grazed than ungrazed pastures because of increased food inputs to soils (Schon et al., 2010a).Disturbance from cultivation can impact negatively on pest and disease levels and associated predators. Age of the pasture has been shown to influence infestation levels, older pasture showing stronger natural biological control (Jackson, 1990; Kalmakoff et al., 1993). Practices driving soil water content, like irrigation or artificial drainage, also impact on the seasonality of soil fauna populations. The application of both organic and inorganic fertilisers to grasslands has been shown to have variable effects on microarthropods and nematode populations, with reports of both reduced and enhanced populations (Bardgett and Cook, 1998). Finally, pesticides are commonly applied to New Zealand pastures to regulate infestation of pests and diseases.

4.5.3 Quantifying the regulation of pest and disease populations:

4.5.3.1 Previous attempts to quantify the regulation of pest and disease populations:

A number of authors have mentioned soils ability to provide habitat to numerous species and to regulate pest and disease populations. De Groot (2006) referred to habitat functions of refugium and nursery. Costanza et al. (1997) consider refugia as an ecosystem service and talked about the provision of habitat for resident and transient populations. Costanza et al.

(1997) and De Groot (2006) also talked about "biological control" and "population control" through trophic-dynamic relations. Moreover, the MEA (2005) mentioned regulating services such as "disease regulation" and "pest regulation". Other authors mention habitat provision (Swinton et al., 2007; Wall et al., 2004; Weber, 2007; Zhang et al., 2007) and population regulation (Barrios, 2007; Lavelle et al., 2006; Porter et al., 2009; Sandhu et al., 2008; Wall et al., 2004; Zhang et al., 2007).

Porter et al. (2009) and Sandhu et al. (2008) assess the biological control of pests in an agro-ecosystem by measuring predation rates of specific insects. To our knowledge, no one has ever tried before to measure and value this service for soils.

In this study, habitat provision is considered as a supporting process not the service. The characteristics of the habitat provided (soil properties like porosity and SWC) are the natural capital stocks from which the regulation of pest and disease populations' service arises. Therefore here, it is not the habitat provision that is quantified, but the dynamics of the efficiency of the regulation of pest and disease populations by soils.

4.5.3.2 Parameters chosen to quantify the regulation of pest and disease populations:

The regulation of pests and diseases by soils depends on soil properties and population dynamics. The dynamics of soil fauna populations are very complex and dependent on a number of factors from soil properties to management practices (Fig. 4.15).

In this study, it was chosen to focus on two pasture pests Porina and Gras grubs. Parasitic nematodes of cattle are not an issue for mature cows, but their regulation should also be taken into account if including the health and well-being of young calves and heifers. This is not included in the current analysis.

To inform the regulation of pest and disease populations by soils, there are two aspects to consider. The habitat conditions for the development of the chosen species need to be identified, and associated to risk of infestation. Schon et al. (2010e) showed that across different dairy pastures, soil porosity appeared to have a larger and more consistent influence on soil invertebrates in comparison to potential food resources.

Process-based models can enable us to follow SWC and Mp dynamics and thereby the risk of infestation by each pest. Secondly, the level of natural biological control (predators and diseases) needs to be assessed. Age of pasture is used here as an indicator of the level of natural biological control.

The measure of the service can then be defined as the difference between maximum levels of infestations (for new pastures with the lowest competition) and the actual infestation rate. This measure would represent the pest population that is regulated by soil conditions (habitat and biodiversity).

4.6 Summary of the quantification of soil regulating services:

The conceptual thinking and information presented in this chapter forms the basis for quantifying the provision of regulating services from soils. The quantification of each service is based on natural capital stocks and the determination of the role played by the soil, as opposed to just recording emissions or losses. Table 4.15 presents a summary of the soil natural capital stocks behind each regulating service and Table 4.16 lists the parameters used to quantify the provision of the services. This information is then used in Chapter Seven, Eight and Nine, to quantify and value the provision of ecosystem services from different soil types, and for different management options.

In the next chapter, **Chapter Five**, it is shown how an existing process-based soil model has been modified to provide the necessary data to calculate the parameters chosen here, and follow their dynamics, for the quantification of soil services.

Table 4-15: Summary of the quantification of soil regulating services.

	Natural capital stocks	Flood mitigation	Filtering of nutrients and contaminants	Detoxification and recycling of wastes	Carbon storage and regulation of N_2O and CH_4	Biological control of pests and diseases
Inherent Properties	Depth	X	X	X		
	Structure	X			X	X
	Texture		X	X		
	Soil strength					
	Stone content	X	X			
	Clay content		X	X	X	X
	Fragipan	X				
	Drainage class of subsoil	X				
	Inherent mineral contents		X	X		
Manageable Properties	Biodiversity		X	X	X	X
	Organic matter		X	X	X	X
	Dissolved organic matter			X		
	Anion storage capacity		X	X		
	Cation exchange capacity		X			
	pH		X	X		X
	Porosity	X	X		X	X
	Bulk density					
	Nutrient status		X	X	X	X
	Trace-elements status					X
	Saturation levels		X	X		
	Temperature		X			X
	Soil water content	X	X		X	X
	Field capacity	X				
	Saturation capacity	X				
	Available water capacity	X				
	Plastic limit					
	Drainage class of topsoil	X			X	X

Table 4-16: Parameters used to quantify the provision of regulating services from soils.

	Flood mitigation	**Filtering of nutrients and contaminants**	**Detoxification and recycling of wastes**	**Carbon storage and regulation of N_2O and CH_4**	**Biological control of pests and diseases**
Parameters chosen	Saturation capacity	N leaching and P runoff for the filtering of nutrients	Dung load	C stocks Net C flows	Porosity
	Rainfall (mm/year)	Timing of grazing events regarding soil water content and runoff for the filtering of pathogens	Timing of grazing regarding soil water content	N_2O emissions	Biodiversity
	Runoff (mm/year)		Soil water content, Macroporosity	CH_4 oxidation	Soil water content

PART TWO

CAPACITY BUILDING

Chapter Five

Methodology for the Dynamic Modelling of Soil Ecosystem Services

This chapter builds on Chapters Three and Four, providing background and a context to the case study dairy farm and the model used to generate the data needed to quantify the provisioning and regulating services from pastoral soils under a dairy farm operation. The chapter also describes the input data required for modelling, the data actually available, as well as data gaps. It also describes the additional information added to several components of the model in order to capture the impact of specific management practices on soil properties and processes behind specific soil services.

5.1 Context of the study:

In this study, the provision of ecosystem services from a pasture soil under a dairy farm operation is examined. The dairy industry is a major driver of New Zealand's economy, and dairy is a widespread land use. New Zealand produces about 2% of the total world milk production at around 16 billion litres per annum (1.4 billion kilograms of milk solids (MS)) but, unlike most other countries, around 95% of its dairy produce is exported, rather than consumed by the domestic market. The Dairy industry is one of New Zealand's largest industries, contributing approximately 25% of total merchandise export earnings ($NZ 10 billion in 2008-09) (LIC and DairyNZ, 2009). The Dairy industry's major markets vary depending on products. Britain and the European Union are New Zealand's most valuable markets for butter. New Zealand is the world's largest butter exporter and accounts for about 44% of all traded butter. The primary markets for casein and cheese are the United States, Japan, and the European Union, with New Zealand being the world's largest exporter of casein and caseinate products. New Zealand's most important milk powder markets are in Central and South America, and Southeast Asia. Exports of skimed and whole milk powders contribute about 27% and 38%, respectively, of world trade (LIC and DairyNZ, 2009).

New Zealand had 11,618 dairy herds and 4.25 million dairy cows and heifers in milk in 2008/09 (LIC and DairyNZ, 2009). Since 1975 the number of dairy cows in New Zealand has doubled (Fig. 5.1). Over time, the number of dairy herds has decreased (Fig. 5.1) (LIC and DairyNZ, 2009).

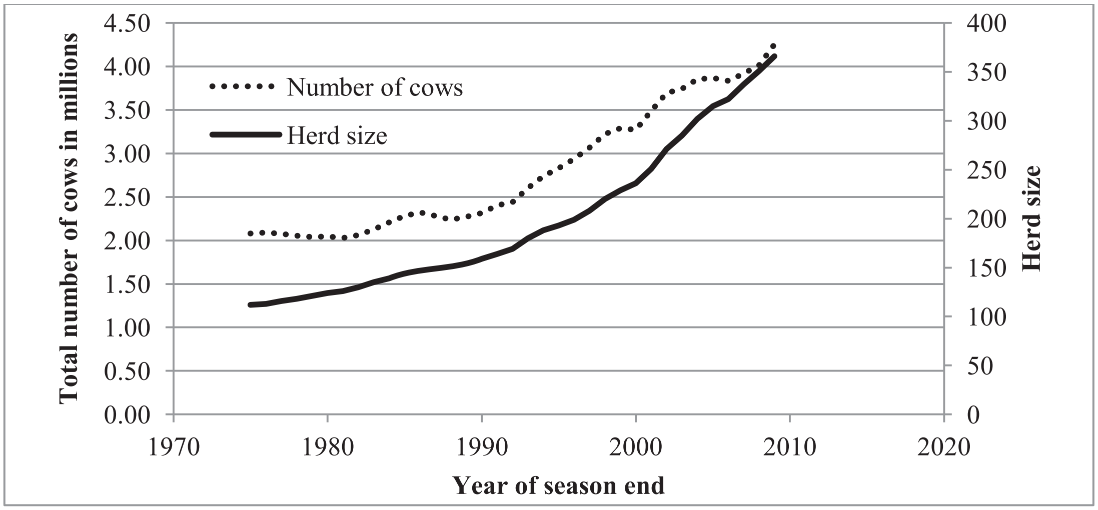

Figure 5-1: Total number of cows in New Zealand and herd size since 1975 (LIC and DairyNZ, 2009).

The New Zealand temperate climate is perfect for outdoor legume-based pasture grazing '*in situ*' by livestock. Around 95% of milk production is from utilisation of the pasture forage base. Few cows are wintered indoors. The national New Zealand dairy herd is made up mainly of Holstein-Friesian (43%), Jersey (14%) and Ayrshire (1%), but also crossbreeds like Holstein-Friesian/Jersey (around 36%) (LIC and DairyNZ, 2009) (Fig.5.2).

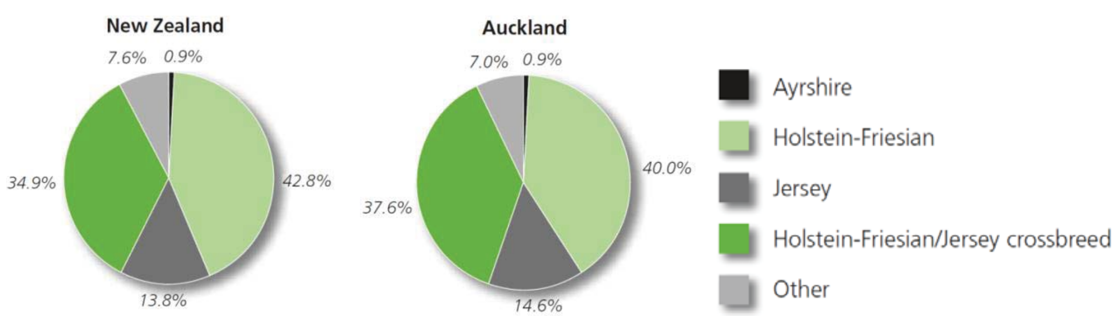

Figure 5-2: New Zealand dairy cattle breeds (LIC and DairyNZ, 2009).

The average cow production is 325 kg of milk solids (MS) per cow per year[3], but production varies between cow breeds (Table 5.1) and regions.

[3] Milk solids (MS) are fat and proteins.

Table 5-1: Dairy cattle breeds characteristics (season 2008/2009) (LIC and DairyNZ, 2009)

Dairy cattle breed	Average live weight (kg)	Days in milk	Milk solids (kg/cow/yr)
Holstein-Friesian	477	214	329
Jersey	380	217	291
Holstein-Friesian/Jersey crossbreed	439	217	328

The main dairying areas are South Auckland/Waikato, Canterbury, Taranaki and Southland (Fig. 5.3). About 77% of dairy farms are located in the North Island, with 31% of all New Zealand dairy farms located in the South Auckland/Waikato region (LIC and DairyNZ, 2009).

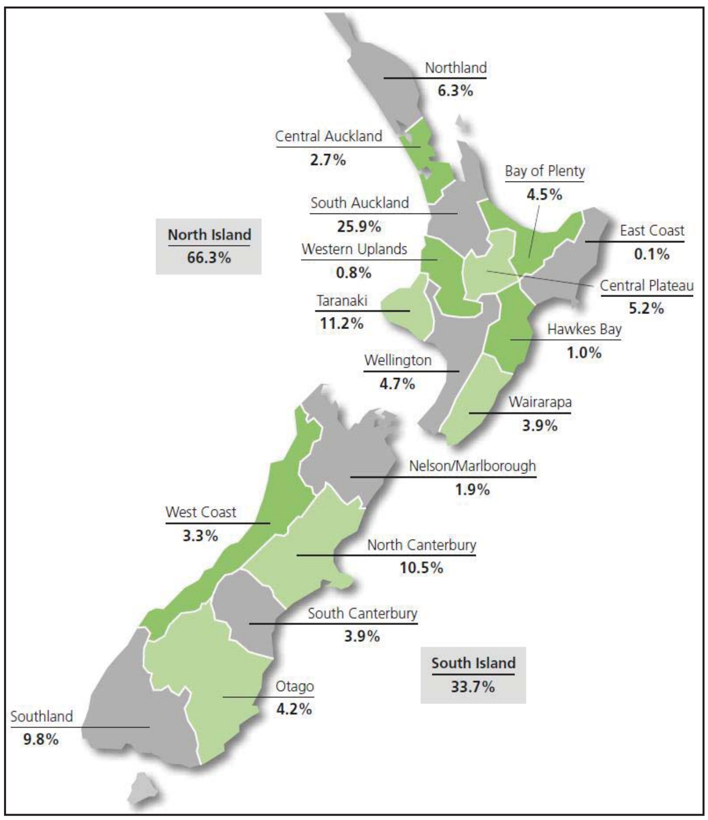

Figure 5-3: Regional distribution of dairy cattle (2008/09) (LIC and DairyNZ, 2009).

The main dairy areas (South Auckland/Waikato, Canterbury, Taranaki and Southland) are also the areas presenting most of New Zealand's high class soils (Fig. 5.4).

Figure 5-4: High class soils in New Zealand (TeAra, 2010)

The country's continued wealth generation is dependent on increasing production, primarily through an increase in the intensity of land use. New Zealand's best soils (versatile or high-class) with the highest natural capital have a limited area, only about 5.5% of New Zealand. They are most common among the Recent and Allophanic soil orders (Hewitt, 1993). If class III soils are also included, the total area of good soils increases to 15% (Rutledge et al., 2010). The majority of soils have medium to low natural capital stocks, requiring more added capital (lime, fertilisers, irrigation, drainage...) to deliver the required production levels.

In this chapter, the dairy farm and soils chosen for this study are described. The chapter also presents the model used, and describes the input data required for modelling and the additional information added to several of the functions of the model in order to capture the impact of specific management practices on soil properties and processes behind specific soil services. The outputs of the modified model then form the basis for valuing the natural capital of a soil in Chapter Seven and the basis for scenario analysis in Chapters Eight and Nine.

5.1.1 A dairy farm in the Waikato:

The Waikato region was chosen for this study because it is one of the major dairy regions of New Zealand, and also one of the regions with the most 'high class' soils, but also other soils with limited natural capital, under the same land use.

A typical Waikato dairy farm is described here (Table 5.2). Such farm is dependent on clover-based pastures grazed *in situ* year round. A typical nutrients program (fertilisers) targets the legume component of the pasture to maintain clover quantity and N status through the fixation of atmospheric N_2. Dairy cows are almost entirely pasture fed with some off-farm grazing (young stock, winter grazing for cows) and some supplements like maize silage brought in, if needed, to sustain milk production. On some farms, animals are kept on feed-pads or wintering platforms for several hours per day, when the soils are too wet, or grazed-off during the winter months (de Klein and Ledgard, 2001; LIC and DairyNZ, 2009).

Table 5-2: Average dairy farm data (season 2008/2009) (LIC and DairyNZ, 2009)

	Country **New Zealand**	**Region** **South Auckland**	**District** **Waikato**	**TDF** **Typical dairy farm**
Average herd size	366	308	313	330
Average effective area (ha)	131	104	109	110
Average cows per hectare	2.83	3.02	2.93	3
Average kg MS per effective hectare	921	952	908	900
Average kg MS per cow per year	323	314	307	300
Days in milk	266	265	270	270

For this study, average data was chosen (Table 5.2) even if big dairy farms of the Waikato can have up to 1000 cows, with stocking rates up to 4-5 cows/ha, and produce up to 525 kg MS/cow/year (LIC and DairyNZ, 2009). The typical dairy farm (TDF) considered in this study (Table 5.2) is a 110 ha farm, on flat land (slope<5%), with 5 ha paddocks. The herd size is 330 cows (3 cows/ha). Pasture silage is made from the farm in spring and fed to the cows as supplements when needed (in winter). The animals are kept on the farm for winter (no grazing off). There is no irrigation or drainage on the farm.

5.1.2 Soils chosen for the study:

For this study, the provision of ecosystem services from two soils with contrasting natural capital, defined by differences in their soil properties, are examined.

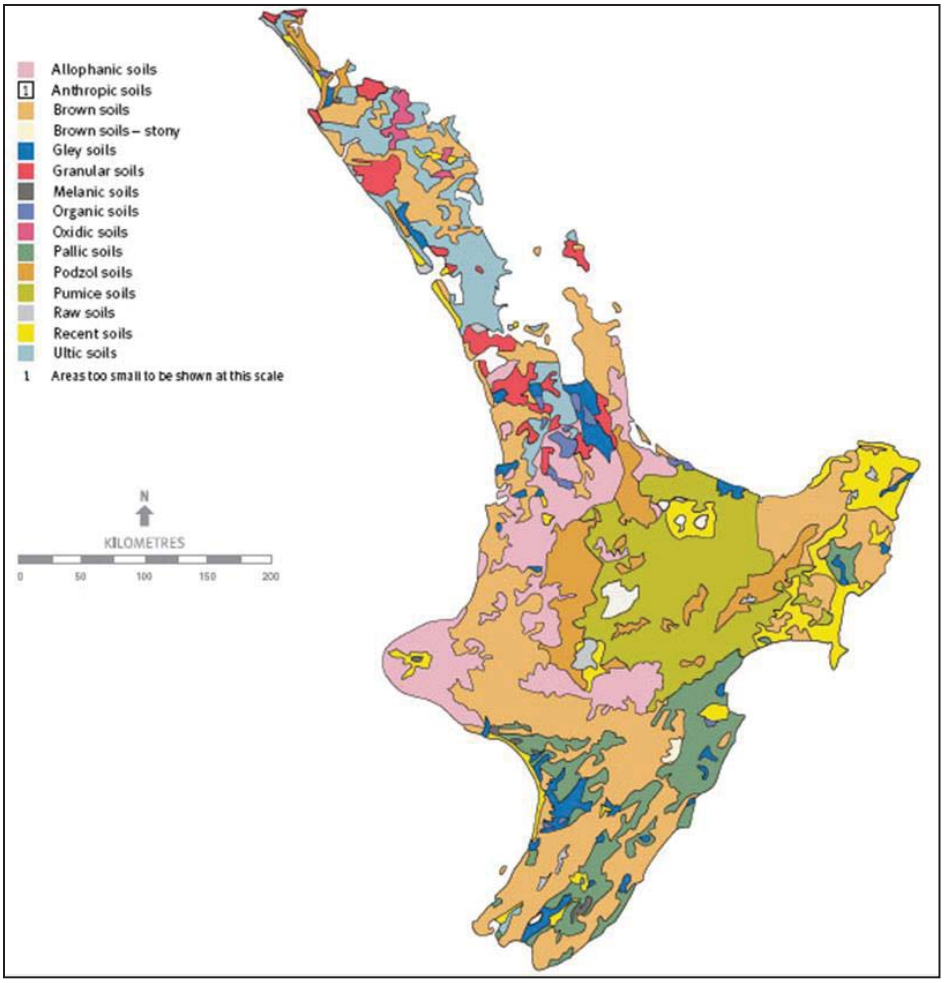

Figure 5-5: Distribution of soil orders in the North Island of New Zealand (MfE, 2010).

5.1.2.1 Typic Orthic Allophanic Soil:

The first soil chosen for this study is a Typic Orthic Allophanic Soil, the Horotiu silt loam (New Zealand soil Database: SB09944), or an Andisol according to the US soil taxonomy. The Horotiu silt loam is a silt loam (90cm) on sand and is well drained (Table 5.3). Allophanic Soils are dominated by allophane minerals which have a high affinity for phosphate. These minerals coat the sand and silt grains and maintain porous, low density structure with weak strength. Allophanic Soils occur predominantly in the North Island volcanic ash, and in the weathering products of other volcanic rocks (Fig.5.5). They also occur in the weathering products of greywacke and schist in the South Island high country. They cover 5% of New Zealand. Allophanic Soils have low bulk density, with little resistance to root growth. Top soils are stable and resist compaction (Table 5.3). Erosion rates are generally low except on steep slopes or exposed sites (Hewitt, 1993). Orthic Allophanic Soils are Allophanic Soils that don't have a perched water table (Perch-gley Allophanic), a groundwater table (Gley Allophanic), or a hard layer (Impeded Allophanic) (LandcareResearch, 2010).

5.1.2.2 Typic Orthic Gley Soil:

The second soil chosen is a Typic Orthic Gley Soil, the Te Kowhai silt loam (New Zealand soil Database: SB09945), or an Inceptisol according to the US soil taxonomy. The Te Kowhai silt loam is a silty clay loam (100cm) on sand and is poorly drained (Table 5.3). Gley Soils are strongly affected by water logging and have been chemically reduced. Waterlogging occurs in winter and spring, and some soils remain wet all year. Gley Soils occur throughout New Zealand in low landscape positions where there are associated high groundwater-tables (Fig.5.5). Large areas of Gley Soils have been artificially drained to form more productive agricultural land. They cover 3% of New Zealand. These soils have shallow potential rooting depth and relatively high bulk density. They are sensitive to compaction (Table 5.3). Organic matter content is usually high (Hewitt, 1993). Orthic Gley Soils are Gley Soils that are not Sulphuric (in marine estuaries), Sandy, Acid, Oxidic, or Recent (young land surfaces, alluvial or estuarine) (LandcareResearch, 2010).

Table 5-3: Soil properties (0-50cm) of Horotiu and Te Kowhai (LandcareResearch, 2010).

Soil properties	Te Kowhai silt loam	Horotiu silt loam
Soil water content: first 50 cm		
Field capacity (FC) (mm)	54	53
Stress point (SP) (mm)	41	38
Wilting point (WP) (mm)	28	25
Available water content (AWC) (mm)	26	28
Soil properties at 10 cm		
Bulk density (g/cm^3)	1.1	0.84
Total C (%)	2.5	5.5
P retention (%)	26	91
K sat (mm/day)	30	86
Drainage class	Poorly drained	Well drained
Sensitivity to compaction	Yes	No

5.2 Modelling soil ecosystem services:

In order to quantify the provision of ecosystem services from the soil natural capital of two different soils, under a dairy farm operation, a large number of datasets describing soil processes, and the dynamics of soil natural capital stocks are required (Chapters Three and Four). To inform the provision of ecosystem services from soils, data had to be collected on over fifteen soil properties, on different soil types under a range of management practices (Chapters Three and Four). Because soil natural capital stocks are influencing, and at the same time, the result of supporting and degradation processes (Fig. 5.6), they are dynamic features of a soil. To inform the provision of soil services, which are therefore also dynamic, soil natural capital stocks (e.g. soil properties) had to be followed over a period of time. To capture the influence of drivers like climate, land use and management, multiple years and different management practices had to be investigated. Because of the necessity to look at the influence of multiple soil properties and processes on soil services under a range of climatic conditions, the data requirements couldn't be sourced from field experiments. Data from the literature provided some assistance, but published studies often only report values on some of the soil properties of interest, or only at one point in time for a limited combination of treatments. Moreover, management practices, study sites, soil types, climate, and so forth, varied between different studies, making it very difficult to build a coherent dataset. To address these limitations, it was decided to use a process-based soil-plant-atmosphere model, which has the capacity to run actual weather data for several years, outputting datasets that include variation in soil properties (stocks) and outputs (flows), and which can run with different management practices.

The modelling option gave us the ability to isolate and explore individual manageable properties (Fig. 5.6). It also provides high flexibility. Process-based models, by allowing us to explore the influence of individual soil properties, e.g. untangle the different natural capital stocks, enables us to operationalise our framework. It also enables an examination of a range of scenarios and their impacts on the provision of soil services.

5.2.1 The SPASMO model:

The SPASMO model (Soil Plant Atmosphere System Model) from Plant and Food Research (Green et al., 2003) was chosen to model soil processes and the dynamics of soil properties needed. This is a soil-plant-atmosphere system model, which describes soil processes, plant growth and aspects of farm management. Supporting and degradation processes make up the core of the SPASMO model (Fig. 5.6). The model uses mathematical functions to describe each of the soil, plant, water and nutrient (N and P) processes and links them dynamically to each other and to soil properties using daily time steps. The model uses, as inputs, soil type (soil properties) and external drivers like climate, land use and management practices (Fig.5.6). It outputs daily measures of chosen soil properties and their dynamics according to these drivers and keeps stock of the flows of nutrients, matter and water. Simple allometric relationships are used to describe the feed, energy and nutrient budgets for the grazing animals, and to parameterize the returns of dung and urine to the grazed pasture.

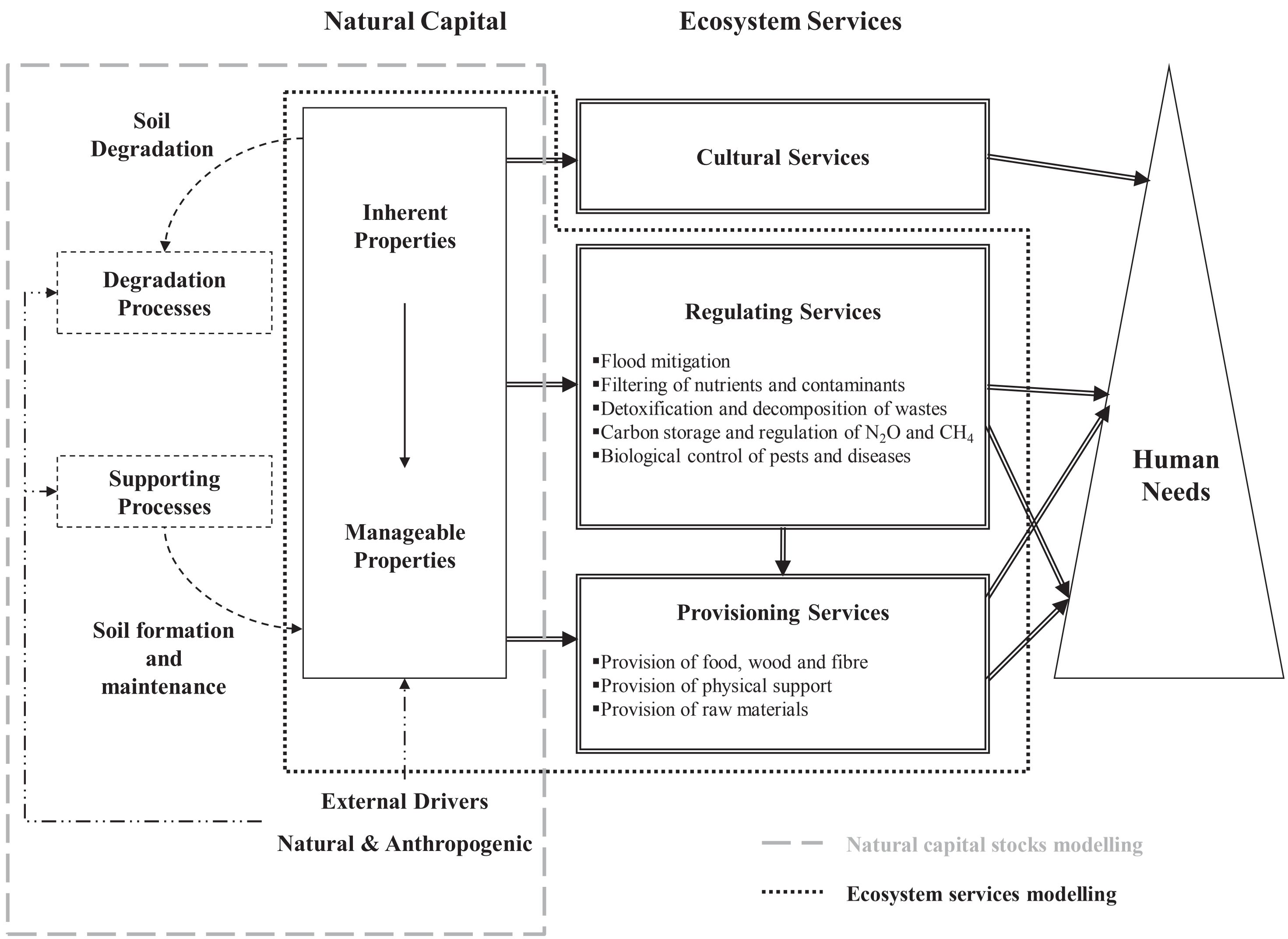

Figure 5-6: Different steps of the modelling process.

The natural capital stocks modelling is done with the SPASMO model. The ecosystem services modelling is done manually.

The generation of the dynamics of soil properties with SPASMO, or "natural capital stocks modelling", constitute the first step in the modelling of soil services (Fig. 5.6). To quantify and value ecosystem services from soils, the dynamics of soil properties obtained with SPASMO then need to be linked to the provision of the services. As described in Chapters Three and Four, this can be done by using soil properties as proxies for ecosystem services. This constitutes the second step of the modelling, the "ecosystem services modelling" (Fig. 5.6), which is done in Chapter Seven. This part of the modelling is not in itself dynamic but is based on dynamic variables obtained from SPASMO. Ideally to develop a complete dynamic model of the provision of ecosystem services from soils, our model would need to be incorporated into a process-based soil model like SPASMO. This was not the primary aim of this study, but it would be the logical next step.

Below, SPASMO is briefly presented, as well as its history and functioning. Additions made to SPASMO in order to adapt it to a dairy grazed system for our study are also detailed below.

5.2.2 History and functioning of the SPASMO model:

SPASMO is a detailed process-based model developed by Plant and Food Research for simulating the interactions in the plant-soil-atmosphere system. It has been developed using international scientific knowledge and has been adapted and tested in New Zealand. Early versions of the model date back to the late 1990s. The model is being continually improved with more detailed routines and procedures to handle various soil processes (Cichota and Snow, 2009). SPASMO is a flexible model but it does not have a well developed end-user interface. It is, thus, not available beyond Plant and Food Research. SPASMO has been widely used in research, such as in the evaluation of N leaching from pastoral and horticultural land (Rosen et al., 2004), the estimation of water use by plants (Green et al., 2002), and the assessment of pesticide transport in soils (Sarmah et al., 2005).

SPASMO considers the movement of water, nutrients (e.g. N and P), microbes (e.g. viruses and bacteria) and dissolved organic matter through a 1-dimensional soil profile (Green, 2008). The soil water cycle is simulated in SPASMO using a water capacity approach. The soil water balance is calculated by considering the inputs (rainfall and irrigation) and losses (plant uptake, evaporation, runoff and drainage) of water from the soil profile (Green, 2008). The model has a simple routine to simulate plant growth and plant water uptake that can be adapted to different crops, from pasture to vegetables or kiwifruit vines. Nutrient cycles are estimated after computing inputs and outputs for each soil process (e.g. plant uptake, mineralisation, volatilisation...). Most of the processes in the soil have weighting factors to account for abiotic influences, such as temperature and soil moisture (Green et al., 2003). Model results for the water balance are expressed in terms of mm[4]. The concentration and leaching losses of nutrients are expressed in terms of mg/L and kg/ha, respectively (Green, 2008).

SPASMO has been used mainly for horticulture, but has been recently modified to include grazing animals. For pastoral systems, the model simulates the nutrients excreted by animals as the result of a balance between intake and the requirement for maintenance, milk production, growth, and gestation. The nutrients excreted are returned to the soil as urine and dung and are assumed to be uniformly distributed over the paddock (Cichota and Snow, 2009). All the outputs of the model are at the paddock scale, and expressed per hectare. The model works at the field scale (Green et al., 2003; Green, 2008) and does not deliver outputs for the whole farm unless integration of several runs is made by the user (Green S.R., 2010, pers. comm.). Management practices like fertilisation, irrigation or grazing regime, as well as soil and weather conditions, are inputs to the model.

For this study, the paddocks of the farm where effluents are spread, which have very different nutrient cycles and grazing regimes from grazing paddocks, are not considered. Effluent blocks usually only represent a small area of the farm. The typical grazing paddock modeled is flat (slope<5%) and assumes negligible erosion. The model runs with milking dairy cows. The paddocks of the farm with the replacement heifers (not in milk) are also not considered in this analysis.

5.2.3 Model inputs:

To run, the model requires the following input files: climate, soil and management data.

Climate Data

Climate data for all simulations were obtained from NIWA's (National Institute of Water & Atmospheric Research) CLIFLO database (NIWA, 2010) using records for 37 years (1972-2009) from Hamilton City weather station (Latitude: -37.825, Longitude: 175.275). CLIFLO is the web system that provides access to New Zealand's National Climate Database. It returns raw data and statistical summaries. Raw data include ten minute, hourly and daily frequencies. The data used in SPASMO includes daily values of incoming global radiation, potential evapotranspiration, maximum and minimum air temperature, wind speed and rainfall (Appendix A).

[4] mm = one litre of water per square metre of ground area.

Soil Data

The soil data used as inputs are mainly inherent soil properties (Fig. 5.6) but also include initial values for manageable properties. The data is from the National Soils Database from Landcare Research (LandcareResearch, 2010) and includes values for every 10 cm of the profile for inherent properties including stone content, anion storage capacity, hydraulic conductivity (Ksat), clay and sand fractions, parameters to fit Van Genuchten water release curves (van Genuchten, 1980) and manageable properties including bulk density, total C and total N, (Appendix A).

Management data

The farm management input data used in SPASMO includes cow's average live weight (450 kg) and herd size (300 cows) (Table 5.4). The cows are milked twice a day for 270 days. Average calving season starts on the 20th July (mid winter). Calving covers an 8-10 week period, with the bulk of cows calving in the first 4 weeks.

Table 5-4: Farm data as inputs to SPASMO.

Farm indicators	Farm data
Herd size (cows)	300
Effective hectares	100
Cows per hectare	3
Kg milk solids per effective hectare	900
Kg milk solids per cow	300
Days in milk per year	270
Cows average live weight (kg)	450
Paddock size (ha)	5

The model also needs data on nutrient use. For this study, best management practices for N and P fertilisers inputs were assumed (Table 5.5), including no N fertiliser inputs during the months when drainage is maximum (May to July) and no P applications between May and October (Houlbrooke, 2008).

Table 5-5: Fertiliser best management practices (Houlbrooke, 2008): example for 100 kgN/ha/yr and 35kgP/ha/yr.

Day of year	Day	Ammonia (kg/ha)	P (kg/ha)
61	2/03	25	0
228	16/08	25	0
275	2/10	25	0
320	16/11	25	35

To inform the provision of ecosystem services from soils, the dynamics of a number of soil properties need to be explored (Chapters Three and Four). Behind each service are a number of parameters that need to be calculated, from soil properties, and from the dynamics of the soil processes (Table 5.6). SPASMO outputs most of the data needed. In order to accurately represent a dairy farm and the pressures applied by management practices (e.g. grazing regime, stock treading damage) to the soil natural capital stocks, additional functionality has been added to SPASMO to accommodate the additional dynamics introduced by these drivers.
The section below examines the additional functionality developed for this study.

5.3 Additions made to the model:

In order to effectively model a dairy farm and gather all the data needed to calculate the parameters behind each soil service (Table 5.6), extra-functionality was added to SPASMO, including the impacts of grazing regime on soil structure and pasture growth, grazing rotation, the use of a standoff pad, and extra routines to the P cycle. The sections below describe how these drivers and processes were incorporated into the model.
A summary of the proxy and parameters chosen to represent each service (from the information presented in Chapters three and four), as well as the data available to calculate or measure the parameters chosen and the data missing that was added to the model are listed in Table 5.6.

Table 5-6: Data used for the quantification of each of the parameters chosen. Note: SWC: soil water content, FC: field capacity; SP: stress point; DM: dry matter.

Services	Proxy for...	Parameters	Data available to measure parameter	Additional data required
Provision of food, wood and fibre	Soil structural health and support to plants	Macroporosity	Mp as model input and literature.	Mp dynamics depending on management practices from model
	Soil sensitivity to treading damages	Field capacity, Saturation	SWC dynamics generated by the model.	None
	Water supply to plants	Number of days/year with available water, when SP<SWC<FC	FC and SP: model inputs and from literature and SWC dynamics generated by the model.	None
	P supply to plants	Native Olsen P (unfertilised pasture)	Native Olsen P of different soil types from literature. Actual P concentrations generated by the model.	None
	N supply to plants	Native N status (from native P on unfertilised pasture)	Native N status of different soil types from literature. Actual N concentrations generated by the model.	None
	Food production	Pasture yield (Kg DM/ha/yr)	Pasture yield generated by the model.	More detailed dynamics of pasture yield depending on management practices
Provision of physical support	The provision of support for human infrastructure	Bulk density (BD), Macroporosity (Mp)	BD and Mp: model inputs and from literature	Mp dynamics depending on management practices from model
	The provision of support to animals	Number of days/year when the soil can support animals (when SWC<(FC+Sat)/2)	FC model inputs and from literature and SWC dynamics generated by the model	None
Filtering of nutrients and contaminants	Sorbing capacity of the soil	Anion storage capacity (ASC)	ASC as model input and from literature. OM cycle generated by the model.	More detailed dynamics of OM cycle depending on management practices
	N the soil couldn't retain/filter	N leaching	N cycle and N leaching generated by the model.	N leaching linked to more detailed dynamics of OM cycle depending on management practices
	P the soil couldn't retain/filter	P runoff	P cycle generated by the model.	Calculate P runoff from model and link it to management.
	Risk of bypass flow	Size of subsoil peds	Size of subsoil peds and risk of bypass flow from literature	None
	Risk of leaching	Timing of grazing or effluents application regarding SWC	SWC dynamics generated by the model	None

Table 5-6: Continued.

Services	Proxy for...	Parameters	Data available to measure parameter	Additional data required
Flood mitigation	Amount of water a soil can store	Saturation capacity	Soil saturation capacity as model input and from the literature	Saturation capacity dynamics linked to Mp dynamics depending on management practices from model
	Period when the soil is saturated so cannot store anymore water	Number of days/year when SWC=Sat	SWC dynamics generated by the model	Saturation capacity dynamics depending on management practices from model
	Amount of water that the soil couldn't store	Runoff	Runoff generated by the model	Runoff dynamics linked to Mp dynamics depending on management practices from model
Detoxification and recycling of wastes	Sorbing capacity of the soil	Anion storage capacity	ASC as model input and from literature. OM cycle generated by the model.	None
	Risk of poor decomposition	Timing of grazing regarding SWC	SWC dynamics generated by the model	None
	Optimum N concentration for soil biota efficiency	$[NO_3^-]$	Actual N concentration of soil solution generated by the model.	None
	Soil structural health	Macroporosity	Mp as model input and literature.	Mp dynamics depending on management practices from model
	Optimum soil moisture for soil biota efficiency	Soil water content	SWC dynamics generated by the model	None
Carbon storage and regulation of N_2O and CH_4	Amount of C stored and lost	C stocks Net C losses	C stocks dynamics generated by the model and literature.	None
	N_2O regulation	N_2O emission	SWC and NO_3^- concentration of the soil solution generated by the model and emission factors from the literature.	None
	CH_4 regulation	CH_4 oxidation	SWC dynamics generated by the model and oxidation factors from the literature.	None
Biological control of pests and diseases	Space available for soil biota	Porosity	Mp as model input and literature	Mp dynamics depending on management practices from model
	Food resources available for soil biota	Organic matter inputs	OM inputs and cycle generated by the model.	More detailed dynamics of OM cycle depending on management practices
	Moisture available to soil biota	Soil water content	SWC dynamics generated by the model	None

5.3.1 Pasture utilisation:

Pasture utilisation and the amount of OM returned to the soil are two processes closely linked to a number of services including the provision of food, the filtering of nutrients and contaminants, and the regulation of pest and disease populations (Table 5.6). In order to quantify these services, it is necessary to be able to quantify the amount of pasture dry matter (DM) produced, the amount of DM eaten by animals, and the amount of DM returned to soils (Table 5.6). These quantities are highly dependent on management practices. To effectively reflect pasture utilisation and OM return to soil in a dairy grazed system, a pasture utilisation function was added to the model. Pasture utilisation is defined here as the proportion of grass ingested by the cows compared to the total amount standing on the pasture.

The model assumes plant growth will achieve a maximum potential only if water, N and P are non-limiting. The uptake of soil nutrients (i.e. N and P) by pasture is determined largely by the growth of the above- and below-ground dry-matter (DM), multiplied by their respective N concentrations. Daily biomass production is modelled using a potential production rate per unit ground area, G (kg/m^2/d) that is related, via a conversion efficiency, ε (kg/MJ), to the amount of solar radiant energy, Φ (MJ/m^2/d), intercepted by the leaves, temperature, plant N and soil water status. Pasture growth is maximised only if soil water and nutrients are non-limiting. A simple relationship is used to partition the daily biomass production into the growth of the foliage, stem material and roots. Plant biomass is expressed in terms of the balance between growth and senescence of the plant organs. A simple mass balance equation is written for each plant organ by considering inputs of DM due to new growth, losses of DM due to senescence, and the removal of DM due to harvest or grazing. The senescent grass is returned to the soil surface where it slowly decomposes (assuming a first-order rate constant) and releases plant nutrients (in the form of dissolved organic nitrogen (DON), dissolved organic phosphorus (DOP) and dissolved organic carbon (DOC)) to the soil profile. The N demand for crop growth is set by the maximum N content of the root, leaf and stem material. During active growth, the plant tries to supply new DM material with N corresponding to these maximum concentrations. Pasture growth parameters have been calibrated from the literature to fit pasture yields in the Waikato for the chosen soils.

The model, as it was, did not take into account the fact that cows only eat part of the standing dry matter available on a pasture and that, depending on SWC they can bury some of the available grass. This has been addressed by adding to the model a function that includes pasture buried by the grazing animals as a function of SWC.

When cows graze, they eat only a part of the available standing DM in the paddock. A certain amount of grass available is not utilised by the cow, and is deposited on the soil surface, and is slowly incorporated into the soil as dead OM. This was added to the model because it influences cow feed intake, as well as nutrients returned to the soil, therefore impacting on many services (Table 5.6). The fraction of the pasture which is not eaten by the cow but deposited on the soil surface was set at 15% of what is available and is independent of SWC.

In New Zealand, dairy cows are not housed but graze pastures year round. This can pose a number of challenges, especially in spring and winter when soils are wet, including N_2O emissions and sediment and nutrient losses. The treading of pastoral soils by dairy cows is a degradation process (Fig.5.6) which impacts on pasture utilisation and soil structure and thereby on the provision of a large number of services contributing directly and indirectly to pasture growth (Table 5.6).

When the soil is wet, cows trample and bury pasture plants. Some of it is lost and returns to the soil organic pool. Some reappears if there is sufficient rain after the treading event to wash off the soil. Often, after a treading event, since some of the grass is buried, not dead but not eaten, the residual grass in the pasture is higher than expected. To illustrate this, as well as the additional OM return, a function linking the fraction of grass eaten (utilised) or buried to SWC was built. Below FC, 85% of the total grass available to the cows (above 1500 kg DM/ha) was considered eaten and none buried (Fig. 5.7).The remaining 15% of it is deposited on the soil surface. Above FC, an increasing amount of grass gets buried by cows' hooves; therefore pasture utilisation decreases (Fig. 5.7). The fraction of grass deposited on the soil surface is mineralised much slower than the fraction buried after a treading event. Therefore, these two fractions have different effects on soil nutrient cycles and especially on processes like NO_3^- leaching and plant uptake for example (Table 5.6). Moreover, the additional OM returned to the soil affects services including the provision of food, the filtering of nutrients and contaminants, and the regulation of pests and diseases (Table 5.6).

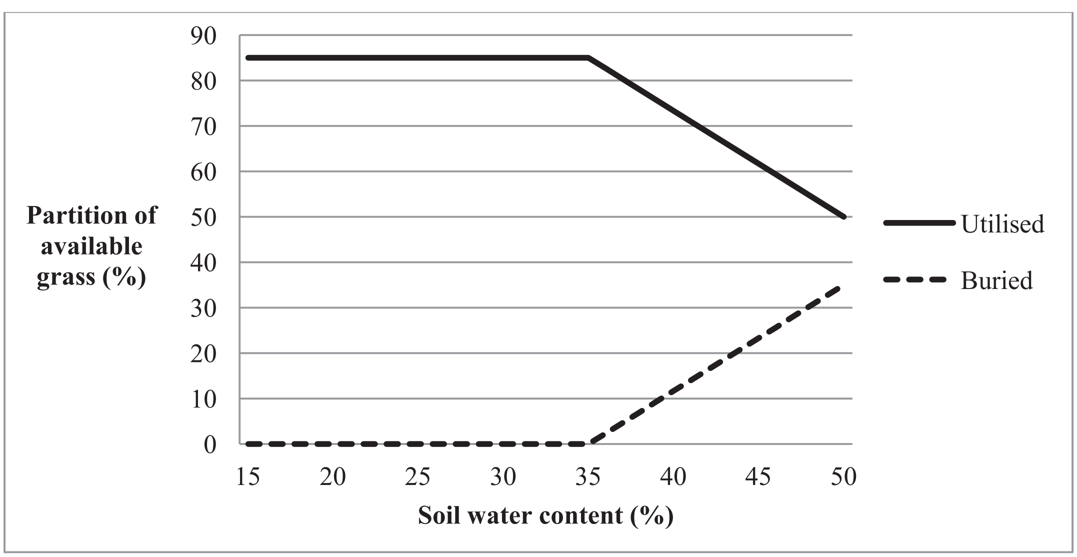

Figure 5-7: Grass utilised and buried according to the volumetric soil water content.

Pande et al. (2000) described the damage and regrowth of pasture after a single, severe cattle treading event during winter. They found that treading damage reduced canopy cover to 43% in dairy pasture, compared to a cover of 90% in undamaged pastures. They concluded that the low spring herbage growth rate following a single, severe winter treading of pasture on wet soil was due mainly to significantly reduced tiller numbers, leaf area index and canopy cover (Pande et al., 2000). They showed a 47% reduction in canopy cover which is comparable to the 35% loss obtained with the function built here.

However, treading not only impacts on pasture utilisation and OM return to soil, it also affects soil structure and especially macroporosity (Mp) and the rate of growth of remaining pasture by impacting on root function. To assess the impact of treading on soil services, routines that describe the effects of treading on soil structure (Mp) and pasture growth were also added (Table 5.6). These additional functions enable to capture of the effects of this degradation process, as well as the benefits of management practices against it, on the dynamics of Mp and pasture growth, and the provision of soil services.

5.3.2 Impacts of cattle treading on soil structure:

Treading, a combination of compaction and deformation on most soils, is a degradation process that impacts on soil structure and associated soil processes. It can be a major issue on intensive livestock systems when soils are wet. In this section, functions are added to SPASMO to describe the impact livestock treading has on soil properties and associated processes.

Other degradation processes mentioned in the conceptual framework (Fig.5.6) are not considered in this study because they are not a major issue for the soil and land use examined in this study. For example, salinisation happens on cropping soils with no vegetation cover and high evaporation, not usually on pastoral soils. Toxification happens on soil where effluents are spread in great quantities like effluent blocks on dairy farms, but these blocks are not considered in this study. Sealing happens on cropping soils with no vegetation cover as a result of rain drops damaging soil aggregates.

Erosion is a major issue in pasture systems in New Zealand hill country. Dymond & Baisden (2010) used an erosion model to calculate losses of particulate organic carbon (POC). They reported that accelerated soil erosion is estimated to export 1.9 (-0.5/+1.0) Mt POC/yr in the North Island of New Zealand and 2.9 (–0.7/+1.5) Mt POC/yr in the South Island. Surface erosion can be triggered following pugging and high rainfall on flat and rolling soils. Pugging destroys soil structure and turns the soil into slurry, very easily mobilised in runoff. In this study, the impact of erosion on nutrient runoff (P, DOC, DON) was considered, but not the amounts of sediments lost, since the farm considered is on flat land (slope<5%).

To model the effects of treading on soil natural capital stocks and soil properties, a number of functions were added to SPASMO, including the effects of livestock hooves on soil Mp depending on SWC, the recovery of Mp after treading and finally, the impacts of Mp changes on the quantity of water that runs off.

5.3.2.1 Macropore loss function:

The physical damage caused by cattle treading on soil structure have several dimensions. Cattle treading causes compaction at low to medium water contents, and pugging (or deformation) at high water contents (Drewry et al., 2008). Compaction and pugging detrimentally affect soil structure and result in surface caps, platy structure, or massive structure (e.g. increased clods) (Drewry et al., 2008). This affects many soil physical properties including porosity, bulk density, aggregate size, stability, penetration resistance and thereby impacts on seedling emergence, root penetration and air and water movement. It also impacts on many biochemical processes, soil biota and plant growth. The incidence of compaction and pugging on pastoral soils is strongly related to soil characteristics, water content, and treading intensity (Drewry et al., 2008), since there is a relationship between soil consistency, water content and the risk of compaction and pugging damage (Fig. 5.8).

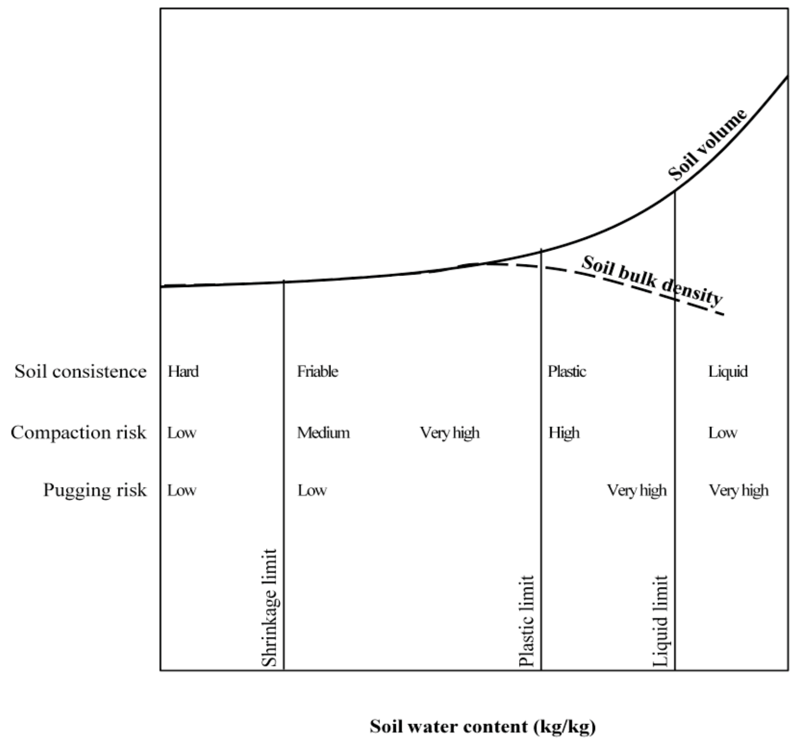

Figure 5-8: Relationship between soil consistency and gravimetric water content (from Drewry et al., 2008) (shape will vary between soil types).

Compaction specifically refers to a reduction in soil porosity. It is due to the application of pressure at the soil surface resulting in the collapse of soil aggregates and the closure of pores. Compaction is described as the compression of an unsaturated soil body resulting in a reduction of the fractional air volume (Hillel, 1980). Compaction decreases the volume of the large inter-aggregate pores (macropores) and occurs more easily when soil is moist (Houlbrooke et al., 2009). Compaction can occur even when the soil is under FC. Maximum compression occurs close to the plastic limit (PL). As a rule of thumb, soil scientists recognise that the maximum bulk density (hence maximum compaction) corresponds to approximately 80% of saturation (Hillel, 1980), but this indicator is not very sensitive to soil type differences.

The plastic limit (PL) of a soil is the gravimetric water content at which the soil changes from being friable to being plastic (Drewry et al., 2008). For some soils the FC is close to the PL. If FC is lower than PL, treading close to FC should be avoided as the soil is likely to be compacted (Drewry et al., 2008).

Pugging is the deformation of topsoil. Soil pugging involves the deformation and remoulding of soil (Houlbrooke et al., 2009). In grazing systems, pugging occurs when the animals' hooves penetrate the topsoil deeply and deform it, at and above the PL (Drewry et al., 2008; Houlbrooke et al., 2009). When the soil is very wet, above FC and especially around PL, macropores are full of water and resist compression, therefore the soil deforms. When pugging occurs pasture plants are damaged, buried or uprooted (Betteridge et al., 2003) because of the soil deformation. Immediately beneath the depth where the hoofs penetration stops, compaction can occur. Therefore, above PL the topsoil deforms and compaction occurs at the base of the hoof (Betteridge et al., 1999). Damage to the soil structure by cattle treading are reversible. Above plastic limit (PL) and around liquid limit (LL), the soil loses bearing strength so differences between soils disappear.

SWC is often used as an indicator to monitor treading. Generally grazing should be avoided near and beyond PL or FC, whichever is less (Drewry et al., 2008) to avoid damage from treading. Table 5.7 presents the gravimetric soil moisture for the two soils used in this study.

Table 5-7: Gravimetric soil moisture (0-10cm) for Horotiu and Te Kowhai (Singleton and Addison, 1999).

Soil type	Field capacity	Plastic limit	Liquid limit
Te kowhai silt loam	54	34	80
Horotiu silt loam	53	57	74

Macroporosity has been mentioned by several authors as a good indicator of soil physical health (Drewry et al., 2008). Macroporosity is very sensitive to structural damage therefore it is useful in assessing the effects of livestock treading on soil properties (Drewry et al., 2004; Singleton and Addison, 1999). Some soil properties are more affected by treading than others. Singleton and Addison (1999) argued that the distribution of pore size is more affected by treading than the volume of pores. Consequently, "measurements related to continuity of pores, such as hydraulic conductivity, and macroporosity, are more affected by treading than were bulk density or total porosity (which are independent of pore continuity)" (Singleton and Addison, 1999, p. 898). Macroporosity is also considered a good indicator of yield with values <10-12% used as a critical level to indicate increasing limiting conditions for plant health and soil aeration (Drewry et al., 2004; Gradwell, 1965; Singleton and Addison, 1999).

Mp can be used as a parameter to assist in the quantification of a number of services including the provision of food and support, flood mitigation and the regulation of pests and diseases (Table 5.6). To take into account the effect of treading on Mp, a function based on SWC and treading intensity was added to the model to calculate the decrease in Mp after a treading event. The differences in the sensitivity of soil types to treading is reflected by SWC and thresholds built around the FC of the soils. Since FC is soil-type specific, the function is adapted for each soil type. The effects of pugging on soil structure, like surface sealing, are not included in this function. However the impacts of pugging on plant growth are taken into account through another function, detailed later (section 5.3.3).

To build the macropore loss function, field data (Keith Betteridge, pers. Com., 2010) was used (Fig. 5.9).

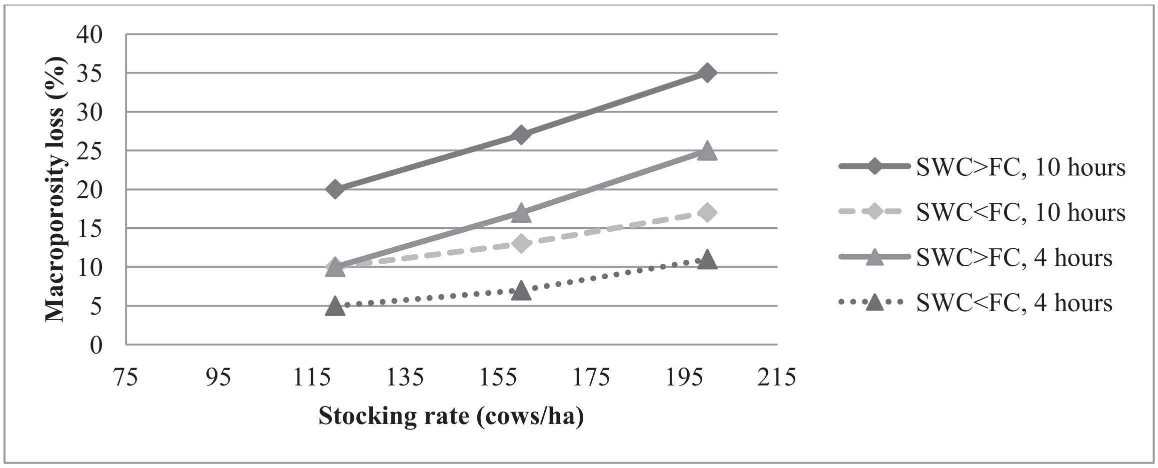

Figure 5-9: Data used to build the macropore loss function (Keith Betteridge, pers. com., 2010). Data were gathered across a range of soil types. Macroporosity loss is a percentage of initial macroporosity. SWC: soil water content, FC: field capacity. 10 or 4 hours: grazing time.

The "macropore loss function" captures the reduction in macroporosity for the top 10 cm (0-100 mm) of the soil profile only. This depth has been chosen because it has been demonstrated that at this depth the maximum difference in mean soil properties occurred (Drewry et al., 2001; Singleton and Addison, 1999; Singleton et al., 2000). Singleton et al. (2000, p.559) argued that "the 0-10 cm depth was best for showing differences between treading regimes". Singleton and Addison (1999) showed that the depth showing most significant differences between treatments in bulk density and total porosity was 5-10 cm. Moreover, the zone of major hoof compaction is known to be situated at 10-15 cm (Drewry, 2006; Singleton et al., 2000). Drewry (2006) argued that the deterioration of macropores commonly occurs at 5–10 or 10–15 cm depth under cattle treading (Drewry et al., 2004). Moreover, changes in soil properties in the first 10cm impact the most on plant processes as this contains the majority of the plant root zone. Macropore deterioration beneath 10 cm may also be smaller because it is beyond the zone of major hoof compaction (Drewry, 2006; Drewry et al., 2003; Singleton and Addison, 1999).

The macropore loss function described below is the same for all treading events. The function includes different components. First, from the data, a function that calculates the maximum macropore loss (MpLmax) depending on stocking rate (SR) and grazing time (GT) was built. Then, the sensitivity of the soil to treading depending on SWC was adjusted.

For any given SR/GT combination the maximum macropore loss function (MpLmax) calculates the macropore loss at the most critical water content that is around FC. The function was built by linear regression from the data presented in Fig. 5.9 (Eq.5.1).

MpLmax = -19.3 + 0.19 * SR + 1.67 * GT [Eq.5.1]

Where SR is the stocking rate in animals per hectare, GT is the grazing time in hours per grazing event and MpLmax is the maximum loss of macroporosity around FC in %.

The values used to build the function may be low compared to the literature (Drewry et al., 2008; Drewry et al., 2002). These values can be adjusted when more data is available. Moreover, these values are not soil specific, but the function is later adjusted to gravimetric SWC which is soil type specific. Moreover, the loss is in % of the initial macroporosity, which is also a soil type specific measure.

The second adjustment to the Mp loss function was the influence of SWC on the sensitivity of the soil to damage. Soil texture and structure impact on soil ability to store water and therefore on SWC. SWC was used to inform soil structural sensitivity to treading damage. A series of SWC at different potentials were used as thresholds to illustrate the soil sensitivity to treading damage. These thresholds and the corresponding SWC for both of the soils studied are presented in Table 5.8. These values were calculated from van Genuchten water retention curves (van Genuchten, 1980) specific to each soil type chosen for this study.

Table 5-8: Thresholds of soil sensitivity to treading damage (mm) by gravimetric (GSWC) and volumetric (VSWC) soil water content.

SWC	Water potential (bars)	Te Kowhai GSWC 0-10 cm	Te Kowhai VSWC 0-10 cm	Horotiu GSWC 0-10 cm	Horotiu VSWC 0-10 cm
Saturation	0	60	66	64	53.8
SP_2	0.05	57	62.7	57	47.8
FC	0.1	54	59.4	53	44.5
SP_1	0.2	51	56.1	48	40.3
SP	1	41	45.1	38	31.9
WP	15	28	30.8	25	21
AWC	FC-WP	26	28.6	28	23.5

Note: SP: stress point, FC: field capacity, WP: wilting point, AWC: Available water capacity.

According to the literature, compaction can occur under FC and is maximal above the PL (Drewry et al., 2008). Two thresholds were chosen, SP_1 and SP_2, around FC, in between which macropore loss is maximal (Eq.5.4). SP_1 is the SWC at twice the potential of FC (Table 5.9). SP_2 is an intermediate value between FC and Saturation. Below SP, when the soil is dry, macropore loss is minimal and approximately equals 45% (Fig. 5.10) of the maximal macropore loss (MpLmax) (Eq.5.2). Above SP_2, the soil is very wet; macropores are full of water and resist compression therefore the soil deforms more than compacts. Macropore loss is still important but decreases the wetter the soil gets. A function was built (Eq.5.5) which calculates the macropore loss as a function of MpLmax and the relative SWC between SP_2 and saturation (Sat): the more SWC is above SP_2, the fewer macropores are lost. To illustrate the increasing vulnerability to treading of soils with increasing SWC, a threshold, SP (Fig. 5.10), was chosen above which macropore loss increases to a maximum, MpLmax, at SP_1. The macropore loss above SP depends on MpLmax and the relative SWC between SP and SP_1 (Eq.5.3).

The different parts making up the macropore loss function are detailed below (Fig. 5.10):

If SWC<SP	Mp Loss = 0.45*MpLmax	[Eq.5.2]
If SP<SWC<SP_1	Mp Loss = 0.45*MpLmax + 0.55*MpLmax*(SWC-SP)/(SP_1-SP)	[Eq.5.3]
If SP_1<SWC<SP_2	Mp Loss = MpLmax	[Eq.5.4]
If SP_2<SWC<Sat	Mp Loss = MpLmax – 0.15*MpLmax* (SWC-SP_2)/(Sat-SP_2)	[Eq.5.5]

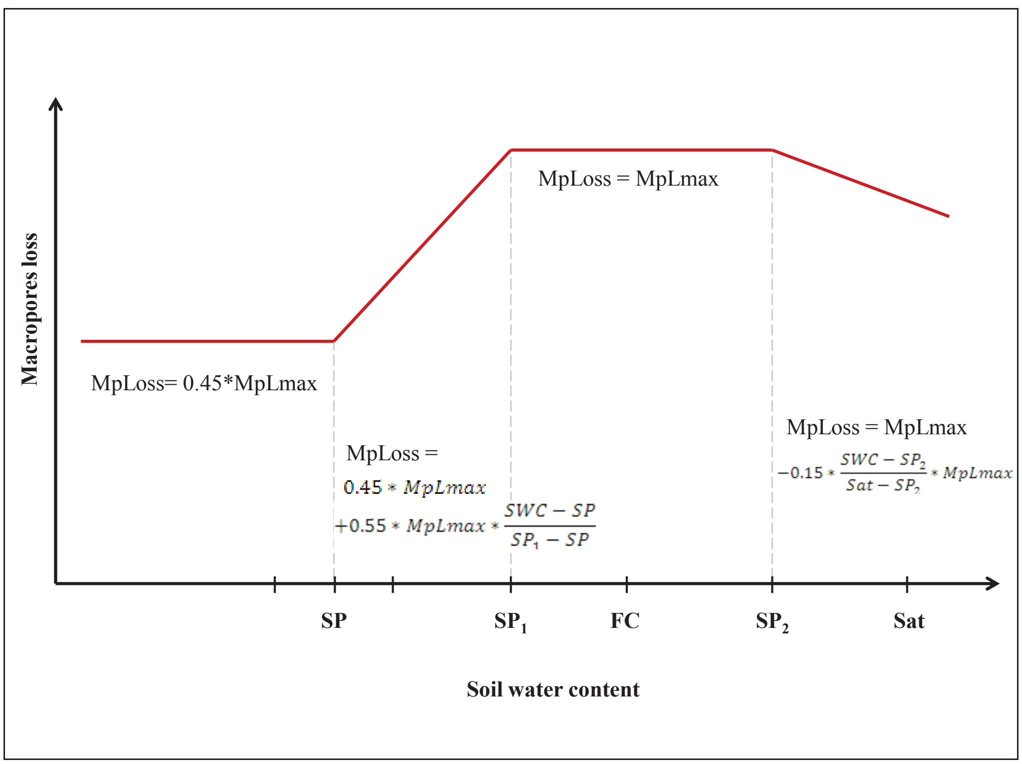

Figure 5-10: Function calculating the macropore loss after a treading event as a function of soil water content.

Outputs of the function, when regressed against the data (Keith Betteridge, pers. com., 2010) (Fig. 5.9), showed a good fit of the model to the data (Fig. 5.11).

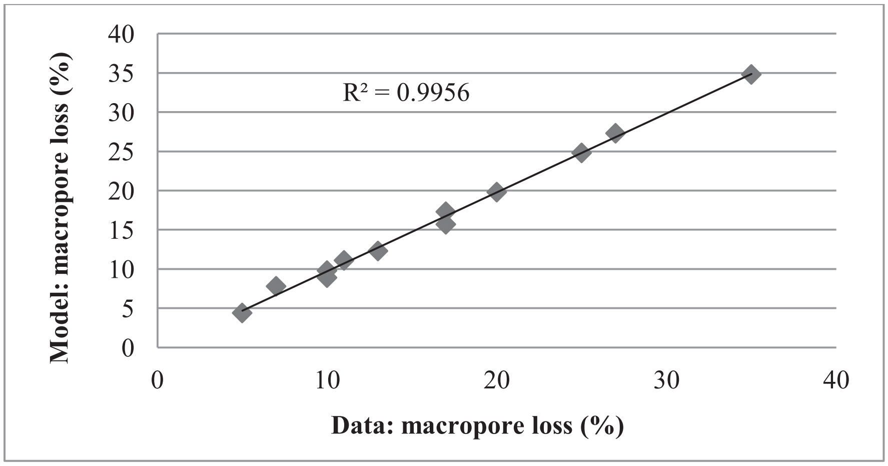

Figure 5-11: Macropore loss (%): data versus simulated model outputs.

For each grazing event, the model calculates the Mploss from SR, GT and SWC with the function described above and applies it to the current soil Mp. After the grazing event, Mp recovers; therefore another function was added to SPASMO to take the recovery process into account.

5.3.2.2 Macroporosity recovery after treading:

After a treading event, deteriorated soil physical conditions have been shown (Drewry, 2006; Drewry et al., 2008) to recover as a consequence of soil fauna (earthworm) activity, plant root growth, freezing and thawing, and wetting and drying cycles (Drewry, 2006). Recovery of soil macroporosity in the short-term under pastoral farming is likely to be limited to 10 cm or at most 15 cm depth (Drewry, 2006). Soil structure has been shown to recover from compaction faster in late spring, summer and autumn than winter. Possible mechanisms explaining this are the drying cycles in summer causing shrinking and cracking, thereby breaking down any compacted layer. Increased plant rooting activity and associated biological activity, with the warmer temperatures, would also have an impact on soil structure (Drewry, 2006; Drewry et al., 2008). The magnitude of recovery for soils prone to damage is of practical interest to farmers and researchers. Farmers may justify using stand-off pads to allow the soil structure and pasture to recover (Drewry et al., 2008).

In this study, the macropore loss was limited to the top 10 cm of the soil profile. Therefore, the function added to take into account the natural recovery of soil physical conditions was also limited to the first 10 cm of the soil profile; this aligns with the literature (Drewry, 2006). In the absence of data sets, a constant recovery rate of macroporosity was used through the year. In the field, depending on when the damage occurred, macroporosity will recover more or less quickly. Damage made just before summer is usually recovered quicker than damage made earlier in the year (Drewry, 2006; Drewry et al., 2008).

After each grazing event, the soil macroporosity recovers up to 95% in 1 year. It recovers to 90% in 3 months and then takes the rest of the year to recover up to around 95%. This rate of recovery is comparable to data found in the literature. For example, Drewry and Paton (2006) found recovery rates between 47% and 127% in the first 4 months after a treading event, for the top 10cm of a silt loam (Table 5.9).

Table 5-9: References for soil macroporosity recovery (Drewry, 2006; Drewry and Paton, 2000; Drewry et al., 2004).

Reference	Soil	Depth	Improvement	Post damage interval
Drewry and Paton, 2000	Silt loam	0-5 cm	88%	4 months
Drewry and Paton, 2000	Silt loam	5-10 cm	47%	4 months
Drewry and Paton, 2000	Silt loam	0-5 cm	76%	4 months
Drewry and Paton, 2000	Silt loam	5-10 cm	127%	4 months
Drewry et al., 2004	Silt loam	0-5 cm	44%	5 months
Drewry et al., 2004	Silt loam	5-10 cm	26%	5 months

The function used to model the recovery of Mp after treading is:

$$Mp^* = Mp - Mploss + Mploss * (1- e^{-\Delta t/\alpha}) \qquad [Eq.5.6]$$

where Mp* is the macroporosity after treading, Mp is the macroporosity before treading, Mploss is the macroporosity loss calculated with the Mploss function (Eq.5.1 to 5.5), Δt is the number of days since treading and α is the mean lifetime[5].

Figure 5.12 shows an example of Mp recovery after a treading event that decreased Mp from 12% to 4% (Mploss = 8).

[5] Here α=120. It should be noted that $\alpha = t_{1/2}/\ln 2$ where $t_{1/2}$ is the half life, which means that Mp recovers 50% after 83 days (=120*ln2). Mp recovers around 96.8% of its original value in 365 days.

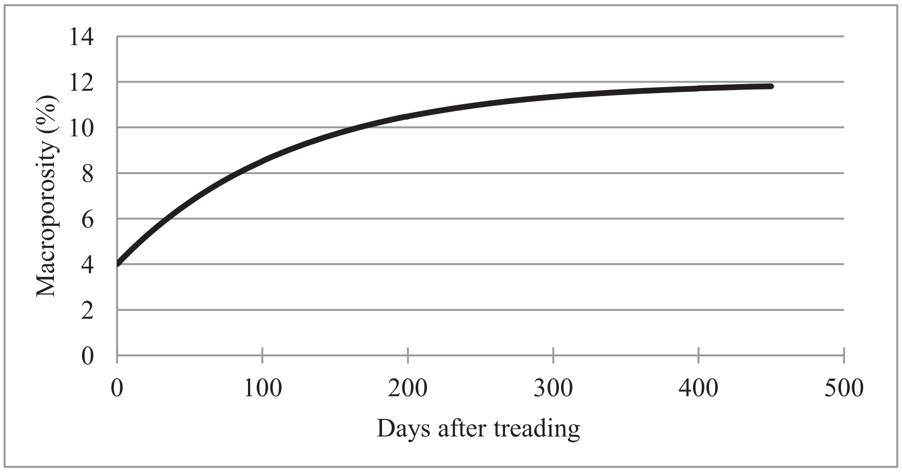

Figure 5-12: Example of model output: Macroporosity recovery after a treading event that decreased macroporosity from 12% to 4%.

This function enables us to follow Mp dynamics and how they affect the provision of ecosystem services from soils.

5.3.2.3 Effects of cattle treading on runoff:

By reducing macroporosity, treading affects the circulation of water and gases through the soil profile. In order to take into account the impacts of reduced macroporosity on the soil water cycle, the repercussion of a macroporosity loss on the different processes of the water cycle were considered and included in the model.

Macropores (>30 µm) are drained at FC and transmit water during infiltration. The bigger macropores (>1 mm) transmit water freely when the soil is saturated (Marshall, 1996). By reducing macroporosity, treading impacts on soil saturation capacity and infiltration and thereby on runoff. After a treading event, when macropores have been lost, the soil is able to store less water and reaches saturation quicker. Runoff therefore starts earlier. In the model, soil saturation capacity is adjusted to reflect the loss of water storage capacity with a macroporosity change as a result of a treading event. Macropore loss also affects infiltration rates. Fewer macropores means slower infiltration and slower drainage, therefore more runoff.
To take into account these effects, a function which modifies the surface runoff routine of SPASMO according to the loss of macroporosity was added to the model.
The surface runoff routine of SPASMO is based on a daily rainfall total. The calculation uses the Soil Conservation Service (SCS) curve number (CN) approach (Williams, 1991). The

curve number approach was selected for SPASMO because: (i) it is based on over 30 years of runoff studies on pasture, arable and forest sites in the USA, (ii) it is computationally simple and efficient, (iii) the required inputs are available, (iv) and the calculation relates runoff to soil type, land use and management practice (Green and Clothier, 2009).

Surface runoff is predicted from daily rainfall plus irrigation, using the SCS curve number equation:

$$Q = (R - 0.2S)^2 , \qquad R > 0.2S$$
$$Q = 0, \qquad R \leq 0.2S \qquad \text{[Eq. 5.7]}$$

where Q [mm] is the daily runoff, R [mm] is the daily rainfall plus irrigation, and S [mm] is the retention parameter that reflects variations among soils, land use and management.

The retention parameter, S, is related to the curve number, CN, using the SCS equation (SCS, 1972), where the constant, 254, gives S in millimetres.

$$S = 254\ ((100/CN)\text{-}1) \qquad \text{[Eq. 5.8]}$$

Curve numbers can be obtained easily for any area of land use type from the SCS Hydrology Handbook (SCS, 1972). An example of CN numbers is given below for a range of pasture and drainage conditions (Table 5.10).

Table 5-10: Soil Conservation Service curve number for a grazed pasture (SCS, 1972).

SCS CN number		**Drainage Condition**			
		Excessive	Good	Fair	Poor
Pasture Condition	Good	39	61	74	80
	Average	49	69	79	84
	Poor	68	79	86	89

A pasture in good condition (well established and developed) that is growing on a free draining soil will have a low CN value (39), while a pasture in poor condition and on a poorly drained soil will have a high CN value (89). A lower CN value implies a bigger retention parameter, S, and so a given soil/pasture combination which yields less runoff for the same daily rainfall total. The SCS runoff calculation also includes an additional adjustment to S, to express the effect of slope and soil water content (Green and Clothier, 2009; Williams, 1991).

For the purpose of this study, the pasture slope is assumed to be always <5% and the pasture sward is assumed to be in good condition. It is noted that this method doesn't take into account the influence of hydrophobicity on runoff (Müller et al., 2010).

A function that modifies CN according to macropore loss was added to the model in order to take into account the impacts of Mp loss on runoff. With treading the macroporosity decreases from Mp to Mp*. If treading is intense macroporosity can decrease up to Mp_{FC}, which is the macroporosity corresponding to FC, an inherent soil property (Fig.5.6). The impact on runoff of macropore loss from treading is accounted for by equation 5.9 below, which modifies the curve number CN:

$$CN^* = CN + 20 * [1- [(Mp^* - Mp_{FC})/(Mp - Mp_{FC})]] \qquad \text{[Eq.5.9]}$$

where CN* is the curve number after treading, CN is the curve number before treading, Mp is the macroporosity before treading, Mp* is the macroporosity after treading and Mp_{FC} is the macroporosity at FC. The constant 20 is the difference between the curve numbers for good and poor drainage conditions (79-61=18), for a pasture in good condition (Table 5.10).
This function allows us to look at the impacts of treading, and treading management on the amount of water running off and thereby on a number of services including flood mitigation and the filtering of nutrients and contaminants (Table 5.6).

5.3.3 Impacts of treading on pasture growth:

Treading not only damages soil structure, it also damages pasture plant numbers and integrity directly, decreasing pasture yield, which impacts on the provision of food (Table 5.6). Moreover, a change in macroporosity can lead indirectly to a reduction in pasture yield especially if macroporosity falls below a critical level where soil physical health is affected, which is considered to be around 10% (Drewry, 2006; Drewry et al., 2008).

A reduction in dry matter production following treading may be due to direct damage to pasture plants, which include both plant damage through hoof action and plant burial in mud, uprooting, crushing and bruising (Drewry et al., 2008) accompanied by disruption of soil aggregates (Menneer et al. 2005; Zegwaard 2006). Greater hoof activity with high stocking rates in semi-arid rangelands directly damages plants and pulverises the soil surface (Greene et al. 1994). Direct damage is often visible to land managers. By reducing macroporosity through compaction, cattle treading also indirectly affects plant growth. It is difficult to separate the effects on pasture production of direct plant damage and pugging from the indirect effect of changes in soil physical properties (Drewry et al., 2008). These effects on plant number and growth however are not permanent and like soil structure, pastures recover after a treading event, if there is no further damage.

To take into account the effects of cattle treading on plant growth and thereby assess the effects of treading on the provision of food (Table 5.6), two functions were added to the SPASMO model: one to calculate the impact of treading on pasture growth potential (expressed in % of maximal pasture growth), and one to predict the recovery of the pasture growth potential after treading.

5.3.3.1 Loss of pasture growth potential:

To predict the reduction in potential pasture growth due to treading, a function was built from data gathered by Betteridge et al. (2003) on treading damage, on wet soils (Fig. 5.13). In that study of Betteridge et al. (2003), the loss of pasture production is linked to stocking rate (SR) and grazing time (GT).

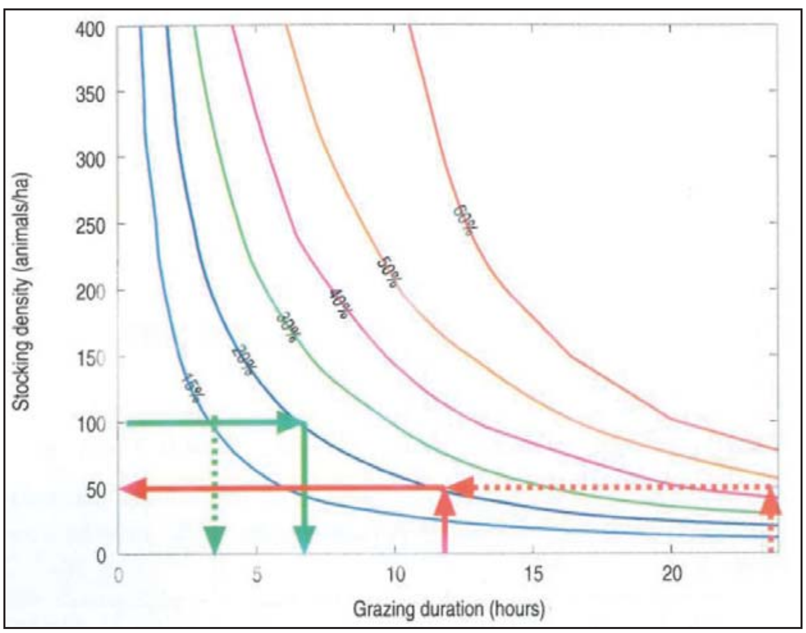

Figure 5-13: Decision tool to determine the loss in pasture yield potential (%) from stocking rate (SR) and grazing time (GT) for a range of soil orders (Betteridge et al., 2003,p. 27)

To build the function predicting the reduction of pasture growth potential with treading, a treading intensity indicator (TI) was built and plotted against the reduction of pasture growth data.

TI = (SR * GT) / 100 [Eq. 5.10]

where TI is the treading intensity, SR is the stocking rate in cows/ha and GT is the grazing time in hours / grazing event.

Two conditions were added to the data: (i) when there are no cows on the paddock, the pasture growth potential is 100%; (ii) the maximum pasture growth potential loss is 60%. A function was then fitted to the data. Figure 5.14 presents the data used and the function fitted.

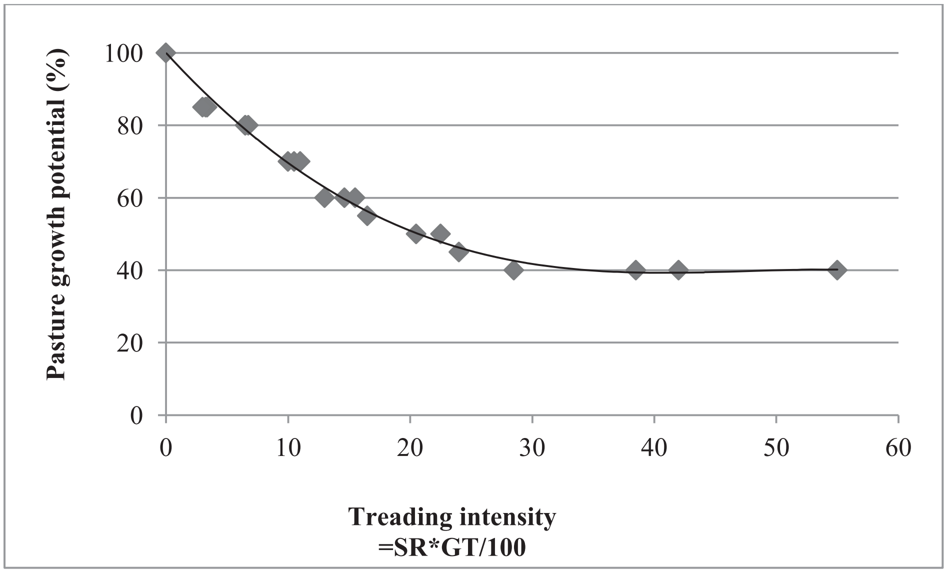

Figure 5-14: Loss of pasture growth potential depending on treading intensity (TI): Data and model fit. SR: stocking rate (cows/ha), GT: grazing time (h).

The pasture growth potential is calculated from TI using the equation:

$$P_p = -3.05 * 10^{-6} * TI^4 - 1.68 * 10^{-4} * TI^3 + 6.55 * 10^{-2} * TI^2 - 3.67 * TI + 100 \qquad \text{[Eq.5.11]}$$

Where Pp is the pasture growth potential (%) and TI is the treading intensity indicator, with TI calculated from Eq. 5.10.

This function (Eq. 5.11) calculates pasture growth potential (in %) and how it decreases due to the treading of grass by cattle during a grazing event. Therefore, the loss of pasture growth potential (LPp) is 1-Pp=LPp. However, this function is valid only when the soil is wet, that is SWC>FC (Betteridge et al., 2003). Below FC, the damage done to pasture plants is much more limited; therefore, to take this into account, a function was built to reflect the sensitivity of pasture growth rate to treading damage depending on SWC (Fig. 5.15).

5.3.3.2 Relative treading damage on pasture:

To reflect the sensitivity of pasture plants to treading damage, SWC needed to be considered. Animal grazing and treading have different impacts on pasture plants depending on SWC:

when the soil is dry, crushing and bruising of the plants occurs, but when the soil is wet and soil structure gets disrupted, pasture plants get buried in mud or even uprooted (Drewry et al., 2008; Menneer et al., 2005). This damage leads to different degrees of reduction of the pasture growth potential.

Therefore, to take into account the sensitivity of pasture to treading damage, the actual loss of pasture growth, ALp, is calculated from SWC and LPp using the relative function below:

If SWC < WP, ALp = LPp * 0.1 [Eq.5.12]

If WP < SWC < FC, ALp = LPp * (0.1 + 0.9 * [(SWC-WP)/(FC-WP)]^2) [Eq.5.13]

If SWC > FC, ALp = LPp [Eq.5.14]

where LPp is the loss of pasture growth potential calculated from Eq. 5.11, SWC is the soil water content at the time of the grazing event and WP is the soil wilting point.

The actual loss of pasture growth, ALp, thereby depends on SR, GT and SWC. Figure 5.15 presents ALp and how it changes with SWC.

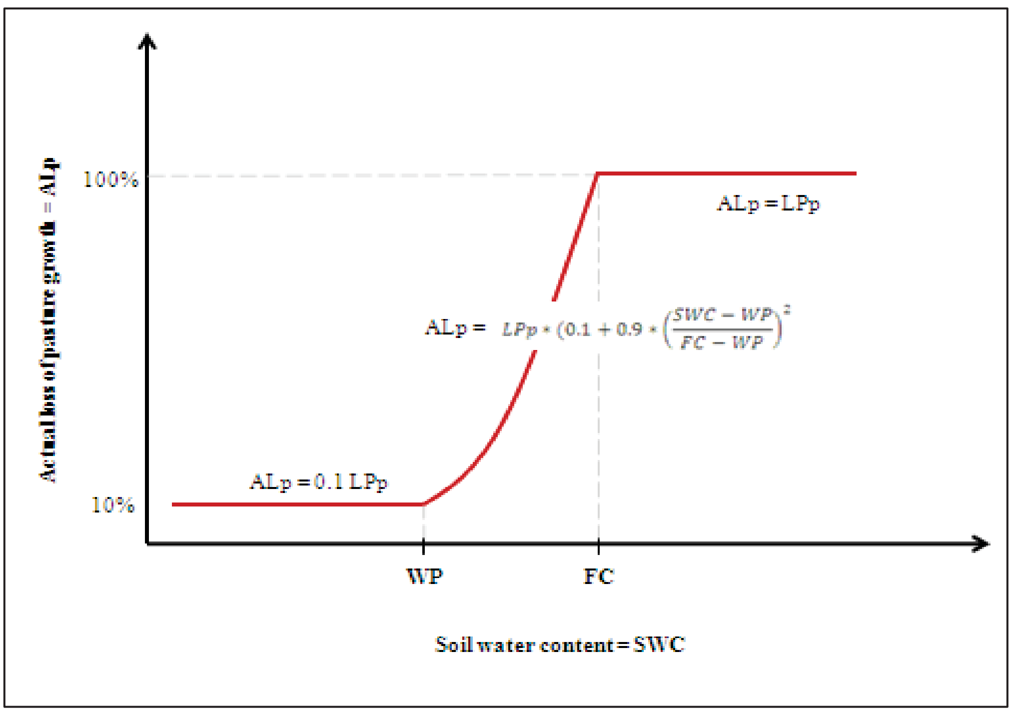

Figure 5-15: Actual loss of pasture growth function, ALp

5.3.3.3 Recovery of pasture growth potential after a grazing event:

After a treading event, pasture growth is slowed down but it has been shown to recover in between grazing events (Betteridge et al., 2003; Betteridge et al., 2002; Pande et al., 2000). In order to take this process into account and quantify its impact on yield and the provision of food (Table 5.6), a function based on the data of Betteridge et al. (2002) and Betteridge et al.

(2003) was built. The actual loss of pasture growth, ALp, is calculated from equations 5.10 to 5.14 mentioned above. ALp stays constant for 3 weeks and then decreases back to nil in around 8 weeks (Betteridge et al., 2003, p.8). Figure 5.16 presents an example of recovery with an actual loss of pasture growth potential (ALp) of 50%.

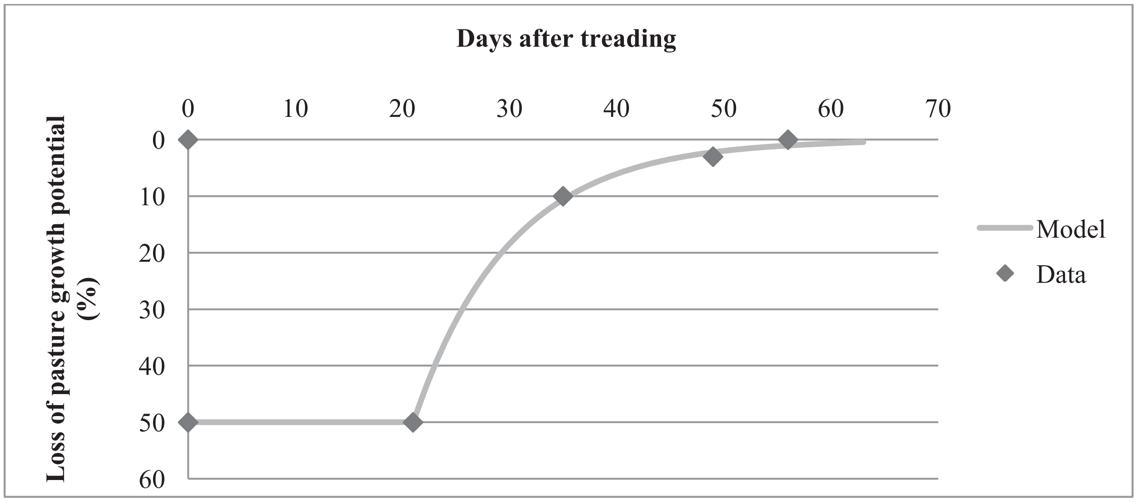

Figure 5-16: Actual loss of pasture growth (ALp) and recovery depending on the number of days after treading. As an example, the figure presents a loss of 50% of pasture growth potential after a treading event and the recovery back to full potential in around 65 days.

The recovery of the pasture growth potential is calculated from the actual loss of pasture growth, ALp, after treading and the time since the treading event using the equation:

$$ALp^* = ALp * e^{-(\Delta t - nR)/\alpha} \qquad [Eq.5.15]$$

Where Δt is the number of days since treading, ALp is the actual loss of pasture growth just after treading, ALp* is the residual loss of pasture growth Δt days after treading, nR is the non-recovery period (here 21 days) and α is the mean life time (here α =10).

The routines used to model pasture utilisation (section 5.3.1) treading damage on pasture (section 5.3.3.2) and pasture recovery (section 5.3.3.3) can be compared to the model of Finlayson et al. (2002). They used gravimetric soil water content, pasture mass, stock number, animal live weight and the duration of grazing to predict the effect of treading on pasture damage and its recovery. The functions used here in comparison are very simple, whereas the model developed by Finlayson et al. (2002) was more complex.

The addition of the different functions detailed above enables us to efficiently model the processes behind treading in a dairy grazed pastoral soil and provides the tools necessary to examine in detail the effects of management practices on soil processes and the provision of ecosystem services from soils (Table 5.6).

To be able to assess the influence of management practices, SR and GT, on natural capital stocks and the provision of ecosystem services, details about the use of standoff-pads and stock rotation (SR and GT) were also added to the model. They are presented in the following sections.

5.3.4 Rotation and stocking rate:

Grazing systems in temperate areas, such as New Zealand, are predominantly pasture-based, with rotational grazing common practice. A rotational dairy grazing system, e.g. with 14–30 days between grazing of the same paddock, is common on New Zealand dairy farms from September to May (i.e. during lactation) (Drewry et al., 2008). For this study, the original grazing routine of the model was modified to fit a typical dairy farm rotation better. Originally in SPASMO, each paddock was 10 ha, the cows stayed on all day minus 4 hours in the milking shed and the time to walk between the paddock and the milking shed.

After modifications, the grazing routine follows the following principles:

- Paddocks are 5 ha each. Half of each paddock is grazed at once (2.5 ha) after each milking for 10 hours. The original size of the paddocks was 10 ha but it was changed to 2.5 ha in order to obtain higher stocking rates, typical of the number of cows on a dairy farm. This provides a realistic level of treading damage on soil structure and pasture. With 33 cows/ha (330 cows on 10ha paddocks), the decrease in pasture growth potential and macroporosity are not representative of reality. The original stocking rate on 10ha would have underestimated the effects of treading. In New Zealand in winter, dry cows are grazed on paddocks divided in blocks or strips and stocking rate can be as high as 300-600 cows/ha (Drewry et al., 2008) for a few hours.
- The paddock where the pasture is the longest is chosen to be grazed after each milking,
- The animals don't return to a paddock if it has been grazed less than 10 days before.
- The cows grazed the paddock when the pasture is >2000 kg DM/ha.
- The cows stay on a paddock until grazed down to 1500 kg DM/ha or until they have consumed what they need (around 20 kg DM/day per animal). If they didn't have enough to eat, they are fed supplements on the same day while milked or on a standoff-pad if available.
- The paddocks not grazed are locked and cut for pasture silage when they reach 3000 kg DM/ha.

5.3.5 Use of a standoff-pad:

To take into account the fact that some famers choose to use stand-off pads or feed-pads to remove animals from wet soils to avoid soil structure and pasture damage (Drewry et al., 2008), a function was added to the model which determines the amount of time per day the animals spend on the pasture depending on SWC and the availability of a standoff-pad.

If there is no standoff-pad on the farm, the animals spend 10h on a 2.5 ha paddock twice a day, after each milking.

If there is a standoff-pad on the farm, the cows are out on the standoff -pad depending on SWC:

When SWC < FC, the cows come on the paddock normally, 10 hours, twice a day.

When SWC > FC, the cows come on the paddock for 4 hours after each milking (8h a day) and otherwise stay on the standoff-pad (16h a day), where they are fed supplements (pasture silage).

FC was the threshold chosen to take the cows off the paddock because compaction was shown to be maximal around PL which for most soil types is around FC.

This function thereby enables us to assess the effects of a standoff-pad on natural capital stocks like Mp, but also on processes like pasture growth and thereby on the provision of all services (Table 5.6).

5.3.6 Calculation of P runoff:

The P cycle is closely linked to the provision of two services: the provision of food, through the supply of the P to plants, and the filtering of nutrients (Table 5.6). To measure these services, the dynamics of the amount of P in solution available to plants needs to be followed, as well as the amount of P in runoff every year. SPASMO already includes a P sorption function to calculate P leaching using a Langmuir adsorption-isotherm (Appendix B). However, a P runoff routine needed to be added to the model.

P runoff is one of the processes to consider when investigating the filtering of nutrients and contaminants by soils (Table 5.6). P loss occurs largely via surface runoff. P is lost in two forms, as soil-bound P with sediments and as dissolved-P, with the former often the dominant form (60-90%) (Parfitt et al., 2009). The quantities of P lost are generally small and a significant proportion of the P lost every year can occur during single-storm events (Parfitt et al., 2009).

P runoff wasn't previously included in the model; therefore a routine was added to calculate P runoff based on storm events and erosion (Steve Green, pers. com.). The modified USLE (universal soil loss equation) (Wischmeier and Smith, 1978) was used to calculate the amount of sediment extracted by storm events and, then, calculate the amount of P lost from these sediments.

The USLE uses five major factors (rainfall pattern, soil type, topography, crop system, and management practices) for computing the expected average annual erosion in the following equation (Kouli et al., 2009):

$A = R * K * L * S * C * P$ [Eq. 5.18]

where A is the average soil loss (t/ha/year), R the rainfall-runoff erosivity factor [MJ mm/(ha h year-1)], K the soil erodibility factor [t ha h/(ha MJ mm)], L the slope length factor, S the slope steepness factor, C the cover management factor, and P the conservation support practice factor. L, S, C, and P are all dimensionless (%).

The R-factor is defined as the mean annual sum of individual storm erosion index values, EI30, where E is the total storm kinetic energy and I30 is the maximum 30 min rainfall intensity. The R-factor can be interpreted as a measure of rainfall erosivity. EI30 is related to the event or daily rainfall amount, RF, by a power function of the form (Yu and Rosewell, 1996):

$EI30 = aRF^b$ [Eq. 5.19]

Where the parameters a and b can be estimated using regression analysis (Fig. 5.17).

Data were gathered, including storm events (hourly rainfall data), the total rainfall per storm (RF), and the maximum intensity of the rainfall (I30) during the storm event to calculate E and I30 (Fig. 5.17).

Storms generate P runoff only if the total rainfall >12mm. Figure 5.19 shows the function fitted to the data, providing the a and b parameters: $EI30 = 0.075*RF^{1.769}$.

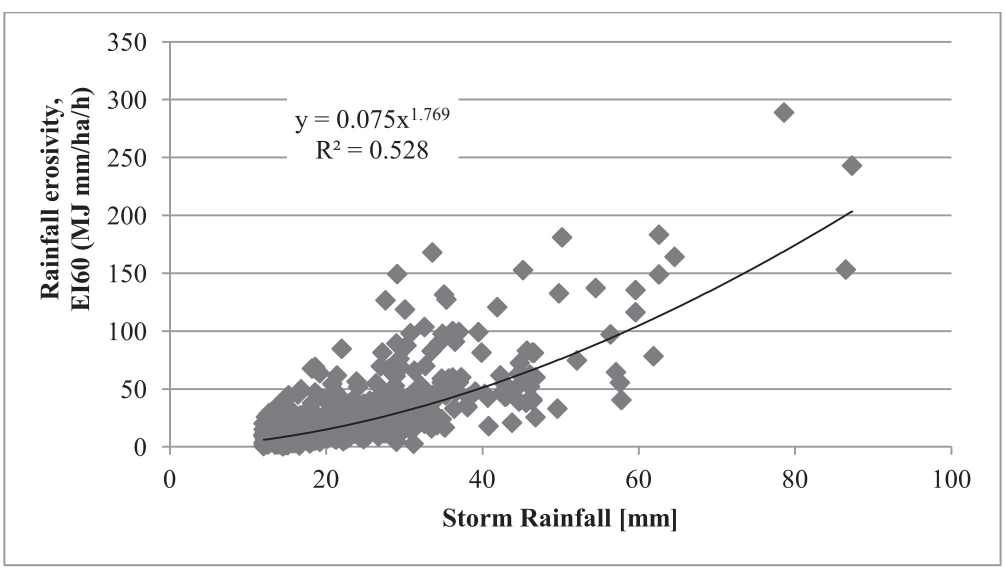

Figure 5-17: Relationship between daily rainfall amount (mm) and rainfall erosivity.

To calculate the K factor, the geometric mean particle size was used for each particle size class (clay, silt, sand) (Kouli et al., 2009). L and S (5%) are set parameters in the model. The P factor was fixed at 1. Different values were used for the C factor, depending on the state of the pasture. The value C=0.025 (2.5%) was used when the pasture was intact, and C=0.1 (10%) when the pasture was damaged after treading. This simple relationship takes into account the fact that treading and pugging increase erodibility and thereby P runoff.
From all these factors, the amount of sediment lost (kg/year) was calculated. Then, a nutrient extraction factor was used to calculate the amount of P lost to runoff water from the sediment. The factor takes the empirical values 2.5% of the organic P of the soil when the soil is not pugged and 10% when pugged.

Adding this routine enables the model to output P runoff (in kg/ha/year) so it can be used as a parameter to measure the filtering of nutrients (Table 5.6). Moreover, linking this routine to the intensity of treading enables us to look at the impacts of management practices on natural capital stocks (P sorbed, P in solution) and the process of P runoff.

5.3.7 Simulation of extreme N and P losses:

To isolate leaching losses due to the soil nutrient retention capacity from inevitable losses from plant turnover and mineralisation, the functions describing the links between N and P losses and cation and anion retention capacities (ASC) were modified in the SPASMO model. This enabled us to determine the potential maximum N loss from each soil by simulating a soil with no nutrient retention capacity. The same thing was done to determine the potential maximum P losses.
In the model the Langmuir adsorption-isotherms driving N and P retention were successively modified to simulate a soil with extremely low ASC, close to zero. All other variables were kept equal. The losses modelled with the low ASC soil were then considered as the potential maximum N and P losses if the soil had no ASC, and were used to quantify the filtering of N and P services.

5.4 Summary of modelling:

This chapter describes the case study dairy farm and the justification for using a process-based modelling approach. It also describes the SPASMO model used to generate the data required to quantify the provisioning and regulating services from soils.
The chapter also details the extra-functionality added to the SPASMO model, in order to model the impacts of management practices on soil attributes and soil processes, to produce all

the data needed to calculate the parameters behind each soil service. The additional functionality includes the impacts of grazing regime on soil structure and pasture growth, grazing rotation, the use of standoff-pads and extra routines to the P cycle. ASC was also set near zero for specific simulations. These additional routines enable us to obtain from the model the data required to calculate parameters and quantify soil services (Table 5.6). The model also allows us to look at properties individually, follow their dynamics, and separate out the contribution the different stocks behind each service have.

An example of the outputs of SPASMO is presented in Appendix C. SPASMO outputs are daily values of chosen properties for 35 consecutive years (1975-2009). These daily values are used to calculate parameters to measure the soil services (Table 5.6), which is detailed in Chapter Seven.

In the next Chapter, the economic valuation of the environment is discussed and the techniques available for the valuation of soil services reviewed.

Chapter Six

Methods for the Economic Valuation of the Environment and Soil Ecosystem Services

Neoclassical Economics is the dominant paradigm in modern-day Economics, dominating the profession and the teaching of Economics in universities worldwide. It is the orthodox view which often goes unchallenged. In this thesis, ideas and measures of value derived from Neoclassical Economics are applied to the quantification of soil ecosystem services.

Nevertheless, it is recognised here, that when it comes to the valuation of soil ecosystem services and the 'environment' in general, there is no universal acceptance of neoclassical theories of value, and several methods of assessing measures of the 'environment' do exist particularly in the field of Ecological Economics.

The purpose of this chapter is to critically review the orthodox valuation methods (based on Neoclassical Economics) as well as the emergence of alternative approaches particularly in the field of Ecological Economics. More attention is given to the Neoclassical Economics methods of valuation because they are the more accepted, as well as the methods chosen for the actual valuation of soil ecosystem services in this thesis. Different valuation methods are examined against a number of criteria specific to the valuation of soil services.

6.1 Different theories for valuing soil ecosystem services:

This section of the thesis critically reviews the different ways Neoclassical Economics[6] and Ecological Economics approach the conceptualization and measurement of value particularly in relation to ecosystem services and the environment.

6.1.1 Value and Neoclassical Economics:

Most of the literature on the value of ecosystem services explicitly uses the Neoclassical Economics approach to valuation, even though such an approach is open to criticism and is far

---

[6] Theories of value are central to all schools of economic thought. Indeed some have argued that the most fundamental differences between different schools of economic thought can be explained by how they approach the question of value (Cole et al., 1991). Historically for example Marxist economics defined value in terms of 'embodied labour' and this sets it apart from modern-day Neoclassical Economics which defines 'value' (price) in terms of ideas of marginal cost (supply curve) and marginal benefit (demand curve).

from being the only approach for the valuation of ecological resources (Cole et al., 1991; Farber et al., 2002).

6.1.1.1 Neoclassical Economics, theory of value and price:

In Neoclassical Economics, the values of preferences of individuals are revealed in their economic behaviour. Individuals are assumed to be the best judge of their own welfare (James, 1994). Economic welfare of individuals is usually defined in terms of the 'utility'[7], a measure of the satisfaction that is derived from actual or potential use and even non-use of goods and services (Patterson, 1998). Individuals spend their income on goods and services to maximise their utility. Utility can also be derived from the consumption of goods and services that are provided 'free', like services from ecosystems (James, 1994). As increasing quantities of a commodity are consumed, total utility increases, at a diminishing rate. The utility that an individual receives by consuming an extra unit of a commodity is known as marginal utility (MU) or marginal benefit. MU is initially high, but decreases as more is consumed. Algebraically, MU consists of the derivative of the total utility function (James, 1994). The value the individual places on each additional unit of the commodity is reflected in the price he or she is prepared to pay. The MU curve gives rise to a demand curve (Fig. 6.1). Typically the demand curve is downward sloping, reflecting a decreasing willingness to pay (WTP) for each marginal unit as an increasing quantity is consumed (James, 1994) (Fig. 6.1).

[7] The founding fathers of utilitarianism are considered to be Jeremy Bentham (1748-1832) and John Stuart Mill (1806-1873).

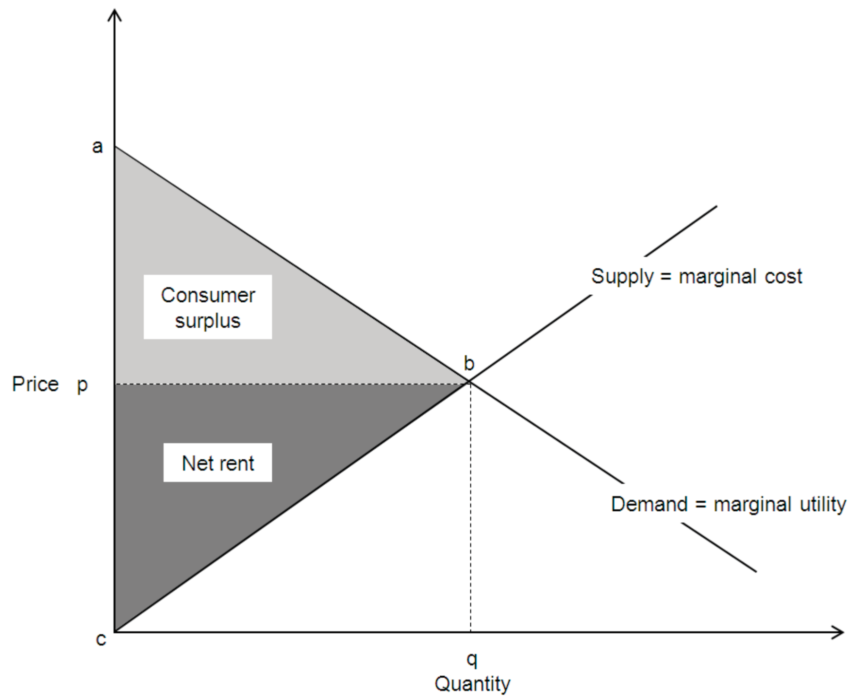

Figure 6-1: Conventional supply and demand curves for a typical marketed good or service (Costanza et al., 1997).

Conventional supply (marginal cost) and demand (marginal utility) curves for a typical marketed good or service (Fig. 6.1) are known as 'Marshallian scissors'[8]. The area abqc represents the total utility obtained for consuming a quantity q of a commodity. It also represents the amount of money the individual is willing to pay if forced to pay separately for each unit consumed (James, 1994). The area pbqc is the amount of money that is actually spent for a quantity q of the commodity, and the triangle abp is known as the 'consumer's surplus'. The consumer surplus is the amount of welfare the consumer receives over and above the price paid in the market. It shows that some individuals might be willing to pay more than the market price (Pearce and Turner, 1990). The cost of production, the area cbq under the supply curve, and the 'producer surplus' or 'net rent' for a commodity, the area pbc between the market price and the supply curve, are also shown in Fig. 6.1. The total economic value (TEV) of a commodity is the sum of the producer and consumer surplus, the area abc, excluding the cost of production (Costanza et al., 1997). TEV can be greater or less than the amount of money actually spent (area pbqc). This theory of value is the standard that dominates Neoclassical Economics and is very widely applied to a range of public policy issues, including ecological problems (Patterson, 1998).

[8] Alfred Marshall (1842-1924) is one of the founders of neoclassical economics, as he brought the concepts of supply and demand, marginal utility and costs of production together into the neoclassical theory of value.

Costanza et al. (1997) challenged this convention by arguing that for goods and services without a market, like ecosystem services, individuals can't express a monetary preference, even though utility is derived. Many ecosystem services are only substitutable up to a point. Their demand curve approaches infinity as the quantity available approaches zero, or some minimum necessary level of services. The consumer surplus (as well as the TEV) approaches infinity (Costanza et al., 1997) (Fig. 6.2). This concurs with the assumption that the natural environment is of infinite value because it supports all life and human activity. Moreover, for a given natural capital stock, some ecosystem services cannot be increased or decreased, their provision is more or less fixed, therefore their supply curves are more nearly vertical (Costanza et al., 1997). In most situations, their production has no cost for society (Fig. 6.2).

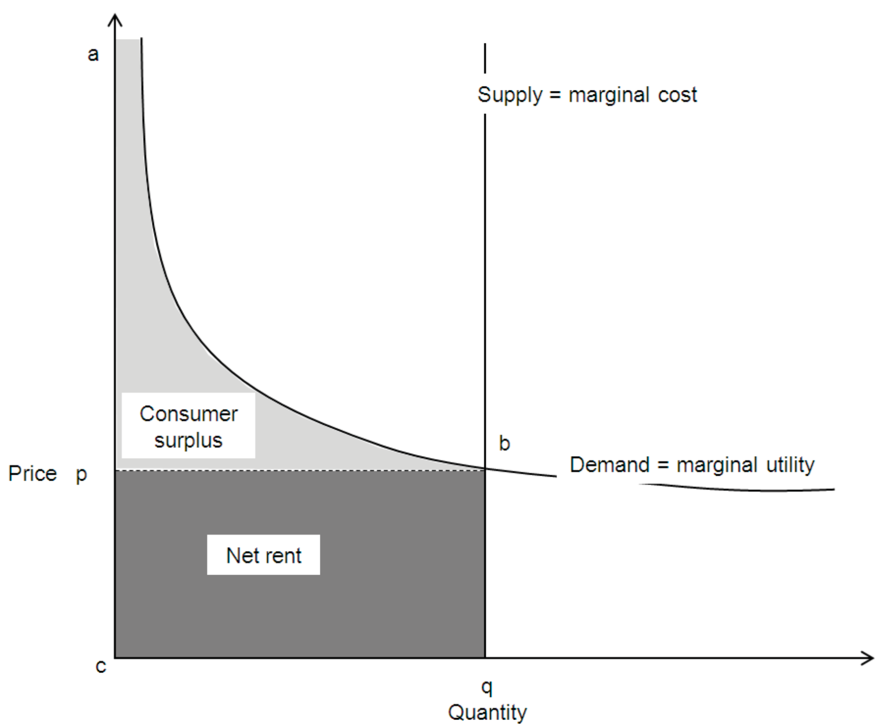

Figure 6-2: Supply and demand curves for ecosystem services (Costanza et al., 1997).

Total economic value can be broken down into several components, which can then be used to describe the value of ecosystems (Fig. 6.3).

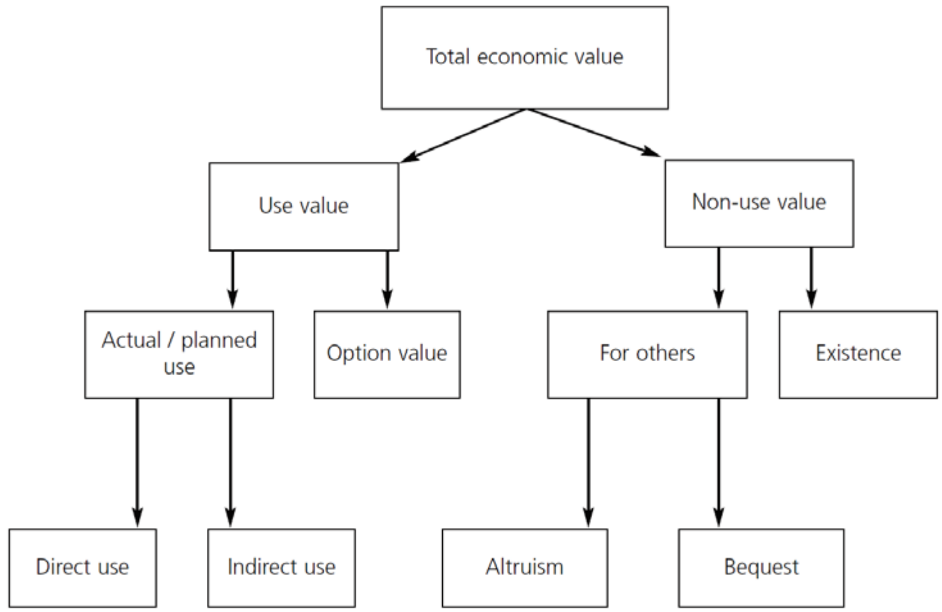

Figure 6-3: Total economic value (Defra, 2007)

TEV can be broken down into use and non-use values (Defra, 2007; Patterson, 1999; Pearce, 1995). Use-values include direct and indirect use, and option values (Fig. 6.3):

- Direct use value: The value of all goods and services derived from the direct or planned use of ecosystems. This can refer to the consumptive use of resources extracted from the ecosystem (e.g. food, timber, parks) or the non-consumptive use of services without extracting any elements from the ecosystem (e.g. recreation, landscape amenity). Some of these goods and services can be traded in markets (e.g. timber, crops), while others are non-marketable, i.e. there is no formal market on which they are traded (e.g. recreation, inspiration) (Defra, 2007). Direct use values are straightforward in concept and generally accepted, but are not necessarily easy to measure in economic terms (Pearce, 1995). They are generally attributed to provisioning and cultural services (Table 6.1).
- Indirect use value: They are derived from ecosystems from processes supporting or regulating the natural capital stocks linked to direct use activities (Defra, 2007; Pearce, 1995). Pearce (1995, p. 42) argued that "indirect use values correspond to the ecologist's concept of 'ecological function". Indirect-use values correspond to regulating services (Table 6.1). Regulating services are often not noticed by people until they are damaged or lost. They include e.g. flood mitigation, greenhouse gases regulation or the recycling of wastes. Because regulating services are generally non-marketable, measuring indirect use values is significantly more challenging than measuring direct use values. Changes in the provision of these services are also difficult to measure.

- Option value: The value that people place on having the option to use, directly or indirectly, a resource or service in the future, even if not currently in use (Costanza et al., 1989; Defra, 2007). It is usually assessed by the amount an individual would be willing to pay (WTP) to conserve a resource or service (Pearce, 1995) like for example, a national park or the mix of species associated with a particular habitat. Option value can also be perceived as insurance for the future. Resources and services could have a value for possible future uses or unforeseen difficulties, which may not yet be known (Defra, 2007). Some authors (Pearce, 1995) consider this a form of non-use value, as it focuses on the option to use something in the future, and not at present. Option value concerns all types of services (Table 6.1).

Non-use value, also referred to as passive value, is not related to the actual use of ecosystem goods and services, but is derived from the knowledge that ecosystems are maintained (Defra, 2007). They concern all types of services (Table 6.1). Non-use value can be further subdivided into three main components (Fig. 6.3):

- Existence value: It relates to the existence of ecosystem goods and services even if an individual has no actual or planned use for it (Costanza et al., 1989). Many people are willing to pay for the preservation of species (whales, rainforest insects) even if they know they might actually never be in contact with them.
- Altruistic value: The value individuals attach to the availability of the ecosystem resources or services to others in the current generation.
- Bequest value: The value that people place on knowing that ecosystem goods and services will be available for future generations. Bequest value is sometimes regarded as part of existence value.

Some authors (Pearce, 1995) debate the merit, and hence application of this classification to neoclassical economic valuation, especially the breaking down into the three non-use value parts. The relevance of existence value to neoclassical economic valuation is questioned since it may be representing counter-preferential values, based on moral concern, obligation, duty or altruism (Pearce, 1995). Some authors (Pearce and Turner, 1990) view altruistic and bequest values as part of option values since the “option” of using the environmental good or service is available for individuals and future generations.
The TEV framework is a useful tool to identify what type of value is being measured, based on the type of good or service concerned (Table 6.1), thereby assisting the selection of the appropriate valuation method (Defra, 2007). Some values, like option values or non-use values, are applicable to all services (Table 6.1).

Table 6-1: Links between ecosystem services and total economic value framework.

Ecosystem services	Use values			Non-use values		
	Direct use	Indirect use	Option	Altruism	Bequest	Existence
Provisioning	X		X	X	X	X
Regulating		X	X	X	X	X
Cultural	X		X	X	X	X

According to Pearce (1995), it is incorrect to think that economists have captured all there is to know about value in the concept of TEV. Often only economic values are captured, while the issue of e.g. intrinsic value remains unresolved.

Neoclassical economic valuation begins with the values of preferences of individuals, as revealed in their economic behaviour, and is based on the Neoclassical Economics model in which value is measured by society's 'willingness to pay' (WTP) for goods and services, that are used as inputs, or consumed as outputs of the economy. TEV is a measure of the utility obtained from consuming a good or service. TEV is measured using individuals WTP for the good or service. The opposite of utility is disutility, which can be thought as a negative utility experienced by an individual exposed to something 'bad' (James, 1994), like pollution or the destruction of natural capital stocks leading to a decrease in the provision of ecosystem services. Disutility may be expressed by an individual as a willingness to accept (WTA) monetary compensation for the 'bad', or WTP for avoidance of the 'bad'. Monetary measures of disutility are classified as costs in an Neoclassical Economics analysis (James, 1994).

According to the neoclassical theory of value, economic valuation in the environmental context, attempts to elicit public preferences in monetary terms for changes in the state of the environment by looking at the individual's WTP or WTA. What is being valued is not 'the environment' but peoples preferences for changes in the state of their environment (Pearce, 1995). Neoclassical economic valuation is thus an entirely anthropocentric concept (Pearce, 1995). Neoclassical economic valuation, in the environment context, is essentially about discovering the demand curve for environmental goods and services (Pearce, 1995). It focuses on the contribution to human welfare of environmental goods and services, which is seen as the most relevant to policy-making (Defra, 2007). It is about accessing the TEV of environmental goods and services that are non-traded in markets to reflect the fact that people indeed derive utility from them.

6.1.1.2 Rationale for neoclassical economic valuation:

Pearce and Barbier (2000, p.7) argued that the rationale of neoclassical economic valuation "lies in the need to ensure that environmental impacts are taken into account in decision-making on the same basis as the conventional costs and benefit of conventional economic activity". Valuing environmental goods and services provides appropriate evidence for use in a benefit-cost analysis (BCA). BCA is an analysis that quantifies in monetary terms as many of the costs and benefits of a proposal as is feasible (Defra, 2007).

Implementing a BCA on a development proposal permits the relevant government agencies to determine whether, in a broad sense, the proposal would use available natural resources efficiently from an economic and community standpoint (James, 1994). Benefit-cost analysis helps to decide whether to accept or reject a development proposal based on the balance of costs and benefits of the project. Pearce and Barbier (2000, p. 54) argued that "benefits will include anything for which people are willing to pay, and costs will be anything for which they are willing to accept compensation or willing to pay to forgo". The relevant comparison when looking at a decision on a development project is between the cost of the project, the benefit of the project, and the TEV that is lost by development. It makes economic sense, and sense for sustainability, to proceed with the development if the benefits of the development minus the costs of the development are greater than the benefits of preserving the environment by not developing the area (Pearce and Turner, 1990).

Many indicators (Defra, 2007; MEA, 2005) suggest that we are using the natural environment in a non-sustainable way. Ecosystems have been under increasing pressure, in recent decades, as a result of human activity and there is evidence that ecosystem services are decreasing (Defra, 2007; MEA, 2005). Valuing ecosystem goods and services for inclusion in BCA could be a useful tool in the advance of sustainable development. The essential role natural ecosystems play in supporting all human activities and their contribution to human wellbeing needs to be fully recognised. Pearce and Barbier (2000, p. 49) argued that "improving our efforts in environmental valuation is important for assessing the economic consequences of natural capital depletion and degradation".

Failure to include ecosystem services in benefit-cost calculations implicitly assigns them a value of zero (Vesely, 2006). For a long time this has been the norm, contributing for some to the major depletion of natural capital stocks and gigantic environmental problems (MEA, 2005).

The inclusion of a value for ecosystem services in BCA should contribute towards better decision-making, ensuring that policy appraisals fully account for the costs and benefits of development proposals on the natural environment (Defra, 2007). For policy-making, the more relevant application of valuation is to marginal changes in the environment (Defra, 2007).

Valuing ecosystem services can help in:

- fully accounting for the environmental impacts in decision making,
- determining whether a development project or the implementation of a new policy that alters natural capital stocks delivers net benefits to society, including the costs of environment degradation,
- choosing between different projects or land uses,
- creating new insights for policy development,
- creating markets for ecosystem services (policy around payments for ecosystem services),
- helping to communicate with the public and land managers on the value of the environment (Defra, 2007).

6.1.1.3 Problems with neoclassical economic valuation:

A number of problems have been raised regarding the use and effectiveness of neoclassical economic valuation in assessing the value of ecosystem goods and services that, as yet, lack markets, and in taking them into account in decision-making.

Optimisation models and compensatory changes underlie both the classical economic utility theory and the traditional cost-benefit analysis[9] (Pearce et al., 2006). However, in practice the specification of a community welfare function requires complete information about all possible options, and the trade-offs between them. Moreover, the externalities involved in environmental management have far-reaching economic and ecological aspects, which cannot always be captured by BCA (Munda et al., 1994).

Three significant problems arise when dealing with neoclassical economic valuation and BCA: the valuation, the discounting and the aggregation of preferences (Gasparatos et al., 2008).
(1) Neoclassical valuation techniques that try to elicit individuals WTP or WTA for a non-market good are confronted to anomalies based on human beings having imperfect knowledge

[9] The "Pareto principle" – whereby a policy is "good" if at least some people actually gains and no-one actually loses – and the Kaldor-Hicks "compensation principle" – whereby gainers can hypothetically compensate losers to achieve a potential Pareto improvement in real-life context – are the two principles underlying Neoclassical Economics (Pearce et al., 2006).

of ecological processes and functions (Patterson, 1998). This is particularly important when valuing soil ecosystem services as they are often invisible and unknown to the non-experts.

Each neoclassical valuation technique is subject to a number of biases, discussed in section 6.2 of this chapter. Human choice is very complex and preferences can be manipulated which make utility calculations the more challenging (Niemeyer and Spash, 2001).

(2) Another argument against neoclassical economic valuation is its non-equitable aspect. BCA is rooted in the concept of economic efficiency and not of distributional equity and justice (Gasparatos et al., 2008). Discounting is an important but very controversial part of BCA which is performed in order to compare present and future costs/benefits (Gasparatos et al., 2008). It represents the trade-off between the enjoyment of present and future benefits (Pearce and Turner, 1990). The greater the discount rate adopted, the greater the devaluation of future costs/benefits. Therefore, in projects with a long time horizon, spreading across several generations, future impacts count for little. It has been argued (Gasparatos et al., 2008) that this is contrary to the interests of future generations, leading to a non equitable distribution of costs and benefits through time by forcing future generations to bear a disproportionate cost (Hanley and Spash, 1993; Niemeyer and Spash, 2001; Pearce and Turner, 1990; Pearce et al., 2006). It has been suggested that low discount rates should be adopted for projects that will greatly affect future generations. In some cases such as species extinction, adoption of a zero discount rate could even be justifiable (Gasparatos et al., 2008; Hanley and Spash, 1993; Pearce and Turner, 1990; Pearce et al., 2006).

(3) The theoretical foundations of BCA assume that individuals' preferences, once determined, can be aggregated so the social benefit is simply the sum of all individuals' benefits (same for costs). However, it has been argued (Pearce et al., 2006) that the externalities associated with environmental issues make it very challenging to determine the geographical boundaries of the aggregation. In some cases, like e.g. GHGs emissions, the boundary may be the world as a whole (Pearce et al., 2006). Recently, it has been understood that welfare is a multidimensional variable that includes a broad set of criteria (e.g. income, environmental quality, equity, public facilities) (Munda et al., 1994). This poses the question of the relevance of preference aggregation in revealing welfare maximisation (Niemeyer and Spash, 2001)

Finally some authors (Martinez-Alier et al., 1998; Munda et al., 1994; Niemeyer and Spash, 2001) have denounced 'monetary reductionism' altogether, that is the use of monetary values as the common standard to measure all sorts of assets, even non marketed ones. Martinez-Alier et al. (1998) showed that the environment has different types of value, and it is misleading to take decisions based on only one type of value. Niemeyer and Spash (2001) argued that implicit ethical choices are made by reducing the value dimension to preference utilitarianism.

It has also been argued that neoclassical economic valuation should not be applied to critical natural capital (Ekins et al., 2003b) which is by definition 'priceless' because not readily substitutable by man-made capital (Martinez-Alier et al., 1998).

6.1.2 Value and Ecological Economics:

Ecological Economics sees the economy as a subsystem of a larger global ecosystem (Martinez-Alier, 2001). Because of population growth, the economy always requires more material and energy and has increasing environmental impacts. The main objective of Ecological Economics is to assess the sustainability of the economy in developing physical indicators and indexes of sustainability (Martinez-Alier, 2001). Costanza (1991) defined the field as 'the science and management of sustainability'. Assigning money values to ecosystem goods and services is one of the techniques used by ecological economists.

Theories of value are at the heart of all economic schools, and are still the object of fundamental disagreements between, for example, Ecological Economics and Neoclassical Environmental / Resource Economics (Patterson, 1998).

Theories of value in Ecological Economics need to be different from neoclassical theory because of the biophysical perspective taken by ecological economists (Patterson, 1998). An 'embodied energy theory of value' has frequently been suggested (Hannon et al., 1986) as being appropriate for Ecological Economics. This has been debated by neoclassical economists who assert that energy is only one of the factors of production. However, in Ecological Economics, there is not really any one theory of value, but rather a mixture of approaches, some of which aren't really 'theories'.

Whatever type of value is chosen, the main problem in Ecological Economics is valuation or how to deal with 'mixed units', or 'commensuration' when quantifying biophysical input and output flows between ecosystems and the economy.

6.1.2.1 EMERGY analysis and ecological pricing:

An 'energy theory of value' has been proposed as being suitable for Ecological Economics (Patterson, 1998), since energy is the fundamental driver of ecological systems and thereby the economy. For example, Odum (1996) has proposed an energy theory of value where the value of a commodity is informed by the amount of energy required to produce that commodity- the idea of EMERGY. Authors like Georgescu-Roegen (1979) rejected a strict energy theory of value, arguing that matter is also important, since it is also subject to the entropy laws and therefore energy should not assume supremacy in any physical view of value (Patterson, 1998).

Research in the area has led to theories of value where prices can be determined for biophysical inputs and outputs, leading to a new type of accounting of the economy: a mass/energy accounting or 'ecological pricing' (Patterson, 1998). For example, Costanza and Hannon (1989) quantified mass and energy flows in the biosphere, and determined prices for various biosphere commodities by using matrix inversion techniques.

This biophysical view of the functioning of the economy comes from the work of Kneese et al. (1970), Georgescu-Roegen (1971) and Daly (1973) in the early 1970's. The basic idea they promoted is that the economy is based on inputs of low entropy matter/energy, and then dissipates, as an output, high entropy matter/energy (Patterson, 1998). Ecological prices are the "weighting factors inferred from models which describe energy and mass flows through ecological and economic systems" (Patterson, 2002, p.457) following the first two laws of thermodynamics.

However, ecological and economic systems are very complex systems, including numerous mass and energy flows. Each process has multiple inputs and multiple outputs. Many accounting frameworks used in Neoclassical Economics and Ecology oversimplify matters and do not reflect the true complexity of ecological or economic systems.

For this reason, ecological economists are constantly working on new accounting frameworks allowing for complexity, joint production and interdependencies, as well as following principles like the conservation of mass and energy, or the linearity of flows. Authors like Costanza and Hannon (1989) and Patterson (1983) have proposed such accounting frameworks (Patterson, 1998).

6.1.2.2 Ecological and contributory values:

Ecological value is a biocentric type of value, different from economic value which is also used in Ecological Economics.

Biophysical processes underpin the production or activity of ecological systems. Ecological value is defined as the value of direct and indirect interactions of a component of an ecosystem, an ecological entity (species) or compartment, with the other components of the same ecosystem (Cordell et al., 2005). Within an ecosystem, each species has a function for the existence of other species of that ecosystem. The loss of species richness and/or abundance eventually leads to loss of ecosystem function. Not all species within an ecosystem are of the same importance. The species which are the more numerous are called dominant species.

Species that have important ecological values, greater than one would expect based on their abundance, are called keystone species. These species are often central to the structure of an ecosystem. Species diversity and redundancies around the same processes increase ecosystem's resilience and ability to adapt and respond to changing environmental conditions.

From an economic point of view, the production of goods and services by the human economy relies on the productivity of natural biotic systems (ecosystem goods and services). Natural ecosystems make contributions to the value of final economic goods and services. This contribution is recognised by the notion of contributory value (CV) (Norton, 1986; Ulanowicz, 1991). CV assigns value to environmental resources "not due to their direct value to humans, but according to their indirect role in maintaining and accentuating the ecosystem processes which support these direct benefits" (Costanza et al., 1989, p. 338). CV provides a counterbalance to the anthropocentric valuation techniques used in Neoclassical Economics (detailed further in this chapter) (Patterson, 1998; Patterson, 2008) because it is based on ecological value. CV attempts to quantify the indirect contributions made by different ecosystem components to an ecosystem's final outputs and, therefore, to economic and human activities (Patterson, 1998; Swift et al., 2004). The concept of CV value focuses on the ecological importance of biodiversity as a component of the total economic value (TEV) or, more precisely, the economic relevance of ecological interactions. CV highlights ecological phenomena which tend to be neglected. It is an estimate of the value individuals would place on ecosystem services if they were fully informed about the functioning of the environment (Costanza et al., 1989; Fromm, 2000; Patterson, 2008).

CV is a 'cost of production' approach similar to the 'energy theory of value' (Patterson, 1998). CV is based on ecological prices, or shadow prices, that can be obtained from data (energy and mass flows) that describe the ecological processes that support human and non-human activity (Patterson, 2008). Ecological prices are based on the biophysical interdependencies in the system (forward and backward linkages), whereas market prices are based on consumer preferences and WTP, determining the exchange value in markets. Ecological prices implicitly exist in nature and do not require human presence (Patterson, 2008). They can be determined by using equivalence factors between different quantities (including monetary values or not). For example, Patterson (2008) determined ecological prices for the global marine system using an input-output model.

6.1.2.3 Multicriteria methods and value pluralism:

Several ecological economists (Martinez-Alier et al., 1998; Niemeyer and Spash, 2001) have rejected methods such as EMERGY analysis and ecological pricing as being inappropriately reductionist.

An environmental good or service can have different types of values, all equally important. Natural resources may be valuable for their biodiversity (richness of species or genetic variety), the beauty of the landscape they form, or their market value (fresh water, wood for timber). Therefore it seems inappropriate to take decisions based on only one type of value, e.g. economic value (Martinez-Alier et al., 1998).

Martinez-Alier et al. (1998) argue that incommensurability (the absence of a common unit of measurement across plural values) is the 'foundation stone for Ecological Economics' (Martinez-Alier et al., 1998, p.279), and that instead of complying to any type of physical reductionism (monetary, energy or other), valuation should push toward multicriteria analysis and the use of weak comparability (comparing options without recourse to a single type of value). Moreover, incommensurability doesn't imply a hierarchy of values.

Environmental management is, as a rule, confronted to qualitative information in evaluation problems (Nijkamp et al., 1990). Thus, there is a clear need for methods that are able to take into account qualitative information, or information of a "mixed" type (both qualitative and quantitative) (Munda et al., 1994). Neoclassical Economics being based on a strong quantitative tradition, including quantitative measures of environmental elements has been fairly easy in conventional models. However, qualitative aspects are harder to deal with in these models (Munda et al., 1994).

A typical multi criteria problem (with a discrete number of alternatives) may be described as following: Given a set A of alternatives (or feasible actions), and a set G of evaluation criteria and assuming the existence of n alternatives (j=1, ..., n) and m criteria (i=1, ...m), it is possible to build an nm matrix P, called evaluation or impact matrix, whose typical element pij (1,…, m; 1, …, n) represents the evaluation of the j-th alternative by means of the i-th criterion. The impact matrix may include quantitative, qualitative or both types of information (Martinez-Alier et al., 1998; Munda et al., 1994; Munda et al., 1995).

When dealing with many alternatives and criteria, there is usually no solution optimising all the criteria at the same time. This implies that environmental management will always be characterised by the search for acceptable compromise solutions (Martinez-Alier et al., 1998; Munda et al., 1994). Multicriteria evaluation techniques provide an adequate evaluation methodology to deal with qualitative multidimensional environmental issues. They provide insight to decision makers into the nature of conflicts and make trade-offs more transparent

(Munda et al., 1994). Martinez-Alier et al. (1998) argued that “as a tool for the understanding of conflicts, and possibly for conflict management multicriteria evaluation has demonstrated its usefulness in many environmental management problems” (Martinez-Alier et al., 1998).

6.1.3 Intrinsic value:

Deep ecologists (Næss, 1989) argue that ecosystem and biodiversity have intrinsic value. Intrinsic value can be defined as the inherent worth of something, independent of its value to anyone or anything else: a value that exists not just because individual human beings have preferences for the environmental asset in question. Intrinsic value is non-anthropocentric, contrarily to any type of value related to human economy. It represents “the value that reside ‘in’ something and that is unrelated to human beings altogether” (Pearce and Turner, 1990, p. 130). This notion is similar to the inalienable right to exist. It is independent of the value that humans may express for ecosystem conservation (Pearce, 1995). The concept of intrinsic value is highly philosophical and controversial. It has been argued that a value cannot exist without an evaluator that is human-beings; therefore intrinsic value does not exist.

The attribution of intrinsic value to other species (and to ecosystem goods and services) is part of an ideology usually called ‘biocentrism’, or ‘ecocentrism’, to be contrasted with ‘anthropocentrism’ (Sarkar, 2005). The biocentric view, forwarded by the deep ecology movement (Næss, 1989), holds that all species have intrinsic value, with humans no more or less important than other species. Therefore, species conservation requires less justification if one accepts that biodiversity has intrinsic value.

Economic and intrinsic values are thus two very different things: the former relates to preferences of people for or against an environmental change, whereas the latter relates to the value that intrinsically resides ‘in’ environmental assets. Economic values can, in principle, be measured. Intrinsic values cannot, because they are not linked to real world choices. However, it is argued by some that humans may capture part of the intrinsic value in their preferences (Pearce and Turner, 1990). Economists (Pearce and Turner, 1990) argue that ‘existence value’ (Fig. 6.3) encompasses the notion of intrinsic value, but the relationships between these two remains contentious (Attfield, 1998). Acceptance of intrinsic value implies that environmental assets should be conserved regardless of the costs associated with conservation. This seems unreal because in the real-world context, there are costs and benefits associated with any type of conservation. It doesn’t mean that intrinsic value is irrelevant or not legitimate: it is as relevant to decision making as economic value (Pearce, 1995).

In any case, the type of theory of value used should be the object of careful considerations taking into account the type of ecosystem concerned, the amount and quality of information available, and most importantly, the use to be made of the values generated.

6.2 Neoclassical economic valuation methods:

For this study, neoclassical economic valuation was chosen to value soil services. This option was chosen because a number of well developed and well documented neoclassical valuation techniques are available. The main challenge of this study was to understand describe and quantify soil services in biophysical terms. Neoclassical economic valuation was used here to take the study one step further and deal with the incommensurability resulting from services quantification, allowing the overall economic value of soil services to be calculated, and used to inform land use sustainability and land management.

This section assesses the techniques available for the monetary valuation of ecosystem services. Neoclassical economic valuation is used to measure public and individual preferences for changes in ecosystem services provision. Valuation techniques for valuing ecosystem services can be distinguished by the type of preferences they elicit: revealed or stated.

(1) Revealed preference techniques obtain values by looking at individuals' preferences and WTP for a marketable good with environmental attributes which influence its price (Defra, 2007; Pearce and Turner, 1990; Pearce and Barbier, 2000). These techniques rely on conventional markets. Conventional markets are actual markets in which the environmental goods and services are already traded (e.g. timber market or CO_2 market). Revealed preferences techniques can also value non-tradable goods and services indirectly through marketed goods and services that embody their values (e.g. air pollution affects the price of houses) (Table 6.2).

(2) Stated preference techniques elicit individuals' preferences for a given change in a natural resource or service through structured questionnaires (Defra, 2007; Pearce and Barbier, 2000). Stated preferences techniques use hypothetical markets which are simulated markets built so individuals can express their WTP for a non-traded environmental good or service (Pearce and Turner, 1990; Pearce and Barbier, 2000). These techniques are the only ones that can estimate non-use values for some natural resource, which can be a significant component of the overall TEV (Defra, 2007) (Table 6.2).

Revealed preference techniques use conventional markets, whereas stated preference techniques use hypothetical markets (Table 6.2).

Table 6-2: Valuation techniques classified according to method and market type (Defra, 2007). NA: Not applicable.

Market type	**Method**	
	Revealed preferences	Stated preferences
Conventional market	Market price Productivity change Defensive expenditures Replacement cost Provision cost Hedonic pricing Travel cost	NA
Hypothetical market	NA	Contingent valuation Conjoint Analysis Choice modelling Group valuation

6.2.1 Criteria for evaluating neoclassical valuation methods:

In order to value the ecosystem services provided by soils, a suite of adequate valuation techniques needs to be identified. Some valuation techniques may be more suited to capture the values of particular ecosystem services than others. For example, market prices are often used for valuing provisioning services, while stated preference methods are well suited to capture the non-use values of all types of services (Table 6.1) (Defra, 2007). To value soil services, a number of criteria need to be examined when selecting valuation techniques:

- Data requirement for the valuation: Appropriate techniques shouldn't be too data intensive to be easy to implement.
- Data availability: The availability of data should be checked before basing a valuation exercise on it.
- Costs of implementation: High costs of implementation can limit the size of the population interrogated for the valuation exercise, thereby limiting the depth of analysis,
- Time line: the amount of time required to implement a method can also be prohibitive,
- Complexity of the design, implementation and treatment of information can lead to errors and false estimations,
- Subjectivity to cognitive burden: Valuation exercises can be compromised by a lack of knowledge about the environmental good or service in question from individuals designing the study or answering questionnaires.
- Subjectivity to joint production: Some valuation techniques can overestimate economic values because the estimates they are considering cover more than one good or service.

In the following section, the techniques available for the neoclassical economic valuation of soil services are described, including their benefits, limitations and examples of implementation. Each method is reviewed against the criteria mentioned above for the valuation of soil services.

6.2.2 Techniques using revealed preferences:

Techniques using revealed preferences are able to capture direct and indirect use values (Defra, 2007) (Fig. 6.3).

6.2.2.1 Market prices:

Market prices can be used directly to capture the value of ecosystem goods and services already traded in markets (Defra, 2007; Pearce and Barbier, 2000). For example, crops, forest products or commercial fish have traded market value. Market prices can be viewed as proxies for direct and indirect use values, but often do not capture the non-use values.

It should be noted that in theory, one should use the total economic value of a commodity instead of its market price. These two values are different by definition (Fig. 6.1). Rigorous analyses sometimes use the producer surplus (or net rent), which is the market price minus the cost of production (Fig. 6.1), instead of the market price. However, the consumer surplus of a commodity usually cannot be determined as easily. For this reason, market prices are usually used as proxies for TEV. One should be aware that even if the market price of a commodity is used widely in the literature to value ecosystem services, this method isn't rigorously following the Neoclassical Economics theory of value.

Market prices can be used to put a value on some provisioning services like the provision of raw materials such as coal or oil, or the provision of food, wood and fibre. Porter et al. (2009) valued the food and raw material production from an agro-ecosystem by using the farm gate prices of grain, pasture and wood.

Market prices can be used to value the provision of marketed goods (raw materials e.g. clay, peat) from soils. For example, the provision of raw materials like peat can be valued with the market price of peat minus costs of extraction and handling. This technique is limited to provisioning services for which a market exists and is sensitive to the amount of good produced. Market prices are readily available and robust, and therefore the valuation is straight forward. They may however need to be adjusted to take account of distortions, such as subsidies (Defra, 2007). Finally, they are not sensitive to individual's knowledge of soil science.

Table 6.3 presents the results of this method against the criteria for soil valuation.

Table 6-3: Market Prices *versus* criteria for soil valuation.

Criteria	Market prices
Data requirements	low
Data availability	high
Cost of implementation	low
Time line	low
Complexity	low
Subjectivity to cognitive burden	no
Subjectivity to joint production	no

Ecosystem goods and services that are marketed are inputs to the economy; therefore the monetary value placed on them can be followed through the economy thanks to input-output multiplier analysis. Input-output multipliers are a useful measure of structural interdependence within an economy. They show the relationship between an additional unit of spending (final demand) and changes in output, income, value added and employment within the economy. Input-output multipliers capture not only the direct effects of additional spending captured, but also the indirect effects resulting from the interdependencies that exist between industries within the economy. Type I multipliers summarise direct and indirect economic impact, while Type II multipliers summarise direct, indirect and induced economic impact.
Different types of multipliers exist (McDonald, 2005):

- Output multipliers relate a unit of spending to an increase in output in the economy.
- Value added multipliers show the relationship between an additional unit of spending and changes in the level of value added generated in an economy.
- Employment multipliers show the relationship between an additional unit of spending and changes in the level of employment in an economy.

For this study, Type II value added multipliers (for backward and forward linkages) could be considered to assess the impact of the dairy industry on New Zealand economy.

6.2.2.2 Productivity change approach:

Productivity change approach (PCA) is worth mentioning because even if it is not a valuation technique per se, it can be used with diverse valuation techniques. PCA is based on the use of production functions. These functions describe the relationship that may exist between a particular ecosystem service, the natural capital stocks behind it and the production of a market good (Defra, 2007). For example, the link between soil moisture and OM content to crop yield are indicators behind the provisioning service “production of food, wood and fibre”. Natural capital stocks are considered as inputs to the production process behind the ecosystem services. Valuation is applied to the outcome quantified through this approach (e.g. the market price of the crop). PCA is very useful to measure actual or modelled changes in production, when natural capital stocks, land use, management (e.g. fertiliser) or external drivers (e.g. climate) vary. The value of the natural capital stocks and ecosystem services (shadow prices) is inferred by considering the changes in production of the market good (market prices) that result from a change in natural capital stocks.

PCA is capable of capturing indirect use value (Defra, 2007), e.g. the value of regulating services like GHGs regulation.

Sparling et al. (2006) linked soil organic matter and climate to the production of milk from pastoral land. Porter et al. (2009) built a production function to calculate carbon accumulation in agricultural soils. The amount of plant and root residue was estimated at 1.5 times the crop grain yield, with 40% of this being carbon. This was used to calculate the economic value of carbon accumulation. The economic value of carbon accumulated by crop and root residue was estimated based on 10 US$ per tonne of carbon accumulated (Porter et al., 2009).

Market data used for PCA is usually available and robust. PCA is quite straight forward if the data to build and run the production function is available. It is sometimes difficult to build production functions for complex ecosystems, and it might require a lot of data on the changes in the service considered, and the impacts on production (Defra, 2007). PCA can also be built into benefit-cost analysis (Vesely, 2006), but can only identify use values. PCA isn’t sensitive to individuals’ knowledge of soil science or joint production. The valuation of the good using a market price is influenced by factors driving the state of the market at the time of the study. Table 6.4 presents the results of PCA against the criteria for soil valuation.

Table 6-4: Productivity change approach versus criteria for soil valuation.

Criteria	Productivity change approach
Data requirements	medium
Data availability	high
Cost of implementation	medium
Time line	medium
Complexity	medium
Subjectivity to cognitive burden	no
Subjectivity to joint production	no

PCA is applicable to agriculture for food, wood and fibre production (Vesely, 2006), therefore it is useful to value soil provisioning services when, for example, production functions linking soil natural capital stocks and crop yields are available. PCA can also be used to value some soil regulating services, like GHGs regulation, since a market exists for CO_2, and models exist which link soil properties and processes to management and land use to GHGs emissions.

The next three methods presented are cost-based approaches. Cost-based approaches to valuing ecosystem goods and services consider the costs, directly observed from markets, arising from the provision of the ecosystem good or service. These methods can measure direct and indirect use values. They provide proxy values and do not directly measure utility, therefore are non-demand curve methods, and hence, need to be used with care (Defra, 2007). Cost-based approaches usually benefit from readily available market data to inform the costs in question. As proxies however, they can easily under- or overestimate the actual value of the ecosystem services in question. Despite these reservations, these approaches can be very useful in validating the scale of values obtained from measurement of direct utility (Defra, 2007). Often the same service can be valued by several different cost-based approaches, each using different costs related to the service in question.

6.2.2.3 Averting behaviour and defensive expenditure:

These methods are based on the theory of consumer behaviour (Birol et al., 2006). They take as their main premise the notion that individuals and households can insulate themselves from a "non-market bad" (e.g. degradation of the environment like air or water quality) by changing their behaviour (averting behaviour) or by purchasing market goods enabling them to avoid or reduce exposure to that non-market bad. These financial outlays are known as defensive expenditures (Pearce et al., 2006). Defensive expenditures can be viewed as the money spent by individuals or households to avoid exposure to a degradation of the environment and a decrease in the provision of ecosystem services. The individual combines quantities of the degraded service (the "non-marketed bad") with a quantity of a market good to produce the previous level of utility. What is measured by defensive expenditures is how much money the individual is willing to pay to keep the same level of utility. Therefore, defensive expenditures do not directly measure utility. The value of the market goods purchased may be used as a proxy, an implicit price, for the value of the degradation of a service (Defra, 2007; Pearce et al., 2006). For example, the cost of water filtration may be used as a proxy for the value of the degradation of drinking water quality (Defra, 2007).

Even if the individual is not purchasing any market good, but only changes their behaviour, e.g. spends more time indoors to avoid the consequences of degraded air quality on their health (Pearce et al., 2006) or boils water for cooking and drinking, or reduces the frequency or length of showers if a volatile organic chemical were present (Birol et al., 2006), the value of the degradation in the environment can be assessed using different costs (e.g. time costs) (Pearce et al., 2006).

The defensive expenditure approach has been implemented in various situations, for example, Kim and Dixon (1986) studied defensive expenditure against soil erosion. Lowland Rice farmers in Korea were prepared to incur costs for the construction of water diversion channels as a defensive measure against siltation and other productivity damage caused by sediment discharge from upland soil erosion.

These methods usually have modest data requirements (Nijkamp et al., 2008) and market data, concerning the market goods purchased, used as defensive expenditures, is usually available and robust. Moreover, defensive expenditures are easy to build into benefit-cost analysis (Table 6.5).

These approaches have a number of limitations (Table 6.5).

1- They require the awareness of the individual who needs to perceive the adverse effect. It must be possible to avoid or reduce the exposure to the adverse effect; and the individual needs to be able to make expenditures for optimal protection.

2- They require data on the magnitude of the change of the ecosystem services or the environmental impact that might not be available.

3- They represent a partial or lower bound estimate of the value of the impact of the non-market bad on wellbeing because they do not measure all the costs related to the degradation of the environment (non-market bad) (Pearce et al., 2006). Moreover, defensive expenditures are limited by income and the value obtained may be conservative.

4- They can create joint products (Pearce et al., 2006) because actual expenditures may be targeted to meet several objectives (Nijkamp et al., 2008) which could overstate the value of the costs avoided. For example, double glazing will help against the noise but will also help with heating and energy conservation.

5- Defensive expenditures are not a continuous decision but a discrete one (e.g. double glazing is either purchased or not) (Nijkamp et al., 2008).

Table 6-5: Defensive expenditures versus criteria for soil valuation.

Criteria	Defensive expenditures
Data requirements	low
Data availability	high
Cost of implementation	low
Time line	low
Complexity	low
Subjectivity to cognitive burden	yes
Subjectivity to joint production	yes

The averting behaviour approach is potentially applicable to the valuation of some soil services. Farmers communities or governments often incur actual expenditures to mitigate or prevent productivity loss or reduce degradation problems (Shiferaw et al., 2005). There are a number of processes, like erosion, compaction or salinisation that degrade soil natural capital and thereby impact on the provision of ecosystem services. Farmers and other land users can take preventive measures to mitigate the impacts of these processes. For example, tree planting on steep land prevents landslides and erosion. Keeping the cows of the paddock, on a standoff pad, in winter when the soil is wet, prevents soil compaction. The value of the defensive expenditures undertaken can be used as a lower bound proxy for the value of the ecosystem services provided by intact soil natural capital stocks.

When applying the averting behaviour approach to the valuation of soil services, the issue of joint production needs to be considered. For example, the money spent on a standoff pad to take dairy cows off wet paddocks in winter prevents soil compaction but also impacts positively on pasture growth rates and water quality. The different costs associated with the use of a standoff pad need to be allocated to the services they impact, including the provision of support, food, or the filtering of nutrients.

6.2.2.4 Replacement cost approach:

The Replacement cost approach (RCA) is a cost-based method that estimates the economic value of natural capital and ecosystem services by using the costs of replacing or restoring damaged ecosystem goods or services to their original state or productivity, using market goods. These costs are used as a measure of the benefits obtained from the intact ecosystem

goods or services (Defra, 2007). RCA tends to implicitly focus on long-term impacts by valuing a permanent change (loss or damage) in natural capital stocks (Vesely, 2006).
The application of this method requires a number of conditions (Shiferaw et al., 2005):

- The magnitude of the damage is measurable,
- There are no secondary benefits associated with the replacement expenditure,
- The substitute provides functions similar to the lost ecosystem service,
- The substitute is the least cost option,
- Affected individuals would be willing to incur these costs if the ecosystem services were no longer provided.

Kim and Dixon (1986) evaluated the benefits of proposed new soil management techniques in Korea, using RCA. Since farming moved into the hilly upland areas due to urban growth and industrial development, poor soil management techniques and errors in field layout and construction have been leading to soil erosion on upland areas. The costs of physically recovering and replacing lost soil, nutrients, and water were taken as a measure of the minimum benefits from preventing erosion and resulting soil, nutrient and water losses. The authors calculated the costs of new preventive management techniques, as well as the cost of compensation, including soil replacement, nutrient replacement, and mulching. The study found that the cost of new management techniques was about half the replacement cost, indicating the proposed preventive steps were worth implementing (Kim and Dixon, 1986).

RCA is widely used because it has modest data requirements. Moreover it is often easy to find market data as estimates of replacement costs (Table 6.6) (Defra, 2007). RCA enables individuals to compare different scenarios to replace the damaged good or service. It also has potential for adjustments if the provision of the good or service keeps changing, since it is based on measures of goods and services.
When using RCA, identifying a replacement for the ecosystem service may not always be possible. Furthermore, the status before damage must be well defined, because it can significantly influence the conclusions on the costs of different scenarios (Vesely, 2006). Thereby, RCA efficiency is subject to the understanding the study designers have of the problem (Table 6.6). Cost assessments can be influenced by factors not related to the good or service being valued, e.g. the full cost of market goods used to replace or restore the service depends on market prices. Moreover, the scenario chosen for replacement might be associated to some externalities not accounted for by the RCA, e.g. repairing damage to soil structure might incur more nutrient losses because of increased drainage. In this case, RCA underestimates the real value of the service lost (Table 6.6).

Table 6-6: Replacement cost approach versus criteria for soil valuation.

Criteria	Replacement cost
Data requirements	low to medium
Data availability	high to medium
Cost of implementation	low
Time line	low
Complexity	low
Subjectivity to cognitive burden	yes
Subjectivity to joint production	yes

RCA is applicable to the valuation of soil services because a number of techniques and human-made systems already exist to mitigate for the loss or damage of soil natural capital and ecosystem services, especially for agricultural activities. When using RCA on farming systems, it is important to be aware that the mitigation strategies employed by farmers can impact on more than one service, therefore their costs should be allocated. Most of the time, mitigation strategies are replacing, or protecting, a service, e.g. a standoff pad compensates for a loss of soil structure but also prevents further damage.
Hence, one needs to be careful not to double count values when using RCA in association to other valuation methods like for example defensive expenditures.

6.2.2.5 Provision cost approach:

The provision cost approach (PC) can be considered as a variant of RCA, but PC does not refer to the replacement or restoration of the ecosystem service in-situ, but to costs of providing the damaged service through alternative means (Shiferaw et al., 2005). This method tries to value the resource in question using actual costs incurred to produce the required good or service. For example, wetlands which provide flood protection may be valued through the cost of building man-made defences of equal function. Since wetlands provide a range of ecosystem services, this costing would be a minimum estimate of the value of a wetland (Defra, 2007). PC relies on the existence of human-made systems and techniques but also markets for major inputs used in the production of the environmental good or service (Vesely, 2006).

The case of the Catskill and Delaware watershed, providing natural unfiltered water to New York City, is a good example of PC. Ninety percent of the city’s water is from the 1,600 sq. mile watershed. The natural filtering abilities of New York’s ecosystems, wetlands and waterways were being threatened by development, runoff from agricultural lands and impervious surfaces, and discharges from wastewater treatment plants at a time when the city faced the potential major investment in a new water treatment facility. New York City chose to implement a comprehensive watershed protection program to preserve and restore natural filtration services as a more cost effective means of maintaining water quality than building a water treatment plant. Watershed management measures included land acquisition (New York City purchased 355,000 acres of land in the watershed between 1997 and 2007) water quality monitoring, disease surveillance, as well as upgrading wastewater treatment plants. The restoration cost to New York City of watershed protection programs was approximately $1.3 billion, whereas investment in a new treatment facility would have cost $6-$8 billion for construction plus $200-$300 million for operation and maintenance costs of the plant. The wastewater treatment service provided by the watershed was valued on the basis of the cost to build and maintain a new treatment facility.

The benefit of the PC is that it uses actual cost outlays which are usually available and robust. PC only gives an estimate of the value of the service, and is only valid if the man-made alternatives are equivalent in quantity and magnitude to the natural functions (Table 6.7). Also, PC requires the existence of markets for the major inputs, which might not be available (Defra, 2007). PC is valid if the alternative considered is the ‘least-cost’ available and individuals are willing to incur the costs to replace the good or service (Defra, 2007). Therefore, PC is subjective to the understanding and willingness of the individuals concerned. PC, like other cost-based approaches, does not measure direct utility (Table 6.7).

Table 6-7: Provision cost approach versus criteria for soil valuation.

Criteria	Provision cost
Data requirements	low to medium
Data availability	high to medium
Cost of implementation	low
Time line	low
Complexity	low
Subjectivity to cognitive burden	yes
Subjectivity to joint production	no

PC is applicable to the valuation of soil services as a number of techniques and human-made systems already exist to provide soil services by other means. For example, hydroponic systems reproduce the provision of food from soils, by providing support, water and nutrients for crops to grow. The cost of a hydroponic system can therefore be used as a lower bound value for the provisioning soil services behind plant growth, but not the other services soil offers.

When choosing cost-based approaches (defensive expenditure, replacement costs, provision costs) to value ecosystem services, the value of built capital is often used as a proxy for the value of the service. Most authors do not annualise the value of the built capital utilised but instead use it as a whole (Costanza et al., 1997; Kim and Dixon, 1986). Annualisation is used as a rule in benefit-cost analysis, and therefore should be also used for the valuation of ecosystem services. Failing to do so is not in line with good accounting and economic theory.
Moreover, the value taken by the discount rate used to calculate annualised value of built capital is also an important but very controversial issue surrounding BCA for the environment (Pearce et al., 2006). The greater the discount rate adopted, the greater the devaluation of future costs/benefits. It has been suggested that low discount rates should be adopted for projects that will greatly affect future generations. The Stern Review on the Economics of Climate Change (Stern, 2007) is a report released for the British government in 2006 which discussed the effect of global warming on the world's economy. It was the first official study using a discount rate of 3% instead of the usual 10 % used, and was therefore extremely controversial. Nonetheless, this document had a great impact on the wider public.

The next two methods presented are revealed preferences techniques, which can be used to value non-tradable goods and services indirectly through marketed goods and services that embody their values.

6.2.2.6 Hedonic pricing method:

The Hedonic Pricing Method (HP) is based on the characteristics theory of value: e.g. an individual's utility for a good is based on the attributes of this good (Garrod and Willis, 1999). HP seeks to find a relationship between environmental characteristics (view levels, air quality) and the price of properties (house, farm land) (Hanley and Spash, 1993). The value of the environmental component (good or service) can therefore be captured by modelling the impact of all possible factors influencing the price of the property (Defra, 2007). The characteristics theory of value states that any given unit (e.g. house) within a commodity class (e.g. housing) can be described by a vector of characteristics. The price of a given unit is a function of these characteristics. HP identifies environmental goods and services as elements of a vector of characteristics describing a market good (Hanley and Spash, 1993).
Two data categories are required for a hedonic pricing study: the specific and the local. The specific data relate to observed property transactions including structural and locational information and details of the purchase such as price, date and particulars of the purchaser. The local data contain details of neighbourhood, amenity, environmental and socio-economic

factors in the area where the property transaction occurs. Some of these data may be more difficult to source.
The choice of explanatory variables (characteristics) in determining the price of the marketed good is crucial. The HP equation can be estimated using ordinary least squares. Implicit prices for each characteristic can be calculated with this equation. The implicit price of a given environmental characteristic is a measure of the value of the marginal change in the environmental quality variable or the WTP for the improvement.

Hedonic pricing can measure direct and indirect use value (Defra, 2007). Moreover, weak complementarity is also assumed, which means that if the level of purchase of the private good is zero, then the marginal WTP for environmental quality is also zero. That's why HP is incapable of estimating non-user values and can only be implemented with environmental quality changes reflected by house pricing (Hanley and Spash, 1993).

HP can be used to examine the effect on property values of issues like:

- Environmental risk, e.g. earthquake damage in residential areas,
- Landscape and water quality: changes in landscape or water quality due to changes in environmental or agricultural policy,
- Environmental protection: policies which protect coastal zone and limit the amount of land available for housing,
- Urban amenity: the presence of urban amenity areas like parks, lakes or green belts,
- Agricultural land values: effect of loss of soil quality and erosion on land values,
- Air and noise pollution,
- Social factors: impact of racial discrimination, effects of schools proximity, and urban railways...

Samarasinghe and Greenhalgh (2009) determine the influence of soil natural capital stocks on farmland values, in the Manawatu catchment (New Zealand). They considered four inherent soil attributes (potential rooting depth, particle size, drainage and profile available water) as well as other attributes (farm size, climate, topography, location and geography) on 4,516 individual farmlands. They found that farmlands with higher potential rooting depth (PRD) were valued higher, on average. WTP was approximately 5% of the average per hectare farmland value to avoid reducing PRD by 25 cm. Similarly, farmlands with higher profile total available water (AW) were valued higher. On average, WTP was approximately 3% of the average per hectare farmland value to avoid reducing AW by 20 mm. On average, compared to farmlands with loamy soil particle size class, farmlands with sandy soil particle size class were valued 8% lower (Samarasinghe and Greenhalgh, 2009), which shows that soil natural capital does influence farmland values.

The hedonic pricing method is useful to investigate the relationships between property prices and environmental attributes and to elicit marginal values for these attributes but has a number of limitations. It has high data requirements and can be very time and resource consuming (Table 6.8). Moreover, several errors can occur when implementing the HP method (Hanley and Spash, 1993):

- Omitted variable bias: if a variable that has significant effect on house pricing and is correlated with some of the included variables is omitted, this will influence the coefficients of the estimated variables and lead to biased estimates for the implicit prices.
- Multicollinearity: Several of the variables included in the hedonic price equation may be correlated with each other. This can result in biased coefficient estimates.
- Market segmentation: Housing markets are often segmented because of exogenous factors (ethnic composition, income, rental versus owner).
- Restrictive assumptions: HP gives an accurate estimate of value of environmental quality only if all buyers are informed of environmental quality level at every possible housing location; all buyers are able to move to utility maximising position; the housing market is in equilibrium. However, these assumptions will never fully describe reality.
- As in all statistical methods, certain statistical problems can affect the results. These include choice of the functional form used to estimate the demand curve, choice of the estimating method, and choice of variables included in the model.

Table 6.8 presents the results of HP against the criteria for soil valuation.

Table 6-8: Hedonic pricing versus criteria for soil valuation.

Criteria	Hedonic pricing
Data requirements	high
Data availability	medium to low
Cost of implementation	medium to high
Time line	medium to high
Complexity	medium to high
Subjectivity to cognitive burden	yes
Subjectivity to joint production	yes

The selling price of farmland can be used in HP, instead of the price of houses which enables access to the valuation of soil natural capital stocks and the ecosystem services they provide (Samarasinghe and Greenhalgh, 2009). However, the use of some production technologies and farming practices to overcome soil limitations and mitigate soil degradation hide differences between soils and may bias HP estimates. For example, two farmlands with very different soil natural capital might sell at the same price even if one of them needs a lot of added capital to reach the same level of production (milk solids per hectare) as the other one.

6.2.2.7 Travel cost method

The Travel Cost Method (TCM) seeks to place a value on non-market environmental goods by using consumption behaviour in related markets (Hanley and Spash, 1993). TCM is predominantly used in outdoor recreation modelling (fishing, hunting, boating, and forest visits). It is a survey-based technique that uses the cost of a trip taken by an individual to a recreation site (e.g. travel costs, entry fees, opportunity cost of time) as a proxy for the recreational value (Defra, 2007). The costs (travel costs, entry fees, on-site expenditure, outlay on capital equipment) of consuming goods and services associated with the use of a recreational site, are used as a proxy for the value of the services.

Questionnaire surveys are used to collect data on the number of visits that a household or individual makes to a site and the cost of gaining access. Travel costs depend on several variables including distance costs (how far the individual has to travel to visit the site and the cost per mile of travelling), time costs (how long it takes to get to the site and the value of an individual's time) and entrance fees to the site. Travel costs are included in a trip generating function (TGF) which predicts how many visits will be undertaken by any individual to a particular site. Socio-economic characteristics (income, age, education) would be also included in the TGF as well as information on the type of trip (holiday, day-trip) (Hanley and Spash, 1993).

TCM captures direct and indirect use value (Defra, 2007). Visitors to the site may hold non-use values, but these cannot be assessed using this valuation method. The method assumes weak complementarity between the environmental asset and consumption expenditure, this is why TCM cannot estimate non-use values (Hanley and Spash, 1993).

For example, tropical rainforests have many values beyond the timber they hold and their potential as sites for agriculture and cattle grazing. Menkhaus and Lober (1996) used the travel cost method to examine one of the additional values of rainforests, places for ecotourism. The study determined the value that tourists from the U.S. place on Costa Rican rainforests as ecotourism destinations, using the Monteverde Cloud Forest Reserve as a sampling site for tourism to Costa Rica's protected areas. Data were collected by a survey of 240 U.S. tourists.

Menkhaus and Lober (1996) found that the value placed by US ecotourists on visiting Costa Rican rainforests was $1150 per visit.

The strength of the TCM is that it is base on observed behaviour and deals with revealed preference data. But TCM has a number of limitations. TCM is generally limited to recreational benefits. The method is data intensive and interviews are time consuming (Table 6.9). Moreover, difficulties arise when data are complex. The choice of the dependent variable (visits from a given zone in visits per capita or visits made by a given individual in visit per annum) will influence substantially the consumer surplus estimates (Hanley and Spash, 1993). Moreover, it is very hard to treat multipurpose, multisite and multi-length trips because their total costs cannot be attributed to one single site and have to be allocated (Hanley and Spash, 1993; Vesely, 2006). Also, holiday-makers have to be distinguished from residents and day-trippers so as not to bias estimates. Finally, time is expended both in travelling to a site and whilst recreating on the site: as a rare commodity, time has an implicit price (opportunity cost) that should be estimated.

Table 6.9 presents the results of TCM against the criteria for soil valuation.

Table 6-9: Travel cost method versus criteria for soil valuation.

Criteria	Travel Cost
Data requirements	high
Data availability	medium to low
Cost of implementation	medium to high
Time line	medium to high
Complexity	medium to high
Subjectivity to cognitive burden	yes
Subjectivity to joint production	yes

It seems difficult to apply directly TCM to the valuation of provisioning and regulating ecosystem services from soils but TCM could be used to explore cultural services like recreation.

At the farm scale, this could be picked up by the differences in productions costs associated with the spatial distribution of natural capital. The extra costs associated with moving production to a distant part of the farm with good soil natural capital could be used: fuel extra-expenses, price of getting irrigation onto a distant land, cost of taking the animals to the area, but this would be a much stretched version of TCM.

6.2.3 Techniques using stated preferences:

Hypothetical markets are simulated markets built so individuals can express their WTP for a non-traded environmental good or service (Pearce and Turner, 1990; Pearce and Barbier, 2000). Contrarily to the techniques explained so far, the techniques using hypothetical markets use individuals stated preferences and not revealed preferences. Techniques using hypothetical markets are, in theory, able to capture all elements of the total economic value even non-use values.

6.2.3.1 Contingent valuation method

The contingent valuation method (CVM) seeks to discover how people would value certain environmental changes, where actual market data are lacking, by directly questioning a sample of the population concerned. The aim of CVM is to obtain an estimate of the economic value of a change in the level of provision of an ecosystem good or service (public good) not traded in markets (Hanley and Spash, 1993), which can then be used in a benefit cost-analysis (Mitchell, 1989). Theoretically, CVM can capture all the elements of TEV, but in practice it is difficult to assess many of the different use and non-use values (Defra, 2007).
CVM is a survey-style approach. It constructs a hypothetical market that involves the description of alternatives to an environmental problem via a social survey questionnaire. Respondents answer questions regarding their willingness to pay (WTP) and/or willingness to accept compensation (WTA) for a particular environmental change (Defra, 2007; Hanley and Spash, 1993). The WTP illustrates how much the respondent will be willing to pay to have an environmental improvement go ahead or to prevent a deterioration in environmental quality. The WTA illustrates how much the respondent will be willing to accept to go without the improvement or to put up with deterioration (Hanley and Spash, 1993).

The application of CVM includes several steps (Garrod and Willis, 1999; Hanley and Spash, 1993; Mitchell, 1989):

1. Define the valuation problem. What are the services and who is the relevant population?
2. Preliminary decisions about the structure of the survey itself: size of the budget, sample size, survey type (mail, phone or in person).
3. Survey design. This is the most important and difficult part of the process and includes several stages: Different focus groups help develop specific questions for the survey, decide what kind of background information is needed and how to present it and what type of payment mechanisms to use for the valuation. The survey is then pre-tested

and feed-back included to ensure that the survey is understood and people answer in a way that reveals their WTP for the service.

4. The survey is implemented on a random selection of the relevant population and bids obtained from respondents through a bidding game, closed-ended referendum, open-ended questions or payment card.
5. Compile, analyze and report findings. Data are analyzed using statistical techniques (estimate mean and median WTP/WTA and bid curves). Biases are looked for and eliminated if possible. WTP per capita are then extrapolated in order to calculate the total benefits from the site.

CVM has been used widely in a number of domains. For example, Colombo et al (2006) estimated the benefits of programmes to mitigate the off-site impacts of soil erosion for a watershed in Andalusia, Spain. They used two stated preference methods, choice experiments and contingent valuation, to obtain estimates of the social benefit from soil erosion reductions under two different methodologies. They conducted face to face interviews and asked respondents their maximum WTP to have, in 50 years time, an environmental situation characterised by a reduction of landscape desertification, a 'medium' quality of the surface and ground waters, a 'medium quality of flora and fauna', the creation of 100 jobs and the area of implementation of 330 km^2 versus the situation at present. The survey was administered to 345 people and 252 responses were gathered for analysis. Bids were expressed as tax payments. The contingent valuation design included an attempt to reduce bias by reminding respondents about substitutes. Results were used to suggest upper limits on per hectare payments for soil conservation programmes (Colombo et al., 2006).

The CVM strength is its flexibility and the fact that it is one of few valuation methods able to estimate non-use values. A number of guidelines of good practice are available in the literature on the use of CVM, as well as methods to check the validity of the results.
There is considerable controversy over whether CVM adequately measures people's WTP for environmental quality, because most people are untrained in placing dollar values on environmental goods and services. CVM uses stated preference data which are generally less accurate than revealed preferences. Conducting a CVM can be expensive, in part because it is also very time-consuming (Table 6.10). This method and the WTP estimates it provides are sensitive to the way information is provided to respondents, including the framing, payment vehicle, assumed functional form and so on. For example, the choice of the welfare measure (WTA or WTP) can influence greatly the results of the study. Quite often, when respondents are asked their WTA for a change in the level of environmental service the amounts they state are much higher than if they were asked their WTP for the same environmental service. Also,

CVM is subject to a number of biases that can be categorised as either psychological or statistical including strategic bias or information bias from the respondents, or design bias resulting from the way information is presented, the order of question, question format, amount and type of information presented, and choice of bid vehicle (Hanley and Spash, 1993).

Table 6.10 presents the results of CVM against the criteria for soil valuation.

Table 6-10: Contingent valuation method versus criteria for soil valuation.

Criteria	Contingent valuation
Data requirements	high
Data availability	medium to low
Cost of implementation	high
Time line	high
Complexity	high
Subjectivity to cognitive burden	yes
Subjectivity to joint production	yes

The use of CVM for soil services valuation is rendered unlikely by the fact that CVM is very time and workforce consuming. Moreover, services provided by soils are less recognised and understood by the general public, than services provided by other ecosystems (e.g. forests, rivers). Care must therefore be taken in designing surveys, describing the hypothetical market and choosing the sample of population to interview if using this approach to value soil services in order to eliminate biases from the respondents' lack of knowledge.

6.2.3.2 Random utility theory and choice modelling:

The random utility theory has an economic and behavioural basis. It postulates that utility is a latent construct that exists in the mind of the consumer but cannot be observed directly by the researcher (Louviere, 2001). Random utility models (RUM) use discrete choices to reveal consumer's utility. Individuals are made to choose between alternatives that differ in their quality, quantity and characteristics (a discrete choice) (Pearce and Barbier, 2000). The individual doesn't maximise their well-being by speaking out their preferences (continuous choice), he/she chooses the alternative which offers them the highest level of satisfaction (utility) out of options available. The random utility theory assumes, as Neoclassical Economics theory, that individuals have a perfect discrimination capability and will pick the option with the characteristics that they prefer, out of all possibilities. Individuals have to make tradeoffs between the characteristics of an option. RUM are particularly suitable to estimate the value of particular features like ecosystem goods and services of interest (Pearce and

Barbier, 2000). RUM can only capture direct use value (Defra, 2007). RUM have been used as an extension of TCM: individuals have to choose between different trip options instead of expressing the number of trips taken in a year to a specific location. For example, Whitehead (2006) estimated the value of king mackerel bag limit changes with both stated (contingent valuation method) and revealed (random utility model) preference methods.

Choice modelling (CM) is based on RUM and therefore also has economic and behavioural basis. Applications of CM follow directly from random utility theory (theory of paired choices) (Louviere, 2001). While CM has been widely used in the market research and transport literatures, it has only relatively recently been applied to other areas such as the environment (Pearce et al., 2006). Like CVM, CM is a stated preference method which can, in theory, capture all elements of TEV, including non-use values (Defra, 2007; Pearce et al., 2006)

Choice modelling is a family of survey-based methodologies for modelling preferences for goods, where goods are described in terms of their attributes (multi-dimensional good) and of the levels that these attributes take, like with RUM. Respondents are presented with several alternative descriptions of a good, differentiated by their attributes and levels, and are asked to rank the various alternatives, to rate them or to choose their most preferred (Pearce et al., 2006). One of the attributes of the good is the price/cost, which is used to indirectly recover people's WTP from their rankings, ratings or choices (Pearce and Barbier, 2000; Pearce et al., 2006).

There are four main variants of the CM approach corresponding to different ways of measuring preferences (Table 6.11). These techniques differ in the quality of information they generate, complexity and ability to produce WTP estimates consistent with the usual measures of welfare change (Pearce et al., 2006).

Table 6-11: Choice modelling variants (adapted from Pearce et al., 2006)

Approach	Tasks	Advantages
Choice experiments (CE)	Choose between two or more alternatives (where one is the status quo).	Consistent with utility maximisation and demand theory when a status quo option is included in the choice set.
Contingent ranking	Rank a series of alternatives.	Ranking data provides more statistical information than CE and therefore more precise implicit prices or measures of WTP.
Contingent rating	Rate each scenario individually on a scale of 1-10.	Ratings are transformed into a utility scale assuming that ratings are comparable across individuals (which may not be valid)
Paired comparisons	Choose the preferred alternative, and indicate the strength of the preference on a similar scale.	Combination between choice experiments and contingent rating.

Wang et al. (2007) estimated using choice modelling the non-market values of the environmental benefits derived from the Conversion of Cropland to Forest and Grassland Program (CCFGP) in the loess plateau region of north west China, both on-site and off-site in Beijing.

Table 6-12: Level of attribute (from Wang et al., 2007)

Attribute	Base level	Alternative levels
Payment per annum for 10 years (CNY)	0	50, 100, 200
Sandstorm days per year	22	20, 18, 16
Landscape (vegetation cover)	10%	20%, 30%, 40%
Water quality (billion tons of annual sediment discharge)	100%	10% less, 15% less, 25% less
Plant species present	1600	1900, 2200, 2400

One-on-one interviews were conducted with five versions of the survey questionnaire (Table 6.12). The payment per annum corresponded to the financial support for the farmers involved in the program to continue growing trees and grass on the loess plateau even after the financial support from the government in implementing the CCFGP has ceased. The average WTP per respondent household in Beijing (off site) was CNY882.56 (USD109.44) each year for the environmental improvements on the loess plateau, whereas WTP onsite was much lower (CNY342.56 (USD42.48) in Xi'an and CNY388.08 (USD48.12) in Ansai) (Wang et al., 2007).

CM is often used instead of CVM because it possesses some advantages. A clear strength of CM lies in the ability to value multi-dimensional environmental changes and trade-offs (Pearce and Barbier, 2000; Pearce et al., 2006). CM, like RUM, is based on observed behaviour and can be used to value policy proposals. Even if it is possible to measure the marginal value of changes with CVM, it would be more costly and troublesome. CM is easier to generalise than CVM, and therefore more appropriate from a benefits transfer point of view (Pearce et al., 2006) or to combine with revealed preferences data (Vesely, 2006). Some CM variants, such as choice experiments, provide much more information than CVM as respondents express their preference several times for a valued good over a range of prices/costs. Moreover, CM may be better at estimating relative values than absolute values, and even if the absolute dollar values estimated are not precise, the relative values elicited are likely to be valid and useful for policy decisions. Since CM relies on ratings, rankings or choices, it avoids an explicit elicitation of respondents' WTP like in CVM, which might be easier for the respondents (Pearce and Barbier, 2000). CM thereby minimises many of the biases that can arise in open-ended CVM studies. Although CM has been widely used in the field of market research, its validity and reliability for valuing non-market commodities is still largely untested.

Several problems can arise with CM:

1- CM studies reflecting intricate environmental changes are themselves complex to design and to analyse (Pearce et al., 2006). They require sophisticated statistical techniques to estimate WTP. To manipulate the data and estimate the model is very time consuming.

2- A major disadvantage of CM is the cognitive difficulty associated with multiple complex choices or rankings between bundles with many attributes and levels, because there is a limit to how much information respondents can meaningfully handle while making a decision (Pearce et al., 2006; Vesely, 2006). While stated preference techniques like CVM and CM can, in principle, be applied to any context, they can be limited by the ability of respondents to understand the nature of the service. When the relationship between cause and effect for complex ecosystem services is not well understood by respondents, they may well not appreciate the impact that an ecosystem service might have on their wellbeing (Defra, 2007). Learning, fatigue or correlation between responses can all introduce bias in the respondent's answers, leading to options being chosen that are good enough although not necessarily the best (no utility-maximisation) (Pearce et al., 2006). It is important to incorporate test of respondents understanding in the CM studies in order to detect these problems.

3- As with all stated preference techniques, welfare estimates obtained with CM are sensitive to study design (Manski, 1977; Whitehead, 2006). There may be additional attributes of the good not included in the design, which generate utility. If the CM does not include substitutions opportunities or alternative attributes (Manski, 1977) that the respondents would have chosen, WTP estimates could be biased. The levels chosen to represent the attributes, the way choices are presented may impact on the estimations of consumer's surplus and marginal utilities (Pearce and Barbier, 2000; Pearce et al., 2006).

Table 6.13 presents the results of CM against the criteria for soil valuation.

Table 6-13: Choice modelling versus criteria for soil valuation.

Criteria	Choice modelling
Data requirements	high
Data availability	medium to low
Cost of implementation	medium to high
Time line	medium to high
Complexity	high
Subjectivity to cognitive burden	yes
Subjectivity to joint production	yes

Ecosystems like soils provide complex multi-attribute services to people. Most of the time ecosystem goods and services total value is not observable, because they are not traded on markets. Nevertheless, each component (natural capital stock) of an ecosystem has a unique shadow value (Farber and Griner, 2000). The usefulness of CM to economists is that it provides a methodology for obtaining relative valuations of each attribute (Farber and Griner, 2000) used to estimate the total value of the change in provision of the environmental good (Pearce et al., 2006). It seems like the choice experiment variant, and to some extent contingent ranking, has become the dominant CM approach in applications to valuation of environmental goods (Pearce et al., 2006). Therefore, CM would be a useful method to value soil services, since soils are complex and multidimensional ecosystems. With soils, the cognitive burden problem could be of consequence, if the respondents have no notions of soil science and the study wants to quantify shadow values of soil properties as in this study. Even if CM is less expensive and time consuming than CVM, survey-based methodologies are complex to design, analyse and also time consuming.

6.2.3.3 Group valuation:

Group valuation methods, also called deliberative or participatory methods or discourse-based methods, are still not typically engaged in the process of ecosystem services valuation. However, it is argued (Wilson and Howarth, 2002) that conventional valuation techniques poorly address questions of social equity raised by the valuation of ecosystem goods and services, because they are based on individual preferences and utility maximization alone. Since ecosystem goods and services are public goods, their valuation should result from a process of free and open public debate, involving the fair treatment of competing social groups, because difficult social decisions must be made to achieve a sustainable ecological future (Wilson and Howarth, 2002).

Group valuation methods are based on principles of deliberative democracy and the assumption that public decision making should not result from the aggregation of separately measured individual preferences, but from open public debate (Wilson and Liu, 2008). Therefore, these methods should be well suited to the valuation of ecosystem goods and services.

A great variety of groups exist, but the most common are (Defra, 2007):

- Focus groups: they aim to discover the positions of participants regarding an issue or set of related issues, and/or explore how participants interact when discussing issues.
- In-depth groups: they are similar to focus groups, but they are less closely facilitated, and focus on how the group creates discourse on the topic.
- Citizens' Juries: they are designed to obtain carefully considered public opinion on a particular issue or set of issues. A sample of citizens considers evidence from experts and other stakeholders, hold group discussion on the issue, and give an informed opinion that is supposed to reflect public opinion.
- Deliberative forums: they spend time listening to the opinions of others (experts, stakeholders or general public) with the aim of forming a collective view.

These methods help to understand peoples preferences and the process of decision-making. They are more about the formalization of procedures and conditions for achieving free and fair deliberation between citizens, than on unanimity or collective action among all citizens (Wilson and Howarth, 2002). These methods tend to explore the group processes surrounding environmental decision-making and how opinions are formed or preferences expressed, not necessarily in terms of explicit economic values. They do not typically fit into the formal process of economic valuation that aims to capture the TEV of ecosystem services (Defra, 2007). However, when they do include reflexions on monetary values they can be considered as stated preference methods, and used to complement and compare with results from more traditional valuation methods used in cost–benefit analysis (Wilson and Howarth, 2002).

The strength of group valuation methods reside in the fact that they use a more constructive approach to value elicitation than classical valuation techniques. By sharing information, members of a group have a higher probability of making more informed decisions than would any single individual (Wilson and Howarth, 2002). A number of authors (Howarth and Wilson, 2006; Wilson and Liu, 2008; Wilson and Howarth, 2002) argued that since the valuation of ecosystem goods and services concerns the allocation of public goods and affects social well-being, it may be well suited to discourse-based methods, which provide fair and equitable environmental value formation. These methods, when they explore explicit economic values for ecosystem goods and services, can be very complementary of classical valuation methodologies (Wilson and Howarth, 2002).

However, a number of problems remain with group valuation methods. First, they are very complex methods that are very time and resource consuming (Table 6.14). It can be very difficult to generate results because of group dynamics. For example, small groups may not be very efficient at sharing information, or there might be issues with social status and power differences that exist between different group members, and interpersonal conflict (Wilson and Howarth, 2002). Moreover, group valuation methods should be designed with a wide variety of citizens/stakeholders and facilitator/moderators in order to make sure that the group is exposed to a wide range of points of view and more diverse resources regarding the environmental valuation task (Wilson and Howarth, 2002). Finally, when group valuation methods fail to generate estimates of explicit economic values, there is no direct means of comparing them with conventional methods of environmental valuation (Wilson and Howarth, 2002).
Table 6.14 presents the results of CA against the criteria for soil valuation.

Table 6-14: Group valuation versus criteria for soil valuation.

Criteria	Group valuation
Data requirements	high
Data availability	medium to low
Cost of implementation	high
Time line	high
Complexity	high
Subjectivity to cognitive burden	yes
Subjectivity to joint production	no

Group valuation techniques like deliberative forum could be well suited to the valuation of soil services. It would be very useful to reach a collective view on the value of soil services after a group listened to different stakeholders dealing with soils like farmers, soil scientists and policy makers.

6.2.4 Benefit transfer:

All the valuation techniques mentioned above can be used in benefit transfer (BT). Benefit transfer consists of taking a value of a non-marketed good estimated in an original or primary site ("study site") and using this estimate as a proxy for values of the good in another site ("policy site") (Defra, 2007; Pearce et al., 2006).

BT studies are increasingly being used to inform new policies, especially environmental policies. The attraction of BT is to avoid conducting a primary study (which may be both time consuming and costly), if existing values are robust enough to be transferred and applied in a new policy assessment (Defra, 2007; Pearce et al., 2006).

There are a number of ways to transfer values from a study to the policy context (Table 6.15). Simply adjusted WTP transfers are easy to implement, once suitable original studies have been identified, but usually fail to capture important differences in characteristics between the original and new policy site. Generally, the more sophisticated the approach the better, in terms of accuracy of the transfer (Pearce et al., 2006). This is why meta-analysis is usually preferred when enough studies are available.

Table 6-15: Types of benefit transfer methods (Pearce et al., 2006). WTP: Willingness to pay.

BT types	Method
Unadjusted WTP transfer	The mean or median WTP of study site is applied to policy site.
WTP transfer with adjustment	WTP values are adjusted by income per capita.
WTP function transfer	The WTP at the study site is described as a function of physical features of the site and socio-economic (and demographic) characteristics of the population.
WTP meta-analysis	A statistical analysis of summary results of a (typically) large group of studies is used to describe WTP.

There are a number of steps involved in undertaking a benefit transfer study (Defra, 2007; Pearce et al., 2006) including:

- A literature review to find appropriate valuation studies to apply to the policy context. A number of databases are available (e.g. the Environmental Valuation Reference Inventory: EVRI).
- Select appropriate studies: the study site should be as close a match as possible to the policy context.
- Adjust the WTP values with one of the methods described above (Table 6.15).
- Aggregate the estimated WTP values over the population relevant to the policy context.

There are a number of advantages of using BT. First depending on the valuation technique used in the original studies used; BT is potentially able to capture all elements of TEV, including non-use values. BT is usually cheaper and less time consuming than doing a new original study. With BT being increasingly used, there are best practice guidelines and protocols available.
One of the limitations of BT is that it has a limited applicability based on existing studies. The validity and accuracy of benefit transfer is a key issue (Colombo et al., 2005; Pearce et al., 2006). Analysts still need to improve their understanding of when transfer works and when it does not, as well as to improve the accuracy of BT because it introduces subjectivity and greater uncertainty into appraisals. The question is whether the added subjectivity and uncertainty around the transfer is acceptable and whether the transfer is still informative (Pearce et al., 2006). Careful consideration also needs to be given to ensuring there is no double-counting of benefits when using BT. This can occur when a number of BT values are applied that relate to services that overlap (Defra, 2007).

BT seems to be the key to a more practical use of environmental values in policy-making. However, there are not many studies available around the value of soil services and natural capital. Therefore it would be difficult to apply BT rigorously or use values obtained for different ecosystems to the valuation of soil services even if the values concern the same services. Notwithstanding this point, even though BT methods might not be able to be directly used in this thesis to estimate the value of soil ecosystem services, literature values could be used to ‘double-check’ the estimates obtained in this thesis.

6.2.5 Final evaluation of neoclassical economic valuation methods:

To value soil services, neoclassical economic valuation was chosen for a number of reasons:

- neoclassical economic valuation methods are widely spread and very well accepted by mainstream economists and the policy community;
- putting economic values on soil ecosystem services makes it easy to compare their provision with marketed goods in common easily understood units (e.g. $);
- there is a lot of literature on neoclassical economic valuation methods, which we could refer to in this thesis;
- The data requirements are not too onerous, as compared with some of the non-neoclassical methods (e.g. ecological pricing or multicriteria analysis).

The economic valuation methods reviewed in this chapter and assessed against the criteria developed to rank their utility for soil services valuation are summarised in Table 6.16. In addition to these criteria, two other challenges need to be acknowledged before valuing soil services.

Because of the complexity of soil functioning, simple measures of soil properties have limited value to gain understanding of the properties and processes behind soil services. To measure adequately the provision of services from soils, sensitive biophysical indicators, complex enough to reflect the dynamics of the provision of each service are needed. For the valuation of soil services, economic values then need to be linked to these biophysical indicators to create a dynamic value of the services provided. For these reasons, the valuation method chosen needs to be simple enough to ensure unnecessary complexity is not added to the whole quantification/valuation exercise.

Secondly, this study focuses on the provision of ecosystem services from soils at the farm scale; the valuation method chosen also needs to be relevant and accurate at this scale.

Table 6-16: Summary of valuation techniques versus criteria for soil valuation.

Valuation methods	Criteria for soil services valuation						
	Data requirement	**Data availability**	**Cost of implementation**	**Time line**	**Complexity**	**Subjectivity to cognitive burden**	**Subjectivity to joint production**
Market prices	low	high	Low	low	low	no	no
Productivity change	medium	high	Medium	medium	medium	no	no
Defensive expenditure	low	high	Low	low	low	yes	yes
Replacement cost	low to medium	high to medium	Low	low	low	yes	yes
Provision cost	low to medium	high to medium	Low	low	low	yes	no
Hedonic pricing	high	medium to low	medium to high	medium to high	medium to high	yes	yes
Travel cost	high	medium to low	medium to high	medium to high	medium to high	yes	yes
Contingent valuation	high	medium to low	High	high	high	yes	yes
Choice modelling	high	medium to low	medium to high	medium to high	high	yes	yes
Group valuation	high	medium to low	High	high	high	yes	no

Amongst neoclassical valuation techniques, techniques using revealed preferences were chosen to value soil services. They include market prices, productivity change, defensive expenditure, replacement cost and provision cost. These methods complied well with the criteria developed for soil services valuation (Table 6.16). Moreover, there are a number of human made infrastructures/management practices that are used commonly by farmers to deal with a lack of natural capital and low ecosystem services provision. Information on the market value of such alternatives is readily available and robust. Moreover, the efficiency of such tools and techniques have been well studied and can be linked to soil natural capital assets and the soil services provided at the farm scale. A discount rate of 10% was chosen to annualise infrastructure costs and used to implement the different revealed preferences techniques. The value of 10% was chosen because it is the value the most commonly used in the literature, enabling the reader to compare our results with other studies. Results using a discount rate of 3%, such as the one used in the Stern report (Stern, 2007), are also presented to provoke discussion.

Techniques using stated preferences would have been more challenging to use for a number of reasons (Table 6-17). First, they require more data that are often not easily available. Because they are based on interviews or group reunions, they are more costly and time consuming. Moreover, they are subject to a number of biases related to respondent's knowledge of the issue in question, understanding of the survey and behaviour. Finally, the analyses of the results of interviews or discussion are usually very complex and subject to a number of biases (Table 6-17).

Based on the analysis in this chapter a pre-selection of methods that could be used for the valuation of each soil service and some details of their implementation are presented in Table 6.17. These methods use the market prices of human made infrastructures or management practices, commonly used by farmers to deal with a lack of natural capital, as proxies for the economic value of soil services.

These methods are only able to measure the use values (direct and indirect) of soil services. It is acknowledged that to access the TEV of soil services, it would be useful to implement techniques using hypothetical markets and stated preferences like choice modelling.

The details of the economic valuation of soil services are presented in the next chapter.

Table 6-17: Possible valuation techniques considered for the valuation of soil services.

Soil services	Possible valuation techniques
Provision of food, wood and fibre	Market price of milk solids produced from pasture
Provision of support to infrastructures	Replacement cost: cost of compaction and foundation building
to animals	Provision cost: cost of building and running a pad for the cows to stand on
Filtering of nutrients and contaminants	Defensive expenditure: cost of mitigation techniques to prevent nutrients and contaminants loss Provision cost: cost of treating contaminated water
Flood mitigation	Provision cost: cost of building dams on the farm to store run-off water
	Replacement cost: restoring artificially soil water holding capacity
C storage and Regulation of GHGs emissions	Market price of carbon
	Defensive expenditure: cost of mitigation techniques to prevent C loss and GHGs emissions
Pest and Diseases regulation	Provision cost: cost of treating the soil against disease or pest
	Defensive expenditures: cost of mitigation techniques to increase natural pests and diseases regulation
Detoxification and recycling of wastes	Provision cost: cost of treating wastes in effluent ponds and fertirrigation
	Replacement cost: cost of mitigation techniques to enhance soil microbial activity and accelerate wastes decomposition

6.3 Conclusion on valuation:

This chapter shows that "the environment" has different types of value and that economic value is only one of them. When it comes to the valuation of ecosystem services in general, and soil services in particular, we see that neoclassical economic valuation is the dominant paradigm in modern-day Economics. However, there is no universal acceptance of the neoclassical theory of value and other methods of assessing the value of the environment exist particularly in the field of Ecological Economics.

This chapter has critically reviewed neoclassical economic valuation techniques for the valuation of soil ecosystem services. It is shown that several techniques using revealed preferences are suitable for the valuation of soil services. They were chosen for this study because they are easy to implement and do not add complexity to the exercise, data are easily available and well suited to valuation at the farm scale. For each soil service, several methods could be used (Table 6.17).

It should be noted that the valuation of soil services is typically a domain where multicriteria analysis could be very interesting to implement, especially when looking at different land-uses which have different criteria in terms of soil services requirement.

The next chapter presents the details of the biophysical quantification and the economic valuation of each one of the services provided by soils under a dairy farm operation.

PART THREE

EMPIRICAL RESULTS FOR DAIRY FARM SOILS

Chapter Seven

Quantification and Economic Valuation of Soil Ecosystem Services

This chapter uses the concepts for the quantification of soil services developed in Chapters Three and Four, and the outputs of the SPASMO model adapted to better represent a dairy operation (Chapter Five), to quantify each of the thirteen ecosystem services (detailed from previous broader categories) provided by a Horotiu silt loam soil under a typical Waikato dairy farm. The economic valuation techniques selected from the review of the literature in Chapter Six are then used to value each of the thirteen soil services.

7.1 Description of the typical dairy farm and base case scenario:

In this chapter, the methodologies described in Chapters Three to Six are used to quantify and value the ecosystem services provided by a Horotiu silt loam (Allophanic Soil) under a typical New Zealand dairy farm operation. The characteristics of the base case typical New Zealand dairy farm include rain fed pastures with no irrigation, no grazing-off, pasture silage made from the farm in spring and fed to the cows as supplements, and no artificial drainage (Table 7.1).

The SPASMO model was used to explore the dynamics of soil properties and processes as they influence the provision of soil services, for the typical dairy farm. SPASMO integrates the soil supporting processes. The extra functions added to SPASMO for this study included pasture utilisation, the impacts of cattle treading on soil structure and pasture growth, stock rotation, the use of a standoff-pad and the calculation of P runoff. They are described in detail in Chapter Five. The SPASMO model runs on a daily time step. Thirty seven years of climate data (1972-2009) were used to explore the impact on soil services of a dairy operation on a Horotiu silt loam. The model runs continuously for thirty seven years, integrating the influence of previous years. The outputs of the model are used to quantify the ecosystem services from the Horotiu soil under a dairy operation. Only the outputs for the years 1975 to 2009 were used, as the model needs a few runs to reach equilibrium. The economic valuation approach adopted to value each of the ecosystem services was based on a critical analysis of all available valuation techniques conducted in Chapter Six.

Table 7-1: Characteristics of the typical dairy farm for the base case scenario.

Characteristics	Data
Effective area (ha)	110
Number of cows	330
SR (cows/ha)	3.0
Milk solids (kg/ha)	900
N fertiliser (kg/ha/yr)	100
P fertiliser (kg/ha/yr)	39
Standoff-pad	No
Soil type	Horotiu silt loam

7.2 General methodology:

The following steps describe the approach developed for the quantification and economic valuation of soil services under a dairy land use:

1- Identify the key soil properties and processes behind each soil service: To determine how soils provide ecosystem services, the soil properties and processes at the origin of the provision of each soil service need to be investigated in details (chapters three and four). Where services depend on dynamic soil properties, the processes driving the changes should be understood (sections 5.3.2 and 5.3.3). External drivers like climate and land use impact on both soil properties and processes and thereby on the flows of soil services. Separating these impacts is important to establish if natural capital stocks are being sustained or degraded (sections 3.2.2, 3.3.2, 4.1.2, 4.2.2, 4.3.2, 4.4.2 and 4.5.2).

2- Analyse the impact of degradation processes on soil natural capital and ecosystem services: Many processes degrade soil natural capital stocks and thereby affect the flows of soil ecosystem services. Knowing where and how degradation processes impact soil natural capital is essential to determine their impact on soil services (sections 3.2.2, 3.3.2, 4.1.2, 4.2.2, 4.3.2, 4.4.2, 4.5.2, 5.3.2 and 5.3.3).

3- Differentiate between natural capital and added or built capital when defining proxies to quantify soil ecosystem services: The definition of each service is crucial to determine a correct proxy to measure it. It is argued here that services not only need to be rigorously identified and defined, but authors should differentiate the part of the service coming from soil natural capital, from that which comes from added or built capital (e.g. infrastructures, inputs such as fertilisers or irrigation water), enabling the contribution of each to be calculated. Proxies to measure each service based on dynamic soil properties need to be based on the part played by the soil in the provision of the service (chapters three and four). Moreover, the soil properties chosen as proxies should be easily measurable and data should be available.

4- Base the economic valuation on measured proxies: the techniques used to value each service should be based on the bio-physical measures of the services and be relevant and appropriate for the chosen scale and land-use (chapter six).

For each of the thirteen soil services considered, a proxy was defined to quantify the service (Table 7.2). Each proxy is based on one or more soil properties (natural capital stock) at the origin of the provision of the service. Each proxy was then calculated from the outputs of the SPASMO model and/or data from the literature (Table 7.2). The SPASMO model was used to explore the dynamics of soil properties and processes regulating some of the soil services, and to quantify services for each of 35 years (1975-2009) using climate records from the Waikato. When reporting the outputs of the SPASMO model or valuation, data are for consecutive years starting in 1975. The quantitative information on each service was then valued using a range of neo-classical economic valuation techniques (Table 7.2). Doing so ensured that the quantification and valuation of each service was as dynamic as the soil properties on which the services depend.

Table 7-2: Overview of the proxies and valuation techniques used for each soil service.

Services (detailed from broader categories)	**Parameters used**	**Proxy for**	**Origin of data**	**Valuation method used**
Provision of food quantity	Pasture yield (kgDM/ha/yr)	Nutrients, water and support provision for plants	Model outputs	Market prices of milk solids
	N fertilisers (kgN/ha/yr)		Set as inputs by scenario	Added value for milk solids
	Native Olsen P	Nutrient levels	Native Olsen P (literature)	Multipliers effect for milk solids for NZ
Provision of food quality	Trace-elements levels	Nutrient levels	Literature	Defensive expenditure (cost of application of trace-elements)
Provision of physical support to humans	Bulk density	Soil compaction	Literature and model inputs	Replacement cost (costs of building foundations)
Provision of physical support to animals	SWC and FC	Macroporosity and drainage	Model outputs	Provision cost (construction and maintenance of a standoff pad)
Flood mitigation	Rainfall and runoff	Macroporosity and drainage	Model outputs	Provision cost (costs of building dams)
Filtering of N	N leaching	Anion storage capacity and drainage	Model outputs	Defensive expenditure (mitigation costs)
Filtering of P	P losses	Anion storage capacity, runoff and drainage	Model outputs	Defensive expenditure (mitigation costs)
Filtering of contaminants	Rainfall and runoff	Infiltration, SWC, drainage	Model outputs	Provision costs (construction costs of a wetland)
Detoxification and recycling of wastes	SWC	Soil conditions for biota activity	Model outputs	Provision costs (costs of construction and maintenance of an effluent pond)
	Timing of dung deposition		Model outputs	
C flows	Net C flows	C cycle	Model outputs	Market prices of C
N_2O regulation	SWC and N_2O emission	Denitrification	Model outputs and IPCC	Market prices of C
CH_4 regulation	SWC and CH_4 oxidation	CH_4 oxidation	Model outputs and literature	Market prices of C
Biological control of pests and diseases	SWC	Soil biota habitat	Model outputs	Provision cost (costs of pesticides)
	Macroporosity		Model outputs	

7.3 Provision of food, wood and fibre (S1):

For a dairy system, the provision of food, wood and fibre from soils (service one = S1) is embodied by pasture production and the production of milk from dairy cows. The service is supported by natural capital stocks including Mp, SWC and N and P status. The use of different species beyond forage plants, including woody species like Pinus Radiata, or fibre production (wool) from grazing animals, is not considered in this study, but could be added.

7.3.1 Quantification of the provision of food:

When quantifying the provision of food from soils, the distinction needs to be made between the part of pasture yield coming from the natural capital stocks, and the contribution from added capital like fertilisers and irrigation. It is important to identify and separate the production levels from these two sources in the quantification of this service. Production technologies and farming practices have been very successful in compensating for the lack of natural capital of many soils.

The SPASMO model was used to calculate annual pasture yield (kg DM/ha/yr) each year for 35 years depending on climate, soil type and farm management. Mean annual pasture production (16.3 tDM/ha/yr) and the range (12.8-21.7 t DM/ha/yr) modelled (Fig.7.1) are within the range of values found in the literature. Crush et al. (2006) reported annual pasture dry matter (DM) yields, in the Waikato for 3 cows/ha, averaging 17.2±0.9 tDM/ha/yr over 4 years, including the influence of 100 kgN/ha/yr. Crush et al. (2006) noticed that "yields from both early and contemporary research pastures are not far below the theoretical potential yield of 22.5 t DM/ha, calculated by Mitchell (1963), for a Waikato ryegrass pasture not suffering any moisture or nutrient deficiencies, and pest and disease free" (Crush et al., 2006, p.120).

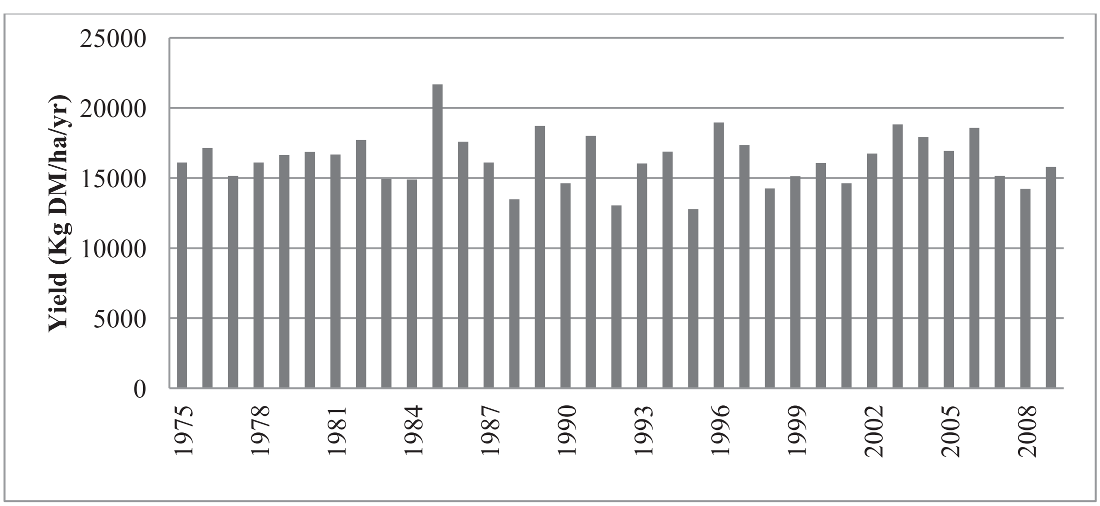

Figure 7-1: Modelled pasture yield for the typical dairy farm on a Horotiu silt loam (kgDM/ha/yr).

SPASMO calculates the total amount of grass grown (standing biomass), as well as the amount of grass eaten by the animals which actually determines the amount of milk solids (MS) produced. The ratio of grass eaten / standing biomass at each grazing is called utilisation and varies from year to year depending on climate and the grazing regime. For the typical dairy farm over 35 years, utilisation varied between 40% and 85%, with an average of 62% (Fig.7.2). Utilisation never goes above 85% because it was assumed in SPASMO that cows always waste at least 15% of the standing DM when grazing (Chapter Five, section 5.3.1).

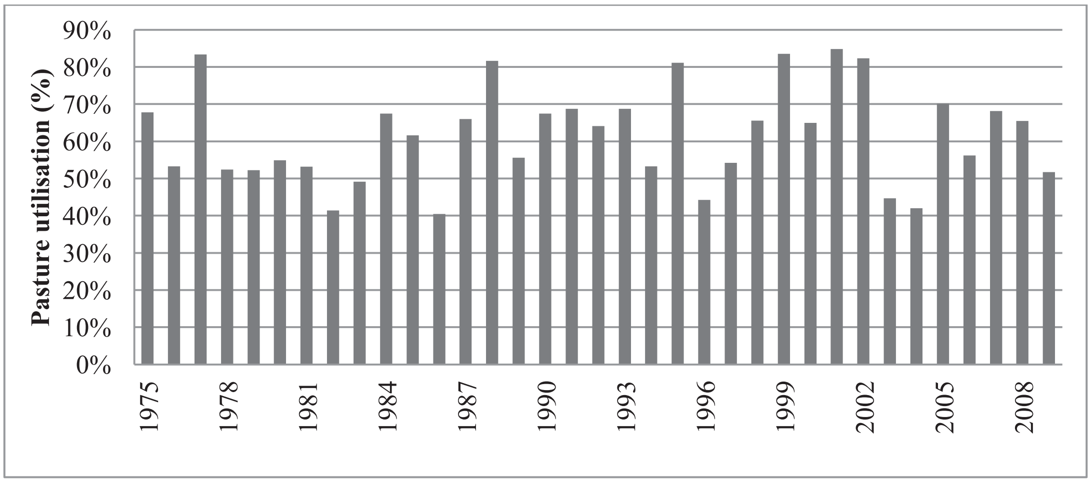

Figure 7-2: Pasture utilisation (grass eaten / total yield) (%).

To estimate the part of the yield coming from soil natural capital, the influence of N and P fertilisers need to be subtracted from the total pasture yield modelled using SPASMO. To do so, the amount of pasture DM due to N fertilisation was subtracted from the total pasture yield modelled. The pasture response to N fertilisers was considered linear: for each kg/ha of N added, the pasture produces 15 kg of extra DM (Gillingham et al., 2007; Gillingham et al., 2008). For the base case scenario, which includes the addition of 100 kgN/ha/yr of fertiliser (Table 7.1), N fertiliser added means an extra 1500 kg DM/ha/yr.

To assess the contribution of P fertilisers to pasture production, the effect of P on legume growth and N_2 fixation needs to be taken into account. Applied P fertilisers drive legume growth, which in turns increases grass growth through the supply of N from biological N_2 fixation.

The basis of the production drive in the New Zealand pastoral system has been through P fertiliser use stimulating legume growth rather than N fertiliser applications directly. New Zealand soils have received P fertiliser additions for over 100 years (Parfitt et al., 2008a). The calibration curves used for current recommended Olsen P levels for maximum pasture production (relationship between Olsen P and relative yield (RY)) are based on an empirical relationship derived from the >3000+ data entries from field studies held in the P data base of AgResearch (Morton and Roberts, 2001) (Fig.7.3). The level of pasture production that could be sustained if no fertiliser was applied was considered to correspond to an Olsen P of 4 (Parfitt et al., 2009). The corresponding relative yield associated with this Olsen P was then calculated from the calibration curves (Fig. 7.3.) (Appendix D).

These data show that for a volcanic soil, as used in this study, an Olsen P level of 4 can only support a RY of about 70%[10]. For the purpose of this study, this is held constant. It would be expected to slowly decline over time. Parfitt et al. (2009) showed that pasture yield was significantly lower on a soil which had not received P fertilisers since 1980 (Olsen P = 8) than on the same soil fertilised annually with P (Parfitt et al., 2009). Numerous studies (Lambert et al., 1989) have shown the impact of withholding fertilisers on pasture production.

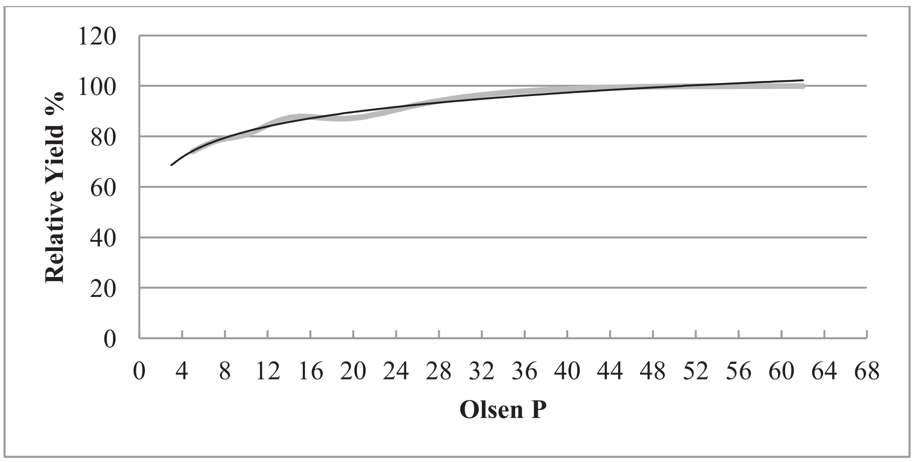

Figure 7-3: Relative pasture yield as a function of Olsen P for an Allophanic Soil.

The part of the total yield coming from natural capital stocks can be calculated as following:

$Y_{NC} = (TY-(15*N))*Pr$

where Y_{NC} is the pasture yield coming from natural capital stocks (kgDM/ha/yr), TY is the total yield modelled by SPASMO (kgDM/ha/yr), 15 is the DM response to N fertilisers (kgDM/kgN), N is the amount of N fertilisers applied (kgN/ha/yr), and Pr is the P ratio associated with low Olsen P levels (70% for volcanic soils). Y_{NC} is the service provided by the soil or the amount of pasture a soil can grow without fertilisers.

SPASMO utilises information about rainfall, Mp and SWC to calculate pasture yield every day, therefore this information is implicitly included in the equation presented above to calculate Y_{NC}.

[10] This measure is deduced from an equation fitted to the shape of the curve. For a Gley Soil an Olsen P of 4 can only support a RY of about 60%.

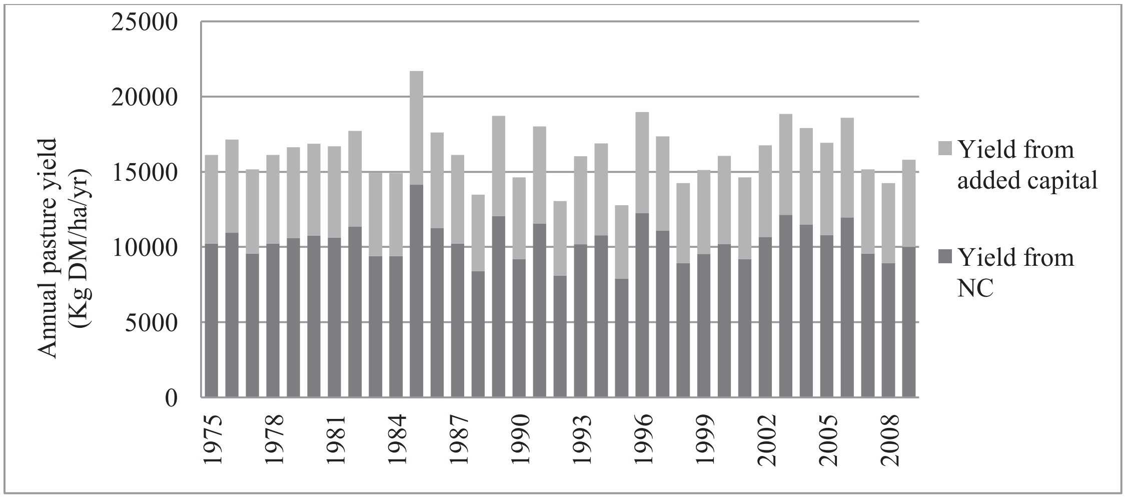

Figure 7-4: Annual pasture yield (kgDM/ha/yr) from natural capital (NC) and added capital.

The results (Fig.7.4) show the total yield modelled as well as the part of the pasture yield due to natural capital for a pasture fertilised each year. The yield from natural capital averages 10.3 tDM/ha/yr (ranging from 7.8 to 14 tDM/ha/yr).

If fertiliser inputs stopped, the total yield would be expected to decline over the years, while natural capital stocks are depleted mainly because of a slow decrease in Olsen P. Lambert et al. (1989) showed that when fertiliser inputs were withheld, pasture production, on a high fertiliser and low fertiliser pasture, declined by 4.6 and 1.7% p.a. respectively (Lambert et al., 1989).

7.3.2 Economic valuation of the provision of food:

7.3.2.1 Value of the food quantity (S1a):

The market price of milk solids (MS) was initially used to value the provision of food from soils. However to follow more rigorously the neo-classical Economics theory of value, the producer surplus (or net rent) should be used instead of market prices (Chapter Six). The net rent of milk solids can be calculated by deducting farm working expenses from the revenue due to the sale of milk solids. For a dairy operation the net rent is around \$2.7/kg MS.

Market prices are commonly used in the literature in studies valuing ecosystem services (Porter et al., 2009; Sandhu et al., 2008). For this reason, it was chosen to use them in this study as well.

In this study, the value of the grass grown from soil natural capital was converted to milk solids using a conversion factor ($F_{Y\text{-}MS}$ = 15 kg DM/kg MS). The market price of MS (P_{MS} = \$6 /kg MS) was applied to the quantity of MS:

$S1a = Y_{NC} / F_{Y\text{-}MS} * P_{MS}$

where S1a is the value of the provision of a quantity of food in $/ha/yr, Y_{NC} is the pasture yield coming from natural capital stocks (kgDM/ha/yr), $F_{Y\text{-}MS}$ is the conversion factor of DM to MS and P_{MS} is the market price of MS.

For example, if Y_{NC} = 8900 kg DM/ha/yr, the value of the provision of food for the base case scenario is S1a = 8900/15*6 = $3,560/ha/yr. If the net rent was used instead of market prices, the value of the provision of food would be $1,602/ha/yr.

The sensitivity of the model to the dynamics of the different soil properties gives us the sensitivity of the food provision service.

Pasture utilisation isn't considered when valuing this service because we are not interested in actual milk production from the pasture grown from natural capital, but instead we are using the price of milk solids to value the service which is embodied by pasture growth.

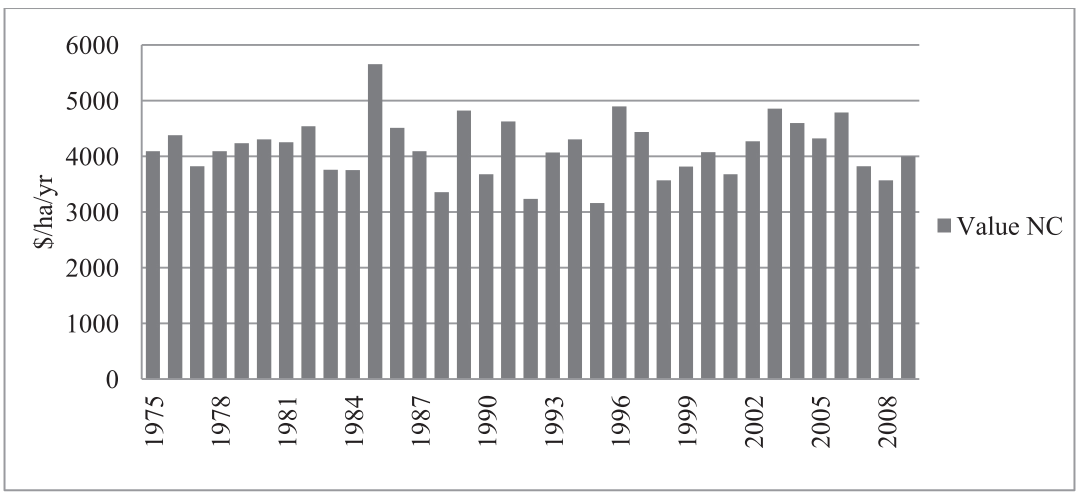

Figure 7-5: Economic value ($/ha/yr) of the provision of food from natural capital (NC).

The average value of the provision of food for the Horotiu soil was $4,155 /ha/yr, ranging from $3,158/ha to $5,655/ha over 35 years (Fig.7.5).

This value is higher than the values found by other authors. Porter et al. (2009) measured yields of grains and pastures in 2006, and calculated their economic value as the farm gate prices of these products to give totals of USD 216 /ha for pasture (NZD 349) (in June 2006 USD 1 = NZD 1.61615) and USD 515 /ha for crops. The pasture yield they measured was 6.1 t DM/ha, and the DM was sold as forage not as added value like milk solids. The value calculated here is more than 10 times that reported by Porter et al. (2009), even though they considered total yields and not yields coming from natural capital.

To assess the impact of the value of the provision of food at the farm scale on the wider Waikato economy, Type II value added multipliers, with backward and forward linkages can be used. Appendix E details the calculations. Since the multipliers (Pers. Comm. Dr Garry McDonald, Market Economics Ltd) used correspond to 2007 NZ$, gross outputs need to be also converted to 2007 NZ$. If the provision of food, that is milk solids, from soil is worth $4,155/ha/yr (NZ$ 2011) or $2,887 (NZ$ 2007), its overall value for the Waikato economy, including backward and forward linkages, is $3,426/ha/yr (NZ$ 2007). It should be noted that the multiplier effects to other regions in New Zealand are not taken into account by these calculations.

The values of other services (regulating services), which are not directly marketed, do not need to be multiplied since they don't generate impacts on the economy, as long as they are provided. It's when they are not provided anymore that humans have to substitute for them using built capital.

7.3.2.2 Value of the food quality (S1b):

Chapter Three mentioned the importance of the provision of trace elements to animals for optimum milk production. The soils considered in this study are not deficient in trace elements (Grace, 1994), providing unrestricted supply. This service can be quantified by establishing the amount of trace-elements necessary to avoid deficiencies in dairy cows (Table 7.3). A replacement cost method was used to value the service (Chapter Six) in case the soil could not provide trace-elements. Trace-elements can be supplied to animals by different means: directly by intramuscular injections (preferred method for iodine), or indirectly through feed by topdressing of pastures (Grace, 1994).

Table 7-3: Costs of prevention of trace-elements deficiencies (Grace, 1994; Pangborn, 2010).

Trace elements	Prevention of deficiency and dose needed	Cost of product	Total cost $/ha/yr
Selenium	Topdressing of pastures with 10 g Selenium/ha/yr	$6.30/kg	0.063
Cobalt	Topdressing of pastures with 350 g Cobalt sulfate/ha/yr	$18.00/kg	6.3
Copper	Topdressing of pastures with 8 kg Copper sulfate/ha/yr	$3.95/kg	31.6
Iodine	Topdressing is not recommended because not efficient		
	Intramuscular injection of iodized oil, 5ml/cow/yr	$22/L	0.33
Total			38.3

The value of the service (S1b) was defined as the difference between the maximum cost of prevention of trace-elements deficiencies if the soil was deficient in all four trace-elements and the actual prevention costs which for the Horotiu soil are nil. Only the cost of products containing trace-element were considered here. Trace-elements are usually applied with other nutrients; therefore no additional transport or spreading costs associated with the mitigation of trace-elements deficiencies were included. If required it could be added to the value.

A soil with adequate quantities of Se, Co, Cu and I, as in this study, is worth an additional S1b = $38.3/ha/yr for providing these four trace-elements. This value reflects feed quality, in addition to the value of the feed quantity grown.

To our knowledge, no one has tried to value the provision of trace-elements from soils as part of an ecosystem services assessment.

The total value of the provision of food for the base case scenario is the sum of food quantity and quality.

Provision of food (S1) = Food quantity (S1a) 4,155 + Food quality (S1b) 38.3 = $4193.3/ha/yr, on average.

7.4 Provision of support for human infrastructure and animals (S2):

The provision of support from soils is determined by natural capital stocks including BD, Mp, and FC, as they link to the soil bearing strength and sensitivity to treading damage.

The provision of support from soils has two dimensions. Soils provide support to human infrastructure (S2a), but also to animals used by humans (e.g. cattle) (S2b). To quantify and value this service, both dimensions need to be considered.

7.4.1 Provision of support for human infrastructure (S2a):

7.4.1.1 Quantification of the provision of support for infrastructure:

For building purposes which includes houses, yards and tracks, compacted soils, that are very stable and don’t sink or deform under the load of a building or road, have the greatest value. Bulk density (BD) is an indicator of soil compaction and bearing strength, and was chosen to measure this soil service. The provision of support for human infrastructure is at its maximum when the soil is already very compacted and dense and there is no risk of deformation. This state corresponds to a high BD. A common practice before building is topsoil removal, so the BD of lower horizons needs to be considered. Parfitt et al. (2010) showed that New Zealand soils present BD between 0.42 and 1.84 g/cm^3. The Horotiu silt loam used for the base case scenario has a BD of 0.83 g/cm^3 below 10 cm (Table 7.4).

Table 7-4: Bulk density (g/cm^3) of a Horotiu silt loam (LandcareResearch, 2010).

Depth (cm)	BD (g/cm^3)
0-10	0.84
10-50	0.83
50-100	0.82

A measure of the service was then defined as the actual BD of the Horotiu soil (Fig.7.6). The measure represents the actual available compaction provided by the soil. The measure of the service is then calculated as following:

$S_I = BD$

Where S_I is the support to infrastructures in g/cm^3, BD is the bulk density of the chosen soil in g/cm^3. For example, for the base case scenario, $S_I = 0.83$ g/cm^3.

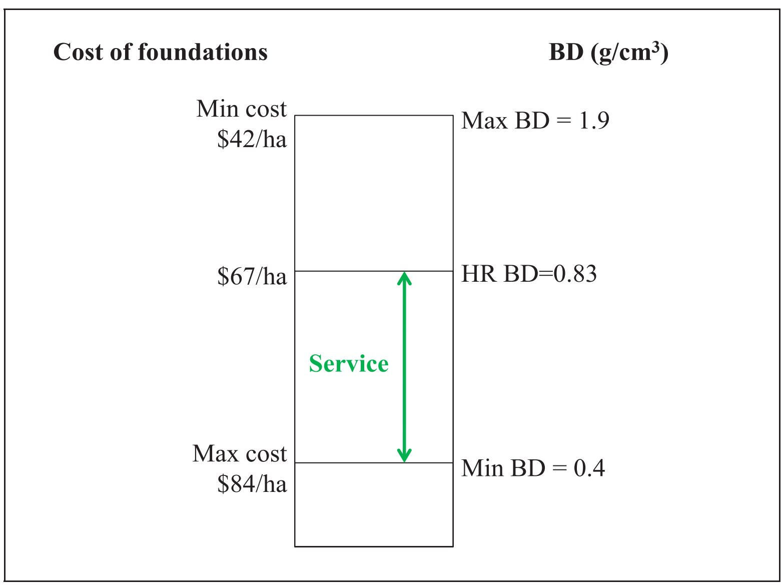

Figure 7-6: Definition of the value of the provision of support for human infrastructure. BD: Bulk density.

7.4.1.2 Economic valuation of the provision of support for infrastructure:

The construction of a farm track was chosen here as an example of the provision of support for human infrastructure. When building tracks, roads or foundations, topsoil and grass are usually removed. The need for, and required thickness of an aggregate surface depends on the type and number of vehicles (or animals) using the road each day and the strength of the underlying soil (Fleming, 2003, p.I-23).

To value the provision of support to human infrastructure a replacement cost method was used (Chapter Six). When the soil fails to provide adequate support for roads, foundations, a compacted aggregate surface, must be built to replace the lack of service. The cost of building these foundations can be seen as a proxy for the value of the service.

The costs of building farm tracks were considered here. The costs of the necessary earthworks depend on soil strength measured by BD (Table 7.4). The construction of farm tracks costs around $10 per linear meter, for a 6m wide, 150mm thick track (Pangborn, 2010, B-67). The thickness of the track's aggregate surface depends on the underlying soil strength. Table 7.5 presents, by soil type, the aggregate surface thickness (Fleming, 2003) and the construction costs based on a cost of $10/linear meter for a 150mm thick track. It was assumed that for a 100ha farm, an average of 3 km of tracks was needed. The total construction cost of farm tracks was then calculated for 100ha, then per hectare (Table 7.5).

Roads are built assets so their capital cost needs to be transformed into annuities over their life time (Chapter Six), here 20 years, to estimate the value of the annual flow of service provided in $/ha/yr (Table 7.5). A discount rate of 10% was chosen here, but results using a discount rate of 3% (Chapter Six) are also presented at the end of the chapter to allow comparison.

The value of the annuities was calculated using the following formula (Holmes, 1998; Pearce et al., 2006):

$$A = \frac{CC}{[\frac{1}{r} - \frac{1}{r(1+r)^n}]}$$

Where A is the present value of the annuity, CC is the capital cost of the asset, r is the discount rate (here 10%), and n is the life time of the asset (here 20 years).

The maintenance cost of the tracks was then added. It was assumed here that maintenance costs are 5% of total construction costs for 100 ha (Fleming, 2003). The annual maintenance cost per hectare was then deducted from this assumption. The annual cost of the tracks is then the sum of the annualised construction cost/ha and the annual maintenance cost/ha.

Table 7-5: Aggregate surface thickness (Fleming, 2003) and construction costs by soil type.

Soil type	Thickness (mm)	Construction costs ($/linear meter)	Construction costs ($ for 3 km for 100ha)	Annualised construction costs ($/yr) for 100ha	Maintenance costs (5%) for 100 ha ($/yr)	Total costs ($/ha/yr)
Pumice, sandy gravels, decayed rock	125	8.3	25000	2936	1250	42
Coarse river sand	150	10.0	30000	3524	1500	50
Silty clays, sandy silts	175	11.7	35000	4111	1750	59
Friable clays*	**200**	**13.3**	**40000**	**4698**	**2000**	**67**
Fine windblown sand	225	15.0	45000	5286	2250	75
Heavy clays	250	16.7	50000	5873	2500	84

* Horotiu silt loam.

It was assumed that the Horotiu silt loam used in this study needed aggregate surfaces of approximately 200 mm, based on its BD and the BD of the reference soil types (Table 7.5). The total cost of replacing the service for the base case scenario was then \$67/ha/yr.

The value of the service was determined as the difference between the worst case scenario (soil with very low BD needing the thickest foundations) and the actual condition of the soils chosen for this study, as presented in Fig.7.6.

The value of the provision of support for human infrastructure was then calculated as following:

S2a= Max cost – Actual cost

where S2a is the value of the provision of support for human infrastructure in \$/ha/yr, Max cost is the maximum cost of building foundations for a very low BD soil, and actual cost is the actual construction cost of foundation for the chosen soil.

For example, the value of the provision of support for human infrastructure for the base case scenario was S2a = 84 - 67 = \$17/ha, or S2a=\$12/ha/yr using a discount rate of 3%.

The value of this service doesn't vary from year to year, because the BD of the subsoil is an inherent soil property, which cannot be changed readily. However, the value of the service will differ between soil types. The costs used here are an approximation; therefore the value of this service could be revised using more detailed costs.

To the knowledge of the author, no study has previously attempted to put a value on the provision of support to infrastructure.

7.4.2 Provision of support for animals (S2b):

The provision of support for farm animals is based on the interaction between soil texture, structure and moisture and the sensitivity of the soil to treading damage.

7.4.2.1 Quantification of the provision of support for animals:

The monitoring of SWC between May and October is recommended to identify periods when soils are sensitive to treading damage (Houlbrooke et al., 2009).

The number of days between May and October when the soil can support animals, that is when SWC<(FC+Sat)/2, was chosen as a measure of the service. To calculate this proxy, the dynamics of SWC, modelled using SPASMO, were followed through each year and across years (Fig. 7.7). The Horotiu silt loam soil provided adequate support for animals for 119 to 160 days, out of 184 days, between May and October for the 35 years modelled.

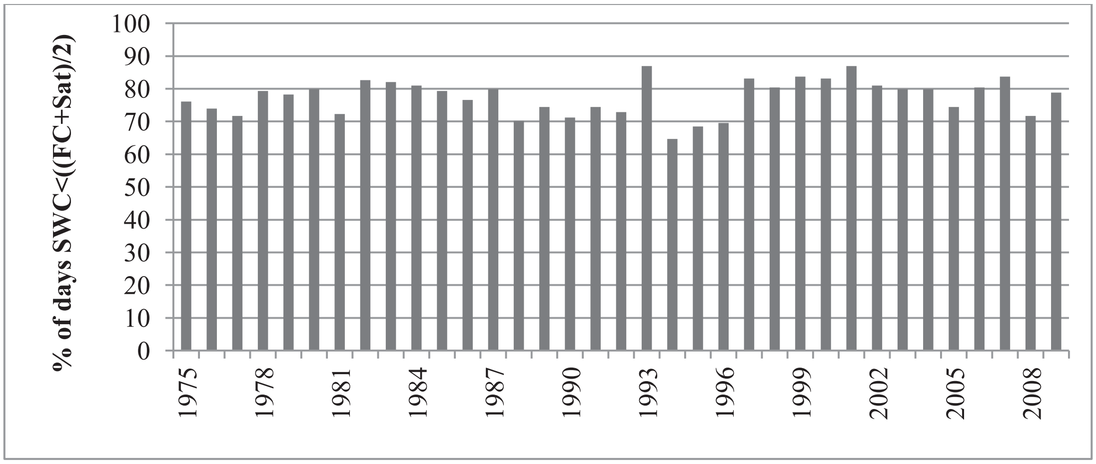

Figure 7-7: Percentage of days between May and October when SWC<(FC+Sat)/2, for the base case scenario (ranges from 65% to 87%).

The measure of the service was determined as the difference between the worst case scenario (SWC>(FC+Sat)/2 for 184 days between May and October) and the actual number of wet days between May and October for the Horotiu soil (Fig.7.7).

The measure of the service is then calculated as following:

$Sup = 184 - W_D$

where Sup is the measure of support to animals, 184 is the number of days between May and October and W_D is the number of wet days when SWC>(FC+Sat)/2 for the chosen soil. Sup is a measure of the number of days when the soil provides adequate support for animals (SWC<(FC+Sat)/2).

For example, if the calculated $W_D = 42$, $Sup = 184-42 = 142$ days.

7.4.2.2 Economic valuation of the provision of support for animals:

The lack of support to farm animals leads to pugging, loss of pasture production and thereby the loss of potential milk production. The loss of production from pugging could have been used to calculate a loss of income but the value the provision of support to animals shouldn't be restricted to a loss of income from pugging, because pugging impacts on all services, not only food production. Therefore valuing this service using only a loss of production would greatly underestimate its value.

To avoid treading damage, New Zealand farmers often use off-pasture standing areas, such as feed pads or standoff pads when the soil is too wet and fails to provide adequate support for animals.

To value the provision of support for farm animals, the provision costs method was chosen (Chapter Six). The value of the support provided by soils to animals can be determined by considering that if the cows cannot stand on the paddock because the soil is too wet, they have to be transferred to for example a standoff pad. The construction and maintenance of a standoff pad is another way to provide the service.

In New Zealand, some farms have standoff pads where cows are fed supplements every day. The function of standoff pads is mainly to increase the efficiency of supplementary feeds.

Standoff pads, the type of pads considered here, are constructed in order to have a place to put cows to avoid damage to soil and pasture when the soils are too wet. The costs of construction and maintenance of a standoff pad were used as a proxy to calculate the value of the provision of support to animals.

Construction of a standoff pad costs \$150 /cow on average (up to \$380 /cow depending on the type of pad) (Dexcel, 2005b). The maintenance of a standoff pad costs \$10/cow/year on average, and can cost up to \$30/cow/year when used year round (Fig. 7.8) (Dexcel, 2005b) (Appendix F).

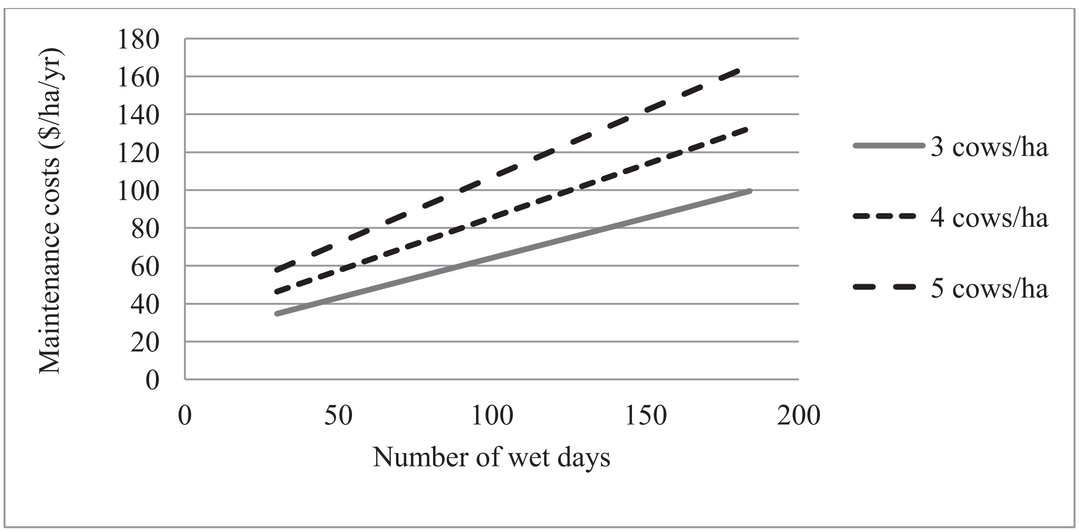

Figure 7-8: Maintenance cost of a standoff pad (\$/ha/yr) depending of the number of days in use and stocking rate (cows/ha).

It is assumed here that farmers won't make the decision to build a standoff pad unless they are unable to put the cows in the paddocks for 30 days or more between May and October. Below that threshold, no costs will be incurred and the value of the support service is assumed to be maximal (Fig 7.9). Above 30 wet days, the costs of using a standoff pad are assumed to be proportional to the number of days the pad is used that is W_D.

If no standoff pad is available, the cows remain on the pasture, even when the soil is too wet. Treading damage will then impact on the provision of all soil services.

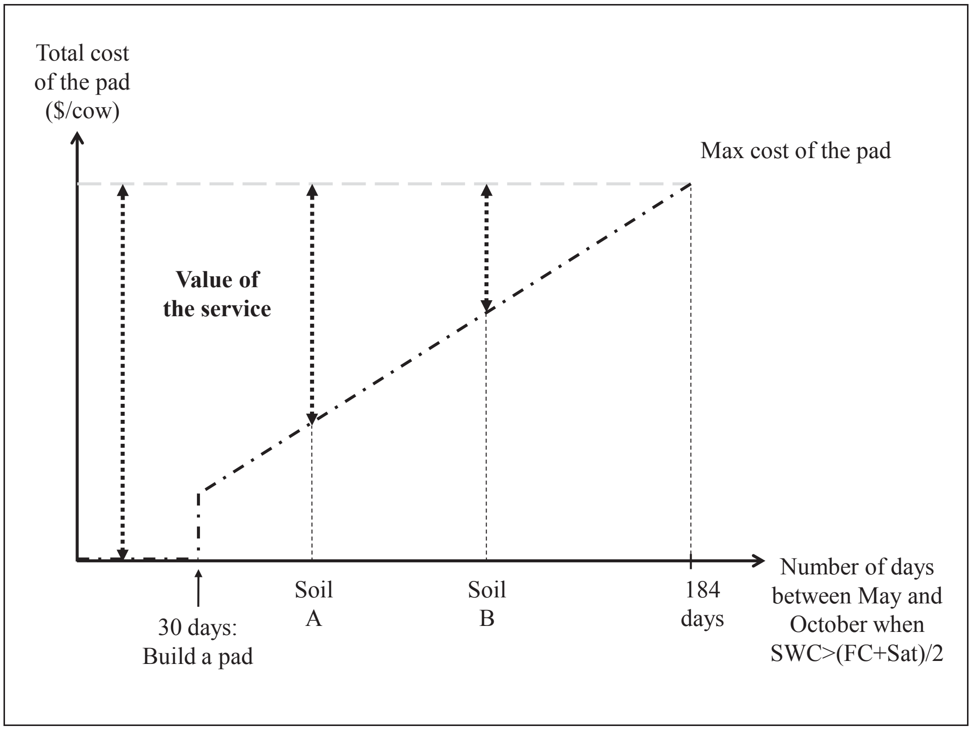

Figure 7-9: Total cost of a standoff pad and value of the provision of support to animals depending on the number of days when SWC>(FC+Sat)/2 between May and October (schematic).

The costs of construction and maintenance of a standoff pad were determined using a number of parameters from the literature (Dexcel, 2005b). Appendix F presents the data from farm study cases used here.

The type of pad considered here is a standoff/wintering pad, a specially built area where animals are withheld from pasture for extended periods when the soil is too wet and supplementary feed is brought to animals on the pad. As the herd may spend several hours per day on the pad for several months, the cows require sufficient area to lie down, as well as additional space for feeding. For this study, it was assumed that when a pad is available on the farm, the cows graze the pasture 8 hours/day, and stay on the pad 16 hours/day if SWC>FC.

The pad area depends on the number of cows using it. It was assumed that a standoff pad requires 6 m^2/cow (Dexcel, 2005a). For a herd of 330 cows the standoff pad is then 1980 m^2. The construction cost of the pad is then given by its size. The construction cost of $24.6/$m^2$ for a bark/sawdust standoff pad (Dexcel, 2005b) (Appendix F) was used here, including earthworks, material and labour. A pad for 330 cows then costs $48,708.

To value the service the construction cost of the pad needs to be annualised over a depreciating period (here 20 years). A discount rate of 10% was used here. The details of the calculation are presented in Appendix F.

The maintenance cost of the pad depends on the number of cows and the number of days per year the pad is used (W_D). Pad maintenance costs were assumed here to be \$0.14/cow/day (Dexcel, 2005b) (Appendix F).

The total costs of a standoff pad in \$/ha/yr are then calculated as the sum of the annualised construction costs (ACC) and the maintenance costs (MC) for the number of days the pad is used, divided by the farm area. For example, the annualised cost of using a standoff pad for a 330 cows herd, for 184 days (every day between May and October) on 110 ha (SR=3 cows/ha) is therefore \$129.3/ha/yr. It would be \$107/ha/yr with a discount rate of 3%.

The value of the service was defined as the total costs of using a standoff pad for the number of days between May and October when SWC<(FC+Sat)/2 for the studied soil (Fig.7.9). This measure represents how much it would cost to take the cows off the pad if the soil was too wet these days and didn't provide any support. It also represents the difference in costs of using the pad every day between May and October (184 days) or only on wet days.

The value of the service was then calculated as following:

$S2b = ACC + MC_{(184\text{-}WD)}$

Where S2b is the value of the provision of support to animals in \$/ha/yr, ACC are the annualised construction costs of the pad, and $MC_{(184\text{-}WD)}$ are the maintenance costs of the pad for the number of days between May and October when SWC<(FC+Sat)/2 for the Horotiu soil. For example, if W_D = 42 days, S2b = \$112/ha/yr. On average S2b=\$112ha/yr with a discount rate of 10% and \$89/ha/yr with a discount rate of 3%.

To the knowledge of the author, no other study as attempted to put a value on the provision of support to animals from soils.

The total value of the provision of support from soils was calculated by adding the value of the support to infrastructure and the value of the support to animals.

Provision of support from soils (S2) = Support to infrastructure (S2a) 17 + Support to animals (S2b) 112 = \$129/ha/yr on average

7.5 Provision of raw materials:

At the farm level, the provision of raw materials from soils (e.g. peat, clay) is often not present or not exploited. In this study, the provision of raw materials from soils was not included, but it is acknowledged that it could make a contribution to the value of ecosystem services provided by soils in different situations, e.g. at a different scale like the catchment scale or the national scale.

7.6 Flood mitigation (S3):

The ability of soils to store water provides a service to humans, buffering excessive rainfall, and in doing so, reducing flood risk. Chapter Four showed that flood mitigation by soils is supported by soil natural capital stocks including soil texture, structure and saturation capacity. Therefore, these natural capital stocks were used to quantify flood mitigation.

7.6.1 Quantification of flood mitigation:

The SPASMO model calculates daily runoff (RO in mm/ha/day) according to plant cover, rainfall, SWC, soil saturation capacity and the management practices impacting on Mp (animal treading) (Fig.7.10).

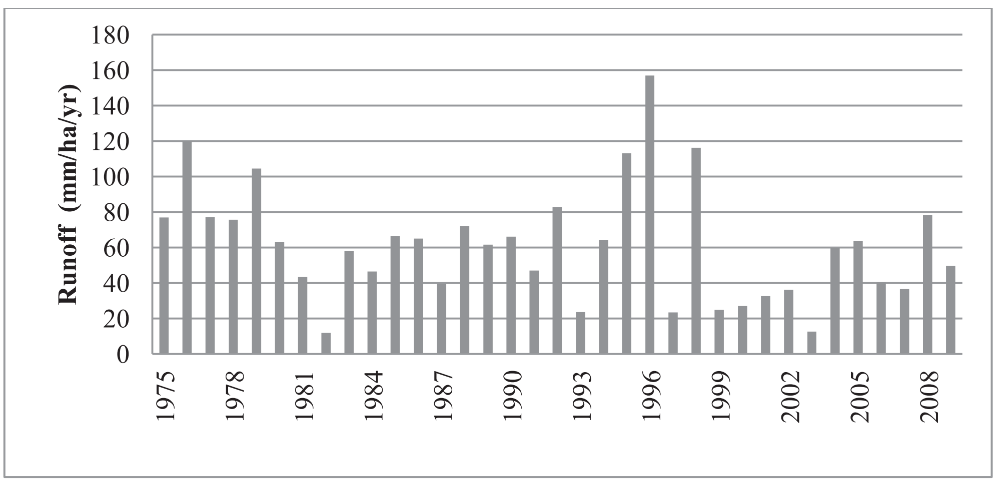

Figure 7-10: Runoff (mm/ha/year) outputs from SPASMO for the base case scenario.

To quantify flood mitigation, it was assumed that in the worst case all the rain falling in a year could potentially run off, as would be the case on an impermeable surface (e.g. concrete). The difference between rainfall and the amount of water that runs off the land is the amount of water absorbed by the soil, or the flood mitigation service (Fig.7.11). The flood mitigation service is defined as the difference between rainfall (RF) and runoff (RO) for each day (Fig.7.11), which is the amount of water that could potentially runoff, but doesn't due to the soil water absorption and retention capacity.

The measure of the service is calculated as following:

$$Fm = RF - RO$$

Where Fm is the measure of the flood mitigation, RF is the daily rainfall (mm) and RO is the daily runoff (mm).

It should be noted that when rainfall is heavy, especially on steep land, water runs off before having time to infiltrate even if soil water storage capacity is available. Therefore, the proxy chosen to measure the service is a lower bound estimate of the soil's flood mitigation potential.

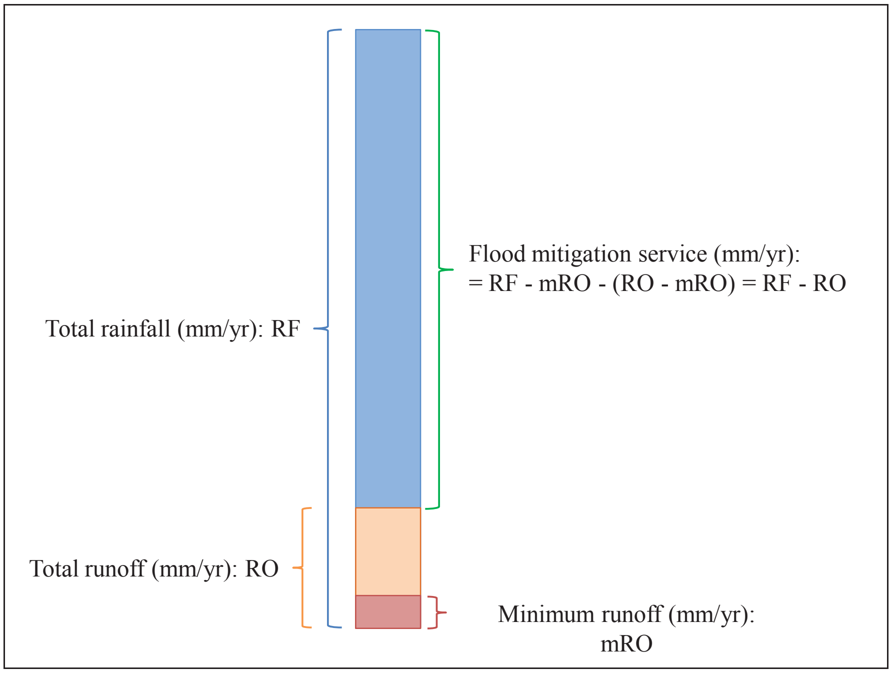

Figure 7-11: Relationship between rainfall (RF), runoff (RO) and the flood mitigation service.

7.6.2 Economic valuation of flood mitigation:

The provision cost valuation method was used to value flood mitigation (Chapter Six). If the soil had no retention capacity, another way of reducing flood risk at the farm scale would be to build dams to store the water presently stored by the soil in order to delay the flood peak. The value of flood mitigation from soils can therefore be assessed by determining the costs of building water-retention dams, on the farm, to store the water that would otherwise run off the land if the soil had no water retention capacity. It was assumed that such a retention dam should be big enough to store the annual maximum of seven consecutive days worth of water stored by the soil, which is RF-RO. This is to mimic the retention of water by the soil profile. This period could be increased or decreased depending on the attributes of a given catchment or specifications of any flood control scheme. For each year, the maximum amount for seven consecutive days of water stored by the soil was calculated using SPASMO outputs (Fig.7.12). The average over 35 consecutive years is 101.8 mm/ha/yr.

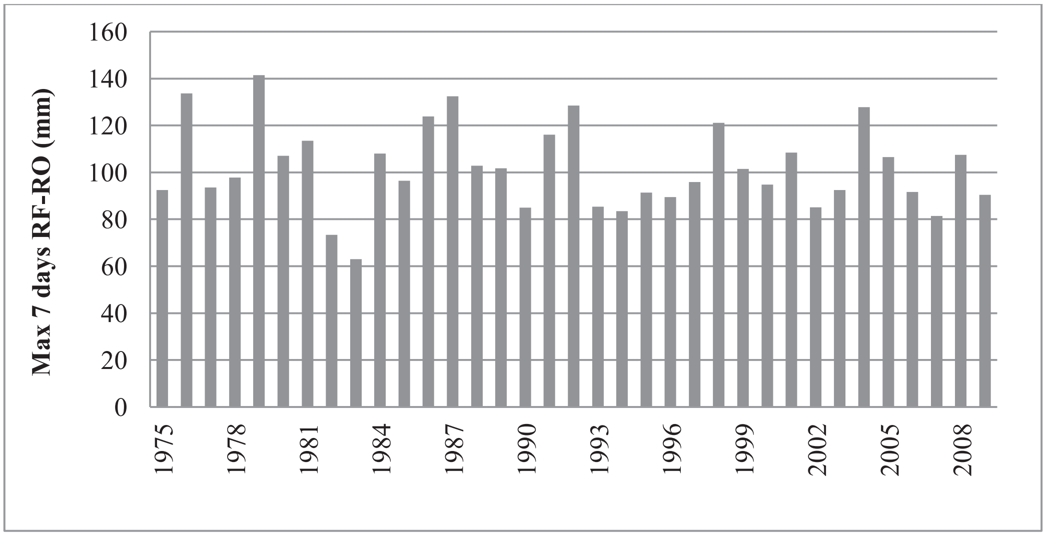

Figure 7-12: Maximum annual amount (mm) of water stored by the soil (rainfall - runoff) for seven consecutive days.

The maximum amount of water stored by the soil in 7 days was used to calculate the size of the dam (volume of storage needed) in m^3/ha. For example, if the maximum amount of water that could runoff in a week (weekly RF-RO) is 100 mm/ha for a given year, a water storage dam of 1,000 m^3 is needed for each hectare to replace the storage capacity of the soil. The cost of construction of a water storage dam was assumed to be \$10/m^3. This value is intermediate between simple excavation costs \$5/m^3 (TRC, 2006) and costs of ponds with lining \$15/m^3 (Pangborn, 2010, p.B-197). The construction of a 1,000 m^3 storage dam costs on average \$10,000. This construction cost was then annualised using a discount rate of 10 % and a depreciation time line of 20 years to calculate the annual value of the service.

The value of flood mitigation from soils was then calculated as following:

$$S3 = \frac{(Max\ weekly\ (RF - RO) * 10 * 10)}{[\frac{1}{r} - \frac{1}{r(1+r)^n}]}$$

Where S3 is the value of flood mitigation in \$/ha/yr and Max weekly RF-RO is the yearly maximum of weekly RF-RO in mm/ha/yr, 10 is the conversion factor of mm to m^3, 10 is the cost of construction in \$/m^3, r is the discount rate (here 10%), and n is the life time of the asset (here 20 years).

For example, if the max weekly is RF-RO = 108 mm, S3 = \$1268.6/ha/yr. On average S3=\$1,196/ha/yr with a discount rate of 10%, or \$685/ha/yr with a discount rate of 3%.

Flood mitigation (S3) = \$1,196/ha/yr on average

Porter et al. (2009) and Sandhu et al. (2008) valued 'hydrological flows' from agro-ecosystems. However, they refer to the water supplied by soils to plants, not as it might influence flood mitigation.

Ming et al. (2007) valued the flood mitigation of wetland soils in China. They quantified the service in a static way, as the difference between FC and saturation. They used the replacement cost valuation method to value flood mitigation by wetlands. They estimated the investment needed in the construction of reservoirs to replace wetlands was US$5,700/ha/yr, if flood mitigation by wetland soil was lost, which is roughly NZ$7,980/ha/yr (with USD 1= NZD 1.4 in 2007). This value wasn't annualised but if it were (over 20 years, at a discount rate of 10%) the annual value of flood mitigation by wetland soils would be $937.3/ha/yr which is comparable to the value calculated in the present study.

7.7 Filtering of nutrients and contaminants (S4):

The soils ability to filter nutrients and contaminants is directly linked to the quality of receiving fresh water bodies and thereby to animal and human health. Chapter Four showed that this service is supported by soil natural capital stocks including ASC, CEC, soil texture and structure and SWC.

SPASMO models soil nutrient retention capacity and its level of saturation, and links it to the dynamics of nutrients in soil solution, drainage and runoff. ASC, CEC, soil texture and structure and SWC are used to determine N and P lost by leaching and runoff from annual N and P inputs to the soil.

To value the soil's ability to filter nutrients and contaminants, two dimensions need to be considered: the retention of soil nutrients (mainly N and P) and the retention of soil contaminants (pathogens, chemicals such as pesticides, endocrine-disrupting chemicals (EDCs)). To quantify the filtering of nutrients, SPASMO outputs were used, including daily SWC, N leaching and P runoff. The model was used to simulate conditions where there was no N or P retention (Chapters Four and Five).

To quantify the retention of soil contaminants, since no detailed data was available on the dynamics of the quantities of contaminants entering and leaving the soil, the risk of contamination by dung pads of runoff water at the time of grazing was considered as a proxy. SWC, runoff and the timing of grazing events from SPASMO were used to quantify this component of the service.

7.7.1 Quantification of the filtering of nutrients:

This study focuses on N and P losses from soils because they are the two problematic nutrients in New Zealand surface water bodies.

7.7.1.1 Quantification of the filtering of N:

The SPASMO model generates N losses (nitrate (NO_3^-) and ammonium (NH_4^+)) according to rainfall, soil properties and processes, and management practices (fertiliser, grazing regime) for the Horotiu silt loam (Fig.7.13).

The measure of the N filtering service was defined as the difference between the potential maximum modelled N loss by leaching when N retention was close to zero, and the modelled N leaching loss when the soil properties describing N retention were activated. This difference represents the N retained by the soil filtering capacity, or the amount of N that was filtered.

The measure of the filtering of N was calculated as following:

F_N = Max N loss – Modelled N loss

Where F_N is the measure of the filtering of N in kg N/ha/yr, Max N loss is the maximum amount of N in kg N/ha/yr that could potentially be leached modelled with SPASMO with soil N retention close to zero, and modelled N loss is the amount of N actually leached each year modelled with SPASMO in kg N/ha/yr.

The SPASMO model was used twice: once to model the potential maximum N loss by leaching when N retention was close to zero, and then to model the amount of N actually leached each year by the Horotiu silt loam (Fig. 7.13).

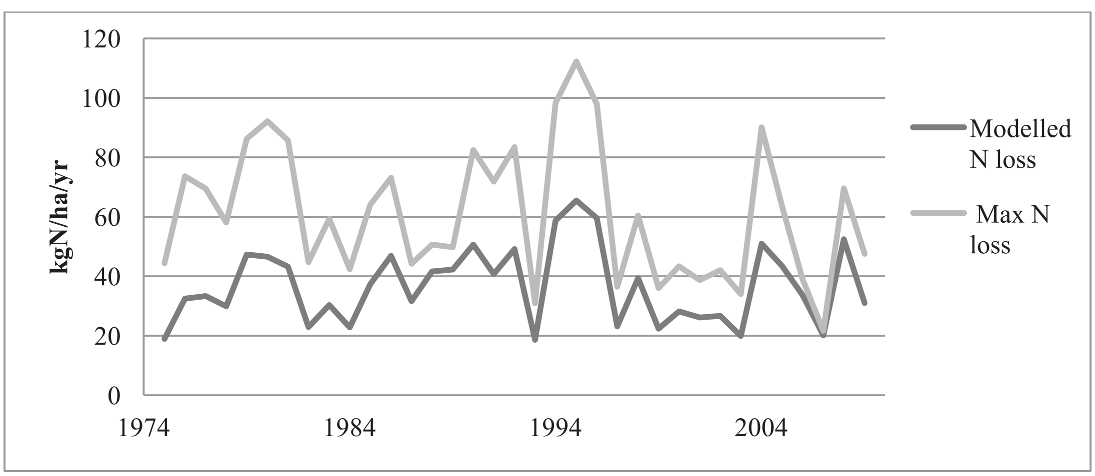

Figure 7-13: Max N loss and modelled N loss outputs (kg N/ha/yr) from SPASMO over 35 years.

The N losses varied between 18.6 and 65.5 kg N/ha/yr with an average over 35 years of 36.8 kg N/ha/yr (Fig.7.13). These values are in agreement with data found in the literature (de

Klein, 2001; Green and Clothier, 2009). It should be noted that in the upper Waikato, the suggestion has been made to limit N leaching losses to 25 kg N/ha/yr (Campbell, 2009).

The difference between Max N loss and modelled N loss outputs from the model over 35 years, that is the measure of the filtering of N, ranged from 1.5 kg N/ha/yr to 46.8 kg N/ha/yr, with an average of 24.3 kg N/ha/yr (Fig. 7.14).

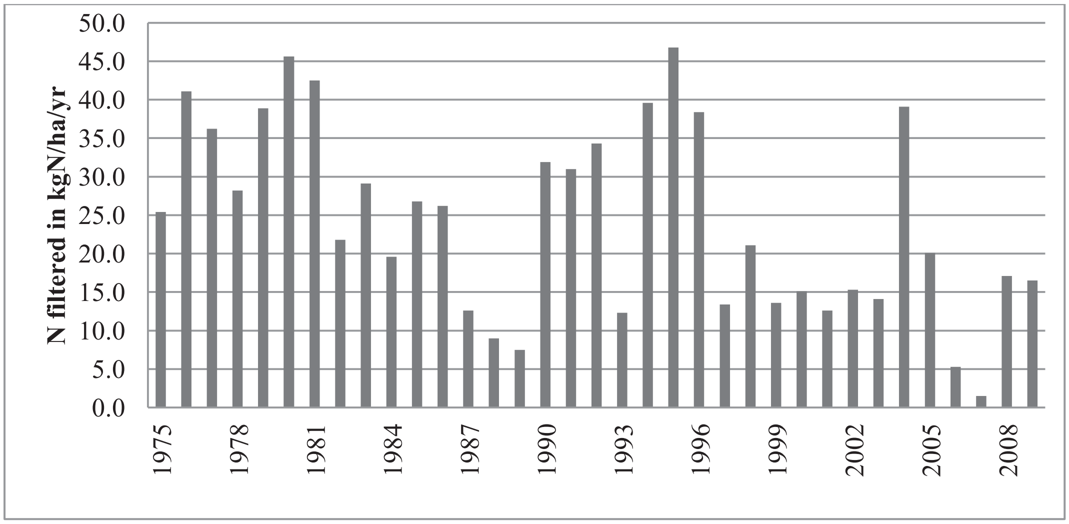

Figure 7-14: Amount of N filtered (kgN/ha/yr).

For example, when the modelled N loss was 28.2 kg N/ha/yr, the measure of the service was F_N = Max N loss – modelled N loss = 43.3 – 28.2 = 15.1 kg N/ha/yr, which is the amount of N the soil filtered.

7.7.1.2 Quantification of the filtering of P:

The SPASMO model generates P losses (runoff and drainage) according to rainfall, management practices and soil properties and processes.

If soil had no P retention capacity, enormous quantities of P would be lost by leaching every year. To quantify this service, the amount of P that would be lost by leaching if the soil had no P retention capacity was modelled using the SPASMO model. The maximum P loss (Max P loss) corresponding to a very low ASC was calculated for each of the 35 consecutive years modelled (Fig 7.15).

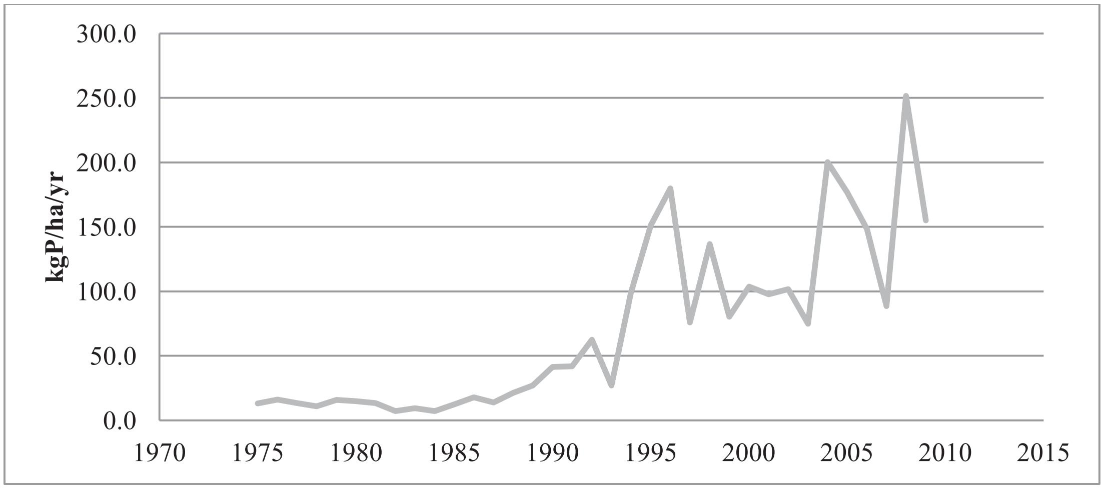

Figure 7-15: Max P loss (P drainage) modelled with SPASMO (kg P/ha/yr).

This Max P loss is specific to each soil and each scenario. For the base case scenario, the Max P loss of a Horotiu silt loam ranged from 6.8 to 250.8 kg P/ha/yr, with an average of 71 kg P/ha/yr (Fig.7.15). These values can seem enormous but can be explained. The amount of P deposited on soils every year can vary between 60 and 70 kg P/ha/yr, coming from P fertilisers but also dung. This P accumulates in the soil profile. Organic P is mineralised and if the soil has no P retention capacity, P travels slowly down the profile and leaches out after a number of years.

For the base case scenario, P runoff losses varied between 0.1 and 1.2 kg P/ha/yr, with an average of 0.6 kg P/ha/yr, which is in agreement with the data found in the literature (Fig.7.16). P losses on New Zealand farms are reported between 0.1 and 4 kg P/ha/yr (McDowell et al., 2005; McDowell et al., 2009). Since, P leached is very small averaging 2g/ha/yr (with the exception of low P sorption soils such as podzols or sands) (Edwards et al., 1994a) it wasn't considered in this study. In dairy systems, P losses are mainly due to soil's sensitivity to erosion and the amount of P lost in runoff. McDowell et al. (2007) found that 10% of P export originated from fertilizer, 30% dung, 20% pasture plants and 40% from P associated with soil.

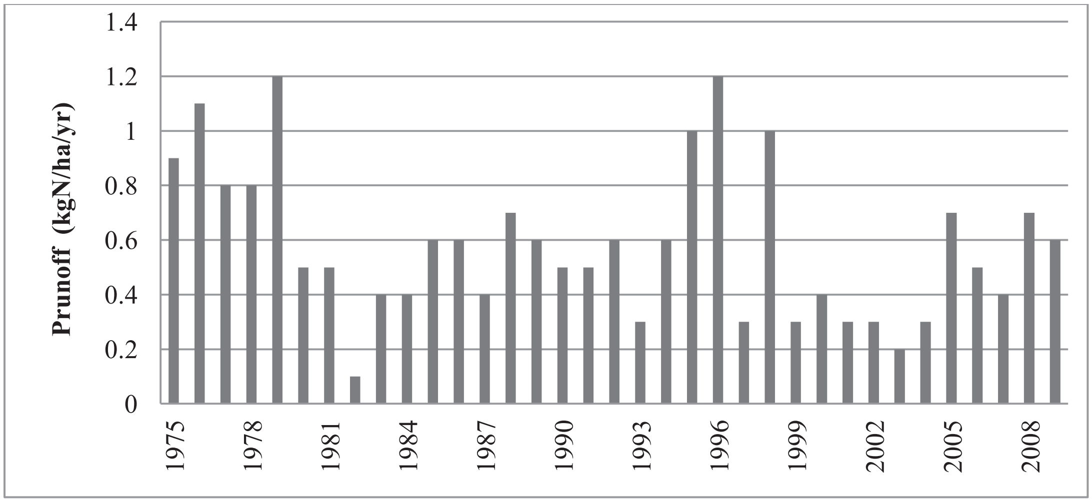

Figure 7-16: P runoff outputs from SPASMO over 35 years (kg P/ha/yr).

The service was then defined as the difference between the potential Max P loss (Fig.7.15), what could potentially be lost by drainage and runoff, and the modelled amount of P runoff, calculated with SPASMO (Fig.7.16). The amount of P that wasn't lost is what the soil retained, the measure of the service.

The measure of the filtering of P is calculated as following:

F_P = Max P loss – Modelled P runoff

Where F_P is the measure of the filtering of P in kg P/ha/yr, Max P loss is the maximum amount of P that could potentially drain modelled with SPASMO in kg P/ha/yr and Modelled P runoff is the amount of P that actually runs off each year modelled with SPASMO in kg P/ha/yr.

For example, when the Modelled P runoff is 0.7 kg P/ha/yr, the measure of the service is:

F_P = Max P runoff – Modelled P runoff = 176.7 - 0.7 = 176 kg P/ha/yr.

The measure of the filtering of P, ranged from 6.8 kg P/ha/yr to 250.8 kg P/ha/yr, with an average of 71 kg P/ha/yr.

7.7.2 Economic valuation of the filtering of nutrients:

7.7.2.1 Valuation of the filtering of N (S4a):

The defensive expenditure method was used to value the filtering of N (Chapter Six). Defensive expenditures are the money spent by individuals, here farmers, to avoid a degradation of the environment and a decrease in the provision of an ecosystem service. The money spent to deal with the lack of N filtering service is used as a proxy for the value of the service. This method was chosen because a number of mitigation techniques exist and data about their costs and efficiency is robust, easily available and well accepted.

It was assumed that the amount of N that wasn't lost by the soil, e.g. the service (Max loss - Modelled loss), would have to be mitigated by farmers otherwise (Fig. 7.17). The cost of the mitigation was used as a proxy for the value of the service.

For each level of production, a maximum and actual nutrient loss can be determined. The measure of the service is defined as the difference between the maximum nutrient loss and the modelled nutrient loss that is the amount of nutrients retained by the soil. The cost of mitigation then depends on the modelled nutrient loss and the level of water quality to reach, which determines the amount of nutrients to mitigate. The more restrictive the water quality target is (threshold A), the greater the need for farmers to mitigate. If the modelled N loss is below the water quality threshold, it means the soil can store more N. Mitigation costs therefore depend on the water quality targeted, whereas the value of the service is fixed (Fig. 7.17).

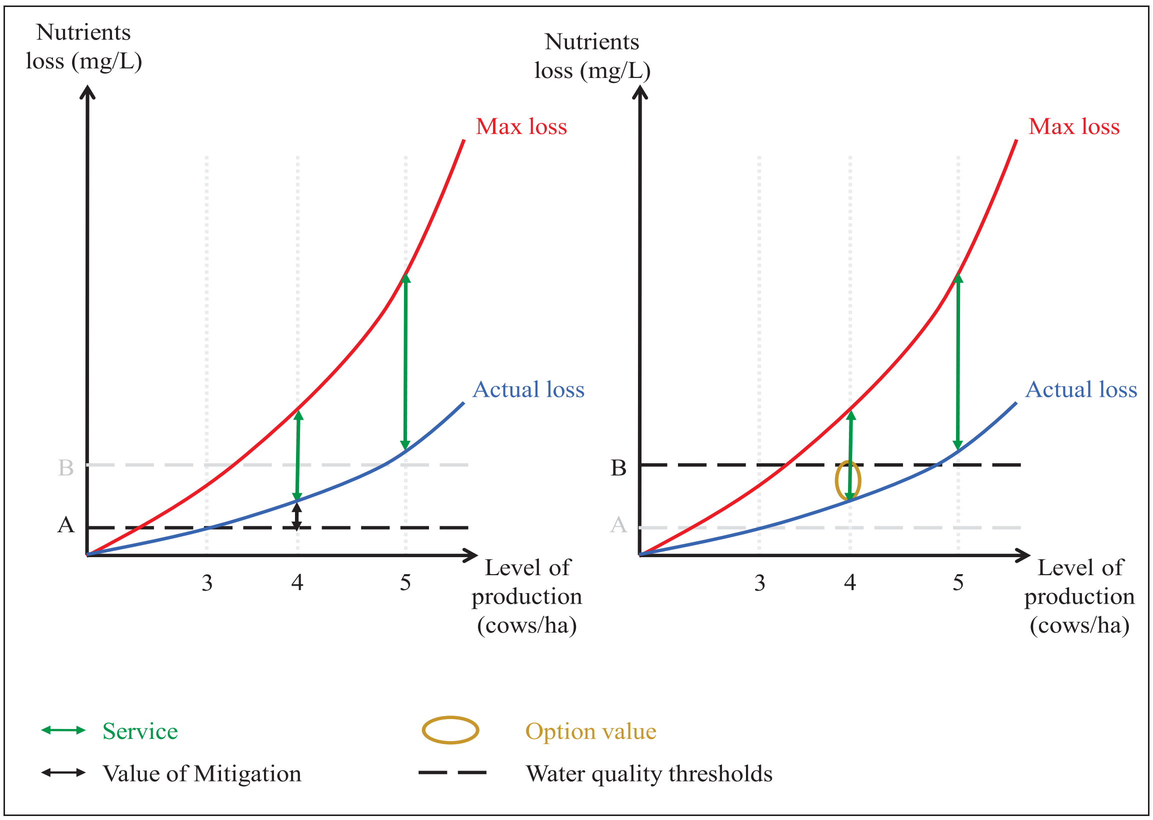

Figure 7-17: Method for the valuation of the filtering of nutrients.

To determine the value of the service, the costs of mitigation need to be applied to the measure of the service, the amount of N the soil didn't lose. The costs and efficiency of different mitigation techniques to reduce N leaching were therefore examined. Farmers have several options to reduce N leaching (de Klein and Ledgard, 2005):

- Reduce the total amount of excreta N returned to pasture,

- Increase the N use efficiency of excreta and/or fertilisers,
- Avoid soil conditions that favour N loss.

This study focused on three techniques commonly used in New Zealand dairy farms, namely a standoff pad to limit urine deposition on pastures, replacing fertilisers with low N feed supplements and using nitrification inhibitors to prevent the transformation of NH_4^+ into NO_3^-. There are a number of other mitigation options available to farmers (de Klein, 2001; de Klein and Ledgard, 2005; Monaghan et al., 2008). The use of a standoff pad and low N supplements aim at reducing the total amount of N returned to pasture and control the timing of application of N regarding SWC. The use of nitrification inhibitors prevents the formation of NO_3^- and N_2O, increasing N use efficiency.

To build a mitigation function the costs of each technique (Table 7.6) were estimated from data found in the literature (Appendix F). The costs of a standoff pad change depending on the type of costs considered: capital costs, annualised at 3% or annualised at 10%. This will therefore also change the equation of the mitigation functions (data not shown here).

Table 7-6: Mitigation options to reduce N losses and costs for 3 cows/ha.

Mitigation option	Comments	Costs ($/ha/yr)
Standoff pad	Max cost of pad, used for 184 days: $31.12/cow/yr (annualised construction costs with 10% discount rate + maintenance)	129.3
Low N supplements	$0.30/kg DM of maize (amount calculated with OVERSEER®)	182.7
Nitrification inhibitors	$90/ha/yr (+GST)*2 applications	207

The efficiency of each technique was determined using the OVERSEER® nutrient budget model (AgResearch, 2005) (Appendix F). SPASMO outputs were fed into OVERSEER® to generate a nutrient budget for the farm considered, and then different OVERSEER scenarios determined the efficiency of each mitigation technique. It was assumed that farmers would use nitrification inhibitors first, because that does not require a change to the management of the farm. If greater quantities of N still need to be mitigated, a standoff pad could be installed and low N supplements fed to the cows (Appendix F). The total cost of mitigation ($/ha/yr) was then determined for each option. A mitigation function was built using mitigation costs for 3, 4 and 5 cows/ha (Fig.7.18) (Appendix F).

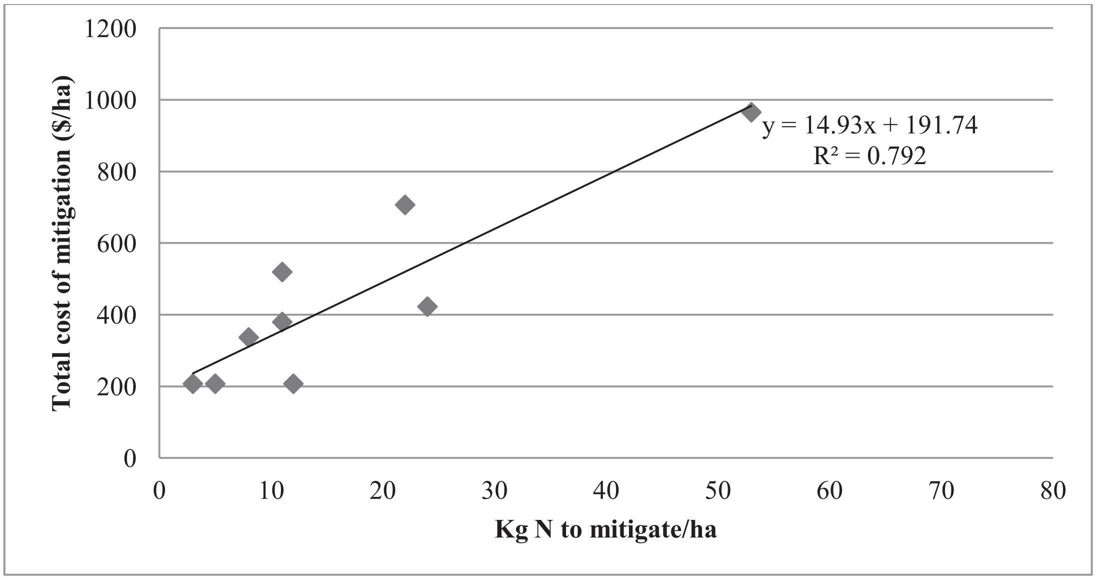

Figure 7-18: Mitigation function for N leaching for a Horotiu silt loam.

The value of the filtering of N was then calculated as following:

$S4a = 14.93 * F_N + 191.74$

Where S4a is the value of the filtering of N and F_N = Max N loss – Modelled N loss, in kg N/ha/yr, is the amount of N the soil filtered, which would have to be mitigated otherwise.

For example if, for the base case scenario $F_N = 43.3 - 28.2 = 15.1$ kg N/ha/yr and S4a = $417/ha. On average S4a=$554/ha/yr with a discount rate of 10%, or $529/ha/yr if using a discount rate of 3%.

Porter et al. (2009) and Sandhu et al. (2008) valued 'N regulation' from agro-ecosystems, which is the provision of N to plants, but not the filtering of N by soils. To our knowledge, no study as attempted to put a value on the filtering of N by soils.

7.7.2.2 Valuation of the filtering of P (S4b):

As for the filtering of N, the defensive expenditure method was chosen to value the filtering of P (Chapter Six). The money spent to avoid P losses is used as a proxy for the value of the service. Again, it was assumed that the amount of P retained by the soil, e.g. the service (Max P loss – Modelled P runoff), would have to be mitigated if lost. Therefore, the cost of the mitigation was used as a proxy for the value of the service.

Farmers have a number of options available to mitigate P losses (McDowell et al., 2009; Monaghan et al., 2008):

- Optimum soil P fertility: Olsen P needs to be at agronomic and economic optimum level. If Olsen P is too high, e.g. too much P fertiliser is/was used, pasture doesn't use the excess P and it is lost. For soils with high Olsen P, mitigation consists of stopping or reducing P fertilisers' application until Olsen P reaches the optimum range.

- Deferred effluent irrigation: Effluents can be stored in ponds and irrigation deferred until SWC<FC to avoid runoff and leaching. Moreover, small amounts/depths of effluent should be applied at low rates to increase infiltration and reduce losses.
- Low solubility P fertiliser can be used to decrease P losses
- Stream protection: streams can be protected by buffers strips, fences, and berms to channel runoff away from stream.
- Use of pads: pugging and dung deposition onto pastures can be minimised by using a standoff pad in winter.

In this study, for the base case scenario, it was assumed that soil Olsen P level was in the optimum agronomic range, effluents were stored and irrigated following best management practices and streams were protected. Therefore, the use of standoff pads to mitigate P losses was chosen to value the filtering of P.

To build the mitigation function the costs of construction and maintenance of a standoff pad was estimated from literature data (Table 7.8, Appendix F) and the efficiency of the use of a pad was determined using SPASMO. The total cost of mitigation ($/ha) was determined for 3, 4 and 5 cows/ha (Fig.7.19). Again, if the costs of a standoff pad change depending on the type of costs considered: capital costs, annualised at 3% or annualised at 10%, it would also change the equation of the mitigation functions (data not shown here).

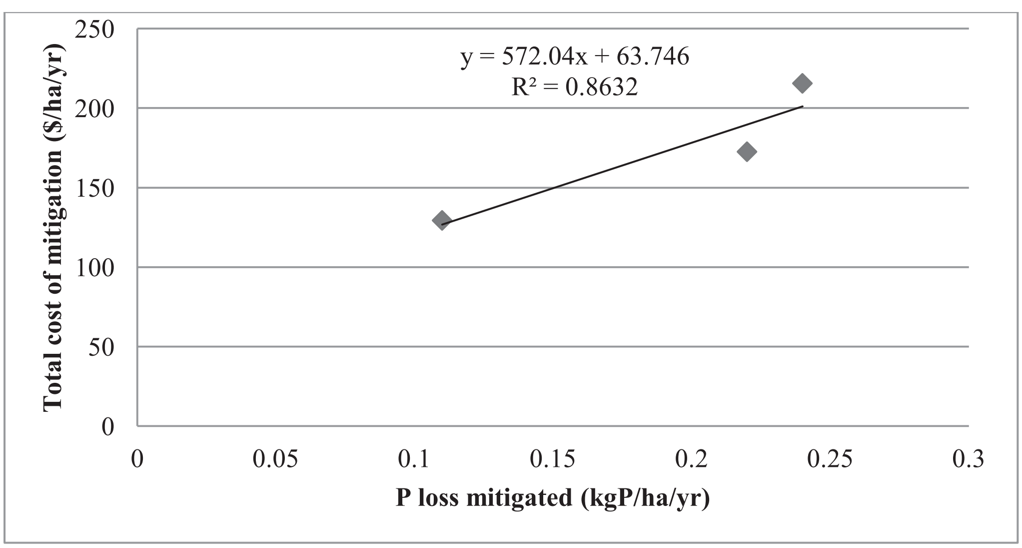

Figure 7-19: Mitigation function for P losses for a Horotiu silt loam.

For the base case scenario, the value of the filtering of P should then be calculated as following:

$S4b = 572.04 * F_P + 63.7$

Where S4b is the value of the filtering of P in \$/ha/yr and F_P = Max P loss – Modelled P runoff, in kg P/ha/yr, is the amount of P the soil retained, which would have to be mitigated otherwise.

However, if this method is applied to the measure of the service calculated previsously (F_P = 176 kg P/ha/yr), it would assume that mitigating 176 kg P/ha/yr is feasible, which is not the case. Using the mitigation function determined (Fig. 7.19), would come to valuing the filtering of P at S4b = \$46,915.3/ha (ranging from \$3,900 to \$143,500/ha/yr).

These results can be seen as a proof that soil P retention capacity (or ASC) is critical natural capital (de Groot et al., 2003; Ekins et al., 2003b). Ekins et al. (2003b) defined critical natural capital as the part of the natural environment that performs important and irreplaceable functions and for which no substitutes in terms of human, manufactured or other natural capital currently exist. The services based on critical natural capital are therefore invaluable (priceless) because non-substitutable by any other type of capital. In economical terms, the demand for these services is so high that their consumer's surplus tends to infinity (Chapter Six).

Therefore, in this study, it was chosen not to apply the mitigation function calculated (Fig. 7.19) to the measure of the service F_P if F_P was taking values above 5 kg P/ha/yr, because such P losses cannot be mitigated with existing technologies with focus mainly on P runoff. Hence, when F_P was greater than 5 kg P/ha/yr, the mitigation function was applied to the maximum value of 5 kg P/ha/yr. This gives a value for the filtering of P of \$2,922/ha/yr with a discount rate of 10%, or \$2,425/ha/yr using a discount rate of 3%.

To our knowledge, no other study as attempted to put a value on the filtering of P by soils.

7.7.3 Quantification of the filtering of contaminants:

ASC, CEC, soil texture and structure and SWC are the natural capital stocks behind the filtering of soil contaminants (pathogens, chemicals (pesticides), endocrine-disrupting chemicals (EDCs)). Unlike for N and P, the filtering of contaminants wasn't directly quantified by looking at contaminant loads and leaching. Instead a proxy was used as a measure of the service: the potential amount of contaminated runoff which reaches waterways. Runoff outputs and the timing of grazing events from SPASMO were used to calculate that proxy.

Dung is deposited on pasture during grazing events. Dung pads are the source of a number of contaminants including pathogens and endocrine-disrupting chemicals. Soil texture, structure and SWC determine how much water runs off pastures after a rainfall event. The timing of dung deposition and rainfall regarding SWC influences how much water washes off fresh dung pads and penetrates the soil, where it can be decontaminated or runs off the land to waterways.
Dung pads take between 40 and 60 days to completely decompose (Aarons et al., 2004). About half of their wet weight disappears in 7 days (Aarons et al., 2004). The assumption was made here that dung can still significantly contaminate runoff water up to 5 days after the grazing event. The amount of contaminated runoff generated in the 5 days following a grazing event modelled with SPASMO was used as a proxy for contaminant loss and was compared to the amount of rain falling up to 5 days after a grazing event (Fig.7.20).
If the soil is too wet to absorb rainfall and filter contaminants, all rainfall could be lost as runoff and potentially be contaminated by dung pads for days after a grazing event. Therefore, the service was defined as, for each year, the difference between the amount of rainfall generated in five consecutive days, and the amount of contaminated runoff generated in the five days after a grazing event. This measure of the service represents the amount of water that could be contaminated but isn't thanks to soil absorption and filtering capacity (Fig.7.20).

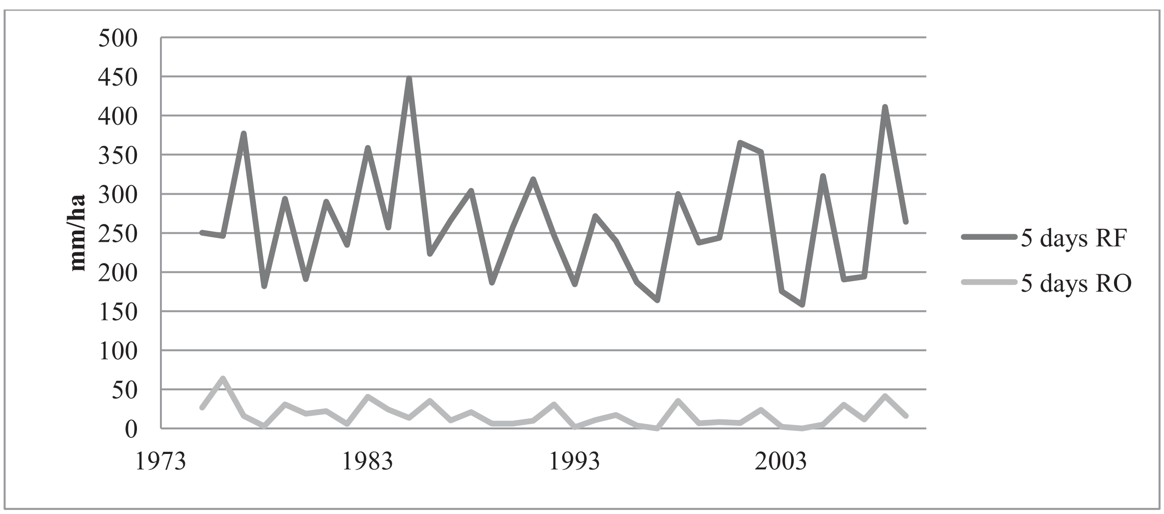

Figure 7-20: Yearly rainfall and runoff (mm/ha) within five days after a grazing event.

The measure of the filtering of contaminants was calculated as follows, for each grazing event:
F_{Ct} = 5 days RF – 5 days contaminated RO
Where F_{Ct} is the measure of the filtering of contaminants in mm/ha, 5days RF is the amount of rain falling within five days after a grazing event modelled with SPASMO in mm/ha and 5 days contaminated RO is the amount of contaminated runoff generated within five days after a grazing event modelled with SPASMO in mm/ha.

For example, for the base case scenario, if 5days RF = 58 mm/ha and 5 days contaminated RO = 5 mm/ha, then the measure of the service is F_{Ct} = 58-5 = 53mm/ha/yr.
Fig. 7.20 presents the sum across the year of 5 days RF and 5 days RO, whereas Fig. 7.21 presents the maximum value taken by F_{Ct} for each year.

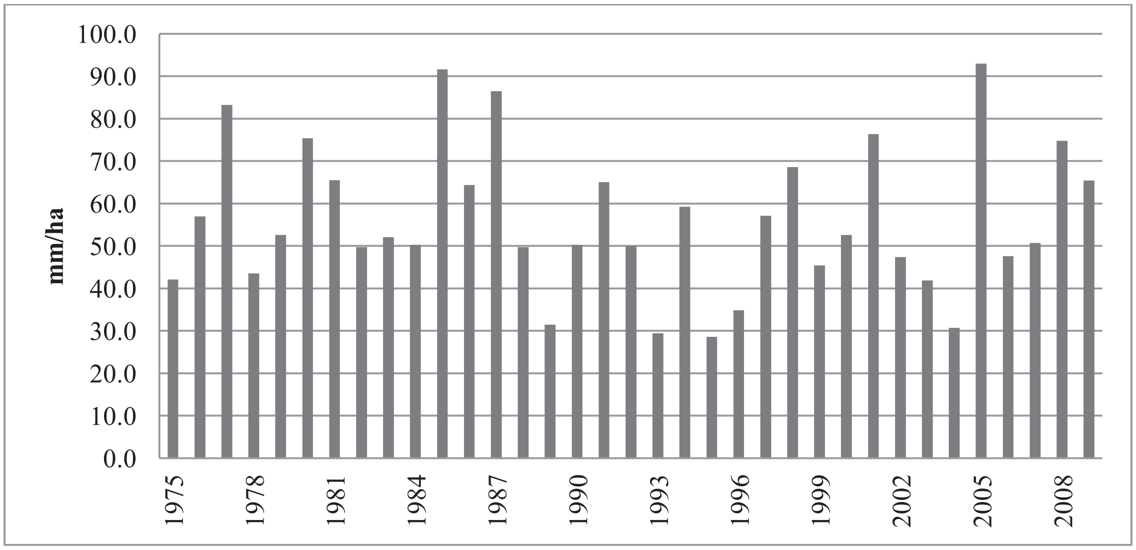

Figure 7-21: Maximum amount (mm/ha) of filtered water (F_{Ct}) for each year.

7.7.4 Valuation of the filtering of contaminants (S4c):

The provision cost method was used to value the filtering of contaminants by soils (Chapter Six). If the soil didn't filter water before it reached water bodies, another way to decontaminate water would need to be used. Here, the construction of a wetland to filter contaminated runoff water was considered as an alternative method to deal with a lack of service. The wetland should be big enough to store, every year, the maximum volume of contaminated water produced in 5 days after a grazing event, which is the maximum for each year of RF-RO within 5 days after each grazing event or the maximum value taken by F_{Ct} for each year.
The cost of building and maintaining a constructed wetland was used as a proxy for the value of the filtering capacity of the soil for contaminants. Wetlands are areas including a variety of plant species densely spaced which together with shallow water, provide good wildlife habitat as well as water purification. They are flooded for part of, or the entire, year. Constructed wetland systems are designed to simulate and optimise the filtering and organic matter breakdown processes that occur in soils and natural wetlands before discharge to a waterway (Sukias and Tanner, 2011; TRC, 2006; Wilcock et al., 2011). Constructed wetlands have the ability to treat a wide range of contaminants.
If the contaminated water had to be only stored, the costs would be similar to those of a storage dam used for the valuation of flood mitigation (between \$5/m^3 for simple excavation and

$15/m^3 for ponds with lining). However, for the contaminated water to be filtered and decontaminated, an active wetland is needed. Such a wetland would be much more expensive to build. The costs of construction of a wetland found in the literature (TRC, 2006) range from $100/m^3 to $150/m^3 (for a standard surface flow constructed wetland, 30cm deep) including earthworks, a clay liner, inlet and outlet structures, gravels, plants and eventually additional establishment costs (site survey, design and resource consent processes). For this study, since we are dealing with volumes bigger than a conventional wetland size (up to 1000 m^3 instead of 150 m^3), it was decided to use the most conservative cost of construction $100/m^3.

To value the filtering of contaminants, this original construction cost then needed to be annualised over 30 years with a discount rate of 10% to obtain the value of the service per year.

Wetlands eventually fill and become inactive after a period of time. Here, this period was estimated to be 30 years, but this could be argued depending on the type of wetland, the amount and quality of material flowing to it and its maintenance.

The maintenance costs of a constructed wetland are generally much lower than maintenance costs of conventional effluent pond systems. They were estimated here to be 1% of the construction costs (6-8% for conventional effluent ponds) (TRC, 2006). The size of the wetland that would be needed if the soil didn't filter contaminants was determined by using the maximum value taken by F_{Ct} for each year. This measure was then converted into a volume (m^3) and thereby area of wetland needed. Annualised construction costs and maintenance costs were determined from the size of the wetland needed.

The value of the filtering of contaminants was then calculated as following:

$$S4c = \frac{(Max\ \mathrm{FCt} * 10 * 100)}{[\frac{1}{r} - \frac{1}{r(1+r)^n}]} + \ (\mathrm{Max\ FCt} * 10 * 100 * 0.01)$$

Where S4c is the value of the filtering of contaminants in $/ha/yr, Max F_{Ct} is the maximum value taken by F_{Ct} for each year in mm/ha, 100 is the cost of construction of a wetland in $/m^3, r is the discount rate (here 10%), and n is the life time of the wetland (here 30 years).

For example, if the measure of the service is 60 mm/ha for a given year, a 600 m^3 wetland is needed, which would cost 600*100 = $60,000. The total value of the service would then be S4c = 6,365 + 600 = $6,965/ha/yr.

On average S4c=$6,513/ha/yr with a discount rate of 10%, or $3,424/ha/yr using a discount rate of 3%.

A number of authors (Swinton et al., 2007; Wall et al., 2004; Weber, 2007; Zhang et al., 2007) have mentioned the ability of soil to filter contaminants as an ecosystem service, but to our knowledge, no one has attempted to quantify and value the provision of this service.

The total value of the filtering of nutrients and contaminants by soils can then be obtained by summing its different components.

Filtering of nutrients and contaminants (S4) = Filtering of N (S4a) 554 + Filtering of P (S4b) 2,924 + Filtering of contaminants (S4c) 6,513 = $9,991/ha/yr on average

7.8 Detoxification and recycling of wastes (S5):

Chapter Four showed that a range of processes enable soils to detoxify, decompose and recycle wastes. The decomposition of wastes is independent from the filtering of nutrients and contaminants treated in the previous section. The focus here is on the ability of soil biota to decompose and recycle waste components. The wastes considered here are animals wastes applied to pastures, namely dairy cows' dung. This section aims to quantify and value the efficiency of microbial populations to degrade and recycle dung. The quantification of the service therefore focuses on soil moisture (SWC) and soil aeration (macroporosity) because they are the key soil properties (natural capital stocks) driving invertebrates and micro-organism populations, which are the main agents of the detoxification and recycling of wastes (Chapter Four).

7.8.1 Quantification of the recycling of wastes:

To quantify the decomposition of wastes, soil conditions were examined regarding their impact on microbial activity. The dynamics of SWC, and thereby implicitly Mp, were followed using SPASMO outputs, and linked to the amount of dung deposited on the pasture for each grazing event. Ideal conditions for optimum decomposition of wastes by soil fauna were associated with a soil neither too dry, nor too wet, that is SP<SWC<FC. Ideally, nitrate concentration $[NO_3^-]$ should be considered as well since micro-organisms need mineral N to decompose N-rich wastes. However, because of the highly dynamic nature of soil $[NO_3^-]$, it is very difficult to relate it to the efficiency of waste decomposition.

The amount of dung deposited in restricting conditions (SWC<SP or SWC>FC) was determined using SPASMO outputs (Fig.7.22).

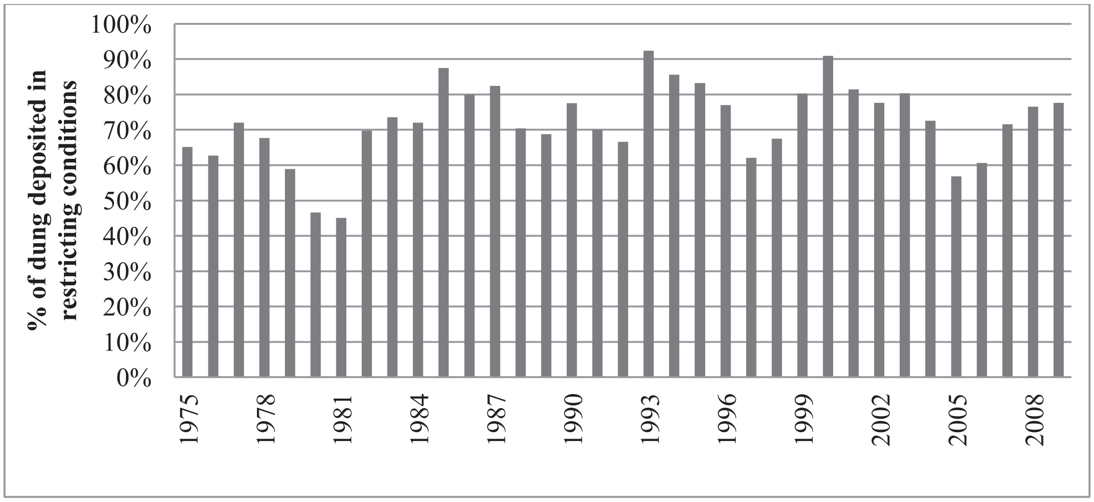

Figure 7-22: Percentage of dung deposited in restricting conditions over 35 years.

The measure of the service was then defined as the difference between the total amount of dung deposited in kg DM/ha/yr and the amount deposited in restricting conditions. The resulting amount represents the dung deposited in ideal conditions that is the amount of waste that would be potentially successfully decomposed.

The measure of the decomposition and recycling of wastes was then calculated as following:

$W_{Dec} = Tw - Wu$

Where W_{Dec} is the amount of dung deposited in ideal conditions in kgDM/ha/yr, Tw is the total amount of dung deposited in kgDM/ha/yr and Wu is the amount of dung deposited in restricting conditions in kgDM/ha/yr.

For example, if Tw = 2826 kgDM/ha/yr and Wu = 1714 kgDM/ha/yr, the measure of the service is W_{Dec} = 2826-1714 = 1112 kgDM/ha/yr.

7.8.2 Valuation of the recycling of wastes:

The provision cost method was used to value the recycling of wastes from soils (Chapter Six). If soil biota didn’t decompose and recycle wastes, the alternate solution would be to use an effluent treatment pond to degrade wastes and fert-irrigation to return the waste nutrients to pasture. To treat dairy effluents, farmers have to collect wastes, usually from a pad, store them in an effluent pond while they breakdown, and reapply them to soil when SWC is favourable. The costs of using an effluent treatment pond, sourced from the literature (Dexcel, 2005b), were used as a proxy for the value of the service.

On a dairy farm, the size of the effluent storage pond is usually determined by herd size and the time the animals spend on the wintering-pad. Standard recommendations (Dexcel, 2005a) exist to determine the adequate volume of storage (Horne et al., 2011). Construction and maintenance costs for effluent ponds then depend on the pond size (Dexcel, 2005b; Horne et al., 2011; Pangborn, 2010; TRC, 2006). Appendix F details the references and method used to calculate the construction and maintenance costs of an effluent pond. On average, the total cost of an effluent pond was $4/m^3, including annualised construction costs (over 20 years with a discount rate of 10%) and maintenance costs of the pond and the irrigation system for effluent application to land (pump, irrigator...). To calculate the volume of effluent corresponding to the measure of the service, it was assumed that a cow produces 50L of effluent per day, which is equivalent to 2 kg of DM (Dexcel, 2005a; Saggar et al., 2003b). Therefore if 1112 kg DM/ha/yr is deposited while soil conditions are inadequate for decomposition, an effluent pond of 1112*50/2 = 27800 L = 27.8 m^3/ha/yr would be needed to properly decompose the waste.

This method, by using effluent pond costs, takes into account the detoxification role of the pond as it substitutes for the soil's detoxification role. To value this service, the price of N and P fertilisers could have been used, since if wastes are not decomposed properly, these nutrients are not returned to the soils. Doing so would have overlooked the detoxification part of the service, and considered only the recycling of nutrients part.

The value of the recycling of wastes was then calculated as following:

$S5 = W_{Dec}*50/2/1000*4$

Where S5 is the value of the recycling of wastes in \$/ha/yr, W_{Dec} is the amount of dung deposited in ideal conditions in kgDM/ha/yr, and 4 is the total cost of an effluent pond in \$/m^3 (including annualised construction and material costs, and maintenance costs).

For example, storing 27.8m^3/ha/yr of effluents would cost S5 = \$111.2/ha/yr. On average S5=\$78/ha/yr with a discount rate of 10%, or \$63/ha/yr using a discount rate of 3%.

Decomposition of wastes (S5) = \$78/ha/yr on average

A number of authors have mentioned soils ability to recycle wastes (Costanza et al., 1997; Daily, 1997; de Groot, 2006; MEA, 2005; Swinton et al., 2007; Wall et al., 2004). Others mentioned "nutrient cycling and mineralisation" (Barrios, 2007; Lavelle et al., 2006; Porter et al., 2009; Sandhu et al., 2008; Weber, 2007; Zhang et al., 2007).

What is valued here is not the nutrient cycling and mineralisation, which is considered as a supporting process behind the provision of food, but the decomposition, detoxification and recycling of wastes which is a service in its own right. Mineralisation isn't a service in itself because humans cannot directly use nutrients, however, the detoxification and recycling of wastes is a service because it directly affects human health.

Authors including Sandhu et al. (2008) and Porter et al. (2009) assessed mineralization of organic matter provided by soil microorganisms and invertebrates. However, these methods do not make the distinction between plant litter decomposition and waste decomposition.
To our knowledge, no one has specifically modelled and valued the recycling of wastes as part of an ecosystem services framework.

7.9 Carbon storage and greenhouse gases regulation (S6):

Soils emit and consume CO_2 and overall have the capacity to store C, which is of interest for signatory countries of the Kyoto Protocol including New Zealand. Soils can also regulate their emissions of GHGs like N_2O and CH_4.
These services are supported by soil natural capital stocks including soil structure and macroporosity, clay content, nutrient status and soil biota diversity. These natural capital stocks were used to quantify carbon storage and greenhouse gases regulation.

7.9.1 Quantification of carbon storage and GHGs regulation:

Regulation of C flows:

To quantify the net C flows from the Horotiu soil, the outputs of the SPASMO model were used. SPASMO models all the processes of the C cycle in soils and thereby calculates soil C stock every day and its net variation. The yearly net variation of C stock was then calculated by making the difference between the stock at the beginning and the end of each year (Fig. 7.23).

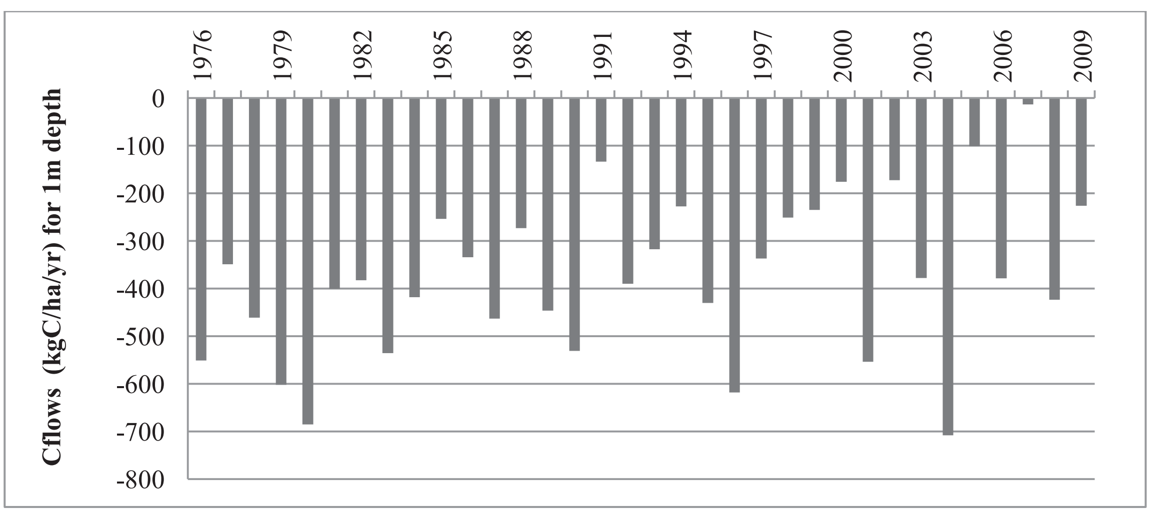

Figure 7-23: Net variations of C stocks for 1m depth (kgC/ha/yr) over 35 years.

These results, averaging losses of 0.3 t C/ha/yr (Fig.7.23), are of the same order as these reported in the literature. Schipper et al.(2010) showed large losses of soil C from soil profiles under pasture averaging 0.8 t C/ha/yr. C losses were not confined to top soils only but observed through the top meter of the profile (Schipper et al., 2007). These results are in accordance with the Trotter et al. (2004) study which showed that improved grasslands loose around 0.9 t C/ha/yr. It is still unknown whether the C losses observed reflect a shift to a new equilibrium of C or whether they are ongoing. These losses could be explained by a number of factors including changes in the amount of litter and litter quality returned to the soil, rates of incorporation or changes in pasture species composition. Data used in this study should be updated when more data is available in the literature.

The service was then defined as the annual net C flows over 35 years for each scenario calculated from SPASMO. When C flows are negative, C is lost from the soil profile which is a degradation process. The impact of this process on other soil properties and thereby on all soil services could be quantified.

The measure of the net C flows was then calculated as following:

C flows = Net C flow *44/12

Where C flows is the measure of the service in kgCO_2eq/ha/yr, Net C flow is the annual net C flow calculated from SPASMO in kgC/ha/yr, and 44/12 is the conversion factor from C to CO_2 equivalent.

For example, for the base case scenario, the average net C flow between 1975 and 2009 was -324 kg C/ha/yr (Fig.7.23), therefore C flows = -324*44/12= -1188 kg CO_2 eq/ha/yr.

Regulation of N_2O emissions

To quantify N_2O emissions from the Horotiu soil, the IPCC (Intergovernmental Panel on Climate Change) methodology (Eggleston et al., 2006) was used as well as SPASMO outputs. The IPCC methodology to calculate N_2O emissions from soils is globally recognised. It uses N inputs to the soil and emission factors to calculate direct and indirect N_2O emissions estimations from N fertiliser inputs and animal wastes deposited on pastures. Figure 7.24 presents the IPCC methodology to calculated N_2O emissions from grazed pastures. Some emission factors have been recalculated to better fit New Zealand conditions (de Klein et al., 2003). Table 7.7 presents the New Zealand factors used in this study.

Table 7-7: IPCC factors used in this study (Eggleston et al., 2006).

Factor	Definition	Value	Unit
Frac GASF	Part of synthetic fertiliser emitted as NO_x or NH_3	0.1	
Frac GASM	Part of N excreted emitted as NO_x or NH_3	0.2	
EF1	Direct emissions from N input to soil	0.01	kg N_2O-N/kg N
EF3PRP	Direct emissions from waste in pasture	0.01	kg N_2O-N/kg N excreted
EF4	Indirect emissions from volatising N	0.01	kg N_2O-N/kg NH_4-N & NO_x-N deposited
EF5	Indirect emissions from leaching N	0.025	kg N_2O-N/kg N leached & runoff

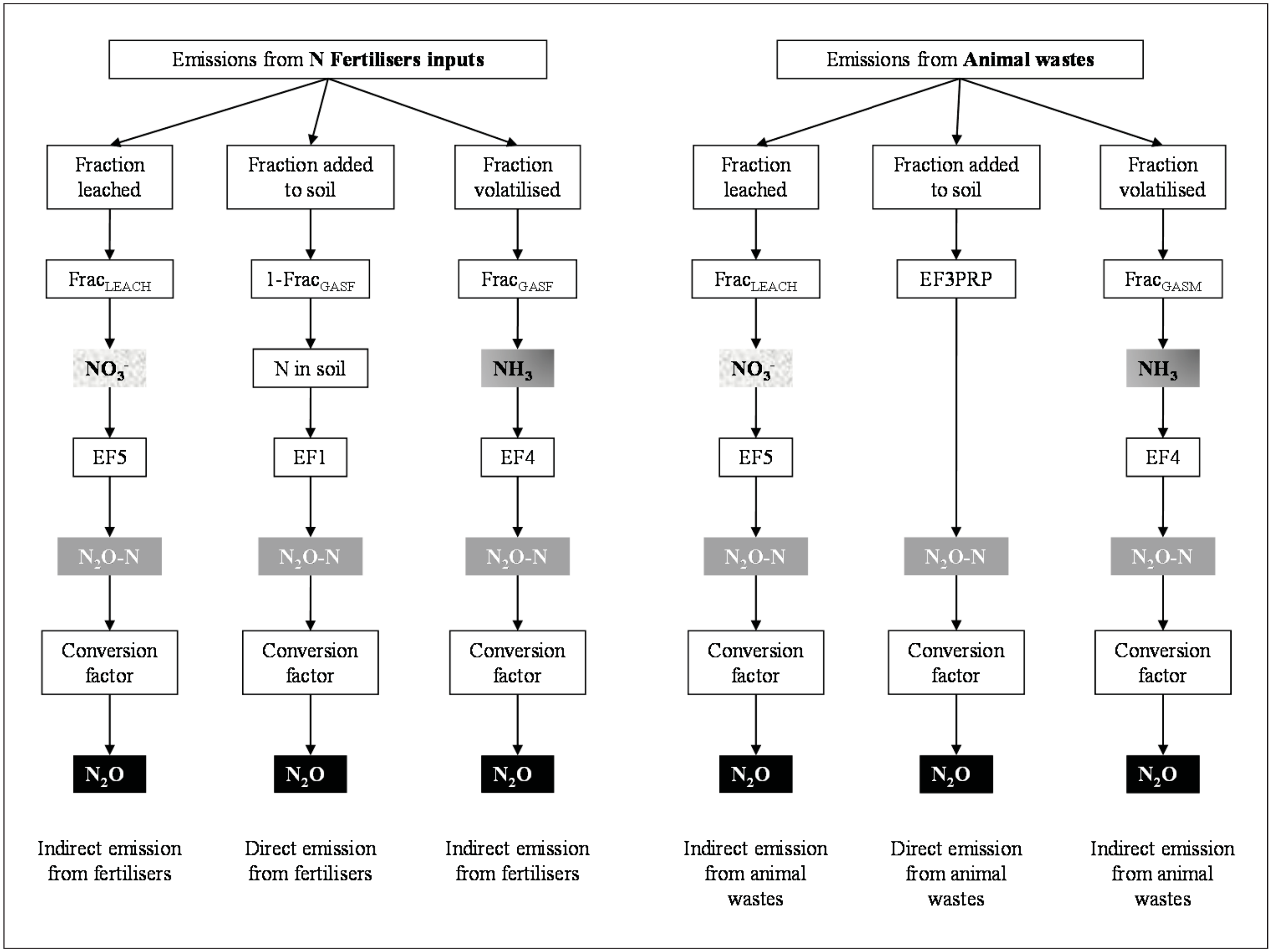

Figure 7-24: IPCC methodology to calculate N_2O emissions from grazed pastures (Eggleston et al., 2006; MfE, 2009b).

The IPCC methodology is unable to account for the distinction between soil types and especially soil moisture dynamics and its impact on N_2O emissions. This is why, in this study, the IPCC methodology was modified to take into account the impact of soil moisture dynamics on N_2O emissions from soils.

To calculate indirect emission from N leached from N fertilisers and animal wastes deposited on pastures, the amount of N leaching every year outputted by the SPASMO model was used instead of using the IPCC 'FracLEACH' and the amounts of N fertiliser and wastes deposited on pasture (Table 7.8). SPASMO calculates the net N leaching every year depending on N inputs to soil as fertilisers and animal wastes, as well as management and climate. Therefore, it was assumed that SPASMO measurement of N leaching would be more accurate than the IPCC methodology using FracLEACH. Yearly N leached outputs in kg of NO_3^--N were then converted to kg of N_2O using the IPCC emission factor (EF5) (Table 7.8).
Indirect emissions from N volatilised from wastes (dung and urine) were calculated using the total amount of wastes deposited on pasture determined with SPASMO and then following the IPCC methodology (Table 7.8). However, to calculate the direct emissions from wastes, the effect of soil moisture on N_2O emissions was taken into account by using different emission factors for the amounts of waste deposited on pasture when the soil was wet or when it was dry. When wastes are deposited when SWC<FC, the standard emission factor EF3PRP (0.01 kg N_2O-N/kg N excreted) was used, but if SWC>FC a greater emission factor of 0.015 kg N_2O-N/kg N excreted was used. De Klein et al. (2003) argued that adopting a single emission factor for New Zealand was inappropriate since they found that poorly drained soil had higher emission factors than well drained soils. The emission factors they calculated for cow urine ranged from 0.3 to 2.5% of the urine N applied. This is why an emission factor of 1.5% was chosen in this study for wet conditions.
Emissions from fertilisers (fraction added to soil and volatilised) were calculated following the IPCC methodology. Since best management practices were assumed, fertilisers are supposed to be applied in dry conditions; therefore it wasn't necessary to take into account the effect of soil moisture on N_2O emissions from fertilisers.

Table 7-8: Detail of the calculation of N_2O emissions from soils.

Emission type	**Definition**	**Origin of data**	**Calculation of N_2O emissions**					
			kg N/ha/yr		kg N/ha/yr		kg N_2O-N/ha/yr		kg N_2O/ha/yr
Emissions from total N leached (fertilisers + wastes)	Indirect emissions from total N leached	SPASMO outputs	Total N leached			*	EF5 (0.025)	*	44/28
Emissions from animals wastes (dung + urine) deposited on pastures	**Direct emissions from wastes added to soil when wet***	**SPASMO outputs**	**N added from wastes when wet**			*	**EF3 PRP modified (0.015)**	*	**44/28**
	Direct emissions from wastes added to soil when dry	SPASMO outputs	N added from wastes when dry			*	EF3 PRP (0.01)	*	44/28
	Indirect emissions from N volatilised from wastes	SPASMO outputs	Total N added from wastes	*	Frac GASM	*	EF4 (0.01)	*	44/28
Emissions from N fertilisers	Direct emissions from N fertiliser added to soil	scenarios	Total N fertiliser added	*	1-Frac GASF	*	EF1 (0.01)	*	44/28
	Indirect emissions from N volatilised from fertiliser	scenarios	Total N fertiliser added	*	Frac GASF	*	EF4 (0.01)	*	44/28

*Modified emission factor for wet conditions.

For the base case scenario, total N_2O emissions varied between 8.3 and 15.3 kgN_2O/ha/yr, which is of the same order as data reported in the literature (Giltrap et al., 2008; Saggar et al., 2004c) (Fig.7.25).

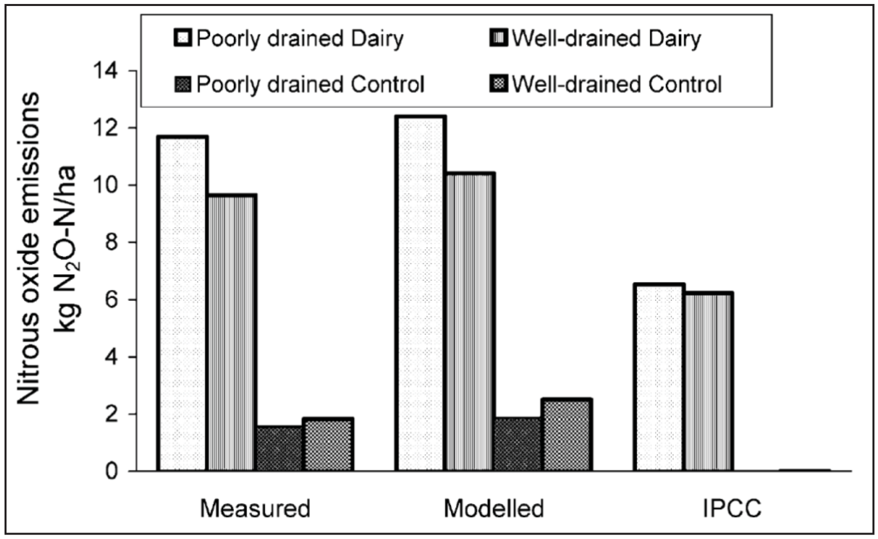

Figure 7-25: Annual-measured, model-predicted and IPCC-calculated N_2O emissions from two ungrazed and dairy-grazed sites (from Saggar et al., 2004c)

The service was then defined as the difference between the maximum potential N_2O emissions simulated using SPASMO, and the Modelled total N_2O emissions for each year. The maximum potential N_2O emission every year was obtained by simulating dung and urine deposition systematically on wet soils (Fig. 7.26). The measure of the service represents for each year the N_2O that could potentially be emitted from the soil, but wasn't thanks to SWC regulation.

The measure of the regulation of N_2O emissions was then calculated as following:

N_2O reg = (Max N_2O emissions – Modelled N_2O emissions) *310

Where N_2O reg is the regulation of N_2O emissions (the N_2O that wasn't emitted from the soil) in kgCO_2 eq/ha/yr, Max N_2O emissions is the maximum potential N_2O emissions for each scenario, modelled with SPASMO in kg N_2O/ha/yr, Modelled N_2O emissions is the total N_2O emissions for each year modelled with SPASMO in kg N_2O/ha/yr, and 310 is the global warming potential for a 100 year time period[11] of N_2O (GWP, 2011) (Fig. 7.26).

For the base case scenario, Max N_2O emissions = 13.6 kg N_2O/ha/yr and Modelled N_2O emissions = 9.6 kg N_2O/ha/yr, therefore N_2O reg = (13.6-9.6)*310 = 1240 kg CO_2 eq/ha/yr.

[11] Depending on the time horizon considered, GHGs have different global warming potentials.

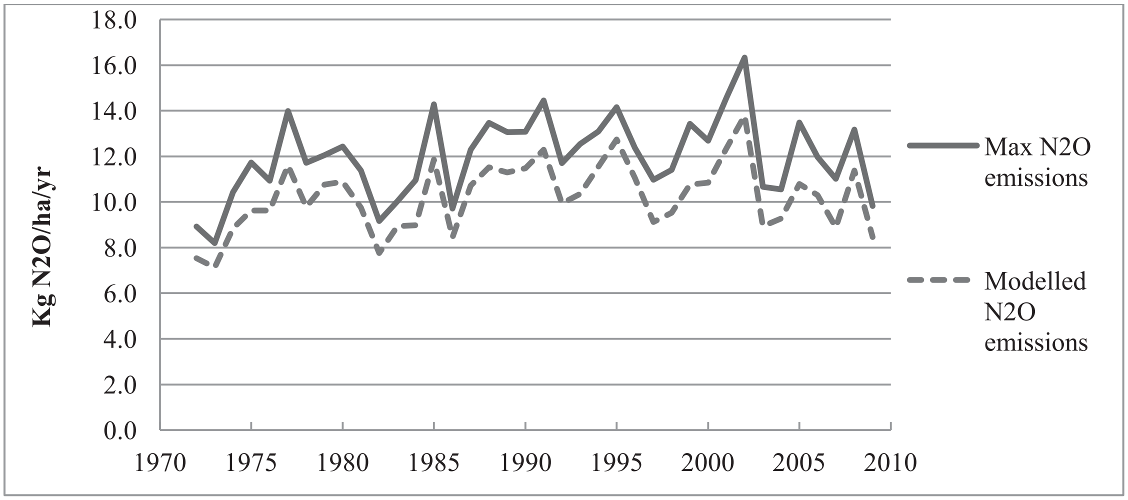

Figure 7-26: Max and modelled N_2O emissions calculated from SPASMO (kg N_2O/ha/yr) over 35 years.

Even when the animals are not on the pasture, their wastes emit some N_2O either from wintering pads or from effluent ponds (Saggar et al., 2003b). It could be argued that these emissions need to be taken into account when measuring the regulation of N_2O emissions from soils. However, what is measured here is the ability of soils to regulate N_2O emissions for a given management and a given amount of N applied. We are not looking at the total GHGs emissions from a farm.

<u>Regulation of CH_4 oxidation:</u>

In order to completely describe the regulation of GHGs by soils, the oxidation of CH_4 was also considered since the degradation of CH_4, a powerful GHG, by soil biota is an ecosystem service. The amount of CH_4 oxidised by pastoral soils at the farm scale is very small, between 0.3 and 2 g CH_4-C/ha/day (Saggar et al., 2008), that is around 0.9 kg CH_4/ha/yr or 19 kg CO_2 eq/ha/yr (using the global warming potential of CH_4 as 21 for 100-year time period). These amounts can seem negligible compared to the amounts of C lost or N_2O emitted. However, when looking at the scale of a country, CH_4 oxidation from soils is worth including in GHGs inventories.

To quantify CH_4 oxidation by soils, the outputs of the SPASMO model as well as data from the literature (Saggar et al., 2008) were used.

Soil water content was followed using SPASMO outputs. Every day, if SWC<FC, it was assumed that an Allophanic Soil would oxidise 2 gCH_4-C /ha/day, and if SWC>FC, 1.3 gCH_4-C /ha/day, according to the data of Saggar et al. (2008)[12] (Table 7.9) (Fig. 7.27).

Table 7-9: Seasonal methane uptake (sink) from two soil types (Saggar et al., 2008).

	Well drained soil gCH_4**-C /ha/d**	**Poorly drained soil** gCH_4**-C /ha/d**
Summer	2	2
Winter	1.3	0.3
Total in kg/ha/yr	0.57	0.48

The service was then defined as the total amount of CH_4 oxidised, which represents the difference between no oxidation and the actual CH_4 oxidation rate.

The measure of the total CH_4 oxidation in CO_2 equivalent is then calculated as follows:

CH_4 oxidation = Total CH_4 oxidation * 1.33 * 21

Where Total CH_4 oxidation is the amount of CH_4 oxidise in kg CH_4-C/ha/yr, 1.33 is the conversion factor from kg CH_4-C to kg of CH_4, and 21 is the global warming potential of CH_4 (for 100-year time period) from CH_4 to CO_2 equivalent (kg CO_2 eq/ha/yr).

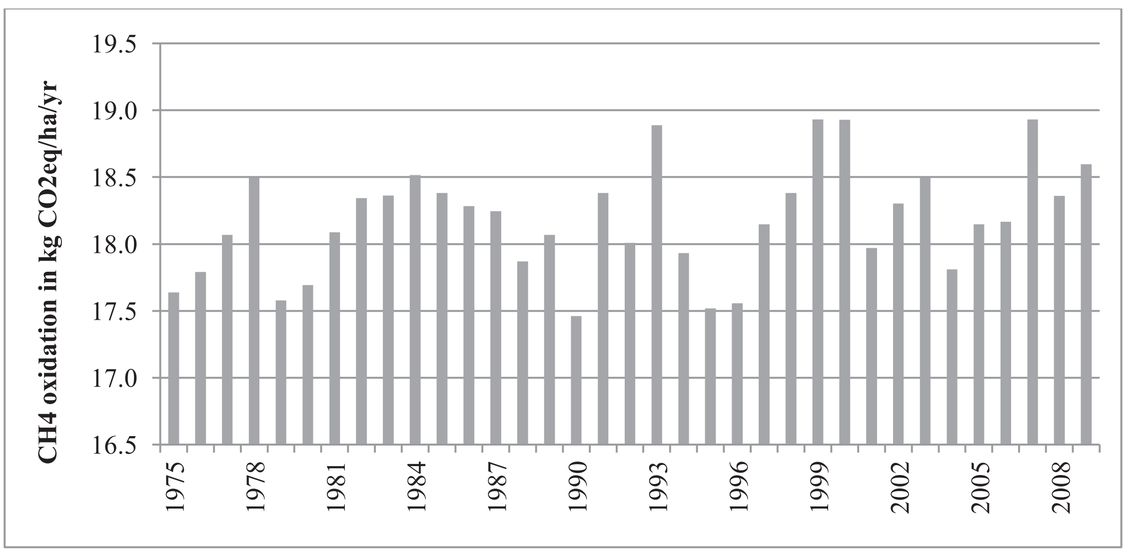

Figure 7-27: CH_4 oxidation (kg CO_2 eq/ha/yr) over 35 years.

For the base case scenario, the Horotiu soil oxidised on average 18.4 kg CO_2 eq/ha/yr. Compared to C flows or N_2O regulation this amount is negligible, it was included here for completeness.

[12] A poorly drained soil was assumed to be able to oxidise 2 gCH_4-C /ha/day if SWC<FC, and 0.3 gCH_4-C /ha/day if SWC>FC.

7.9.2 Valuation of carbon storage and GHGs regulation:

To value carbon storage and Greenhouse gases regulation from soils a market price of CO_2 was used. The measures of C flows, N_2O regulation and CH_4 oxidation services converted to CO_2 equivalents were summed and multiplied by the market price of CO_2. The market price of CO_2 used here was NZ$ 26 /t CO_2. This value is highly controversial and market prices change every day but it corresponds to the commonly used value of €15/t CO_2 (2010).

The value of the carbon storage and Greenhouse gases regulation was then calculated as following:

S6 = (C flow + N_2O regulation + CH_4 oxidation) * 26/1000

Where S6 is the value of the carbon storage and Greenhouse gases regulation from soils in NZ$/ha/yr, C flow is the Net C flow for 35 years in kg CO_2 eq/ha/yr, N_2O reg is the regulation of N_2O emissions in kg CO_2 eq/ha/yr, CH_4 oxidation is the amount of CH_4 oxidise in kg CO_2 eq/ha/yr, 26 is the market price of CO_2 in $/t CO_2 and 1000 is the factor to convert kg in t of CO_2 equivalents.

For example, for the base case scenario, C flow = -1188 kg CO_2eq/ha/yr, N_2O reg = 1240 kg CO_2eq/ha/yr and CH_4 oxidation = 18.4 kg CO_2eq/ha/yr; therefore S6 = $1.8/ha/yr

Carbon storage and GHGs regulation (S6) = Net C flow (S6a) -36 + N_2O regulation (S6b) 15 + CH_4 oxidation (S6c) 0.47 = - $20.5/ha/yr on average

Porter et al. (2009) and Sandhu et al. (2008) attempted to model C storage. Both studies mentioned C accumulation as an ecosystem service. They estimated the C accumulated in plant and root residues from crops but this methodology fails to consider net C flows. They also used C price to value the service. To our knowledge, no one has tried to value N_2O regulation and CH_4 oxidation from as part of an ecosystem services framework.

7.10 Regulation of pest and disease populations (S7):

For a dairy farm system, soils play a major role in the regulation of a number of pest and disease populations but only three were considered in this study: two pasture pests, namely Porina caterpillars and grass grub, and an internal pest of dairy cows, parasitic nematodes.

The biological regulation of pest and disease populations is supported by natural capital stocks including Mp, SWC and food sources (e.g. OM inputs to the soil), influencing soil biodiversity. The regulation of porina caterpillar and grass grubs populations were examined regarding these stocks.

7.10.1 Quantification of the regulation of pest populations:

To inform the part played by soils in the regulation of Porina caterpillars and grass grub populations, the dynamics of soil properties (SWC and Mp) were considered, as well as data from the literature.

Kalmakoff et al. (1993) reported a dramatic rise in the porina (*Wiseana*) population in the first year after sowing a new pasture, followed by a decrease in population in year 2 and 3. Larval density of a new pasture in the first year (47.8 larvae/m^2) was four to five times the density of an old pasture (9.2 larvae/m^2) where viruses and parasites of the larvae were well established. Similarly, Jackson (1990) reported initially low numbers of grass grub (*Costelytra zealandica*) in young pastures, that commonly rose to a peak 4-6 years after sowing, before declining. He also noted that grass grub numbers in older pastures rarely reach the same levels as the early peak, thanks to natural biological control agents.
Therefore, the value of the biological control provided by biodiversity in well-established pastures (older than five years) also needed to be taken into account when valuing pests' regulation by soils.

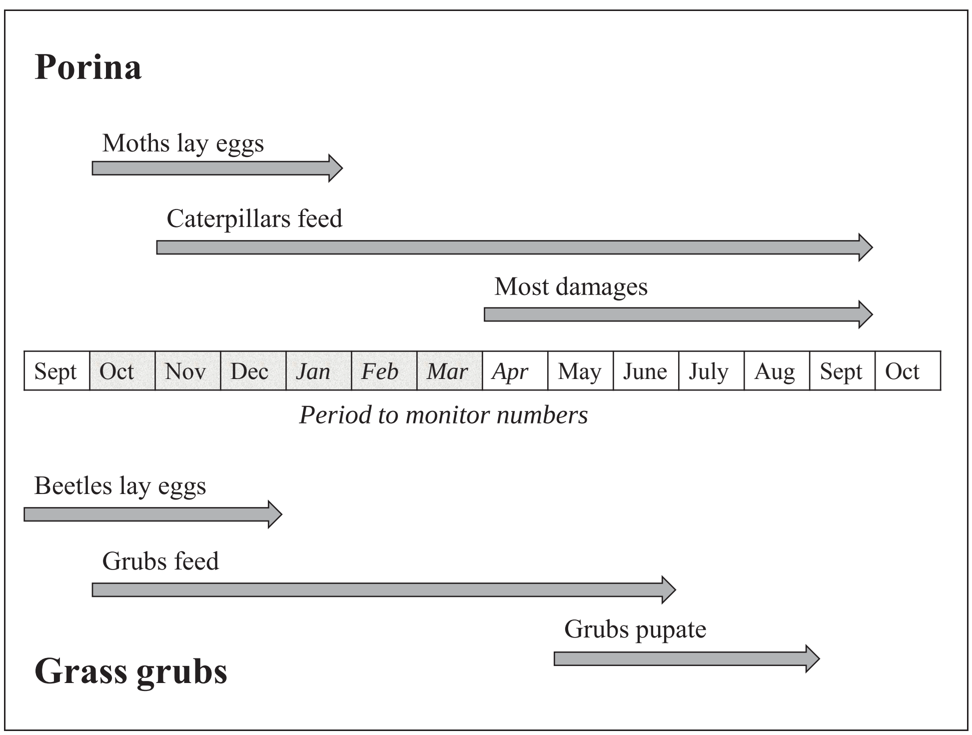

Figure 7-28: Life cycles of porina and grass grub (PGGwrightson, 2010).

Eggs and young larvae of porina and grass grubs are very sensitive to SWC extremes (Fig. 7.28). Farmers are recommended to maintain low pasture cover during late spring to reduce survival of eggs and young larvae. More mature larvae are sensitive to cattle treading and low Mp. Therefore, the dynamics of soil properties (SWC and Mp) were followed and associated with a risk of pest development. Ideal conditions for pests' development were assumed for each pest (Table 7.10).

Table 7-10: Ideal conditions for pest development.

Pest	Soil water content	Macroporosity
	Oct - Dec	Jan - Mar
Porina caterpillar	SP<SWC<FC	>9
Grass grubs	SP<SWC<FC	>9

The outputs from SPASMO were used to follow SWC and Mp across the year. The number of days between October and December (92 days) when SWC was meeting ideal conditions for eggs and young larvae development was calculated first, and then the number of days between January and March (90 days) when Mp was ideal for mature larvae. For pest control to be effective, farmers usually assess the number of pests in early autumn (March-April) before too much damage is done to the pasture. Pesticide applications are usually recommended for the end of April, therefore the risk of pest infestation needs to be assessed before April.

The total number of favorable days between October and March (FavD) was then linked to a level of infestation (Table 7.11) to serve as a proxy for the pest regulation. These levels correspond to infestation rates for well established pastures, that is pastures where biological control agents are already well established. It was assumed here that high infestation levels for well-established pastures are at most half of the initial infestation rates on new pastures (Jackson, 1990; Kalmakoff et al., 1993). It was also assumed that the link between soil properties and infestation levels are the same for both Porina and grass grubs, since the larvae of these two species are sensitive to the same soil properties.

Table 7-11: Number of favorable days (FavD) for pest development between October and March and infestation levels.

Infestation levels	FavD	% of favorable days
Low	<37	<20%
Medium	37-90	20-50%
High	>90	>50%

The service was then defined as the difference between the worst case scenario, all days (182) between October and March are favourable to pest development, and the modelled number of favourable days calculated from SPASMO outputs. The service corresponds to the number of days unfavourable to pest development, which is the number of days when pest population is regulated by soil properties.

The measure of the regulation of pest populations was then calculated as following:

P reg = 182- FavD

where P reg is the number of days unfavourable to pest development between October and March, 182 is the total number of days between October and March, and FavD is the number of favorable days for pest development between October and March.

For example, for the base case scenario, FavD = 22 days which corresponds to a low level of infestation, therefore P reg = 182-22 = 160 days.

As mentioned in Chapter Four, parasitic nematodes are a problem for young livestock but not for well fed mature dairy cows, who develop immunity over time. If writing about a complete dairy system, with grazing calves, the value of the regulation of nematodes population by soils would need to be taken into account. However, this study only considers the paddocks in the milking platform. Therefore, nematodes are mentioned here for completeness, but no value is placed on their regulation.

7.10.2 Valuation of the regulation of pest populations:

Pests and disease infestation in dairy grazed systems can cause severe production losses by either loss of pasture production due to plant destruction, or by loss of milk production due to animal health issues. Therefore, the cost of pest and disease infestations could be valued by the loss of production they incurred. However, the regulation of pest populations by soils impacts on a number of soil properties, like e.g. macroporosity, OM levels, and thereby on a number of soil services. Therefore the regulation of pest populations by soils cannot be valued only by loss of production, because it would underestimate the value of this service.

To value the regulation of pest and disease populations from soils, the provision cost method was used (Chapter Six). If the soil fails to regulate pest and disease, insecticides can be used. They are a way to provide the service by other means. Therefore the cost of applying insecticides to regulate pest populations in a well-established pasture was used as a proxy for the value of the service.

Several products are available. A broad spectrum insecticide was chosen, efficient on porina and grass grubs, because it was assumed that it is what farmers would use.

The dose of insecticide needed depends on the level of infestation. It was assumed that for a well-established pasture, and a high infestation rate, the dose of insecticide that needed to be applied was 50% more than for a medium infestation rate (according to the product label) (Table 7.12). It was also assumed that for a low infestation rate, pesticide wouldn't be applied therefore the costs are nil (Table 7.12). Application costs need to be added to the cost of the product. They are about $20/ha for a liquid insecticide ($20 at water rate of 200L/ha) (Pangborn, 2010) (Table 7.12).

Table 7-12: Costs of application for the insecticide Diazinon 800 EC ($460.45 for 20L) for a well-established pasture.

Pest	Infestation rate	Dose	Cost	Application cost	Total cost	Initial infestation costs
		L/ha	$/ha/yr	$/ha/yr	$/ha/yr	$/ha/yr
Grass grub	Low	0	0	0	0	
	Medium	3	69.1	20	89.1	
	High	4.5	103.6	20	123.6	227.2
Porina caterpillar	Low	0	0	0	0	
	Medium	1	23	20	43	
	High	1.5	34.5	20	54.5	89

The value of the biological control provided in well-established pastures also needed to be taken into account and compared to high initial infestation levels in new pastures. Initial infestation rates in new pastures are usually much higher than infestation rates in well-established pastures because in new pastures, the predators and diseases of pasture pests are not yet present. Therefore, the costs of insecticide application for initial infestation rates in new pastures was assumed to be twice the cost of insecticide application for high levels of infestation in a well-established pasture, that is $227.2/ha/yr for grass grubs (twice the dose + application costs) and $89/ha/yr for Porina (twice the dose + application costs). In doing so, the regulation of pests and diseases due to biocontrol agents and inter-species competition is accounted for, on top of the control by soil properties.

For each pest, the value of the service was then defined as the difference between the cost of insecticide application for initial infestation rates (I high), and the cost of the insecticide application at the infestation rate determined from SPASMO outputs (corresponding to P reg) for a well-established pasture (I actual). The total value of the service is then calculated by adding values for both pests.

The value of the regulation of pest populations is then calculated as following:

S7 = Σ (I high – I actual)

Where S7 is the value of the regulation of pest populations in \$/ha/yr, I high is the cost, for each pest considered of insecticide application at the initial infestation rate in \$/ha/yr, and I actual is the cost, for each pest considered, of insecticide application at the modelled infestation rate in \$/ha/yr. The costs for each pest need to be summed to get the total value of S7.

For example, for the base case scenario, FavD = 22 days which corresponds to a low level of infestation, for grass grubs and Porina, therefore the total value of the service is S7 = (227.2 - 0) + (89 – 0) = \$316.2/ha/yr (Fig. 7.29).

Regulation of pest populations (S7) = \$210 /ha/yr on average

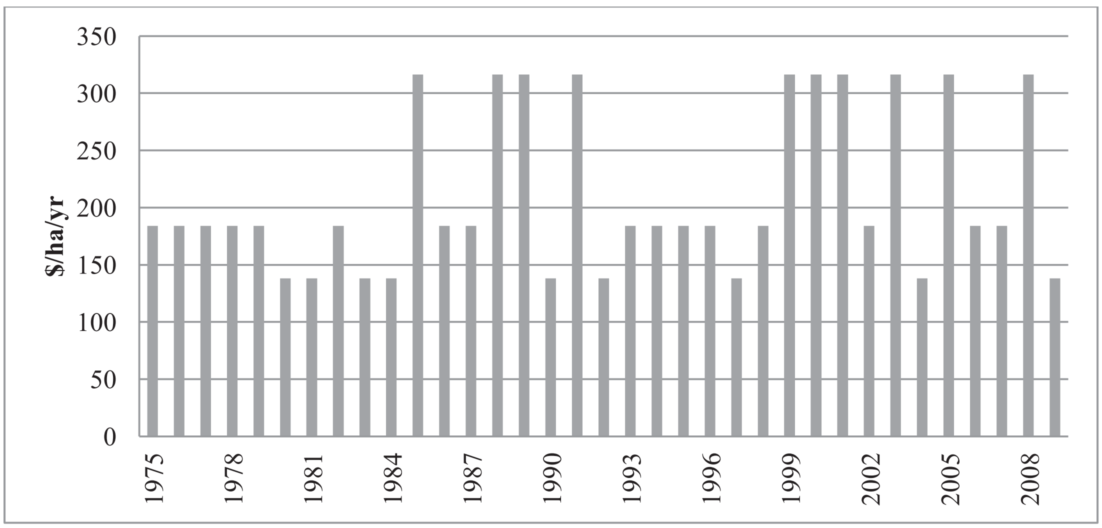

Figure 7-29: Value (\$/ha/yr) of the regulation of pest populations by soils.

Porter et al. (2009) and Sandhu et al. (2008) assessed the biological control of pests in an agro-ecosystem by measuring predation rates of specific insects, but they didn't consider the regulation of pest and disease populations by soils. They value the biological control of pests by using costs of pesticide application and found a value of USD24/ha/yr (\$32.7/ha/yr) which is about 10 times less than the value found in this study.

7.11 Summary of quantification and valuation:

The value of soil services have been calculated here using either market prices, when available, or the construction and maintenance costs of built infrastructures which could provide the services concerned (Table 7.13). Construction costs of built infrastructure were annualised in order to represent the annual value of the flows of services provided each year. Changing from a 10% discount rate to a 3% discount rate when calculating annualisation generally decreased the value of the service by a quarter. This reflects the fact that the higher the discount rate, the higher the value of the annuities corresponding to the present value of the infrastructure (Appendix F) (Holmes, 1998). It is comparable with annual repayments for a mortgage: annual repayments are higher, the higher the interest rate.

The total value of the ecosystem services provided by soils can be calculated by summing up the value of all services (Table 7.13). The average value of soil services from a Horotiu silt loam under a dairy operation over 35 years was $15,777 /ha/yr, ranging from $11,737 /ha/yr to $21,455 /ha/yr, using a 10 % discount rate for annualisation (Table 7.13). The range in the value of the services reflects the interaction between climate and soil properties for the 35 years of continuous weather records used in SPASMO to quantify the soil services.
The aggregation of the values of each service could be criticised because of the issues of joint production and double counting. By using the costs of built infrastructures like a standoff pad or effluent ponds, to value soil services, the values obtained are subject to joint production, as the use of infrastructures, such as a standoff pad, impacts on a number of soil properties (Mp, OM content, nutrient content) and thereby on a number of soil services. Different methods were used to value different services; therefore it is recommended to examine the values of each soil service separately and compare the value of one service between scenarios rather than compare total values.

The study showed that regulating services have a much greater value than provisioning services. Of these the filtering (63.3% of the total value of services) and flood mitigation (7.6% of the total value of services) services had the highest value. Loss of these services would have a major impact on the wider environment and the community by increasing flood risk and the risk of contaminants entering the ground and surface water. Land management at the moment has a strong focus on making maximum use of the provisioning services, such as the provision of food and physical support. This is not surprising as these are the services that are recognised and valued by the market. While the provision of support for animals isn’t marketed as such, it is increasingly valued indirectly through the recognition of the additional costs incurred on soils where the service is poor. Inclusion of the regulating services in the analysis adds a new dimension when exploring the interaction between land use and resource management.

It should be noted here that the value of the ecosystem services provided by soils (annual flows) is different from the value of soil natural capital (stocks). These shouldn’t be confused. An ecosystem services valuation exercise gives us an idea of the value of the flows coming from natural capital stocks, but by no means indicates the value of the stocks. A good example would be the differences in value between soil C stocks ($11,010/ha on average for a Horotiu silt loam at 1m depth) and net C flows ($-36/ha/yr on average) (Table 7.13).

It could be argued that the non-annualised costs of infrastructures therefore correspond to the value of the natural capital stocks they replace. However, this value is a lower bound estimate since built infrastructures are in no way as dynamic, renewable and inter-connected as natural capital stocks. However, in the literature authors (Costanza et al., 1997; Kim and Dixon, 1986) often use the "lump sum" value of built infrastructure as a proxy for ecosystem services valuation. This is not in line with accounting and economic theory, where lump sum value should be amortised or annualised.

Valuing the filtering of P is challenging. The amounts of simulated P lost are very large which shows how strongly some soils retain P. However, since no techniques exist to mitigate losses of P over 5 kg/ha/yr, it was impossible to value this service realistically.

Not all soil services can be provided by built infrastructures. For example, no human infrastructure exists to store C or regulate GHGs emissions. For some relevant services (Table 7.13), the costs of built infrastructures able to provide the services could be used as proxies for the value of the natural capital stocks behind these services. However, when no human infrastructure are able to provide a service, the value of the natural capital stocks behind this service cannot be determined easily.

If these natural capital values are summed up for this study (Table 7.13), it can be argued that the value of the natural capital of the Horotiu silt loam under a typical dairy farm operation is at least $78,198/ha/yr. This value doesn't include all natural capital stocks and doesn't report the value of the interconnectivity, renewability and dynamism of natural capital stocks. Interestingly, it is above (more than double) the current market price of farm land.

Table 7-13: Different values of soil services for the Horotiu silt loam, under a dairy operation, and capital values.

Soil services		**Capital value**	**Average value of service** (3% dr)	**Average value of service (10% dr)**			
				Average	*%*	*Max*	*Min*
Provision of food Quantity	S1a	NA	4,155	**4,155**	*26.3*	*5,655*	*3,158*
Provision of food Quality	S1b	NA	38	**38**	*0.2*	*38*	*38*
Provision of support for human infrastructure	S2a	NA	12	**17**	*0.1*	*17*	*17*
Provision of support for farm animals	S2b	503	89	**112**	*0.7*	*119*	*102*
Provision of raw materials	NC	NC	NC	**NC**	*NC*	*NC*	*NC*
Flood mitigation	S3	10,185	685	**1,196**	*7.6*	*1,661*	*741*
Filtering of N	S4a	NA	529	**554**	*3.5*	*890*	*214*
Filtering of P	S4b	NA	2,426	**2,924**	*18.5*	*2,924*	*2,924*
Filtering of contaminants	S4c	56,112	3,424	**6,513**	*41.3*	*10,787*	*3,320*
Recycling of wastes	S5	388	63	**78**	*0.5*	*143*	*24*
Carbon flows	S6a	11,010 (C stock at 1m)	-36	**-36**	*-0.2*	*-1.2*	*-67.5*
N_2O regulation	S6b	NA	15	**15**	*0.1*	*21.8*	*8.8*
CH_4 oxidation	S6c	NA	0.47	**0.47**	*0.003*	*0.49*	*0.45*
Regulation of pest and disease populations	S7	NA	210	**210**	*1.3*	*316*	*138*
TOTAL value $/ha/yr		78,198	11,610	**15,777**			

NC: not considered, NA: not applicable, dr: discount rate.

Land in the Waikato in 2010 was valued around $45,000/ha for a dairy farm. Farm infrastructures are generally worth, new, about $15,000/ha (Table 7.14), which puts the value of the land down to $30,000/ha. Considering that this land provides every year ecosystem services worth around $14,899/ha/yr, it is safe to say that the actual market value of farm land is currently on the low side.

Table 7-14: Value of some of the infrastructure of a dairy farm.

Asset	Price	Measure for 110 ha	Value ($)
Fontera shares	$4,52/kg MS	99000 kg MS	447,480
Fences	$15/linear meter	25 km	375,000
Tracks	$16/m	2,5 km	40,000
Milking shed (complete shed including rotary milking system)	$14000/bail	50 bails	700,000
Standoff pad	$25/m^2	2000 m^2	50,000
Irrigation system for effluents			15,000
Effluent pond	$15/m^3	3000 m^3	45,000
Troughs (water for animals)	$600/trough	22	13,200
Total		$ for 110 ha	1,685,680
		$/ha	15,324

MS: Milk solids.

The valuation exercise realised here could be improved. This is discussed further in Chapter Ten. The quantification of some services (e.g. the filtering of N and P, the recycling of wastes and the regulation of pest populations) are highly dependent on parameters and thresholds chosen for this study, that wouldn’t be applicable to e.g. a different scale like a catchment. Therefore, the quantification of soil services should always be context and scale driven.

In the next chapter, different scenarios are run with different management practices, and on a different soil.

Chapter Eight

Impact of Soil Type on the Provision of Soil Ecosystem Services

This chapter extends the quantification and valuation exercise in Chapter Seven to a comparison of the ecosystem services from two different soil types under a dairy operation.

8.1 Effect of soil type on the provision of soil services:

The quantification and valuation of soil services from an Allophanic Soil, the Horotiu silt loam (HR) explored in Chapter Seven is extended here to include an examination and comparison of the provision of services from a Gley Soil, the Te Kowhai silt loam (TK), again under a typical New Zealand dairy farm operation. These two soils differ in a number of ways including their available water contents, bulk densities, carbon contents, anion storage capacity and hydraulic conductivity (Table 8.1), and were chosen for this reason to provide a contrast in the natural capital stocks of soils.

The typical New Zealand dairy farm used in this study covers 100 ha, runs 330 milking cows producing 900 kg MS/ha/yr. Fertiliser N use is 100 kg N/ha/yr for both soils. P fertiliser use is 39 kg P/ha/yr for the Horotiu silt loam, and 35 kg P/ha/yr for the Te Kowhai silt loam. The operation does not have a stand-off pad. Pasture silage is made from the farm in spring and fed to the cows as supplements; there is no grazing-off, pastures are rain fed with no irrigation or artificial drainage.

Table 8-1: Soil properties (0-10cm) of the Horotiu and Te Kowhai silt loams (New Zealand National Soil Database).

Soil properties	Te Kowhai silt loam (TK)	Horotiu silt loam (HR)
Soil water content: first 50 cm (in mm)		
Field capacity	54	53
Stress point	41	38
Wilting point	28	25
Available water content	26	28
Soil properties at 10 cm		
Bulk density	1.1	0.84
Total C (%)	2.5	5.5
Anion Storage Capacity (%)	26	91
K sat (mm/day)	30	86
Drainage class	Poorly drained	Well drained
Sensitivity to compaction	Yes	No

The differences in the provision of ecosystem services from theses two contrasting soils for a dairy operation are examined below.

All the values of ecosystem services presented have been calculated using annualised costs of infrastructure (discounted with a 10% discount rate when necessary) and market prices for the provision of food (not TEV). Multipliers used and discussed in the previous chapter haven't been used here. The methods used here enable the reader to compare the results with other studies in the literature.

8.2 Soil services quantification and valuation:

8.2.1 Provision of food quantity:

Pasture yield was modelled over 35 years using the SPASMO model from Plant and Food. The average yields sustained by soil natural capital were 10.4 t DM/ha/yr for HR, and 7.8 t DM/ha/yr for TK (Fig. 8.1). On average, the part of the yield coming from natural capital, modelled with SPASMO for a typical dairy farm, was around 63% for HR, and 54% for TK. These values are consistent with the notion that HR is a more productive soil than TK, with a higher AWC and base fertility supporting higher plant growth.

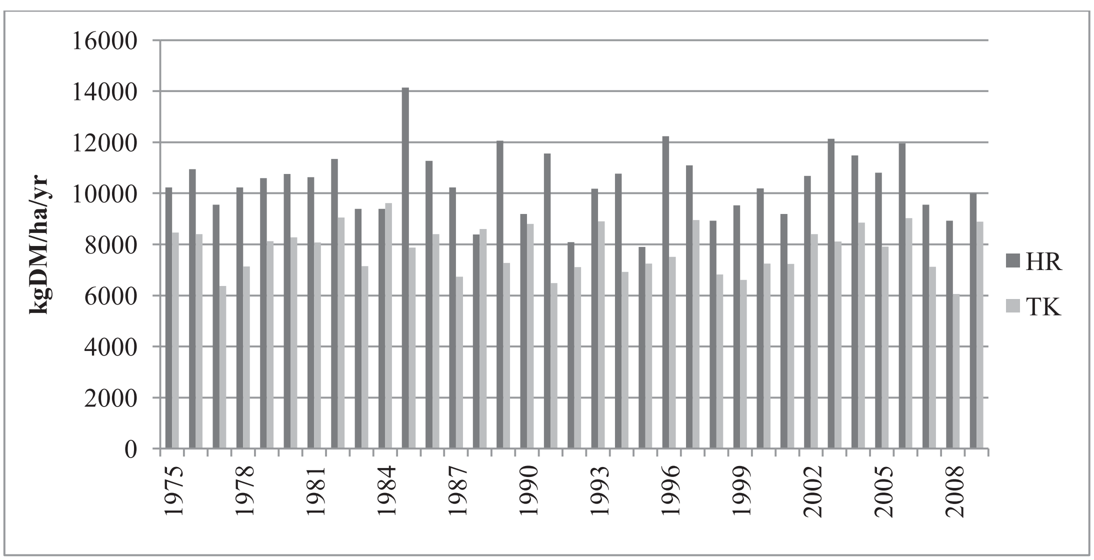

Figure 8-1: Modelled pastures yield (kg DM/ha/yr) for the two soils over 35 years. HR: Horotiu Silt Loam; TK: Te Kowhai Silt Loam.

The provision of food, valued using the market price of milk solids, averaged \$4,155/ha/yr for HR, and \$3,129 /ha/yr for TK over 35 years (Fig. 8.2). It was decided to limit analysis to market prices for the valuation of this service, as it aligns with the literature and is easy to implement.

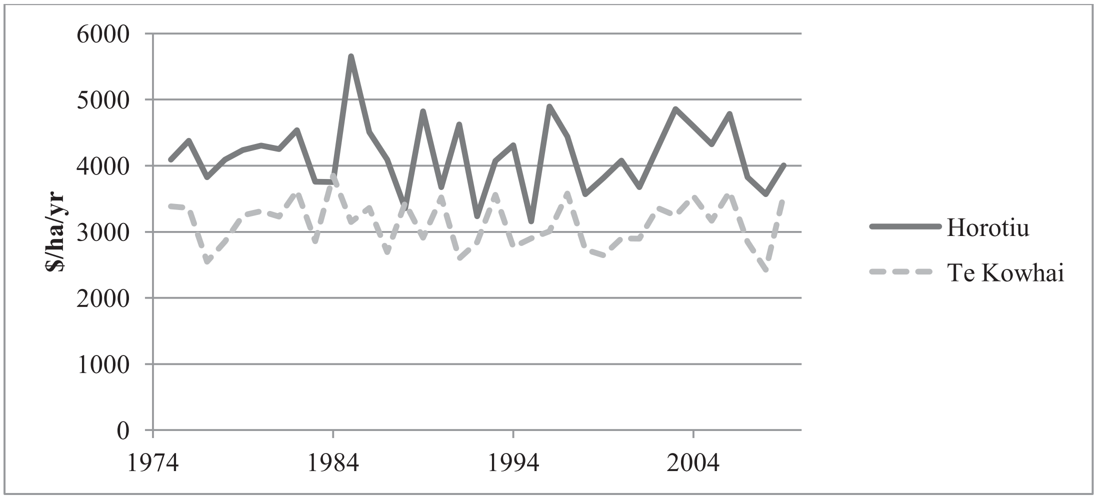

Figure 8-2: Value ($/ha/yr) of the provision of food from the two soils over 35 years.

Annual rainfall does not have a significant effect on the provision of the service, although it is suspected that the timing of rainfall would have been significant (Fig. 8.3).

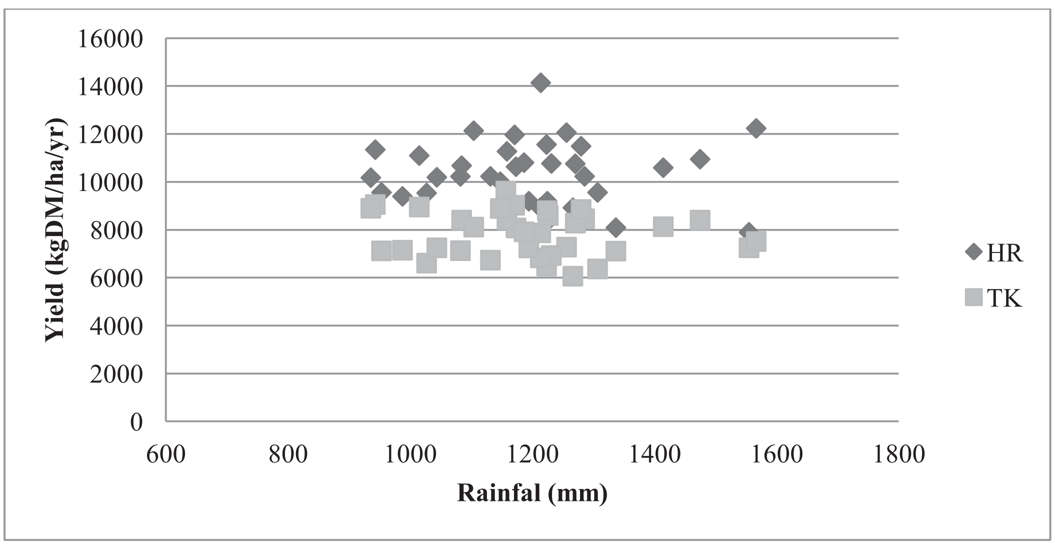

Figure 8-3: Relationship between yield from natural capital and rainfall for the two soils. HR: Horotiu Silt Loam; TK: Te Kowhai Silt Loam.

8.2.2 Provision of food quality:

Neither of the soils considered in this study are deficient in trace elements for pasture agriculture (Grace, 1994). Both therefore have the same value for the provision of food quality $38 /ha/yr. This value corresponds to the cost of applying trace-elements if the soil was deficient. The unrestricted supply of trace elements is often given little thought in decisions on land use or practices.

In this study, the value of the food quality was kept constant over time but in practice, if a soil is farmed long enough it will eventually become deficient in some trace-elements. Nutrient depletion is regarded as a degradation process, directly affecting soil natural capital stocks.

8.2.3 Provision of support for human infrastructure:

The service depends on soil BD below 10 cm, an inherent property for a soil under dairy use and therefore considered to be constant. The BD of HR and TK were 0.84 and 1.1 respectively at 10 cm (Table 8.1). The value of the support for human infrastructure was calculated at $17 /ha/yr for HR, and $25 /ha/yr for TK.

The Te Kowhai silt loam has a lower Mp and a higher BD than the Horotiu silt loam. Therefore it is more compact, providing better support for human infrastructure and has a greater value.

8.2.4 Provision of support to animals:

Soil water content is a major factor in the provision of support to animals determining soil bearing strength and soil sensitivity to the pressure from animal hooves. SWC was followed between May and October (the wet season) using SPASMO to measure the support to animals, or lack of it. The average number of days per year when SWC<(FC+Sat)/2, that is when the soil provides adequate support for animals, was 143 days for HR (78% of the wet season between May and October), and 134 days for TK (73% of the wet season) (Fig. 8.4).

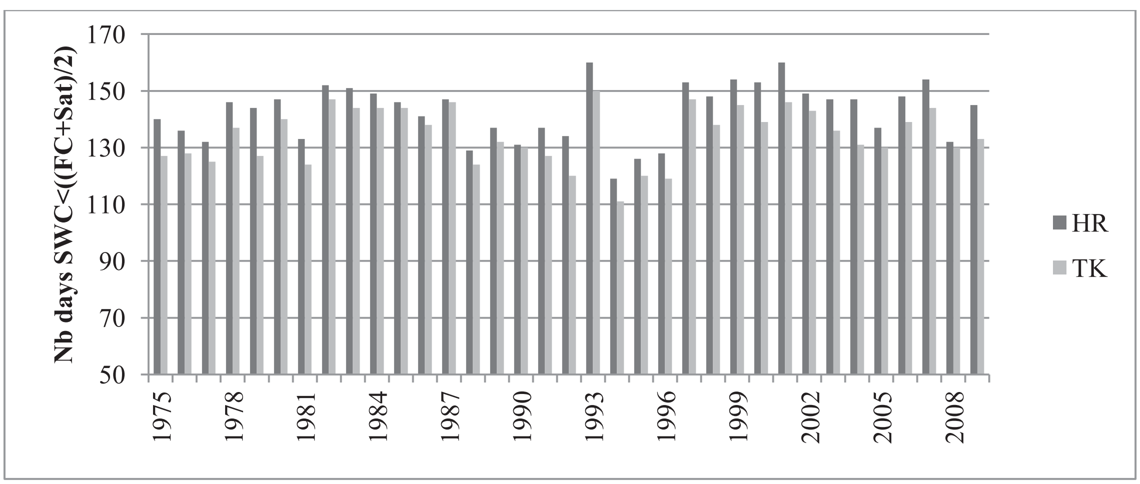

Figure 8-4: Number of days per year when SWC<(FC+Sat)/2 for the two soils over 35 years. HR: Horotiu Silt Loam; TK: Te Kowhai Silt Loam.

On average the value of the service was $112/ha/yr for HR (ranging from $102 to $119/ha/yr) and $108/ha/yr for TK (ranging from $99 to $115 /ha/yr) (Fig. 8.5).

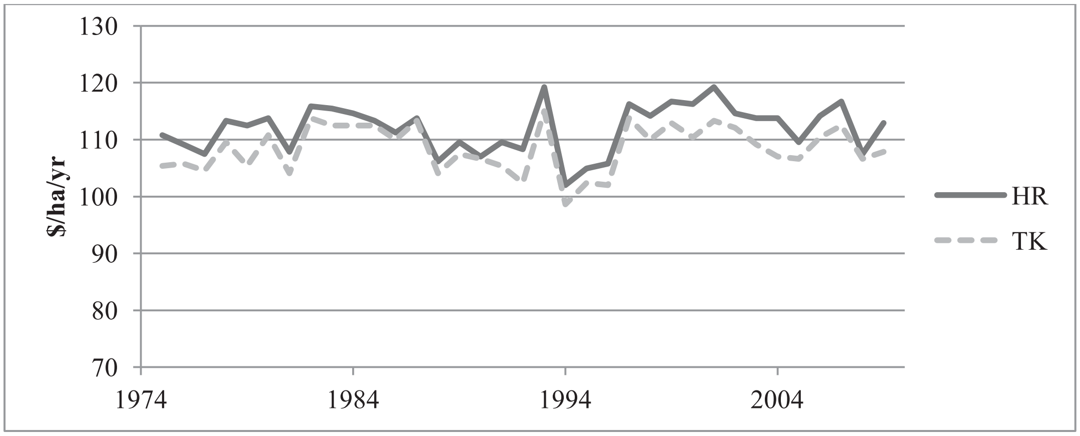

Figure 8-5: Value ($/ha/yr) of the provision of support to animals for the two soils over 35 years. HR: Horotiu Silt Loam; TK: Te Kowhai Silt Loam.

8.2.5 Provision of raw materials:

The provision of raw materials from soils was not included in this study, because it wasn't considered relevant at the farm scale. Therefore a value of 0 is attributed to it by default. It is acknowledge that the provision of raw materials could make a significant contribution to the value of ecosystem services provided by soils in different situations, e.g. at a different scale like the catchment scale or the national scale.

8.2.6 Flood mitigation:

The two soils have very marked differences in their hydraulic properties, reflected in the differences in water conductivities (Table 8.1). Water infiltrates much more quickly in HR than TK (Table 8.1), producing less runoff. The average runoff modelled with SPASMO was 61mm/ha/yr for HR and 248mm/ha/yr for TK. Moreover, the saturation capacities of the soils are also quite different. Over 35 years it averages 61 mm for HR and 58 mm for TK. Saturation capacity changes with macroporosity and thereby is sensitive to compaction and livestock treading damage. On average the water storage capacities of the soils above FC are Sat-FC = 60.5-53 = 7.5mm for HR, and 57.8-54 = 3.8mm for TK, which means that HR is able to store more water than TK before runoff starts, leading to the provision of different ecosystem services.

Averaged over 35 years, the maximum weekly RF-RO every year was 102 mm/ha/yr for HR, and 82 mm/ha/yr for TK. This measure of the service represents the maximum amount of water that is stored by the soil after a rainfall event for each year (Fig. 8.6).

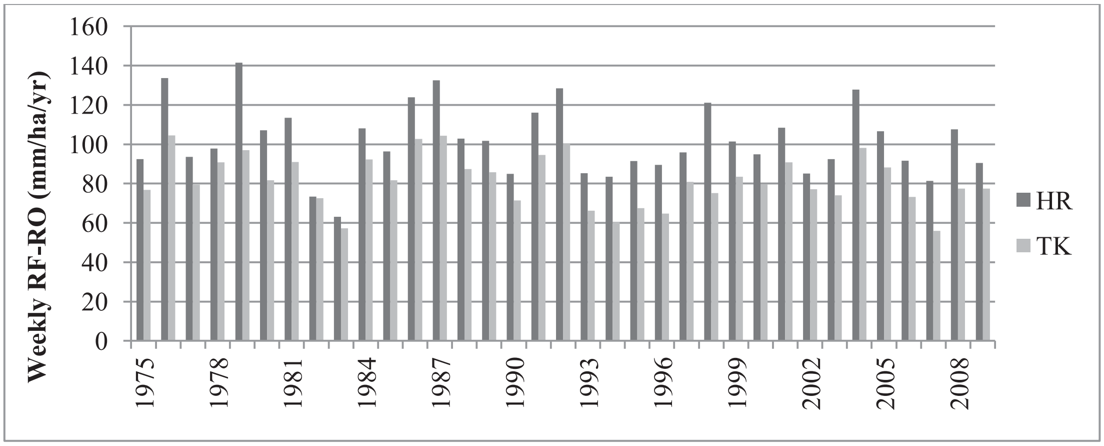

Figure 8-6: Maximum weekly RF-RO per year (mm/ha/yr) for the two soils over 35 years. HR: Horotiu Silt Loam; TK: Te Kowhai Silt Loam; RF: rainfall; RO: runoff.

Averaged over 35 years, the value of the flood mitigation was $1,196 /ha/yr for HR and $960/ha/yr for TK (Fig. 8.7).

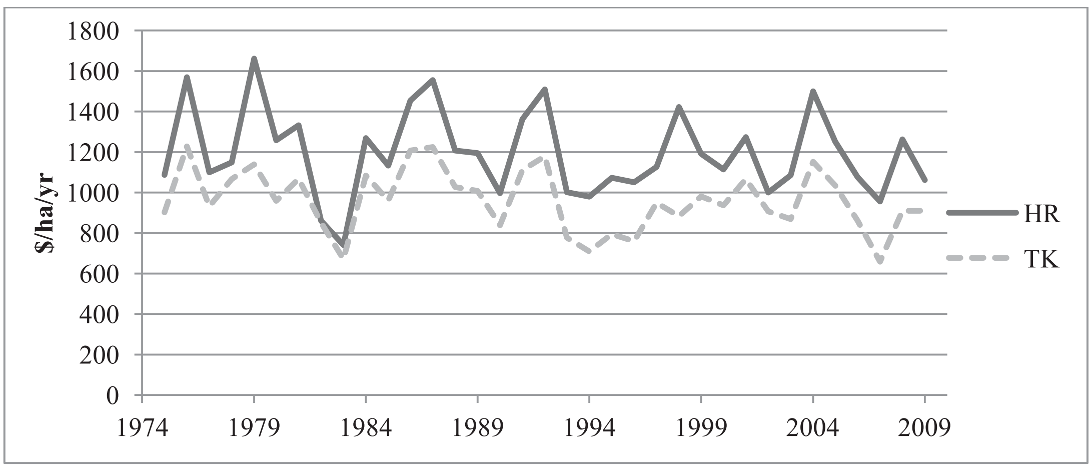

Figure 8-7: Value ($/ha/yr) of flood mitigation for the two soils over 35 years. HR: Horotiu Silt Loam; TK: Te Kowhai Silt Loam.

8.2.7 Filtering of N:

The two soils belong to different drainage classes, with the HR classified as a well drained soil, while the TK is classified as a poorly drained soil.

The potential maximum N loss assessed by running the SPASMO model with very low ASC showed that HR and TK could potentially lose, on average, 61.1 and 24.9 kg N/ha/yr, respectively. Averaged over 35 years, the modelled N leaching losses from HR were 36.8 kg N/ha/yr, whereas TK lost only 20.2 kg N/ha/yr (Fig. 8.8).

The difference between modelled potential maximum losses and actual modelled losses is a quantification of how much N was retained and hence a measure of the filtering service. On average this amounted to 24.3 kg N/ha/yr for HR and only 4.7 kg N/ha/yr for TK (Fig. 8.8). These results show that even though TK leaches less N, it also filters less N than the HR.

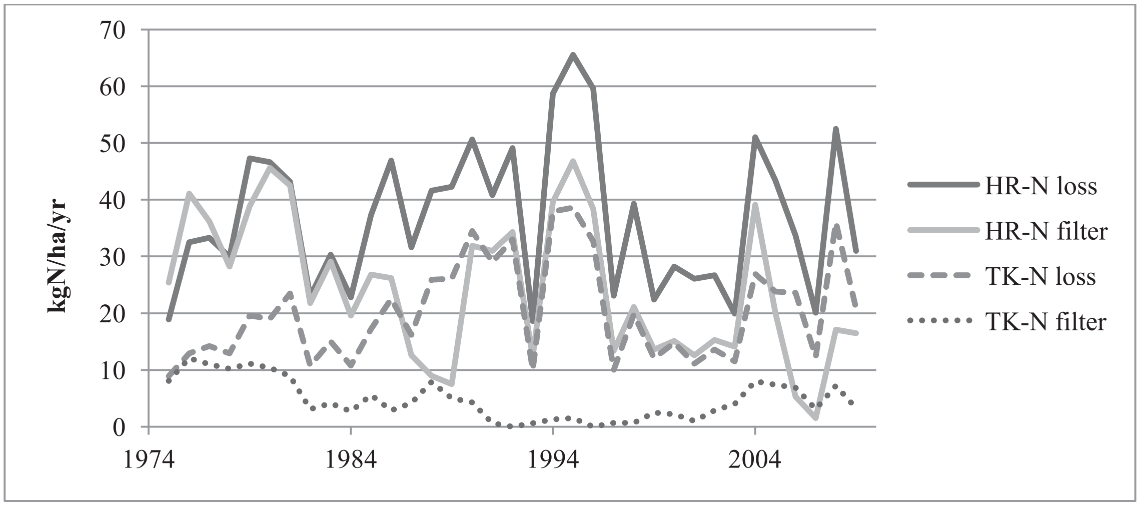

Figure 8-8: Amount of modelled N (kg N/ha/yr) leached and filtered by the two soils over 35 years. HR: Horotiu Silt Loam; TK: Te Kowhai Silt Loam.

Averaged over 35 years, the value of the filtering of N was \$554 /ha/yr for HR and \$328 /ha/yr for TK. These results reinforce the notion that Allophanic Soils are more efficient at filtering and absorbing N than the Gley Soils (Fig. 8.9) (McLaren and Cameron, 1990; Webb and Purves, 1983).

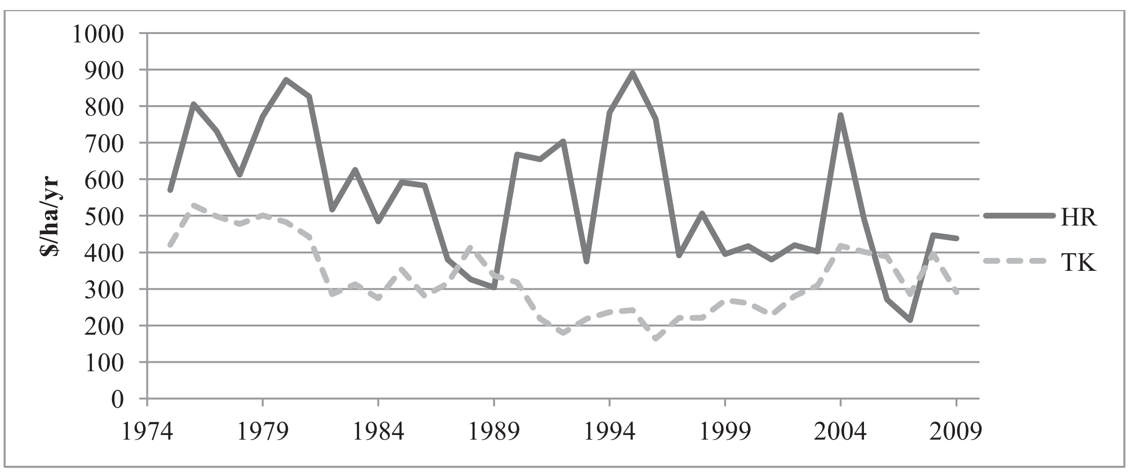

Figure 8-9: Value (\$/ha/yr) of the filtering of N for the two soils over 35 years. HR: Horotiu Silt Loam; TK: Te Kowhai Silt Loam.

As rainfall increases the filtering of N provided by HR also increases, whereas, the amount that TK could filter shows little change (Fig. 8.10).

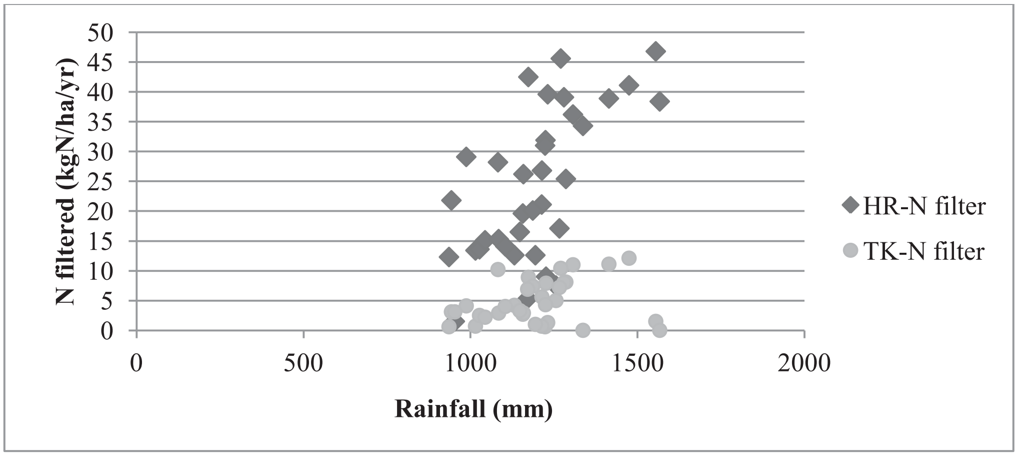

Figure 8-10: Relationship between, the modelled N filtered (kg N/ha/yr) and rainfall (mm) for the two soils. HR: Horotiu Silt Loam; TK: Te Kowhai Silt Loam.

8.2.8 Filtering of P:

The anion or P storage capacity (ASC) of the two soils represent extremes, with the allophanic HR soil above 90% and regarded as high, while the TK soil is regarded as only medium, reflective of most Gley Soils in New Zealand. To quantify the filtering of P, potential maximum P losses were simulated with SPASMO using an extremely low ASC.

For HR, P coming from P fertilisers and animal wastes slowly accumulates in the profile. If there was no filtering service, P would slowly move down the profile and eventually leach out to the wider environment (Fig. 8.11).

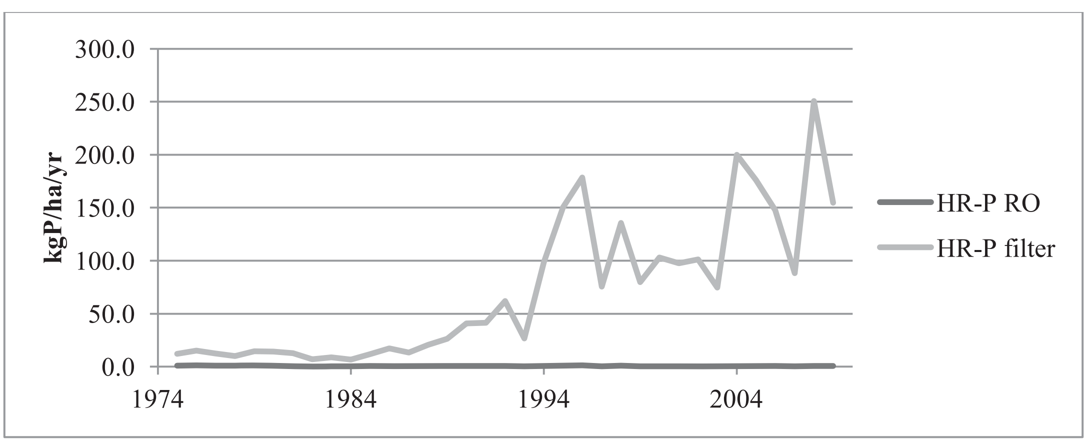

Figure 8-11: Measures of modelled P runoff (P RO) and P filtered (P filter) (kg P/ha/yr) by the Horotiu silt loam over 35 years. HR: Horotiu Silt Loam; TK: Te Kowhai Silt Loam.

For TK, the potential maximum P losses modelled could be as high as 11 kg P/ha/yr, but averaged 6 kg P/ha/yr. In comparison, the potential maximum P losses modelled for HR averaged 72 kg P/ha/yr (Fig. 8.11 and 8.12).

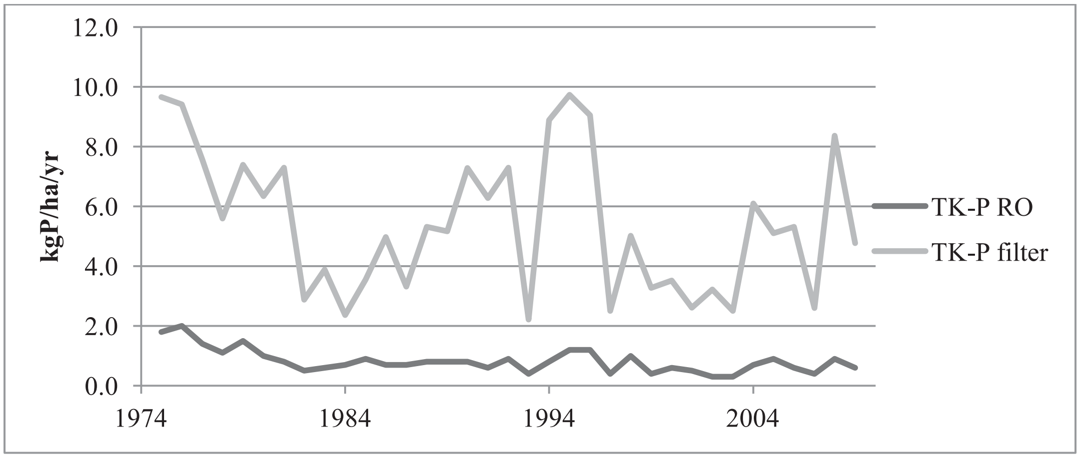

Figure 8-12: Measures of modelled P runoff (P RO) and P filtered (P filter) (kg P/ha/yr) by the Te Kowhai silt loam over 35 years. HR: Horotiu Silt Loam; TK: Te Kowhai Silt Loam.

The large amounts of P filtered by both soils (on average, 71kg P/ha/yr for HR and 5.4kg P/ha/yr for TK), and especially the Allophanic Soil HR, makes it difficult to value this service, as there are very few mitigation strategies available to mitigate P losses above 5 kg P/ha/yr. Using valuation techniques such as defensive expenditures is therefore limited as discussed in Chapter Six and Seven. Mitigation costs were used to value the service up to 5 kg P/ha/yr, but not beyond (Fig. 8.13).

The maximum value that could therefore be put on the filtering of P for HR was $2,924 /ha/yr, corresponding to the costs of mitigation of 5 kg P/ha/yr. The average value of the service for TK was $1,188/ha/yr. The mitigation of 5 kg P/ha/yr for TK was only $1,392/ha/yr (Fig. 8.13), because mitigating P on Gley Soils is more efficient than on Allophanic Soils (Appendix F).

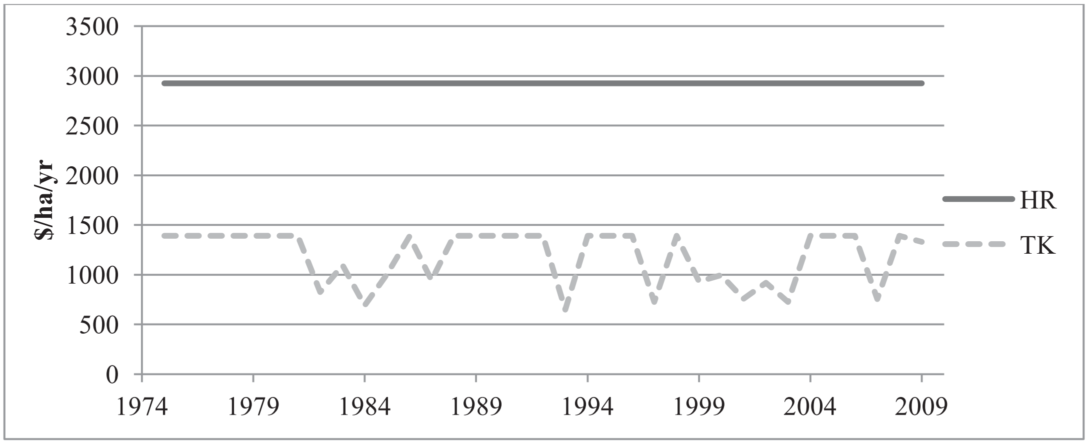

Figure 8-13: Value ($/ha/yr) of the filtering of P for the two soils over 35 years. HR: Horotiu Silt Loam; TK: Te Kowhai Silt Loam.

8.2.9 Filtering of contaminants:

For each year the amount of runoff generated within 5 days of a grazing event, and thereby potentially contaminated by fresh dung pad was used to investigate the filtering of contaminants. The maximum amount of water infiltrating within 5 days of a grazing event (5 days rainfall-runoff) was calculated with SPASMO for every year in mm/ha/yr. Averaged over 35 years, this proxy of the filtering of contaminants was 56 mm/ha/yr for HR and 47 mm/ha/yr for TK (Fig. 8.14).

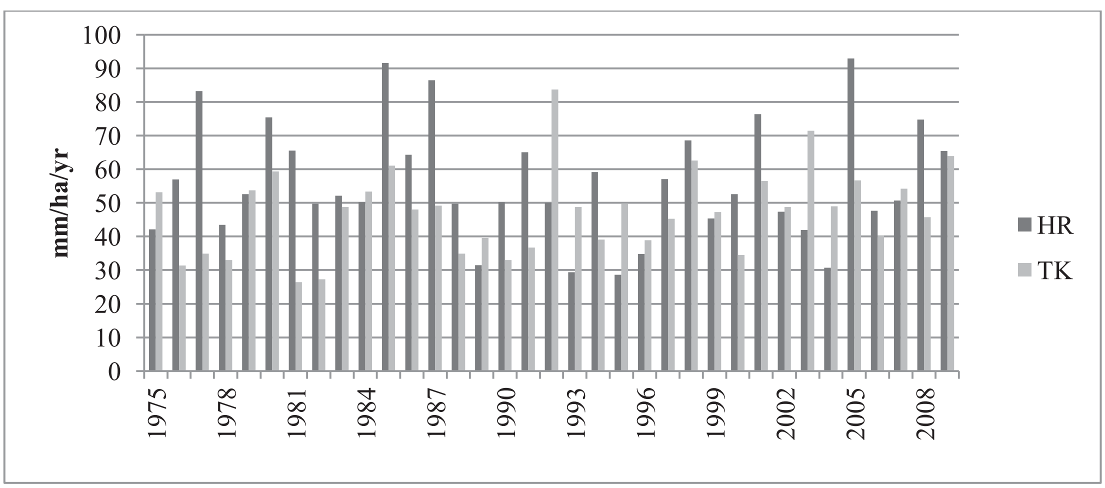

Figure 8-14: Measure of the maximum 5 days RF-RO every year (mm/ha/yr) for the two soils over 35 years. HR: Horotiu Silt Loam; TK: Te Kowhai Silt Loam.

The service was then valued using the costs of a constructed wetland that would filter contaminants instead of the soil. Averaged over 35 years, the value of the filtering of contaminants was $6,513/ha/yr for HR and $5,506 /ha/yr for TK (Fig. 8.15).

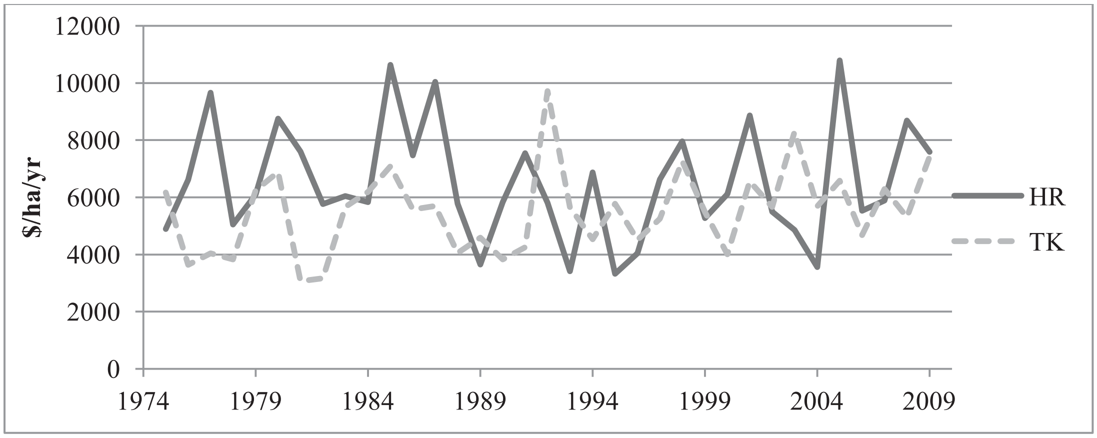

Figure 8-15: Value ($/ha/yr) of the filtering of contaminants for the two soils over 35 years. HR: Horotiu Silt Loam; TK: Te Kowhai Silt Loam.

8.2.10 Recycling of wastes:

To quantify this service the amount of dung deposited in restricting conditions (SWC<SP or SWC>FC) was determined using SPASMO outputs and compared to the total amount of dung deposited annually. Averaged over the 35 years, the percentage of dung deposited that was potentially degraded in optimum conditions was 27.2% for HR and 28% for TK (Fig. 8.16). For this service, the two soils behaved similarly, even in wet years (e.g. 1995, 1996).

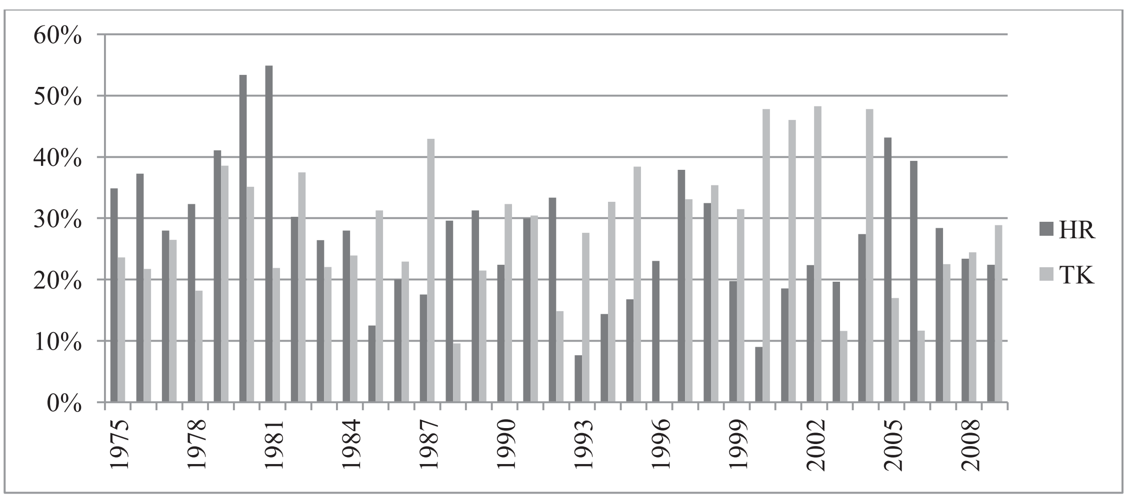

Figure 8-16: Percentage of dung deposited in ideal degradation conditions for the two soils over 35 years. HR: Horotiu Silt Loam; TK: Te Kowhai Silt Loam.

The costs of using an effluent treatment pond to treat the dung currently treated by the soil were used as a proxy for the value of the recycling of wastes. Averaged over 35 years, the value of the recycling of wastes was $78 /ha/yr for HR and $82 /ha/yr for TK.

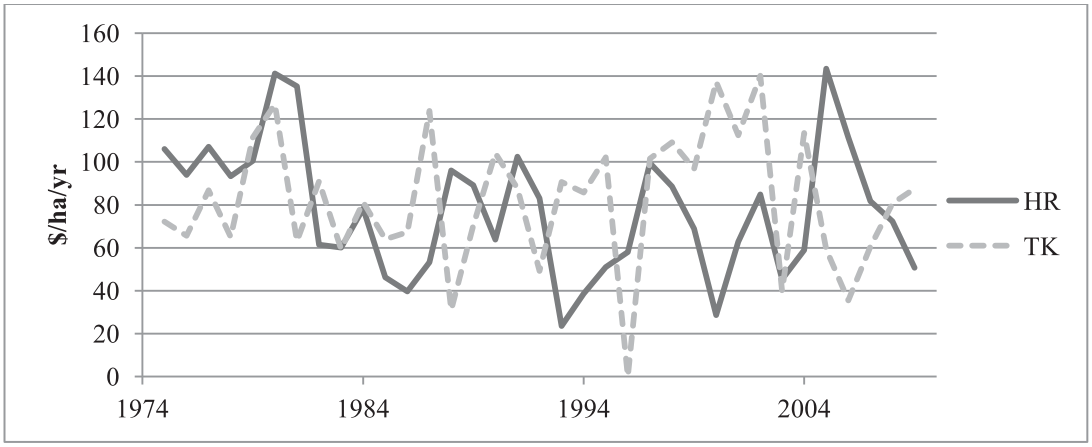

Figure 8-17: Value ($/ha/yr) of the recycling of wastes for the two soils over 35 years. HR: Horotiu Silt Loam; TK: Te Kowhai Silt Loam.

8.2.11 Carbon flows:

The net flows of C modelled with SPASMO averaged -375 kgC/ha/yr for HR, and 13 kgC/ha/yr for TK, over 35 years (Fig. 8.18). The C flows outputs from the SPASMO model were very sensitive to a number of parameters. Current research on the influence of land use and management practices on the variables influencing soil C stocks will provide insights into the relative importance of these parameters.

Until recently soils under old pastures were considered to be at equilibrium, but the study of Schipper et al. (2007) showed that in intensive pasture systems, soil C is a volatile fraction and as such can increase or decrease.

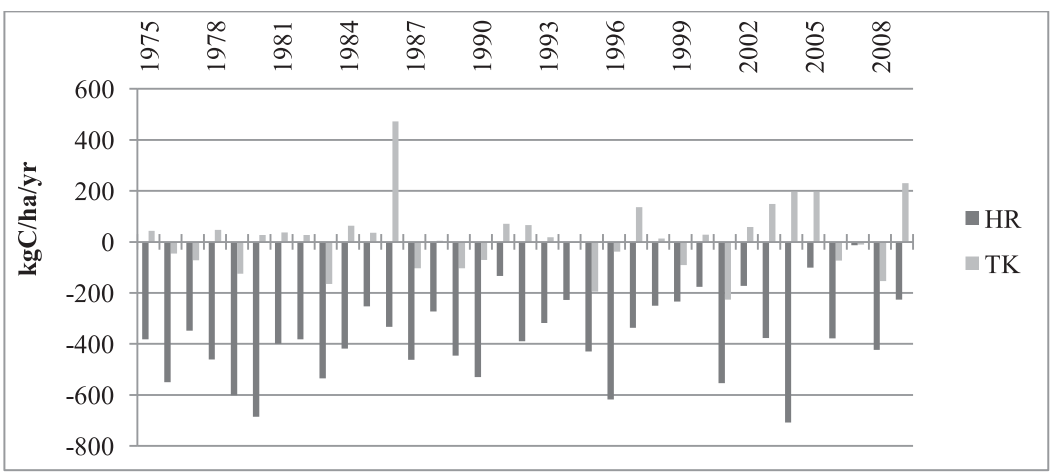

Figure 8-18: Net C flows (kg C/ha/yr) for the two soils over 35 years. HR: Horotiu Silt Loam; TK: Te Kowhai Silt Loam.

Carbon flows, valued using CO_2 market price, averaged $-35.8 /ha/yr for HR and $1.2 /ha/yr for TK over 35 years (Fig. 8.19).

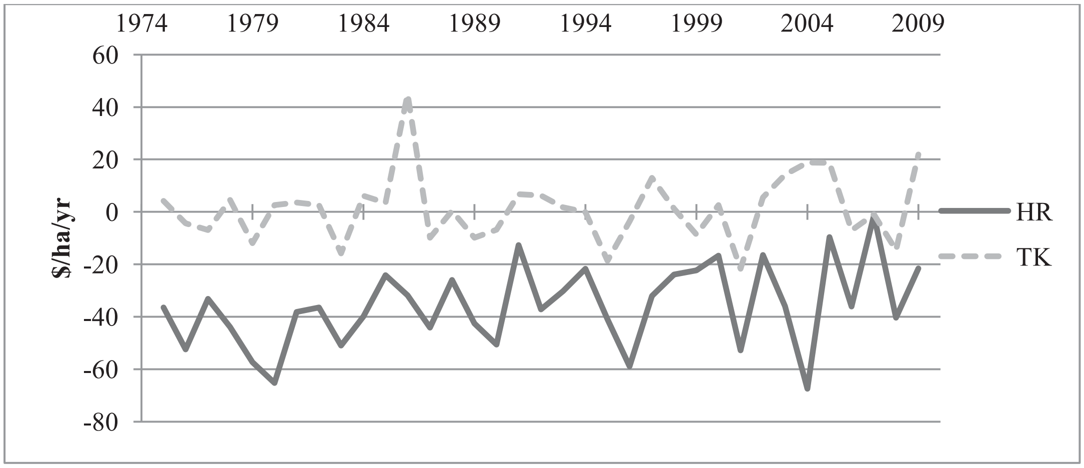

Figure 8-19: Value ($/ha/yr) of C flows for the two soils over 35 years. HR: Horotiu Silt Loam; TK: Te Kowhai Silt Loam.

8.2.12 Nitrous oxide regulation:

This service was measured using a method inspired by the IPCC methodology and based on SWC and N leaching outputs from SPASMO. Averaged over 35 years, N_2O emissions were calculated to be 10.4 kg N_2O/ha/yr for HR and 9.4 kg N_2O/ha/yr for TK (Table 8.2). These include direct and indirect emissions. HR is well drained, leaching more nitrates, which means more indirect N_2O emissions. Moreover, HR also grows more pasture, which means the cows come back more often on the paddock to graze and thereby deposit more dung, creating a greater source of N for direct N_2O emissions (Table 8.2).

Table 8-2: Average direct and indirect emissions of N_2O (kg N_2O/ha/yr) from the two soils over 35 years.

	Horotiu silt loam	**Te Kowhai silt loam**
Direct emissions	7.8	7.4
Indirect emissions	2.7	2.0
Total	10.4	9.4

Regulation of N_2O emissions is influenced by the soil's ability to deal with all anaerobic conditions, including waterlogging and poor drainage. A well drained soil will be less likely to produce N_2O. A measure of the service was defined as the difference between the maximum potential N_2O emissions (modelled for a permanently waterlogged soil), and the actual modelled N_2O emissions for each year. The average N_2O emissions regulated by SWC were 1.81 kg N_2O/ha/yr for HR and 1.78 kg N_2O/ha/yr for TK over 35 years (Fig. 8.20).

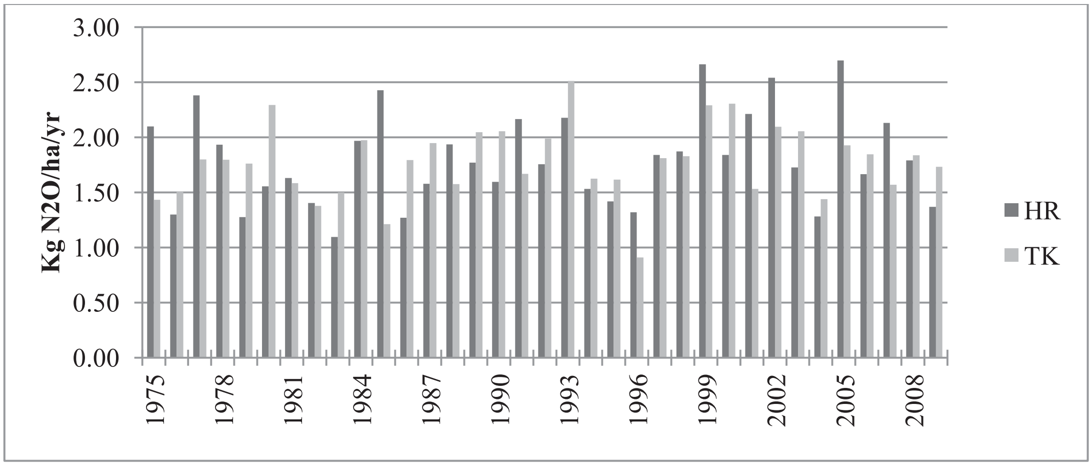

Figure 8-20: N_2O emissions (kg N_2O/ha/yr) regulated by the two soils over 35 years. HR: Horotiu Silt Loam; TK: Te Kowhai Silt Loam.

Regulated N_2O emissions were valued using CO_2 equivalents and C market prices. Over 35 years, they averaged \$14.6 /ha/yr for HR (ranging from \$8.8 to \$21.8) and \$14.3 /ha/yr for TK (ranging from \$7.3 to \$20.2) (Fig. 8.21).

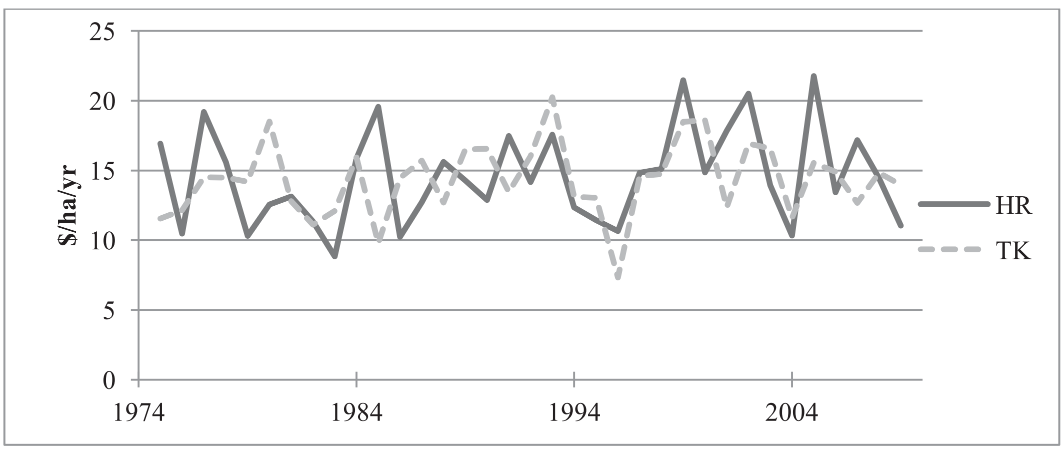

Figure 8-21: Value (\$/ha/yr) of N_2O regulation for the two soils over 35 years. HR: Horotiu Silt Loam; TK: Te Kowhai Silt Loam.

8.2.13 Methane oxidation:

Methane oxidation was quantified using SPASMO outputs for SWC and data from the literature. Averaged over 35 years, the amount of CH_4 oxidised was 0.87 kg CH_4/ha/yr for HR and 0.73 kg CH_4/ha/yr for TK. These amounts are small in comparison with livestock emissions, but even a very small methane-sink capacity could impact strongly on New Zealand's national methane inventory, as nearly all (10.6 million ha) of agriculture land in New Zealand is pasture for livestock farming. Assuming all New Zealand's grazed pastoral soils oxidise methane at a similar rate as the dairy pasture soils in this study, they could potentially oxidise 10 kilotonnes CH_4/yr, 0.8% of New Zealand annual methane emissions of 1264 kilotonnes CH_4 (MfE, 2009b; Saggar et al., 2003a).

Methane oxidation, valued using CO_2 equivalents and C market prices, averaged \$0.47 /ha/yr for HR and \$0.4 /ha/yr for TK, over 35 years (Fig. 8.22). This is negligible at the farm scale, but was added here for completeness. It is significant at larger scales.

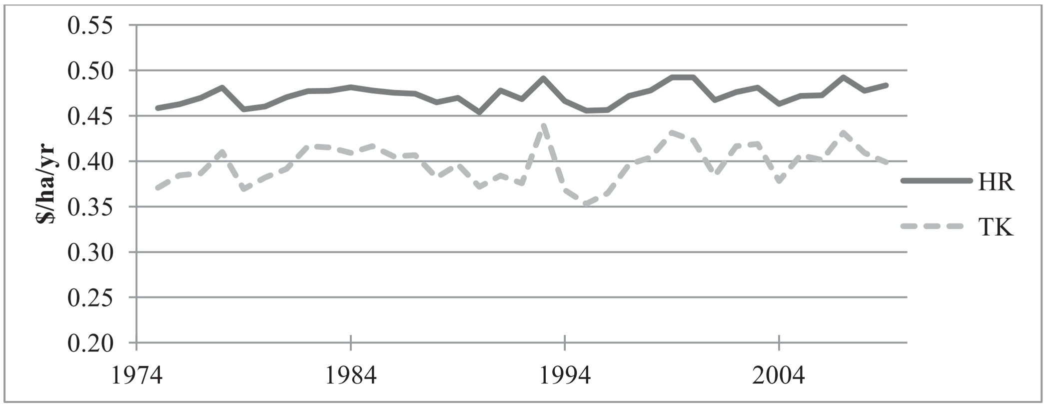

Figure 8-22: Value (\$/ha/yr) of methane oxidation for the two soils over 35 years. HR: Horotiu Silt Loam; TK: Te Kowhai Silt Loam.

8.2.14 Regulation of pest and disease populations:

The number of days between October and December (92 days) favourable for pest development (when SP<SWC<FC and Mp>9), that is the number of days when pest population is not regulated by soil properties were quantified using SPASMO outputs (SWC and Mp). Averaged over 35 years, the number of favourable days for pest development over 6 months was 61 days for HR and 25 days for TK (Fig. 8.23). This means that TK provides better control of pest development than HR.

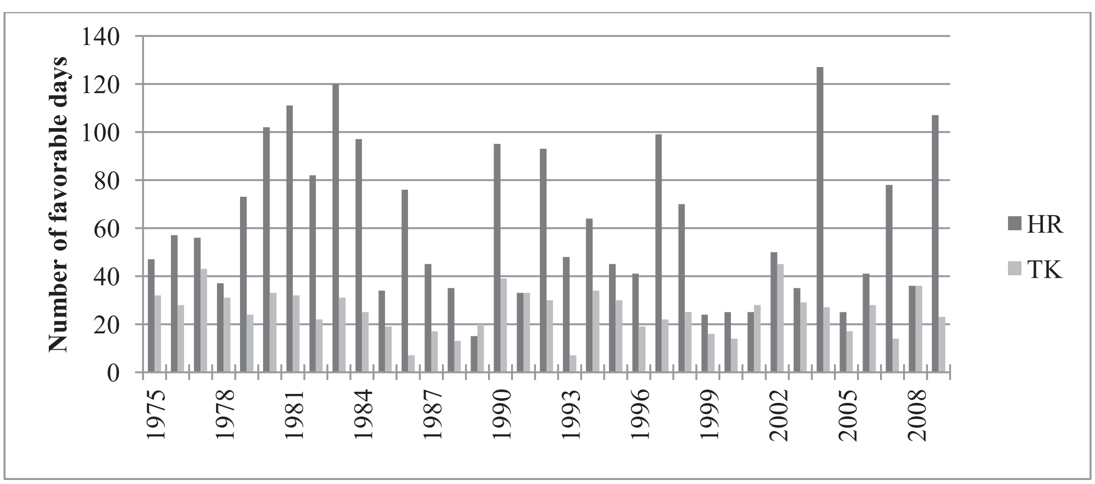

Figure 8-23: Number of favourable days for pest development from October to March for the two soils over 35 years. HR: Horotiu Silt Loam; TK: Te Kowhai Silt Loam.

The market prices of insecticide applications were used to value the regulation of pest and disease populations. Averaged over 35 years, the value of the service was $210 /ha/yr for HR and $305 /ha/yr for TK (Fig. 8.24).

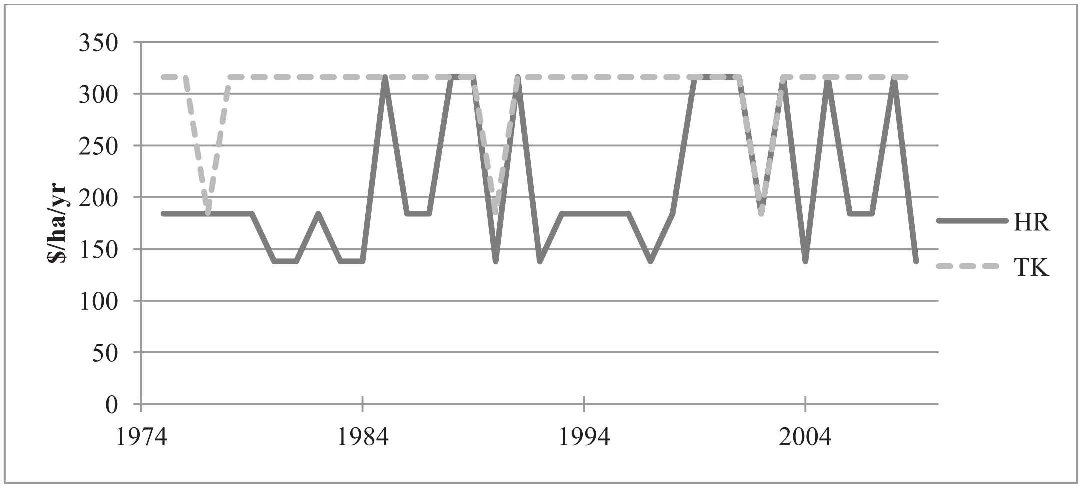

Figure 8-24: Value ($/ha/yr) of the regulation of pest and disease populations for the two soils over 35 years. HR: Horotiu Silt Loam; TK: Te Kowhai Silt Loam.

TK is poorly drained and more compacted than HR, which provides less than ideal condition for pests to develop.

8.3 Overview of the influence of soil type on the provision of soil services:

A summary of the results of the valuation of the services provided by a Horotiu silt loam (HR), and a Te Kowhai silt loam (TK) under a typical dairy farm operation are provided in Table 8.3. Averaged over 35 years, the soil services under a dairy operation were worth $15,777 /ha/yr from HR (ranging from $11,737 /ha/yr to $21,455 /ha/yr), and $11,687 /ha/yr from TK (ranging from $9,347/ha/yr to $15,886/ha/yr) (Fig 8.25). The value of the services provided by HR was on average 35% greater than the value of the services provided by TK.

The major differences in the value of ecosystem services between the two soils were in the provision of food, flood mitigation and the filtering of nutrients and contaminants (Table 8.3). It is interesting to note that for some services (the provision of support for human infrastructure, the recycling of wastes, carbon flows and the regulation of pest and disease populations) the TK soil presented a greater value than the HR (Table 8.3).

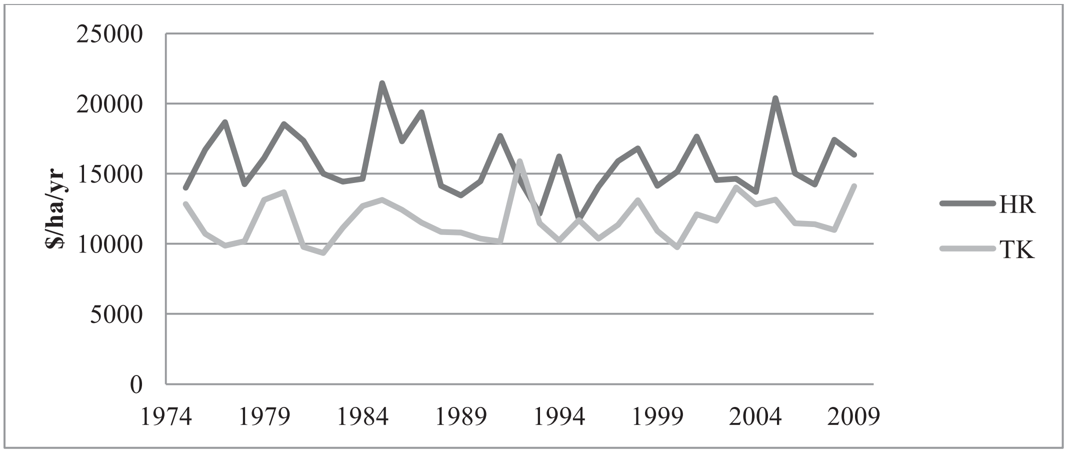

Figure 8-25: Total value ($/ha/yr) of soil services for the two soil types over 35 years. HR: Horotiu Silt Loam; TK: Te Kowhai Silt Loam.

These results reflect the differences in soil structure and soil water dynamics between the two soils, as physical structure is the natural capital stock at the heart of the provision of many of these services under a dairy use. TK is poorly drained, as opposed to HR which is well drained. In contrast, TK is known to have a lower Mp and higher bulk density than HR, providing better support to human infrastructure and better regulation of pest populations.

Table 8-3: Average value ($/ha/yr and % of total value) of soil services for the two soils, under a typical dairy farm operation, over 35 years.

Soil service	**Horotiu silt loam**		**Te Kowhai silt loam**	
	$/ha/yr	**%**	**$/ha/yr**	**%**
Food Quantity	4155	26.3	3129	26.8
Food Quality	38	0.2	38	0.3
Support for human infrastructure	17	0.1	25	0.2
Support for animals	112	0.7	108	0.9
Raw materials	NV	NV	NV	NV
Flood mitigation	1196	7.6	960	8.2
Filtering of N	554	3.5	329	2.8
Filtering of P	2924	18.5	1188	10.2
Filtering of contaminants	6513	41.3	5506	47.1
Recycling of wastes	78	0.5	82	0.7
Carbon flows	-36	-0.2	1	0.0
N_2O regulation	15	0.1	14	0.1
CH_4 oxidation	0.47	0.0	0.40	0.0
Regulation of pest and disease populations	210	1.3	305	2.6
Total	15,777		11,687	

NV: Not valued.

Other authors (Porter et al., 2009; Sandhu et al., 2008) have tried to value the ecosystem services from agro-ecosystems (Table 8.4). Only a limited number of soil services were however considered in any one of these published studies. A major limitation with many of these studies was double counting due to attempts made to value supporting processes rather than services. For example, they considered water and N supply to plants, as well as yields, and C inputs, instead of net C flows. The values found in the literature are presented in Table 8.4 for comparison.

Traditionally soils are compared in terms of their productive capacity and versatility. For the first time, this study enables land managers and policy makers to compare the total utility of soils, not just their productivity and versatility for different land uses. To the author's knowledge, previous attempts to combine regulating services with production were only based on the use of scales and weighted values (Hewitt et al., 2010; Webb and Wilson, 1994). Hewitt et al. (2010) investigated land use suitability by looking at a soil adequacy index of soil services, for a specific land use. The framework used in this study not only enables soil types to be compared, but also a wide range of land uses, including recreational uses.

Table 8-4: Comparison of the value (NZ$/ha/yr) of soil services between different studies.

Soil services	Horotiu	Te Kowhai	Porter et al. (2009)	Sandhu et al. (2008)
Food Quantity	4155	3129	349 (fodder)	6424 (grains)
Food Quality	38	38	NC	NC
Support for human infrastructure	17	25	NC	NC
Support for animals	112	108	NC	NC
Raw materials	NV	NV	0 (wood)	35 (wood)
Flood mitigation	1196	960	122 (water supply)	172 (water supply)
Filtering of N	554	329	699 (N supply)	483 (N supply)
Filtering of P	2924	1188	NC	NC
Filtering of contaminants	6513	5506	NC	NC
Recycling of wastes	78	82	NC	109 (litter)
Carbon flows	-36	1	37 (storage)	22 (storage)
N_2O regulation	15	14	NC	NC
CH_4 oxidation	0.47	0.40	NC	NC
Regulation of pest and disease populations	210	305	NC	NC
Total	15,777	11,687		

NV: Not valued, NC: Not considered.

In the next chapter, the effects of different stocking rates and the use of a standoff pad on the provision of ecosystem services from soils are examined.

Chapter Nine

Effect of Dairy Cow stocking rates and the Use of a Standoff Pad on the Provision of Soil Ecosystem Services

This chapter utilises the quantification and valuation methods developed in Chapters Five and Six and used in Chapter Seven and Eight to extend the quantification and valuation of soil ecosystem services to include an examination of the influence of a range of different management practices under a dairy operation.

The impact on the provision of soil services of management practices, like an increase in dairy cow stocking rate, and the use of a standoff pad, are examined for each of the services provided by a Horotiu silt loam and a Te Kowhai silt loam, under a dairy farm operation.

9.1 Twelve scenarios to investigate farm management:

To determine the impact of dairy cow stocking rates, phosphorus and nitrogen fertiliser practices and the use of a standoff pad on the provision and value of ecosystem services from the Horotiu silt loam and Te Kowhai silt loam soils, under a dairy farm operation, twelve scenarios were constructed and tested. The scenarios considered are listed in Table 9.1. Specifically, the study examined the influence of three dairy cow stocking rates (3,4 and 5 cows/ha) with corresponding N and P fertiliser inputs, and two pasture management options: cows are on the paddock (cows ON) or are taken off the paddock onto a standoff pad when the soils are too wet (cows OFF).

The influence of these management factors was examined as they influenced both the quantity and value of each soil service.

Data reported are averages of the outputs of the SPASMO model for 35 consecutive years starting in 1975. Box plots were used to present the data for each scenario. The range of values for each service reflects the interaction between climate and soil properties, for the 35 years of climate data (1975-2009), for the Waikato, used as inputs to SPASMO to quantify the soil services.

Table 9-1: Detail of the twelve scenarios studied.

Name of scenario	Soil type	Number of cows	Stocking rate (cows/ha)	Milksolids (kg/ha)	N (kg/ha/yr)	P (kg/ha/yr)	Stand off pad
HR-BR	Horotiu	330	3.0	900	100	39	no
HR1	Horotiu	440	4.0	1200	180	53	no
HR2	Horotiu	550	5.0	1500	300	66	no
HR3	Horotiu	330	3.0	900	100	39	yes
HR4	Horotiu	440	4.0	1200	180	53	yes
HR5	Horotiu	550	5.0	1500	300	66	yes
TK-BR	Te Kowhai	330	3.0	900	100	35	no
TK1	Te Kowhai	440	4.0	1200	180	46	no
TK2	Te Kowhai	550	5.0	1500	300	58	no
TK3	Te Kowhai	330	3.0	900	100	35	yes
TK4	Te Kowhai	440	4.0	1200	180	46	yes
TK5	Te Kowhai	550	5.0	1500	300	58	yes

9.2 Soil services quantification and valuation:

9.2.1 Provision of food quantity:

The amount of grass grown from the natural capital stocks decreased with increasing stocking rate. Since plant growth depends on the provision of water, nutrients and support from soils, it is a good indicator of the overall status of the soil. The presence of a standoff pad slightly increased pasture yield for both soils, regardless of stocking rate (Fig. 9.1).

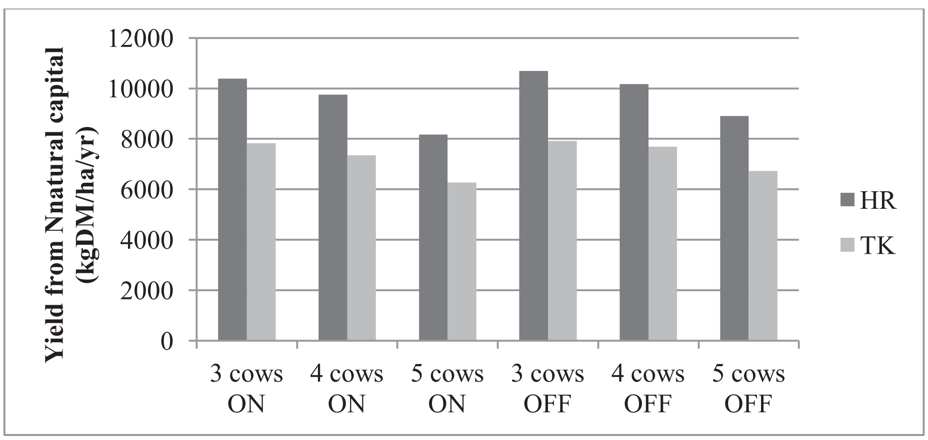

Figure 9-1: Average yield (kgDM/ha/yr) from natural capital stocks for all scenarios. HR: Horotiu Silt Loam; TK: Te Kowhai Silt Loam.

The percentage of pasture production coming from natural capital decreased with increasing stocking rate, reflecting the greater quantities of N and P fertilisers applied (Table 9.1) to generate enough feed for the animals, at the higher stocking rates (Fig. 9.2).

The grazing rotation was determined by the model depending on the amount of grass available on a paddock (Chapter Five). Some pasture silage was produced from the farm and fed back to the cows, but for high stocking rates, silage had to be imported if not enough grass was grown to meet the animals' feed requirements. The model outputs presented in Fig. 9.1 and 9.2 only include the pasture grown, not the total amount of pasture DM fed to the animals.

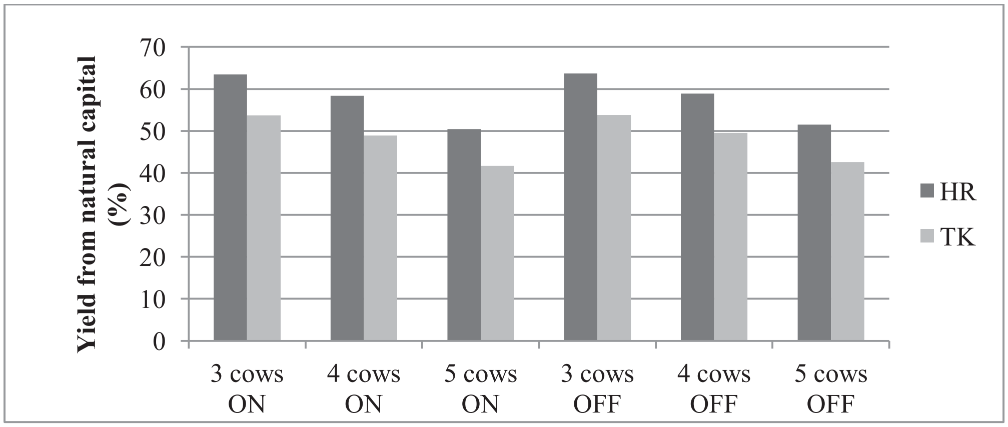

Figure 9-2: Average percentage of total yield from natural capital for all scenarios. HR: Horotiu Silt Loam; TK: Te Kowhai Silt Loam.

The value of the provision of food decreased with increasing fertiliser inputs and associated increasing stocking rate, but increased with the use of a standoff pad for both soils and all three stocking rates (Fig. 9.3). When a standoff pad is used cows spend less time on the paddock when the soil is wet and sensitive to cattle treading. Therefore soil structure (macroporosity) is more efficiently preserved, leading to increased pasture growth.

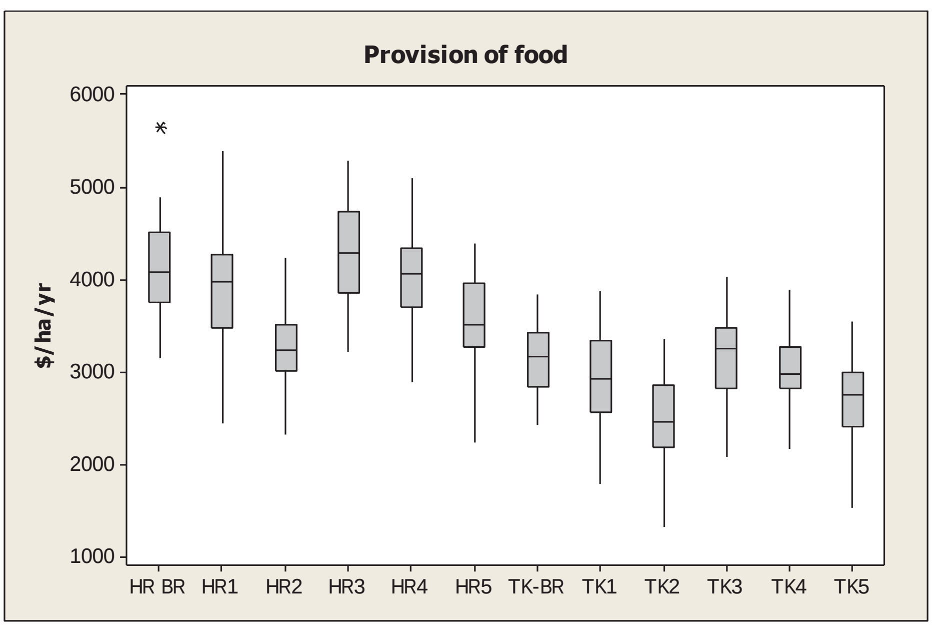

Figure 9-3: Value ($/ha/yr) of the provision of food for all scenarios.

The outlier[13] (*) represents a year (1985) for which pasture production was way above average.

9.2.2 Provision of food quality:

Neither of the soils considered in this study were deficient in trace elements, therefore they presented the same value for the provision of food quality, that is $38.5/ha/yr. This value was considered constant for different intensities and managements because the trace-element content of a soil is an inherent property which doesn't change readily with management. In practice, if a soil is farmed long enough it will eventually become deficient in some trace-elements.

9.2.3 Provision of support for human infrastructure:

This service depends on soil BD below 10 cm, which is an inherent property for a soil under dairy use, regardless of management. Therefore the value of the support for human infrastructure was considered to be constant at $17/ha/yr for HR, and $25/ha/yr for TK.

9.2.4 Provision of support to animals:

When stocking rates increased the number of days when the soils can support the animals without damage ('dry days') decreased slightly (Fig. 9.4) regardless of the soil type or the presence of a standoff pad. On average, HR provided 8 more 'dry days' a year than TK without a standoff pad, and 10 with a standoff pad. Although consistent with the notion that HR is a well drained soil, with higher macroporosity than TK, these differences are smaller than might have been expected.

This might be due to a lack of sensitivity of the functions of the SPASMO model linking macroporosity dynamics to drainage. The modelling of these functions would be improved, should more data become available.

---

[13] Box plot display consists of the following:
Outlier (*) - Observation that is beyond the upper or lower whisker.
Upper whisker - Extends to the maximum data point within 1.5 box heights from the top of the box.
Interquartile range box - Middle 50% of the data:
- Top line - Q3 (third quartile). 75% of the data are less than or equal to this value.
- Middle line - Q2 (median). 50% of the data are less than or equal to this value.
- Bottom line - Q1 (first quartile). 25% of the data are less than or equal to this value.

Lower whisker - Extends to the minimum data point within 1.5 box heights from the bottom of the box.

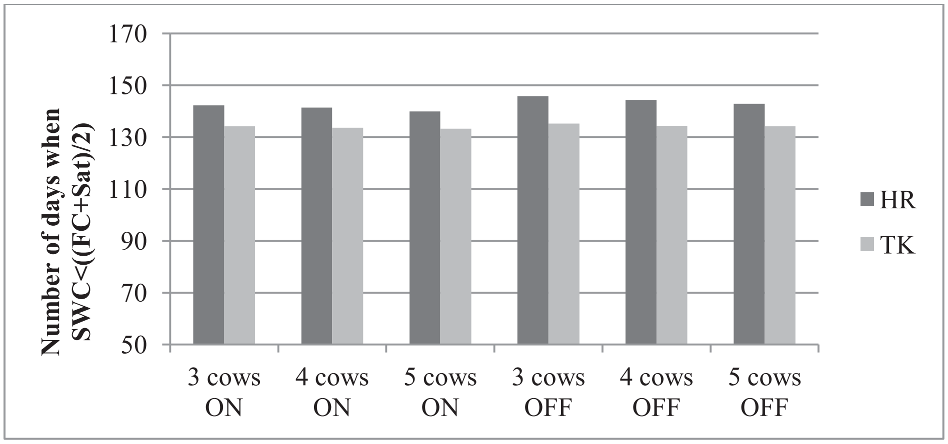

Figure 9-4: Average number of days between May and October when SWC<(FC+Sat)/2 over 35 years for all scenarios. HR: Horotiu Silt Loam; TK: Te Kowhai Silt Loam.

The value of the provision of support for farm animals increased (on averaged by 30%) with stocking rate, again regardless of the soil type or the presence of a standoff pad (Fig 9.5). Higher stocking rate means more animals to deal with. If the farmer had to put the cows on a standoff pad when SWC>(FC+Sat)/2 , it would be more expensive for 550 cows than for 300 cows because a bigger standoff pad would be needed. The value of the service per extra wet day therefore increased with stocking rate.

The presence of a standoff pad, by preserving Mp, and increasing the number of 'dry days', also increased slightly the value of the service, for both soil types.

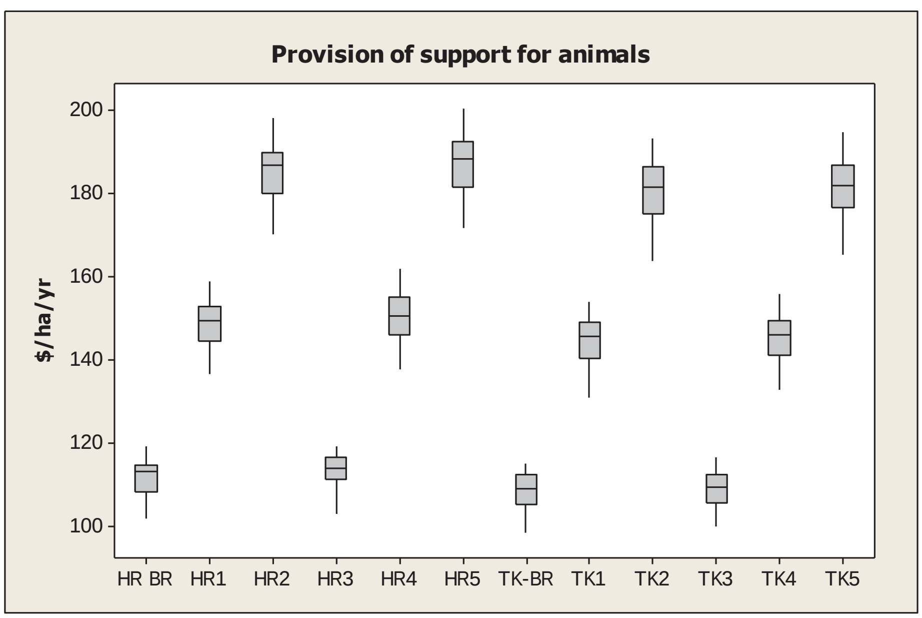

Figure 9-5: Value ($/ha/yr) of the provision of support to animals for all scenarios.

9.2.5 Provision of raw materials:

The provision of raw materials from soils wasn't considered relevant at the farm scale. Therefore a value of 0 was attributed to it by default.

9.2.6 Flood mitigation:

Flood mitigation is highly dependent on soil available water storage capacity and is thereby linked to runoff. Runoff increased with increasing stocking rate regardless of soil type or the presence of a standoff pad (Fig 9.6). This was due to the increased impact of cattle treading on macroporosity.

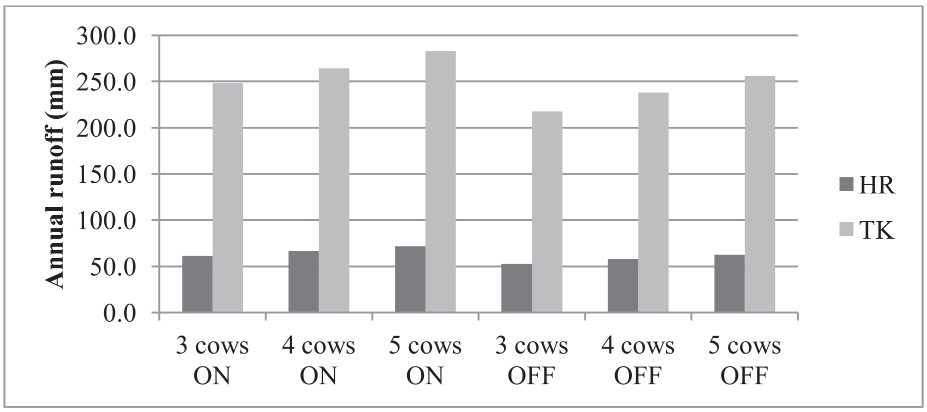

Figure 9-6: Average of modelled annual runoff (mm) over 35 years for all scenarios. HR: Horotiu Silt Loam; TK: Te Kowhai Silt Loam.

The measure of the service (Fig. 9.7), the water stored by the soil in seven days slightly decreased with increasing stocking rate, regardless of soil type or the presence of a standoff pad.

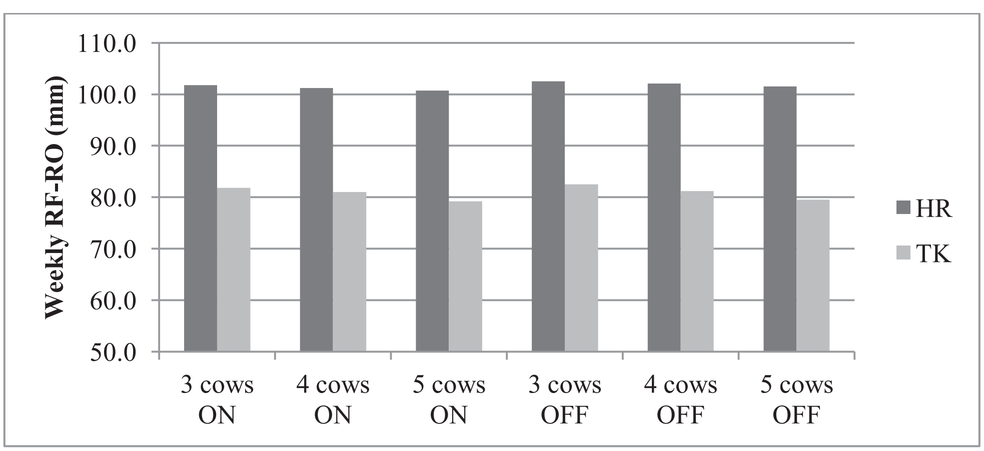

Figure 9-7: Average annual maximum of 7 days rainfall- runoff (mm/ha) for all scenarios over 35 years. HR: Horotiu Silt Loam; TK: Te Kowhai Silt Loam.

The different soil types had a marked effect on the measure of flood mitigation. However, the measure of the service was not increased by the presence of a standoff pad, which is contrary to what might have been expected. Again, this might be due to a lack of sensitivity of the functions in the SPASMO model linking macroporosity under treading to drainage and runoff. Extra-functionality was added to the model for this study to capture the effects of treading and macroporosity loss on runoff (Chapter Five). However, the method used (the soil conservation service curve number approach (Williams, 1991)) might not be sensitive enough. The modelling of the links between macroporosity and runoff would be improved, should more data become available.

Consequently, the value of flood mitigation (Fig. 9.8) was different between soil types, but there was no obvious impact of stocking rate or the use of a standoff pad.

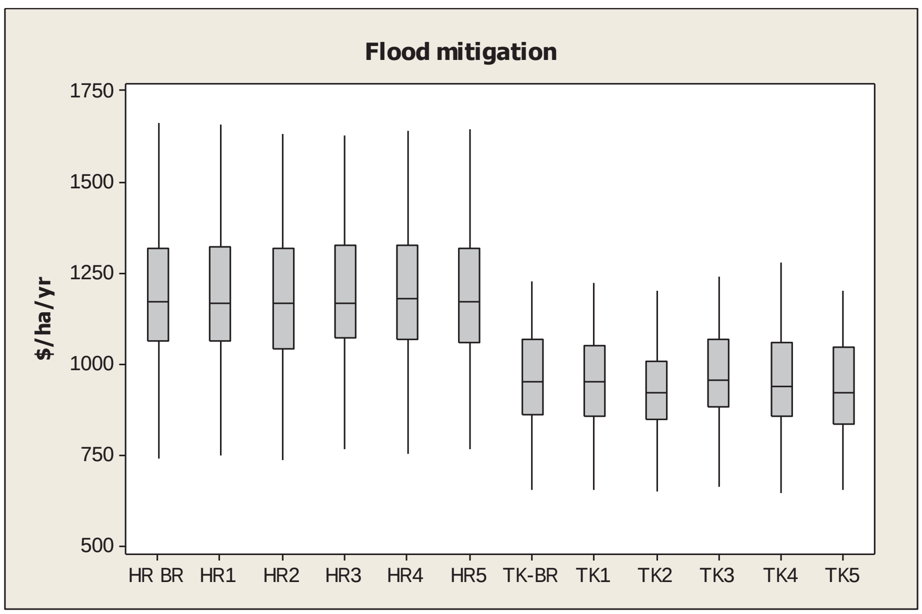

Figure 9-8: Value ($/ha/yr) of flood mitigation for all scenarios over 35 years.

9.2.7 Filtering of N:

The outputs of the SPASMO model for the potential maximum N loss (assessed by running the SPASMO model with very low ASC) are not shown here. Only the measure of the filtering of N (potential maximum modelled N loss - modelled N leaching loss) is displayed (Fig. 9.9).

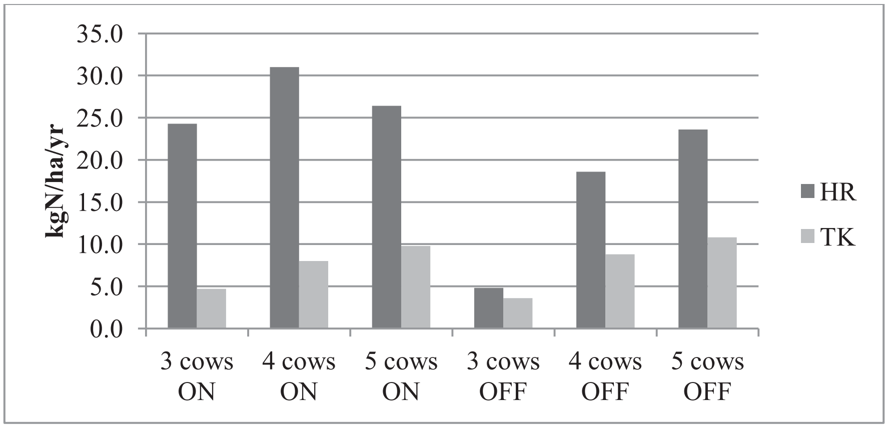

Figure 9-9: Average modelled N filtered (kgN/ha/yr) for all scenarios over 35 years. HR: Horotiu Silt Loam; TK: Te Kowhai Silt Loam.

The modelled annual N leached increased with increasing stocking rate, regardless of soil type or the presence of a standoff pad (Fig. 9.10). However, HR, a well drained soil, lost up to 3 times more N than TK. The presence of a standoff pad decreased N losses at all stocking rates, the animals returning less N to pastures. N losses were decreased by 11 to 28% for HR, and by 3 to 20% for TK.

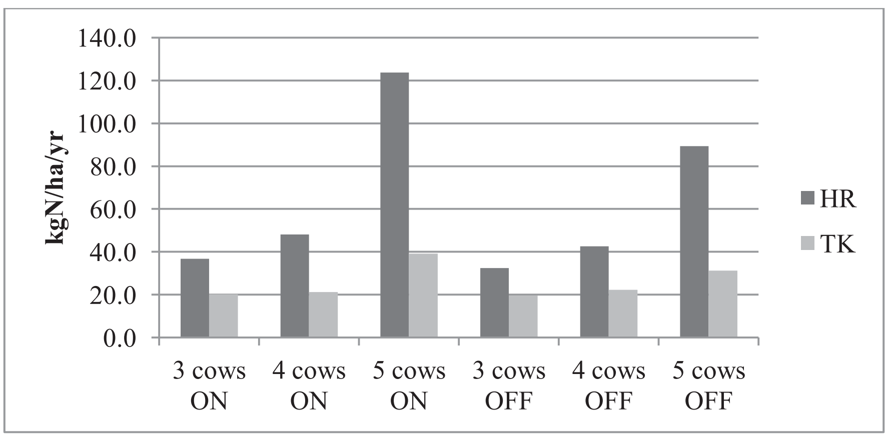

Figure 9-10: Average modelled N leached (kgN/ha/yr) for all scenarios over 35 years. HR: Horotiu Silt Loam; TK: Te Kowhai Silt Loam.

The measure of the service, the amount of N filtered by the soil, was greater for HR than TK (Fig. 9.9) regardless of stocking rates or the presence of a standoff pad.

To measure the service the scenarios were run by simulating very low ASC which lead to more N losses but also less pasture grown. The way the service was measured (Max N loss – Actual N loss) meant that for some years the service was nil because the maximum N losses modelled were similar to actual ones or even lower. This was the case for the following scenarios: HR2, HR3, HR5, TK2, TK3, TK4 and TK5.

For HR, the use of a standoff pad decreased the amount of N filtered by the soil, which was logical, a lot less N being returned to the paddock when the cows are spending time on a pad. It seemed like the filtering capacity of the HR soil wasn't saturated even with the cows on the paddock all year round.

However, for TK, for high stocking rates (4 and 5 cows/ha), the use of a standoff pad increased the amount of N filtered by the soil, even with less N deposited on the soil by the animals. This might be due to increased Mp and increased access to ASC.

The average value of the filtering of N slightly increased with stocking rate for scenarios with a standoff pad. For scenarios without a standoff pad, at 5 cows/ha, both soils seem to filter N less efficiently (data are more spread over 35 years), which is coherent with degraded soil conditions (Fig. 9.11). The use of a standoff pad decreased the average value of the filtering of N for HR, but increased it for TK at high stocking rates (4 and 5 cows/ha).

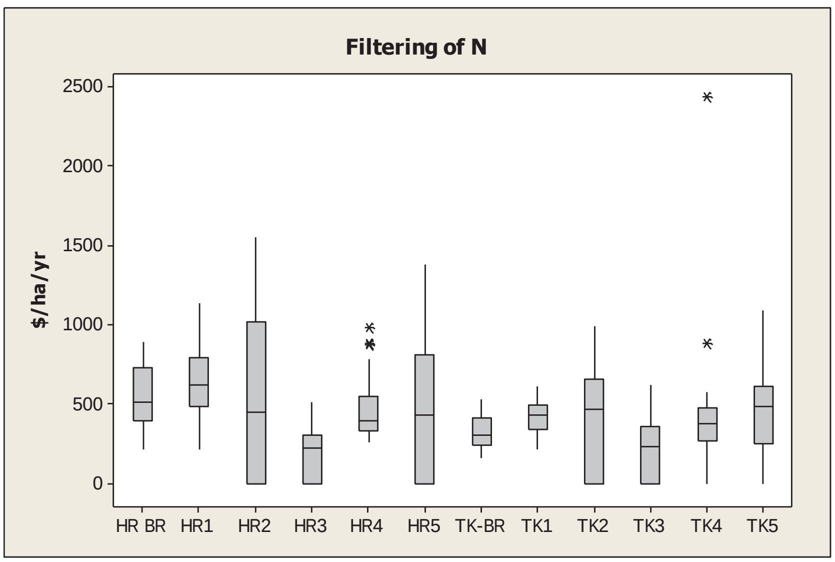

Figure 9-11: Value ($/ha/yr) of filtering of N for all scenarios over 35 years.

Outliers (*) represent observations that are beyond the upper or lower whisker, that is over or below 1.5 box heights from the top or bottom of the box. For the filtering of N they represent years for which the maximum potential N losses were much greater than the modelled N losses, leading to very high amounts of N being filtered by the soil.

9.2.8 Filtering of P:

The measures of the service, the P filtered by the soil, were very high averaging 70 kg P/ha/yr for HR and 5 kg P/ha/yr for TK (Fig. 9.12). P filtered decreased with increasing stocking rate, which is consistent with the increase in P losses.

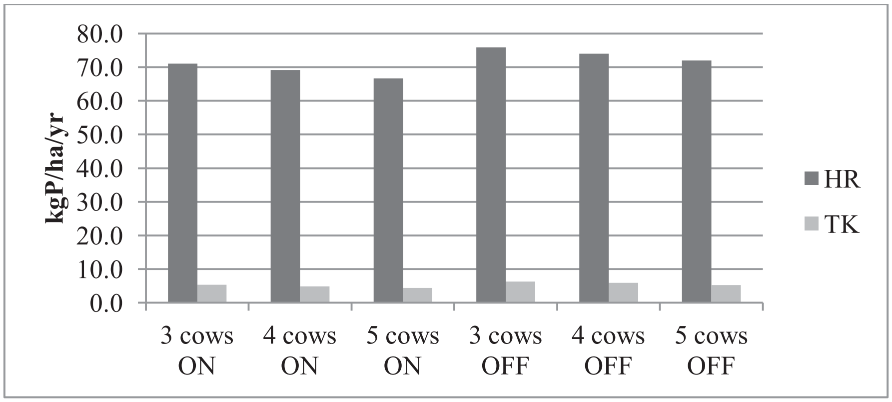

Figure 9-12: Average modelled annual P filtered (kg P/ha/yr) for all scenarios over 35 years. HR: Horotiu Silt Loam; TK: Te Kowhai Silt Loam.

The modelled annual P runoff increased with increasing stocking rate regardless of soil type or the presence of a standoff pad (Fig. 9.13). This can be explained by the fact that P runoff depends on soil surface integrity and the level of disturbance, both influenced by cattle hooves. TK lost more P than HR for all scenarios. Allophanic Soils like HR are known to adsorb P very strongly.

The use of a standoff pad decreased P losses by up to 50% for HR and 30% for TK.

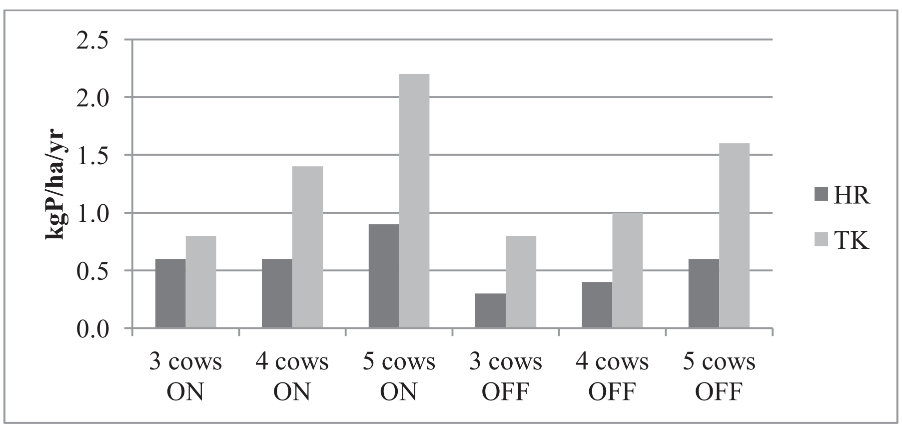

Figure 9-13: Average modelled annual P runoff (kg P/ha/yr) for all scenarios over 35 years. HR: Horotiu Silt Loam; TK: Te Kowhai Silt Loam.

Since no technologies exist to enable farmers to mitigate P losses over 5 kg P/ha/yr, whenever the measure of the service was equal to or above 5 kg P/ha/yr, the value of the service was limited to the costs of this quantity of P, determined using mitigation functions specific to each soil type. For this reason, the value of the filtering of P was equal to $2,924/ha/yr for all scenarios for HR, and limited to $1,392/ha/yr for TK (Fig. 9.14).

The average value over 35 years of the filtering of P for TK decreased with increasing stocking rate because of decreasing amount of P filtered (Fig. 9.14).

The use of a standoff pad increased the value of the filtering of P for TK by 12% (from TK2 to TK5).

It was impossible in this study to capture the great value of the filtering of P from HR. The value used, \$2,924/ha/yr, does not reflect the “true value” of the service, since there are no mitigation strategies able to cope with P losses above 5 kg P/ha/yr, the costs of which could be used to value the service.

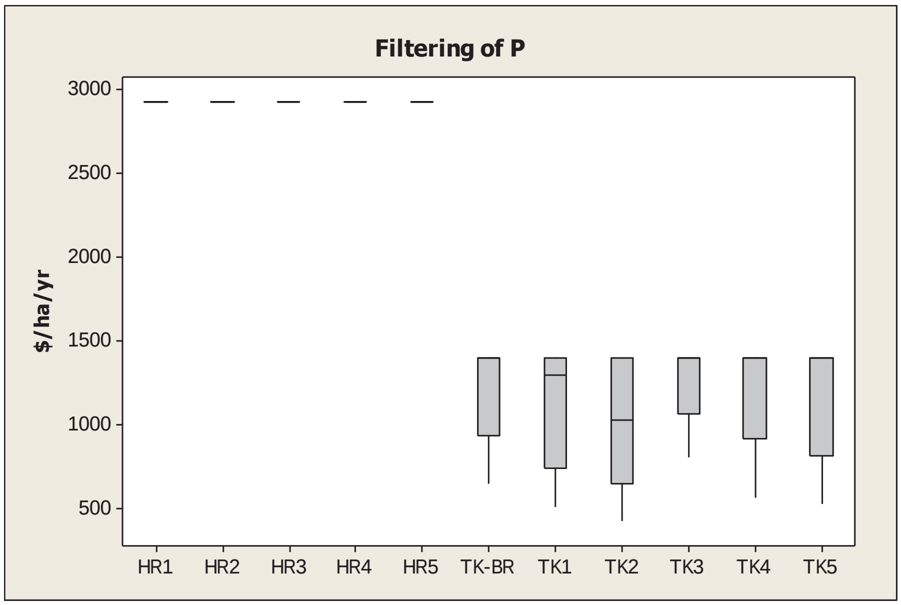

Figure 9-14: Value (\$/ha/yr) of the filtering of P for all scenarios over 35 years. For the Horotiu silt loam the only value represented is \$2,924/ha/yr which is the cost of mitigating P losses of 5 kg P/ha/yr.

9.2.9 Filtering of contaminants:

For each year the amount of runoff generated within 5 days of a grazing event, and thereby potentially contaminated by fresh dung pad was used to investigate the filtering of contaminants (Fig. 9.15). Generally, contaminated runoff increased with increasing stocking rate which is consistent with increased total runoff (Fig 9.6). TK, being a poorly drained soil, produced much more runoff than HR. The use of a standoff pad, by increasing soil Mp, decreased the amount of contaminated runoff produced for both soils.

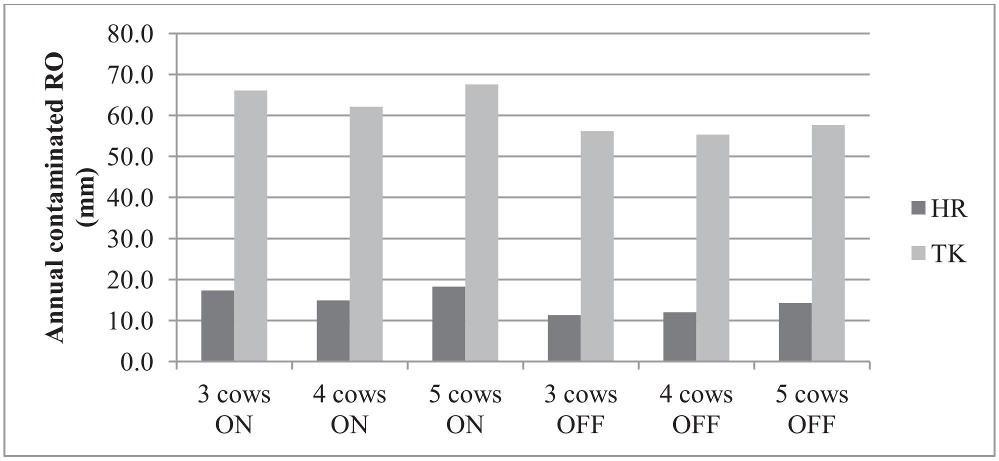

Figure 9-15: Average annual contaminated runoff (mm) 5 days after a grazing event for all scenarios over 35 years. HR: Horotiu Silt Loam; TK: Te Kowhai Silt Loam.

The measure of the service was then defined as the difference between the rain falling within 5 days of a grazing event, that is the amount of water that could potentially be contaminated, and the actual runoff generated during this period (5 days RF-RO) modelled with SPASMO. For HR, around 5% of the rain falling ran off and got contaminated, whereas it was around 20% for TK (Fig 9.16). The amount of contaminated runoff increased with increasing stocking rate, whereas the use of a standoff pad decreased it for both soils.

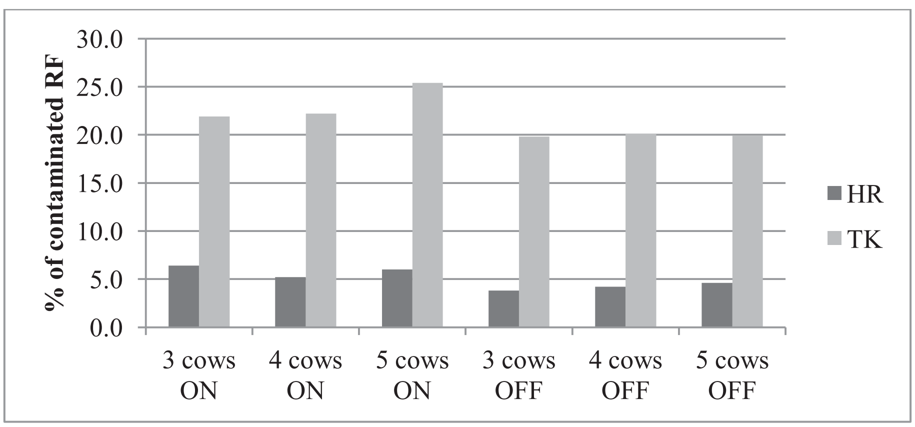

Figure 9-16: Average percentage of annual contaminated rainfall 5 days after a grazing event for all scenarios over 35 years. HR: Horotiu Silt Loam; TK: Te Kowhai Silt Loam.

To value the filtering of contaminants, the costs of using a constructed wetland to treat the water currently treated by the soil (5 days RF-RO) was considered. The annual maximum of 5 days RF-RO was considered to dimension the wetland that would be needed. This measure was very dependent on rain patterns; therefore it was difficult to find noticeable trends in the value of the service (Fig 9.17). The average value seemed to generally decrease with increasing stocking rate, especially for the HR with a pad, and the TK without a pad. Overall, the value of the service was greater for HR (around \$6,000/ha/yr) than TK (around \$5,000/ha/yr).

The use of a pad means that the animals are off the pasture when the soil is too wet. Less dung is deposited on the pasture when wet, a higher proportion is deposited on dryer soil, less inclined to generate runoff in the event of rainfall. However these facts do not show much in the value of the service calculated.

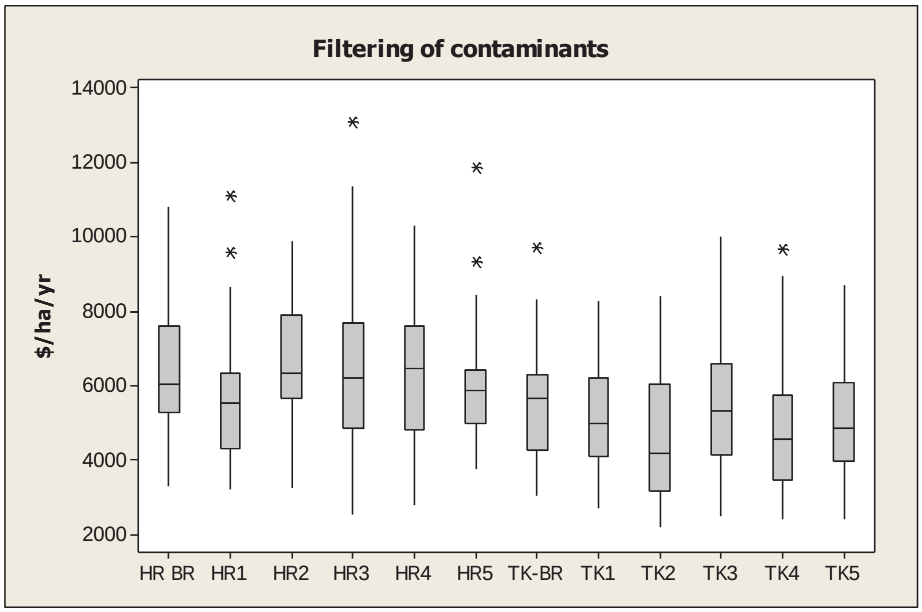

Figure 9-17: Value (\$/ha/yr) of filtering of contaminants for all scenarios over 35 years.

Outliers (*) correspond to years when a great amount of rain fell within 5 days of a grazing event, but little runoff was generated.

9.2.10 Recycling of wastes

Soil conditions (SWC) were investigated to determine if they were optimal for dung decomposition and recycling. The use of a standoff pad means that less dung is deposited on the pasture when the soil is wet. For this reason, more dung is potentially well decomposed when a pad is used (Fig 9.18).

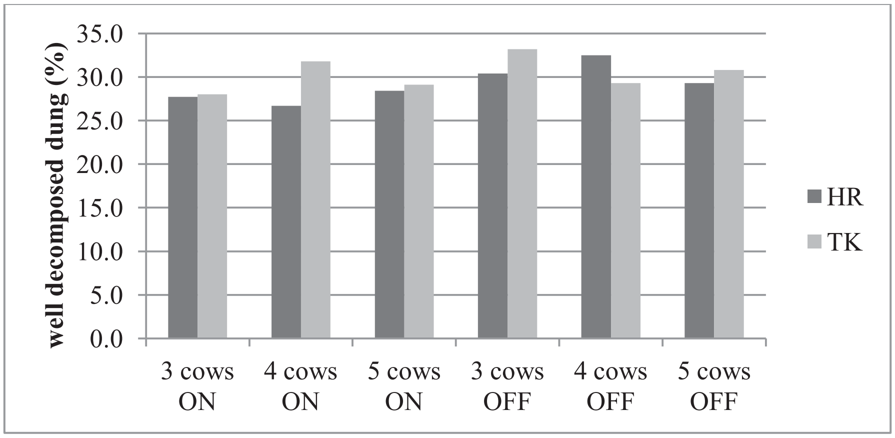

Figure 9-18: Average percentage of dung potentially well decomposed for all scenarios over 35 years. HR: Horotiu Silt Loam; TK: Te Kowhai Silt Loam.

The costs of using an effluent treatment pond to decompose the wastes currently treated by the soil were used as a proxy for the value of the recycling of wastes. The value of the recycling of wastes increased with increasing stocking rate because more cows make more waste for the soil to treat (Fig. 9.19). The soil types studied seemed to behave the same way. The presence of a standoff pad generally slightly improved the value of the recycling of waste.

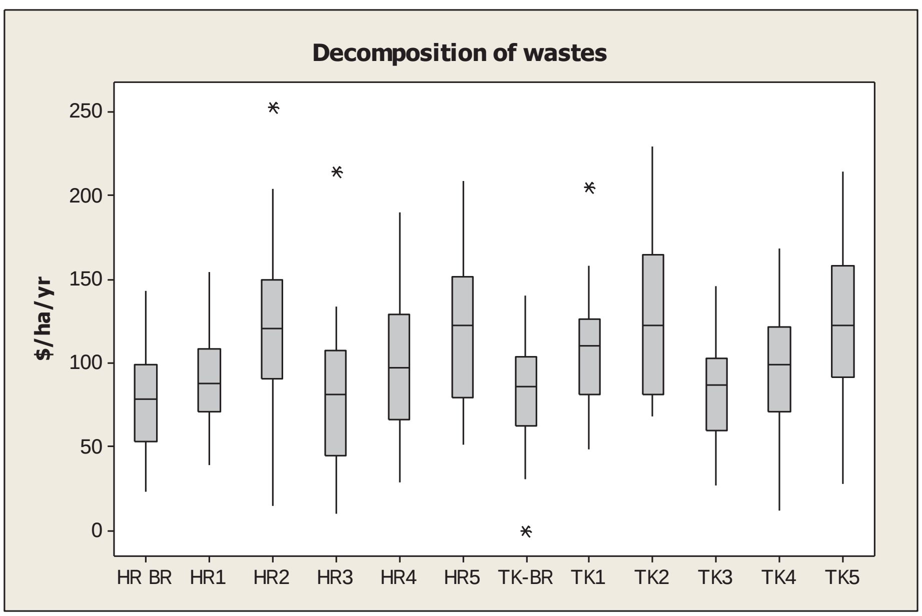

Figure 9-19: Value ($/ha/yr) of the recycling of wastes for all scenarios over 35 years.

High outliers (*) correspond to years when great amounts of dung were deposited in ideal conditions for decomposition (up to 55% of dung deposited), leading to a greater value for the service. The low outlier (*) corresponds to a year (1996) when all dung was deposited in restricting conditions, either too dry or too wet.

9.2.11 Carbon flows:

Average net C flows over 35 years, modelled with SPASMO, showed that HR tended to lose C, whereas TK was at quasi-steady state (Fig. 9.20). These values were very sensitive to modelling parameters therefore field data should be used to measure this service.

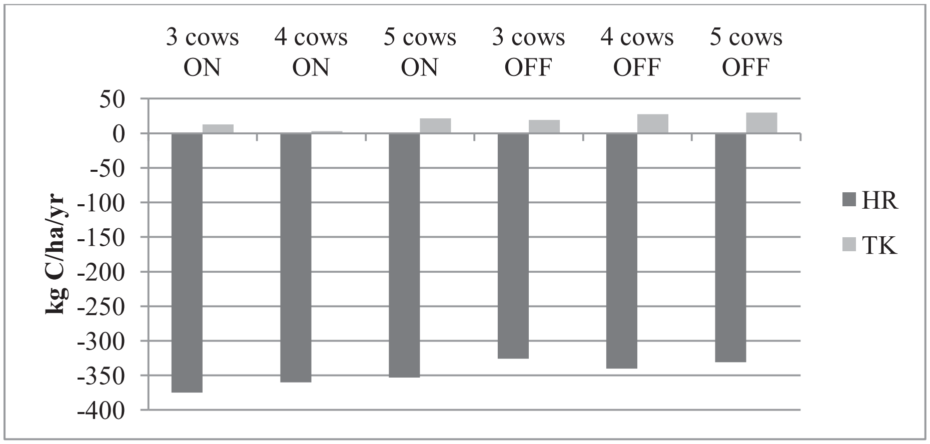

Figure 9-20: Average net C flows (kg C/ha/yr) for all scenarios over 35 years. HR: Horotiu Silt Loam; TK: Te Kowhai Silt Loam.

For both soils the use of a standoff pad slightly increased the net C stored (Fig. 9.20). Increasing stocking rate generally slightly increased the net C stored probably thanks to the additional C returned to pasture from animal dung.

Net C flows were valued using C market price. Since HR was losing C, the value of the service provided was therefore negative, which means that the soil is losing value. This confirms that C loss is a degradation process depleting soil natural capital and decreasing its value.
TK was at steady state, therefore the value of the service was quite small around $2/ha/yr (Fig. 9.21).

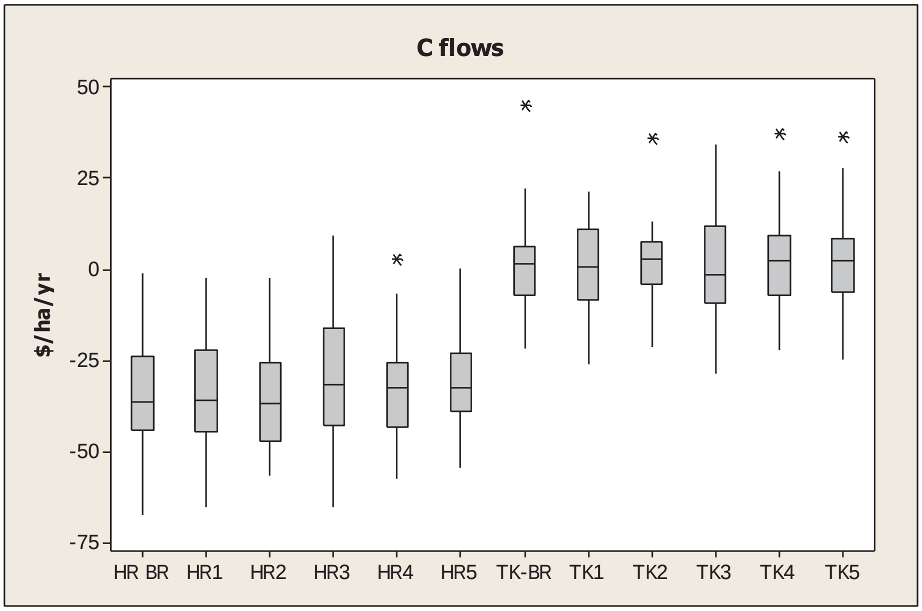

Figure 9-21: Value ($/ha/yr) of net C flows for all scenarios over 35 years.

Outliers (*) correspond to years when the net flow of C was greatly positive, leading to accumulation of C that year.

9.2.12 Nitrous oxide regulation:

Nitrous oxide emissions were calculated using an IPCC inspired method. Nitrous oxide emissions were higher for HR than TK. This can be explained by the fact that HR leached more nitrates which meant more indirect N_2O emissions. Moreover, HR also grows pastures with better N fixation; when grazed this leads to richer urine returned to the soil and more direct N_2O emissions (Fig. 9.22).

Modelled N_2O emissions increased with increasing stocking rate because more cows deposited more waste on pastures. The use of a standoff pad decreased N_2O emissions by 10 to 14% for HR and 7 to 18% for TK. This was mainly due to less waste being deposited on pastures.

The average amount of N_2O mitigated over 35 years increased with increasing stocking rate for both soils. The use of a standoff pad slightly increased the average amount of N_2O mitigated over 35 years for HR, and slightly decreased it for TK, but these trends were not very marked at the stocking rate studied. N_2O mitigated was very similar between the two soils (Fig. 9.22).

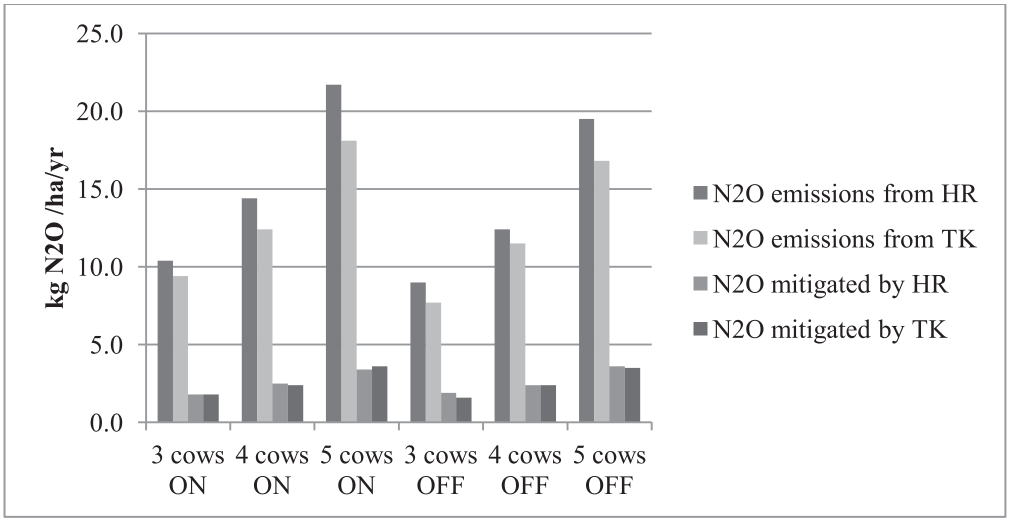

Figure 9-22: Average N_2O emissions and N_2O mitigated (kg N_2O/ha/yr) for all scenarios over 35 years. HR: Horotiu Silt Loam; TK: Te Kowhai Silt Loam.

Nitrous oxide regulation was valued using CO_2 equivalents and C market price. The value of nitrous oxide regulation increased with increasing stocking rate because more animals return more N to pastures (Fig. 9.23). The value of nitrous oxide regulation was very similar between the two soils and with and without a pad.

This can suggest that at the stocking rates studied both soils had the capacity to deal with N inputs. It would be interesting to look at even more intensive systems.

Outliers (*) correspond to either dry (upper outliers) or wet (lower outliers) years when modelled N_2O emissions were either very different or very close to the potential maximal N_2O emissions for a saturated soil (Fig. 9.23).

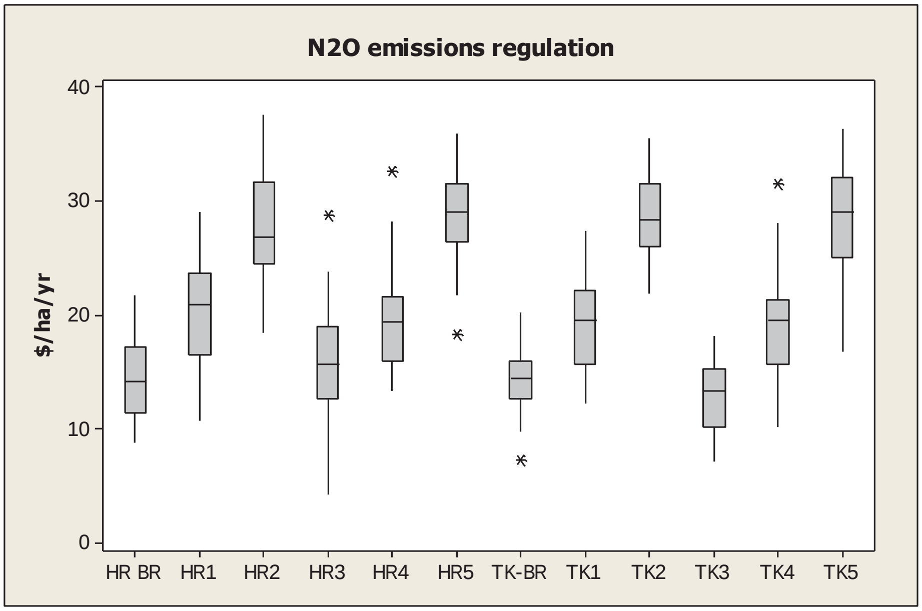

Figure 9-23: Value ($/ha/yr) of N_2O emissions regulation for all scenarios over 35 years.

9.2.13 Methane oxidation:

Methane oxidation was modelled from SWC. On average over 35 years, HR oxidised 0.14 kg CH_4/ha/yr more than TK for all scenarios (Fig. 9.24). CH_4 oxidation was constant for HR with or without a pad and at all stocking rates. For TK, CH_4 oxidation slightly increased with increasing stocking rate and slightly decreased with the use of a standoff pad (Fig. 9.24). However, the amounts of CH_4 concerned were so small that they might be considered insignificant even at the farm scale.

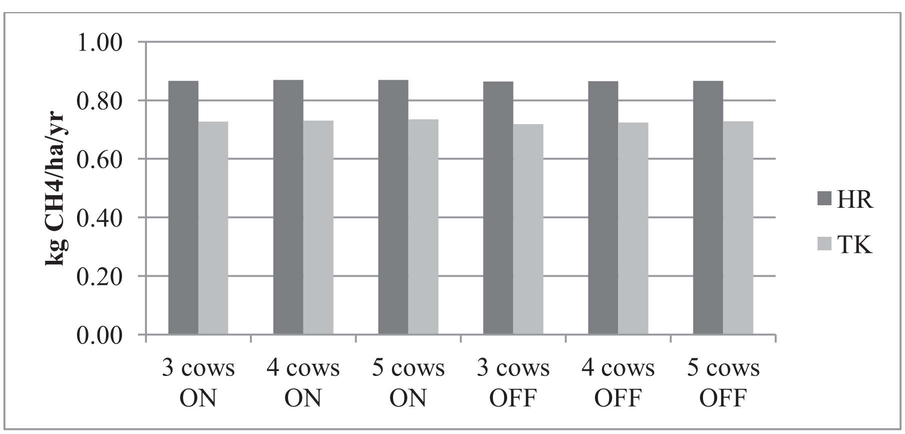

Figure 9-24: Average CH_4 oxidation (kg CH_4/ha/yr) for all scenarios over 35 years. HR: Horotiu Silt Loam; TK: Te Kowhai Silt Loam.

Methane oxidation was valued using CO_2 equivalents and C market price. The value of methane oxidation was stable at $0.47/ha/yr for HR and $0.39/ha/yr for TK across stocking rate and with or without pad (Fig. 9.25).

It would be interesting to look at different soil types to check if CH_4 oxidation is more reactive to management. It would also be better to use field data as CH_4 oxidation can vary greatly within a landscape.

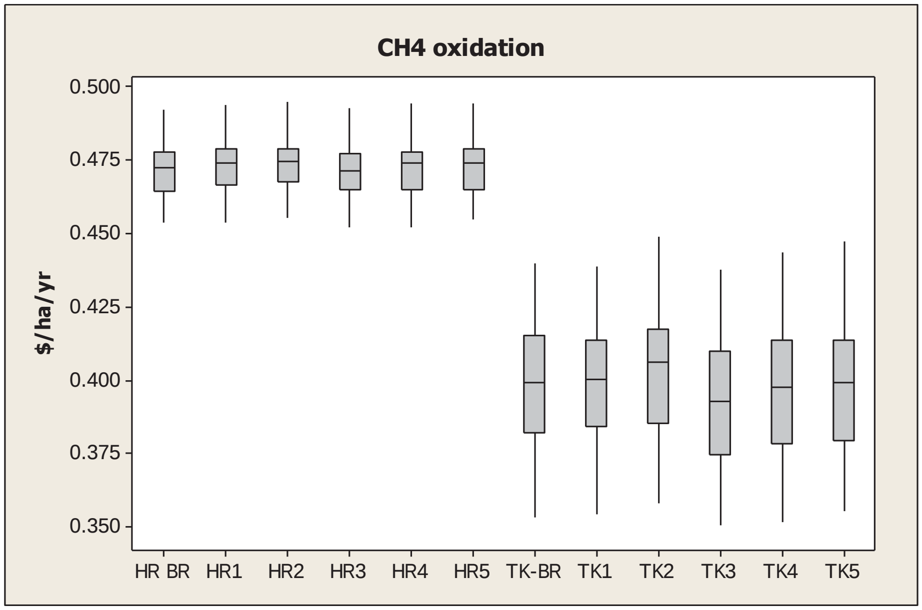

Figure 9-25: Value ($/ha/yr) of CH_4 oxidation for all scenarios over 35 years.

9.2.14 Regulation of pest and disease populations:

The number of favourable days to pasture pests' development was measured between October and December (92 days) using SPASMO outputs for SWC and Mp. This measure was used as a proxy for the regulation of pest and disease populations from soils. HR was generally more favourable for pest development than TK. The number of favourable days decreased with increasing stocking rate, which was especially marked for HR. This was due to decreased Mp between May and March due to increased cattle treading (Fig. 9.26). TK's macroporosity was below 9 from the start. For this reason, TK was less sensitive to changes in stocking rates. The use of a standoff pad increased soil Mp and thereby the number of favourable days for pests at all stocking rates.

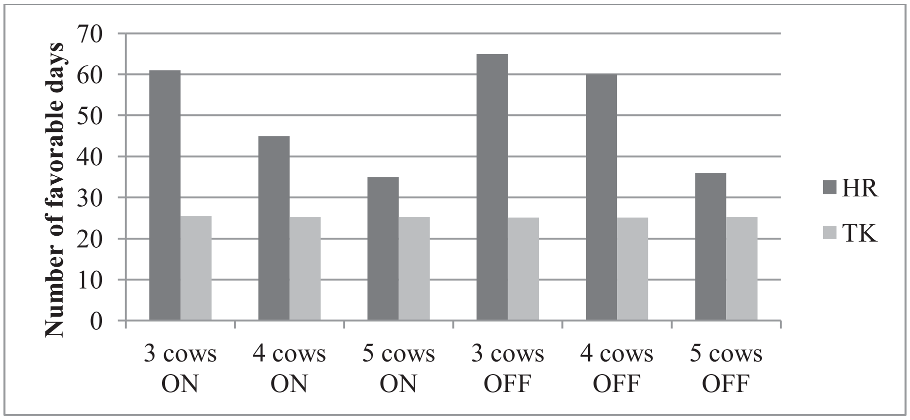

Figure 9-26: Average number of favourable days to pasture pest development between October and December for all scenarios over 35 years. HR: Horotiu Silt Loam; TK: Te Kowhai Silt Loam.

The regulation of pest and disease populations was valued using the cost of pesticides that would be needed if the soil didn't limit pest development. The data on the regulation of pests taking only 3 values, box plots weren't informative, hence intervals of confidence were used instead (Fig. 9.27).

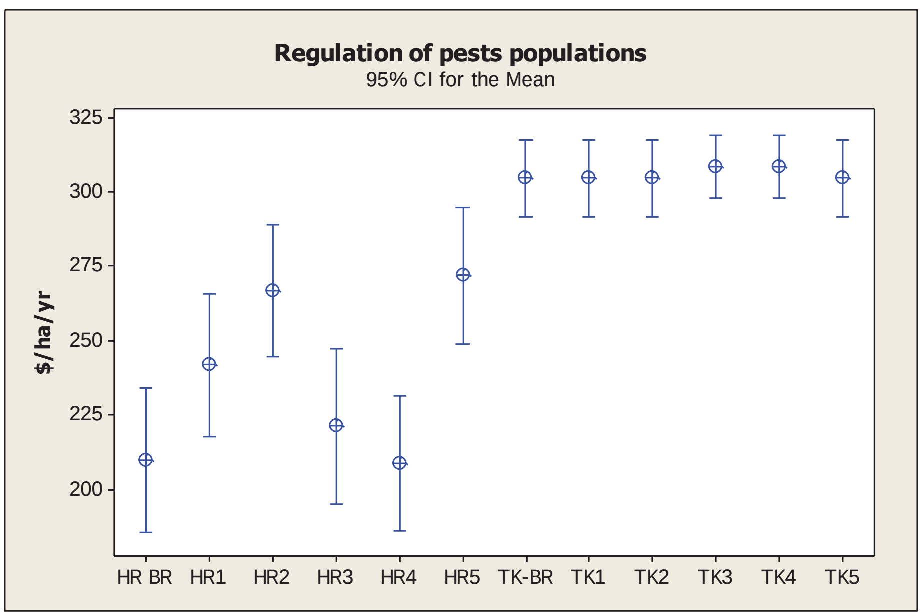

Figure 9-27: Value ($/ha/yr) of the regulation of pest and disease populations for all scenarios over 35 years. 95% CI: 95% confidence intervals.

The value of the regulation of pest and disease populations generally increased with stocking rate as the number of favourable days decreased. TK, which was already quite compacted, provided better regulation of pest populations than HR. The use of a standoff pad didn't significantly affect the average value of the service except for HR with 4 cows/ha for which the value of the service decreased by 14% (Fig. 9.27).

9.3 Overview of the influence of farm management on the provision of soil services:

The average value over 35 years of each soil service for each scenario is presented in Table 9.2 and Fig 9.28. The total value of soil services was significantly greater for HR than for TK for all scenarios (Fig 9.28 and Fig. 9.29).

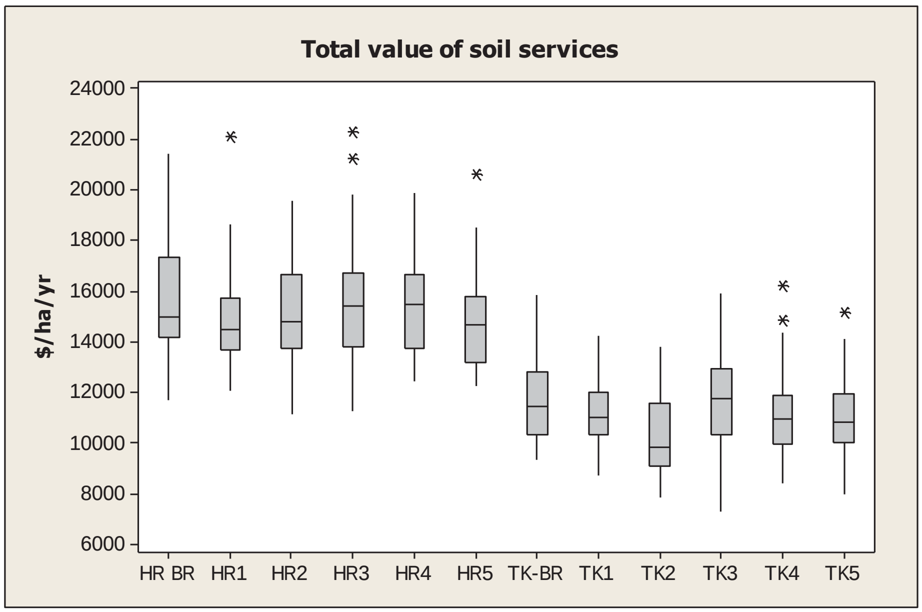

Figure 9-28: Total value ($/ha/yr) of soil services for the twelve scenarios studied over 35 years.

Outliers (*) represent years for which the total value of the services provided by the soil was much greater than the rest of the data.

Table 9-2: Average value ($/ha/yr) over 35 years of each soil service for each scenario. HR: Horotiu Silt Loam; TK: Te Kowhai Silt Loam.

	HR BR	**HR1**	**HR2**	**HR3**	**HR4**	**HR5**	**TK-BR**	**TK1**	**TK2**	**TK3**	**TK4**	**TK5**
	3 cows ON	4 cows ON	5 cows ON	3 cows OFF	4 cows OFF	5 cows OFF	3 cows ON	4 cows ON	5 cows ON	3 cows OFF	4 cows OFF	5 cows OFF
Provision of food Quantity	4155	3903	3265	4279	4070	3564	3129	2939	2507	3166	3076	2688
Provision of food Quality	38.3	38.3	38.3	38.3	38.3	38.3	38.3	38.3	38.3	38.3	38.3	38.3
Provision of support for human infrastructure	17	17	17	17	17	17	25	25	25	25	25	25
Provision of support for farm animals	112	149	185	113	150	187	108	144	180	109	145	181
Flood mitigation	1196	1189	1183	1204	1200	1192	960	952	931	969	954	934
Filtering of N	554	655	526	175	470	479	328	419	410	223	432	452
Filtering of P	2924	2924	2924	2924	2924	2924	1188	1114	1016	1252	1195	1141
Filtering of contaminants	6513	5700	6582	6635	6348	6079	5506	5222	4623	5413	4966	5207
Recycling of wastes	78	91	124	81	99	119	82	107	129	84	94	125
Carbon flows	-35.8	-34.4	-33.7	-31.0	-32.4	-31.6	1.2	0.3	2.1	1.8	2.6	2.8
N_2O regulation	15	20	28	15	20	29	14	19	29	13	19	28
CH_4 oxidation	0.473	0.473	0.474	0.472	0.472	0.473	0.397	0.399	0.402	0.392	0.395	0.398
Regulation of pest and disease populations	210	242	267	221	209	272	305	305	305	309	309	305
Total ($/ha/yr)	15777	14895	15105	15673	15512	14868	11685	11284	10195	11602	11255	11129

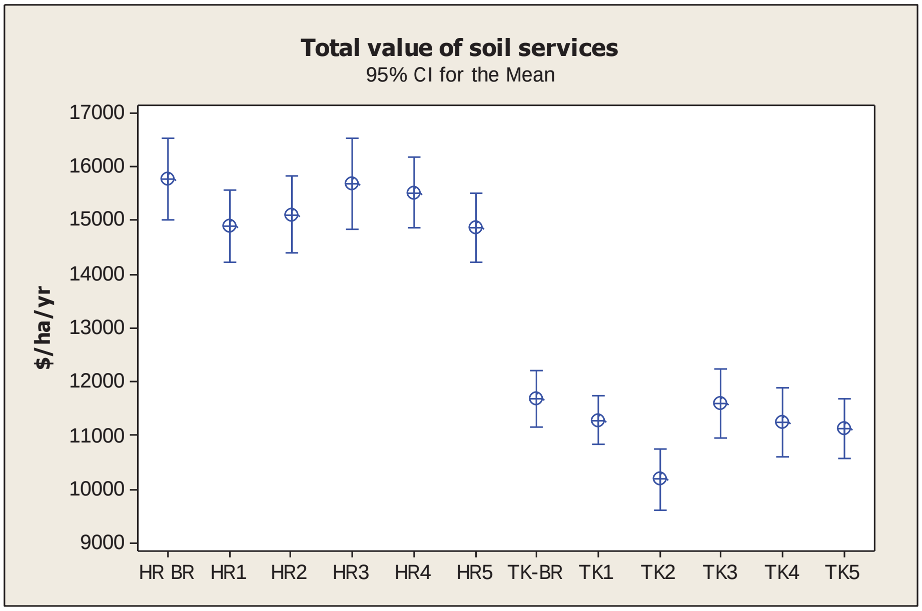

Figure 9-29: Total value ($/ha/yr) of soil services for all scenarios (P<0.05).

For all scenarios, regulating services were more valuable than provisioning services, accounting for 70 to 77% of the total value of soil services (Table 9.3).

Table 9-3: Average value of soil services ($/ha/yr) over 35 years for all scenarios.

Scenario	Provisioning services	Regulating services	Total ($/ha/yr)
HR BR	4,322	11,455	15,777
HR1	4,107	10,787	14,895
HR2	3,505	11,600	15,105
HR3	4,447	11,225	15,673
HR4	4,275	11,237	15,512
HR5	3,806	11,063	14,868
TK-BR	3,301	8,385	11,685
TK1	3,146	8,138	11,284
TK2	2,750	7,445	10,195
TK3	3,338	8,264	11,602
TK4	3,284	7,972	11,255
TK5	2,932	8,197	11,129

The services having the most value were the filtering of contaminants (43.6% of total on average across all scenarios), the provision of food quantity (25.7% of total on average across all scenarios), the filtering of P (14.7% of total on average across all scenarios), flood mitigation (8.2% of total on average across all scenarios) and the filtering of N (3.32% of total on average across all scenarios) (Table 9.4).

Table 9-4: Contribution (in %) of each service to the total value of soil services for all scenarios.

	Average for all scenarios	**HR-BR** 3 cows ON	**HR1** 4 cows ON	**HR2** 5 cows ON	**HR3** 3 cows OFF	**HR4** 4 cows OFF	**HR5** 5 cows OFF	**TK-BR** 3 cows ON	**TK1** 4 cows ON	**TK2** 5 cows ON	**TK3** 3 cows OFF	**TK4** 4 cows OFF	**TK5** 5 cows OFF
Provision of food Quantity	25.7%	26.3%	26.2%	21.6%	27.3%	26.2%	24.0%	26.8%	26.0%	24.6%	27.3%	27.3%	24.2%
Provision of food Quality	0.3%	0.2%	0.3%	0.3%	0.2%	0.2%	0.3%	0.3%	0.3%	0.4%	0.3%	0.3%	0.3%
Provision of support for human infrastructure	0.2%	0.1%	0.1%	0.1%	0.1%	0.1%	0.1%	0.2%	0.2%	0.2%	0.2%	0.2%	0.2%
Provision of support for farm animals	1.1%	0.7%	1.0%	1.2%	0.7%	1.0%	1.3%	0.9%	1.3%	1.8%	0.9%	1.3%	1.6%
Flood mitigation	8.2%	7.6%	8.0%	7.8%	7.7%	7.7%	8.0%	8.2%	8.4%	9.1%	8.3%	8.5%	8.4%
Filtering of N	3.3%	3.5%	4.4%	3.5%	1.1%	3.0%	3.2%	2.8%	3.7%	4.0%	1.9%	3.8%	4.1%
Filtering of P	14.7%	18.5%	19.6%	19.4%	18.7%	18.8%	19.7%	10.2%	9.9%	10.0%	10.8%	10.6%	10.3%
Filtering of contaminants	43.6%	41.3%	38.3%	43.6%	42.3%	40.9%	40.9%	47.1%	46.3%	45.3%	46.7%	44.1%	46.8%
Recycling of wastes	0.8%	0.5%	0.6%	0.8%	0.5%	0.6%	0.8%	0.7%	0.9%	1.3%	0.7%	0.8%	1.1%
Carbon flows	-0.1%	-0.2%	-0.2%	-0.2%	-0.2%	-0.2%	-0.2%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%
N_2O regulation	0.2%	0.1%	0.1%	0.2%	0.1%	0.1%	0.2%	0.1%	0.2%	0.3%	0.1%	0.2%	0.3%
CH_4 oxidation	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%
Regulation of pest and disease populations	2.1%	1.3%	1.6%	1.8%	1.4%	1.3%	1.8%	2.6%	2.7%	3.0%	2.7%	2.7%	2.7%

The average total value of soil services generally decreased with increasing stocking rates for both soil types and both management practices (Fig. 9.28). This trend was the most marked on the TK without a pad, the total value of soil services provided by TK2 being significantly ($P<0.05$) different from TK-BR and TK1 (Fig. 9.29). In making that statement, it is worth noticing that the value of the service did not differ markedly across the range of management systems for either HR ($14,868 to $15,777/ha/yr) or TK ($10,195 to $11,685). In both soils, the highest value was under the base case scenario. The lowest value was obtained for TK with no standoff pad and 5 cows/ha.

The increase in stocking rate had the most effect on the values of the provision of food, provision of support to animals, the recycling of wastes, N_2O regulation for both soils, as well as the regulation of pest populations for HR (Table 9.2). These reflect the impacts of stocking rates on soil structure and nutrient inputs to soils, as physical structure and nutrient concentrations are the natural capital stocks at the origin of the provision of these services under a dairy use.

Surprisingly, the increase in stocking rate didn't have much effect on the value of the services regulated by soil water dynamics (filtering services and flood mitigation). This might be due to a lack of sensitivity of the functions of the SPASMO model linking macroporosity dynamics to drainage and runoff. Even if extra-functionality was added to the model to capture these effects (Chapter Five), it would be necessary to try to improve this part of the model when more detailed data becomes available. The quantification of the filtering and flood mitigation services might also be inappropriate to capture the dynamics of their provision.

The influence of the use of a standoff pad had the most effect on the values of the provision of food from HR, the filtering of nutrients and contaminants from both soils, and the regulation of pest and disease populations from HR (Table 9.2).

The influence of the use of a standoff pad had on the overall value of the services was only significant ($P<0.05$) (Fig. 9.29) for one scenario TK2 (Te Kowhai soil with 5 cows/ha and no pad) (Fig 9.28 and Fig. 9.29).

For HR, the decrease in value of the services due to the use of a pad (Fig. 9.30) for scenarios with 3 and 5 cows/ha (HR3 and HR5) were due to the fact that when the cows are on the pad, less dung and urine is deposited on the paddock therefore the soils have less nutrients and contaminants to deal with, reducing the necessity for the filtering services. The gain in value for TK with 5 cows/ha with a pad on (Fig. 9.30) is due to increased pasture yield as well as better filtering of N, P and contaminants, which means that for this soil at 5 cows/ha, the presence of a pad improves soil properties greatly.

The difference in the value of soil services between TK2 and TK5 was on average \$934/ha/yr (Fig. 9.30). The costs of a standoff pad for 500 cows are around \$215/ha/yr (annualised construction costs + maintenance costs). Therefore, investing in a pad increases ecosystem services from a TK soil by 934-215 = \$719/ha/yr even after the costs of a pad are deducted.

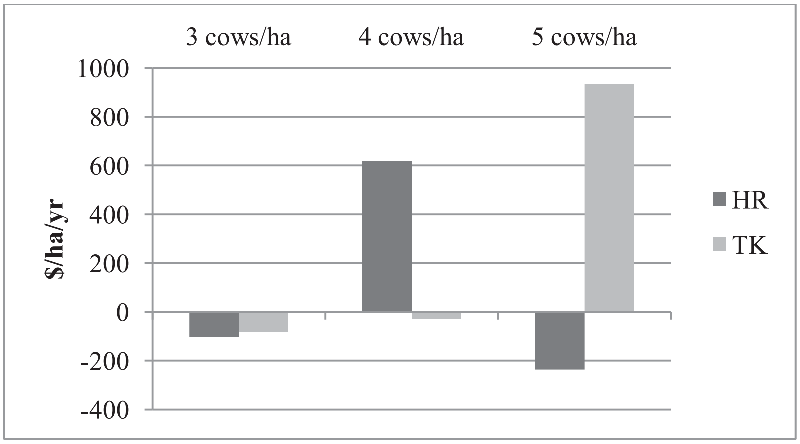

Figure 9-30: Difference in value (\$/ha/yr) between scenarios with and without a standoff pad. HR: Horotiu Silt Loam; TK: Te Kowhai Silt Loam.

The way the grazing rotation was modelled might also be responsible for the lack of great distinction between management practices. The grazing rotation used was based on the amount of grass available on the paddock. The cows returned to a paddock only if the pasture cover was >2000 kg DM/ha. The cows stayed on a paddock until grazed down to 1500 kg DM/ha or until they have consumed what they need (around 20 kg DM/day). The paddocks not grazed were locked and cut for pasture silage when they reached 3000 kg DM/ha. Such grazing rotation may mask the differences between soils in their response to management practices because it meant that the cows only came on the paddock when the pasture cover was sufficient. For a soil growing pasture slowly, it means fewer grazing events, and thereby fewer nutrients to deal with, and less physical disturbance. For the studied scenarios, the number of grazing events per year for the two studied soils was rarely identical. Using a fixed grazing rotation and set number of grazing events per year might affect the results presented here. It would be interesting to use real farm data and compare it to the model outputs.

For the first time, this study provides a comparison of the ecosystem services of two soil types of contrasting natural capital under a range of common management practices, for a dairy farm operation.

The approach used in this study allows land managers to assess on which soil ecosystem services a management practice will have the most impact, by identifying the soil natural capital stocks likely to be affected by the practice.

Chapter Ten

Thesis Summary and Conclusions

This thesis has developed a framework for quantifying and valuing the ecosystem services provided by soils and tested the framework at the farm scale on two contrasting soils under a range of dairy farm operations.

The purpose of this chapter is to (1) explicitly identify the key contribution of this thesis, in particular, how it contributes to and extends current knowledge on soil natural capital and ecosystem services; and (2) identify areas for further research and development.

10.1 Thesis contributions:

The key contribution this thesis has made to current knowledge has been to integrate current thinking in Soil Science and Ecological Economics for the development and implementation of a conceptual soil ecosystem services framework that links for the first time our understanding of soil formation and processes, with soil classification systems, current ecosystem services thinking, and existing frameworks.

These different links were explored through a series of theoretical steps (soil science and ecological economics concepts), static modelling or capacity building, dynamic modelling, and economic valuation.

The major theoretical, methodological and empirical contributions of the thesis are summarised below.

10.1.1 Theoretical contributions:

The concepts of natural capital and ecosystem services forms the core of this thesis, and are the basis for the quantification, modelling and valuation of soil ecosystem services that followed. The development of a conceptual framework linking Soil Science concepts to key Ecological Economics concepts, such as natural capital and ecosystem services (Chapter Two), constitutes part of the theoretical basis of this thesis.

The major contribution of this thesis to current thinking on soil ecosystem services is the new framework. In the development of the framework, several major milestones were achieved:

- *Settlement of the limitations of existing natural capital and ecosystem services frameworks*. The theoretical basis of the thesis built on frameworks developed previously including Costanza et al. (1997), de Groot et al. (2002) and the Millennium Ecosystem Assessment (2005), as well as soil specific frameworks like the ones developed by Daily at al. (1997a), Wall et al. (2004) and Robinson et al. (2009). Each of these frameworks contained a number of common limitations. These are identified and discussed in Chapter Two. The major limitations of existing frameworks included:

- They didn't inform in detail the part played by soils in the provision of ecosystem services,
- They didn't link ecosystem services back to natural capital stocks,
- They were difficult to implement practically.

This thesis has addressed these limitations and provides for the first time a clear picture of how Ecological Economics concepts can be integrated with Soil Science thinking into a soil ecosystem services framework.

- *Applicable definition of the concept of natural capital and links to Soil Science concepts*. The existing definitions of natural capital were reviewed and this concept was applied to soils by linking it to well-know Soil Science concepts. It is argued that soil natural capital are stocks, embodied by soil properties. A major contribution of this thesis is the recognition of the difference between inherent and manageable soil natural capital, a distinction well known to soil scientists and land managers but never used within an ecosystem services framework before. Such a distinction allows land managers to identify where and how land use impacts on the provision of soil services.
- *Applicable definition of the concept of ecosystem services and links to Soil Science concepts:* The existing definitions of ecosystem services were reviewed and this concept was applied to soils. Soil ecosystem services were defined as flows coming to or from soil natural capital stocks fulfilling human needs. Making the difference between natural capital stocks and ecosystem service flows is critical for land managers if they are to understand how climate and land uses impact on land resources, and the ecosystem services they provide.
- *Establishment of the difference between soil processes and ecosystem services within a natural capital and ecosystem services framework*: the place, within ecosystem services frameworks, of processes underpinning soil formation, functioning and degradation, which is strongly debated in the literature, was discussed and integrated to the conceptual framework. The difference was made for the first time between supporting (e.g. soil formation) and degradation (e.g. erosion) processes and ecosystem services. Such a distinction is critical in linking soil science knowledge to the concepts of natural capital and ecosystem services. The distinction between processes and services is also important when it comes to valuation, because it prevents overlaps and double counting.
- *Development of a new methodology to quantify ecosystem services:* A major contribution of this thesis to the fundamental understanding of ecosystem services is

the new approach adopted for the quantification of individual soil services. Proxies to measure each service are defined as the difference between potential loss/emission if the soil didn't provide the service, and actual measures. For example a measure of flood mitigation was defined as the difference between rainfall and runoff; a measure of N_2O regulation was defined as the difference between potential maximum emissions and actual emissions, and so forth. This new approach is a major advance in defining soil ecosystem services compared to just stating the status of soil natural capital stocks as it's been done so far. Such methodology bridges the gap between the concept of ecosystem services and its application at different scales, and enables land valuation to be detached from productive capacity or versatility.

- *Establishment of the place of external drivers within a natural capital and ecosystem services framework*: the place and role of external drivers such as climate, geomorphology or land use within an ecosystem services framework was very unclear until, in this thesis, it is shown that such drivers impact on natural capital stocks and thereby on the provision of ecosystem services.
- *Determination of the difference between the value of soil natural capital and the value of soil ecosystem services*: the difference is made between the value of soil natural capital and the value of soil ecosystem services. Such distinction is critical to value ecosystem services rigorously. In the literature authors (Costanza et al., 1997; Kim and Dixon, 1986) have used indifferently the non-annualised value of built infrastructure as a proxy for the value of ecosystem services, which is in our opinion, not in line with good accounting and economic theory.

10.1.2 Methodological contributions:

Another major contribution of this thesis has been in the development of methodologies and tools for quantifying and valuing ecosystem services. The methods developed here can be applied elsewhere, for different land uses or at different scales.

These methodological contributions include:

- *Methodology to identify the key soil properties and processes behind each service provided by soils.* Chapters Three and Four have discussed in detail what are the soil properties and processes at the origin of the provision of each soil service. The information presented in this thesis is specific to dairy grazed systems and the farm scale, but the methodology is applicable to any land use or other scales. Chapters Three and Four also identified what soil properties should be followed as proxies to measure each service.

- *Methodology to identify where and how external drivers impact on the provision of soil services.* Chapters Three and Four also discussed the impacts of external drivers like climate and land use on soil natural capital embodied by soil properties, and how they affect the provision of soil services. Again the analysis is specific to dairy grazed systems and the farm scale, but the methodology is once again applicable to any land use or other scales. Such methodology now enables land managers to predict the impacts of management or land use on outcomes at different scales including the provision of soil services, by identifying which natural capital stocks (soil properties) are affected.
- *Methodology to quantify the provision of soil ecosystem services at the farm scale.* Chapter Seven proposes a methodology to design proxies, based on dynamic soil properties, to quantify and measure each soil service. The proxies are built from the properties identified in Chapters Three and Four. These proxies are specific to dairy grazed systems and the farm scale, but the methodology is applicable to any land use or other scale. Moreover, in this thesis, proxies were calculated from the outputs of a dynamic model, but field data could be used to calculate them, and thereby measure the provision of soil services.
- *Methodology to model the provision of ecosystem services from soils.* Chapter Five detailed the methodology to include into a dynamic model the impacts of external drivers on soil properties and thereby on the provision of soil services. In this thesis, the impacts of cattle treading were examined but the same methodology could be used to inform the impacts of e.g. erosion or hydrophobicity at a larger scale.
- *Methodology to value soil ecosystem services at the farm scale.* Chapter Six carried out a critical review of the valuation techniques available for the economic valuation of ecosystem services. This review was then used in Chapter Seven to develop a methodology to value each soil service at the farm scale, under a dairy operation, based on the proxies determined in the same chapter. Basing economic valuation on dynamic proxies is a very innovative and powerful technique to apply the concepts of ecosystem services. Once again, the methodology could be used for different land uses and at different scales.

10.1.3 Knowledge contributions:

This thesis has produced information that has improved knowledge and insight into the provision of ecosystem services from soils.

- *Economic value of soil services for a Horotiu silt loam under a dairy operation:* The key contribution of this thesis is the data generated on the value of soil services for different soil types and different management practices. The average value of the ecosystem services for a Horotiu silt loam under a dairy operation was $15,777/ha/yr, with a range spanning from $10,189/ha/yr to $21,105/ha/yr, reflecting the interaction between climate and soil properties over the 35 years modelled. This data can now be compared to the value of ecosystem services from different ecosystems, or scaled up, or used in the valuation of services from agro-ecosystems. The thesis also tried for the first time to put a value on some soil natural capital stocks, and showed that such value was different from the value of soil services. It was argued that the value of the natural capital of the Horotiu silt loam under a typical dairy farm operation is at least $78,198/ha/yr, including only some easily valued natural capital stocks.

- *Value of provisioning versus regulating soil services*: The study showed that regulating services were more valuable than provisioning services, accounting for 70 to 77% of the total value of soil services. Of the provisioning services, the provision of food had the highest value (25.7% of the total value of services across all scenarios). Of the regulating services, the filtering (61.6% of the total value of services across all scenarios) and flood mitigation (8.2% of the total value of services across all scenarios) services had the highest value.
- *Impact of soil type on the value of soil services:* Averaged over 35 years, soil services under a dairy operation were worth $15,777/ha/yr from a Horotiu silt loam, and $11,687/ha/yr from a Te Kowhai silt loam. Soil type influenced greatly the value of soil services especially for the provision of food, flood mitigation and the filtering of nutrients and contaminants. The value of the services provided by the Horotiu silt loam was on average 35% greater than the value of the services provided by the Te Kowhai silt loam. These results reinforce the notion that Allophanic Soils are more valuable than Gley Soils. This study showed that this is true not only for production but also for all other soil services.
- *Impact of management practices on the value of soil services:* The increase in stocking rate had the most effect on the values of the provision of food, provision of support to animals, the recycling of wastes, N_2O regulation for both soils, and the regulation of pest populations for Horotiu silt loam. The influence of the use of a standoff pad had the most effect on the value of the provision of food from Horotiu silt loam, the filtering of nutrient and contaminants from both soils, and the regulation of pest and disease populations from Horotiu silt loam. Stocking rates and the use of a standoff pad impact mainly on soil structure and nutrient inputs to soils. This study showed that it is possible for land managers to predict on which soil ecosystem services a management practice will have the most impact by identifying the soil natural capital stocks that are most affected by the practice.

- *Identification of new research fields needed:* This study has identified soil science areas where information is missing, notably to link the impact of degradation processes on soil properties to the provision of ecosystem services. Such knowledge could be used to develop new research programs around land evaluation and planning, gaining new insight into for example climate change, investigating the impacts of different land uses, or extend the utility of soil quality indicators.
- *Use of field data:* One of the major strengths of the quantification of soil services realised in this study is that it is based on dynamic soil properties. The knowledge developed around what properties to consider when measuring ecosystem services enables managers to use real field data to measure the services instead of model outputs. The measures of some services can also be scaled up easily.

Before soils were considered as a black-box within ecosystem services frameworks and valuation has never been implemented for soil services. This thesis showed that not only it is feasible, but also that it is compatible with existing soil science knowledge.

10.2 Limitations of the study:

As outlined in the relevant chapters, several limitations could be addressed in future extensions of this research. They are discussed below.

10.2.1 Of the quantification:

The method used to measure each soil service in this study is specific to the measurement of ecosystem services from soils under a dairy land-use. As a consequence, some of the proxies used here to quantify each service are not directly transferable or applicable to other land uses. Further uncertainty exists in the quantification of ecosystem services from soils, from gaps in our knowledge of soil processes. Uncertainty also surrounds the outputs from process-based models. For example, the way cattle treading was modelled in this study was conservative, e.g. the decrease in macroporosity found in the field after treading is greater than model outputs. If time and resources are available, it will be worth trying to improve this part of the model with more field data to obtain better description of the actual impact of treading. The first thing to do would be to account for differences in behaviour of different soil types when calculating variables such as the maximum macropores loss, macroporosity recovery rate or the actual loss of pasture growth. The spatial component is also missing from this study and is part of the next step of the analysis.

Provision of food quantity: The method used to determine the part of the pasture yield due to natural capital is robust, because it uses data that has been proven to be accurate and is currently used for technical advice on pasture nutrient requirements. However, for different land uses, the provision of marketed goods is embodied by different products (trees, fruits, crops) and therefore the method to determine the part of the yield coming from natural capital would need to be adapted to that land use.

Provision of food quality: the provision of trace elements from soils can affect any agricultural activity; therefore determining the impact of trace element deficiencies on yields for different land uses is a robust method.

Provision of support for human infrastructure: in this study soil bulk density was used as a proxy to measure the provision of support for human infrastructure, which is relevant at the farm scale. However, if looking at a different scale (landscape, catchment, region) the shape of a landscape and the position of soil types in this landscape would have to be considered in addition to BD, as well as the nature of deep soil horizons and underlying regolith.

Provision of support for farm animals: Using SWC as a proxy for soil physical resistance to loading is robust at the farm scale which is used by farmers on a regular basis. At a different scale, the shape of a landscape and the position of soil types in this landscape would have to be considered.

Provision of raw materials: this service wasn't quantified here because it was considered non-relevant, at the farm scale however, at a different scale (catchment, region, country), it would be necessary to quantify it by considering the net flows of raw materials from soils and their renewability and sustainability.

Flood mitigation: the method used to quantify flood mitigation here (RF-RO) is fairly robust. However, at a different scale, landscape and most importantly slope, would have to be considered since on steep land, during heavy rainfall, water runs off before having time to infiltrate even if soil water storage capacity is available. Similarly, the influence of hydrophobicity, a degradation process, on this service, hasn't been investigated but should be included since it can have a major impact on this service.

Filtering of N: the method used here for the quantification of the filtering of N was enabled by the existence of a dynamic model (SPASMO) which was modified in order to determine N leaching for a soil with very low ASC. The method used was not ideal since lower ASC also means less pasture grown and therefore fewer wastes deposited on the paddock. By sticking with the general principle, it would be possible to gather data on N leaching on different soil types under the same management and compare them, the differences in N leaching being due to differences in soil ASC.

<u>Filtering of P</u>: Again, the method used here for the quantification of the filtering of P was enabled by the existence of a dynamic model (SPASMO) which was modified in order to determine P runoff and leaching for a soil with very low P retention. The amounts of simulated P lost for the Allophanic Soil studied were very large which shows how strongly some soils retain P. However, since no techniques exist to mitigate losses of P over 5 kg/ha/yr, it was impossible to value this service realistically.

<u>Filtering of contaminants</u>: In this study, to measure the filtering of contaminants, a proxy was used (contaminated runoff) because of a lack of detailed data on relevant contaminants. Instead, if data about the dynamics of each contaminant (e-coli, pesticides, EDCs) was available for the studied soils, the difference between amounts applied and leached could be used to measure the service. Such methodology would be relevant at different scales.

<u>Recycling of wastes</u>: In this study, to measure the recycling of wastes, a proxy was used (the amount of dung deposited in unrestricted conditions) because of the complexity of the dynamics of dung decomposition and the recycling and transformation of OM. Instead, if, for the studied soils, more data was available on the recycling of dung pads as a function of the season, Mp and SWC for example, the difference between amounts applied and efficiently decomposed could be used to measure the service. Typically this service would be better informed by using field data than model outputs. Such methodology would be relevant at different scales. Moreover, for scenarios including the use of a standoff pad, dung was deposited only when SWC<FC, which would have influenced the measure of the service.

<u>Carbon flows</u>: The net flows of C modelled with SPASMO are very sensitive to a number of parameters, therefore the outputs of the model should be considered with extreme caution. Actual data on measured C flows could be used instead if available for the studied soils. The question was raised of valuing C stocks, but here it’s argued that the service is the net flow of C. If positive it is truly a service since soils are storing C. If net flows are negative, then they can be considered as a degradation process and the impact of C losses on other soil properties and on the provision of soil services should be investigated.

N_2O regulation: the use of the IPCC methodology to calculate N_2O emissions from soils has been heavily criticised but is still a reference. The methodology used here is inspired from the IPCC methodology. It is argued here that the use of model outputs (N leaching and SWC) to calculate N_2O emissions, as well as the addition of an extra emission factor taking into account wet soil conditions makes the calculation more accurate. For even more accurate calculation a process-based model, such as the DNDC (Giltrap et al., 2008; Saggar et al., 2007a; Saggar et al., 2007b), specialised in GHGs emission could be used.

CH_4 oxidation: At the farm scale, CH_4 oxidation is quite small, therefore it would be more relevant to consider this service at a bigger scale. Our estimation of CH_4 oxidation from soils was based on data from the literature as well as model outputs (SWC) and is therefore approximate. For more accurate calculation a process-based model, such as the DNDC (Giltrap et al., 2008; Saggar et al., 2007a; Saggar et al., 2007b), specialised in GHGs emission could be used.

Regulation of pest and disease populations: Following soil conditions to assess pest infestation risk is a robust method used by farmers. However, the information used in this study is quite approximate. The quantification of this service could be improved by using more accurate data on the impact of soil conditions on the different stages of pest development. Moreover, animal pests (e.g. parasitic nematodes) weren't considered in this study because they aren't posing much of a problem in mature cows, but they should be considered if investigating a complete dairy system, with grazing calves.

The quantification of some services was limited by the availability or existence of relevant data. The identification of missing information could be used in the future to develop new research programs.

10.2.2 Of the economic valuation:

A number of issues are associated with the economic valuation of ecosystem services. General problems with neoclassical economic valuation were discussed in Chapter Six. However, the valuation undertaken in this study presents a number of specific limitations discussed below.

Joint production: The aggregation of the values of each service could be criticised because of the issues of joint production and double counting. As long as the ecosystem services are entirely independent, adding up the values is possible. However, the interconnectivity and interdependencies of ecosystem services may increase the likelihood of double-counting ecosystem services (Barbier et al., 1994). Moreover, a number of the methods used here to value soil services are subject to joint production (defensive expenditure, replacement cost, provision costs) (Pearce et al., 2006). By using the costs of built infrastructures like a standoff pad or effluent ponds, to value soil services, the values obtained are subject to joint production, as the use of infrastructures, such as a standoff pad, impacts on a number of soil properties (Mp, OM content, nutrient content) and thereby on a number of soil services. The use of the cost of built infrastructures to value the provision of specific services could potentially result in an overestimation of the value of the services. Lastly, different methods were used to value different services; therefore comparing values of different services or adding up values can be risky. This study hasn't dealt with these issues, therefore it is recommended to examine the values of each soil service separately and compare the value of one service between scenarios rather than compare total values.

Annualisation: when valuing ecosystem services, one wants to put a value on the flows coming from natural capital stocks and not the stocks in themselves. This is why when using the cost of infrastructure to value ecosystem services, one needs to annualise these costs in order to determine the annual flows of value that can be attributed to flows of ecosystem services. Annualisation is used as a rule in benefit-cost analysis. Nonetheless, in the literature some authors have used the value of built infrastructure for ecosystem service valuation without annualising it. In our opinion this approach is not in line with good accounting and economic theory.

Discount rate: The value of the discount rate used for annualisation is one of the major issues around BCA. Ecosystem service valuation being a very young field, there is no standard method generally accepted by scholars and the value of the discount rate to use for environmental studies is still highly controversial. In this study, it was shown that (Chapter Seven) changing from a 10% discount rate to a 3% discount rate when calculating annualised costs decreased the value of the service by around a third. Such information could be used to choose an appropriate discount rate depending on the project considered.

Mitigation functions: The mitigation functions build to value the filtering of N and P were constructed with a restricted number of model outputs (Appendix F), and therefore assumed linear. In the future, it would be recommended to use a great number of field data if available to build these curves.

The values obtained here are a lower bound estimate of ecosystem services from soils, because the valuation techniques used are not able to account for the dynamism, renewability and interconnectivity of soil natural capital stocks and the ecosystem services they provide. Combining valuation models and different economic valuation techniques (such as contingent valuation or group valuation) may be the way forward to minimise technical uncertainty in the application of valuation methods.

10.2.3 Of the framework and the whole exercise:

The focus of this study is the valuation of soil ecosystem services. The distinction is made between the value of the ecosystem services provided by soils (annual flows from natural capital stocks) and the value of soil natural capital (stocks). It is suggested that the construction costs of built infrastructure which provides the same services as a given natural capital stock could be used as a proxy for this stock. However, this study shows that some natural capital stocks and ecosystem services are difficult to value (e.g. the filtering of P) because no built infrastructure currently exists to replace specific natural capital stocks (e.g. C storage).
This poses the question of the relevance of natural capital and ecosystem services valuation as a tool to assess land value. It has been argued that natural capital and ecosystem services valuation is unable to capture the value that resides in the complexity, resilience, interconnectivity and renewability of natural ecosystems. However, even if the economic valuation of natural capital and ecosystem services provides only a lower bound estimate of the real value of land, such information will be useful to land managers and policy makers, who are familiar with costs and benefits. It would at least allow land managers and policy makers to take into account the total value of land more efficiently than currently, which is limited to value for production.

10.3 Future research:

10.3.1 Immediate future of this work:

<u>Wider natural capital and ecosystem services valuation:</u>
This study focuses on the valuation of ecosystem services from soils and starts discussion on the challenges of valuing soil natural capital and how it could be advanced. This study focused on one land use; therefore the next logical step would be to measure and value soil services for different land uses as well as different land use changes. The framework developed can be adapted to different land uses and can help identify which natural capital stocks to inventory and follow.

<u>Evaluation and up scaling:</u>

The study focuses at the farm scale, but to be useful as a decision support tool for land management, the quantification and valuation of soil ecosystem services should also be available at the catchment, regional and national scales.

10.3.2 Applications of a soil services framework for land management around the world:

A number of projects are already implemented around the world, and aim at developing decision support tools for land management and policy based on ecosystem services frameworks. These projects reflect the global realisation of the need for a better appreciation of the value of land in land use governance.

Defra, the Department for Environment, Food and Rural Affairs in the UK is taking a more systematic approach to the assessment of impacts on the natural environment (Defra, 2007), in order to ensure that the true value of ecosystems and the services provided are taken into account in policy decision-making (Beddington, 2010).

The European Union (EU) identified in the European Union Soil Thematic Strategy soil ecosystem services as a priority research area. The EU is financing a number of projects based on soil ecosystem services including:

- The SoilTrEC project (Soil transformations in European catchments). This project started in February 2010 for 5 years and involves 10 countries from the EU plus the USA and China. Its challenge is to understand and predict how soil provides ecosystem services, and how to protect soils against threats like erosion, loss of organic matter and loss of biodiversity. This research focuses on 12 field sites as Critical Zone Observatories around the world, where international scientific effort is concentrated.
- SOIL SERVICE project: This project started in September 2008 for 3.5 years and involves 8 countries from the EU. It is a collaborative, medium scale focussed research project. The aim of the project is to understand how economic drivers will change current and future use of soil-related ecosystem services and how they affect diversity and sustainability of agricultural soils. The project will construct quantitative scenarios of long-term land use change across Europe and determine how soil nutrients can be retained - even after extensive use.
- EcoFINDERS project (Ecological Function and Biodiversity Indicators in European Soils): This project started in February 2011 and gathers 23 partners from 10 European

countries plus China. It aims to increase our knowledge of soil biodiversity and its role in ecosystem services across different soils, climate types and land uses, to standardize methods and operating procedures for characterizing soil biodiversity and functioning, and develop bio-indicators, and to assess the added value brought by cost-effective bio-indicators, and of cost effectiveness of alternative ecosystem service maintenance policies.

10.3.3 Applications of a soil services framework for land management In New Zealand:

This study is part of the SLURI (Sustainable Land Use Research Initiative) project which started in October 2004 as a partnership between AgResearch, (then) Crop & Food Research, (then) HortResearch and Landcare Research. SLURI conducts research on the sustainable management and use of soil resources, develops new tools for regulators and land managers and works with key stakeholders and other research organisations. One of the priorities of the project is to quantify and value soil natural capital and ecosystem services.

The capacity developed within SLURI is being used in a number of projects with different stakeholders including Regional Councils.

10.3.3.1 Natural capital based land management:

Regional Councils are the organisations responsible for land management in New Zealand in accordance with the requirements of the Resource Management Act 1991 (MfE, 1991). The RMA sets out legislation on how to manage the environment in New Zealand. In the RMA "sustainable management" is defined as "managing the use, development, and protection of natural and physical resources in a way, or at a rate, which enables people and communities to provide for their social, economic, and cultural wellbeing and for their health and safety while:

(a) Sustaining the potential of natural and physical resources (excluding minerals) to meet the reasonably foreseeable needs of future generations; and

(b) Safeguarding the life-supporting capacity of air, water, soil and ecosystems; and

(c) Avoiding, remedying or mitigating any adverse effects on the environment."

The objective (b) refers broadly to natural capital stocks and the provision of ecosystem services, but recently, regional councils in New Zealand started using these concepts to inform land management through policy.

The Horizons Regional Council released in August 2010 their "One plan", a new regional plan to guide the management of natural resources in the Manawatu. One very noticeable feature of the One Plan is that nitrogen leaching allowed from new dairy farming land is based on rates depending on the natural capital of each Land Use Capability (LUC) class[14] of land (Lynn et al., 2009). It is the first time in New Zealand that a policy to reduce non-point source pollution by nutrients from farms is based on soil natural capital and the ecosystem services they provide and made independent of land use.

The Waikato Regional Council has recently produces its first “Regional Policy Statement” which has also been developed in accordance with the requirements of the RMA (MfE, 1991). This Regional Policy Statement provides an overview of the resource management issues in the region and policies and methods to achieve integrated management of natural and physical resources. This statement seeks a more integrated planning approach, with clear connections between air, land, water, and coastal resource management, based on new scientific research underpinning new policy and rules. The work realised in this study is going to be used by the Waikato Regional Council to inform their new policies.

10.3.3.2 Improving soil quality indicators:

Currently, Regional Councils’ state-of-environment monitoring and reporting for soil quality is based on target ranges for soil quality indicators (Sparling and Schipper, 2004). The soil quality indicators: New generation project started in June 2010. It aims at linking soil quality indicators to outcomes at the paddock, farm and catchment scale using the soil natural capital and ecosystem services framework developed in this thesis. This project involves different CRI’s (AgResearch, Plant and food research and Landcare research) and Regional councils around New Zealand.

Linking soil quality indicators to the provision of ecosystem services will increase their value to managers and policy makers by enabling a change in soil quality indicators to be linked to outcomes at a farm or catchment scale. Regional and national managers will then have a tool which can assess whether land use and land-use changes align with regional policy statements. The project has the potential to offer a nationally consistent approach.

10.3.3.3 Informing the debate on land use and land use change:

The methods developed in this study can also provide new insights to inform the debate on land use and land use change and the best use of the land resource in New Zealand. A presentation on the potential value of an ecosystem services approach to land management was made at a one-day Forum, “Collision of land use”, organised by the Royal Society of New

[14] Land Use Capability (LUC) classes are based on a assessment of the physical factors of the land and its long-term capability to sustain one or more land uses.

Zealand in August 2010 to raise debate around soils and land use within New Zealand with a view to establishing a policy for New Zealand land use (Mackay et al., 2011).

The main recommendations coming out of this forum were:

- New Zealand should move towards a national framework of interest and associated national standards on land to assist regions and districts to provide guidelines and limits for policy development on land management and land use changes at local and regional level.
- An integrated approach to the use of land is necessary to assess the wider implications of ongoing land use change on society. New Zealand should therefore transition from a sector-based approach to land use on farms to a systems-based approach that considers natural capital values and ecosystem services.
- Land use development should switch from overcoming limitations to increasing the natural capital of soil and enhancing ecosystem services from the land, and matching land use with land capability.
- Science must have an increased role in contributing to providing solutions to minimise potential damage instead of trying to support unproductive areas.
- Such new imperatives will require a significant investment in new and emerging research themes to inform policy development.

Most of all these recommendations can be informed by the framework developed in this thesis.

10.3.3.4 Investing in ecological infrastructure:

Improving the valuation of soil ecosystem services and advancing further the techniques to value soil natural capital can also add fuel to discussions around investments in ecological infrastructure (that is natural capital stocks) and how they can improve the yield of ecosystem services coming from land (Bristow et al., 2010) and increase the sustainability of land uses.

The framework developed in this thesis could be used to provide new insights into land development. Over the last 100 years science has been at the forefront of the development of production technologies to overcome soil limitations (low nutrient status (e.g. fertilisers, legumes), wetness (e.g. drainage, flipping), low water holding (e.g. irrigation) and stoniness (removal or burial)). The future focus on land development needs to increasingly be on the efficiency of use of both natural resources (land, climate (e.g. rainfall) and scarce inputs (e.g. nutrients).

In New Zealand the area of high class soils (i.e. Class I and II land) is confined to 1.4 million ha. This represents <10% of the pastoral land farmed. Investigating the feasibility and costs associated with the transformation of Class III land (2.4 million ha) into Class I or II land, and the potential impact on their provision of ecosystems services is made possible by the framework developed in this thesis.

List of PhD Outputs

During the course of this research two papers were accepted for publication in journals (lead author in both cases), Two papers are manuscripts in progress (lead author), four papers were presented at conferences (three by the student, one by MG Patterson on behalf of the student). All of these research outputs build on methodologies created in this thesis, and report its key findings. The full list of outputs is presented below:

Published Papers

Dominati EJ, Patterson MG, Mackay AD (2010) A framework for classifying and quantifying the natural capital and ecosystem services of soils. Ecological Economics 69, 9, 1858-1868.

Dominati EJ, Patterson MG, Mackay AD (2010) Response to Robinson and Lebron - Learning from complementary approaches to soil natural capital and ecosystem services. Ecological Economics, 70, 2, 139-140.

Forthcoming Papers

Chapters Five, Seven, Eight and Nine are being rewritten as three papers to be submitted to the Soil Science Society of America Journal, and to Ecological Economics. A contextual paper, written with a number of authors has been submitted to the Vadoze Zone Journal.

Robinson D.A., Hockley N., **Dominati E.J.**, Lebron I., Scow K.M., Reynolds B., Emmett B.A., Keith A., de Jonge L.W., Schjønning P., Moldrup P., Jones S.B., Tuller M. (2011) Natural Capital, Ecosystem Services and Soil Change: Why Soil Science must Embrace an Ecosystems Approach. Vadose Zone Journal (to be published).

Conference Papers

Dominati EJ, Mackay AD, Green S, Patterson MG (2011) The value of Soil Services for Nutrients Management. In “the 24th Annual Fertilizer & Lime Research Centre Workshop: Adding to the knowledge base for the nutrient manager”, 8-10 February 2011, Massey University, Palmerston North, New Zealand.

http://www.massey.ac.nz/~flrc/workshops/11/paperlist11.htm

Dominati EJ, Mackay AD, Patterson MG (2010) Modelling Ecosystem Services from Soils: A study case in New Zealand dairy farms. In “11th Biennial conference of the International Society for Ecological Economics: Advancing sustainability in a time of crisis”, 22-25 August

2010, Oldenburg and Bremen, Germany. *(Presented by MG Patterson on behalf of EJ Dominati).*

Dominati EJ, Patterson MG, Mackay AD (2010) Modelling the provision of ecosystem services from soil natural capital. In "19th World Congress of Soil Science: Soil Solutions for a Changing World", 1-6 August 2010, Brisbane, Australia. *(Lead talk in the symposium on Soil Ecosystem services)*

Dominati EJ, Patterson MG, Mackay AD (2009) A framework for the ecosystem services provided by soils. In "8th International Conference of the European Society for Ecological Economics: Transformation, innovation and adaptation for sustainability", 29th June - 2nd July 2009, Biotechnical Faculty, Ljubljana, Slovenia.

Dominati EJ, Mackay AD, Patterson MG (2008) Valuing the ecosystem services provided by the natural capital of soils. In "Australian and New Zealand Society's of Soil Science Conference: Soil – the living skin of planet earth", 1-5 December 2008, Massey University, Palmerston North, New Zealand.

References

(2010) Point Carbon, http://www.pointcarbon.com, Carbon prices.

Aarons S.R., O'Connor C.R., Gourley C.J.P. (2004) Dung decomposition in temperate dairy pastures I. Changes in soil chemical properties. Australian Journal of Soil Research 42:107-114.

AgResearch. (2005) Overseer® nutrient budgets model, www.agresearch.co.nz/overseerweb/.

Aislabie J.M., Richards N.K., Boul H.L. (1997) Microbial degradation of DDT and its residues-A review. New Zealand Journal of Agricultural Research 40:269-282.

Aslam T., Choudhary M.A., Saggar S. (2000) Influence of land-use management on CO2 emissions from a silt loam soil in New Zealand. Agriculture, Ecosystems & Environment 77:257-262.

Aslam T., Deurer M., Müller K., Clothier B.E., Rahman A., Northcott G., Ghani A. (2009) Does an increase in soil organic carbon improve the filtering capacity of aggregated soils for organic pesticides? A case study. Geoderma 152:187-193.

Attfield R. (1998) Existence value and intrinsic value. Ecological Economics 24:163-168.

Balmford A., Fisher B., Green R.E., Naidoo R., Strassburg B., Turner R.K., Rodrigues A.S.L. (2011) Bringing ecosystem services into the real world: An operational framework for assessing the economic consequences of losing wild nature. Environmental and Resource Economics 48:161-175.

Balmford A., Bruner A., Cooper P., Costanza R., Farber S., Green R.E., Jenkins M., Jefferiss P., Jessamy V., Madden J., Munro K., Myers N., Naeem S., Paavola J., Rayment M., Rosendo S., Roughgarden J., Trumper K., Turner R.K. (2002) Ecology - Economic reasons for conserving wild nature. Science 297:950-953.

Barber S.A. (1995) Soil Nutrient Bioavailability: a mechanistic approach John Wiley & Sons, Inc, New York.

Barbier E.B., Burgess J., Folke C. (1994) Paradise lost? The Ecological Economics of Biodiversity Earthscan, London.

Bardgett R.D. (2005) The Biology of Soil: A community and ecosystem approach Oxford University Press, Oxford.

Bardgett R.D., Cook R. (1998) Functional aspects of soil animal diversity in agricultural grasslands. Applied Soil Ecology 10:263-276.

Barrios E. (2007) Soil biota, ecosystem services and land productivity. Ecological Economics 64:269-285.

Beddington J. (2010) Foresight Land Use Futures Project: Making the most of land in the 21st century, The Government Office for Science, London.

Betteridge K., Mackay A.D., Drewry J., Singleton P. (2003) Managing treading damage on dairy and beef farms in New Zealand. AgResearch.

Betteridge K., Mackay A.D., Shepherd T.G., Barker D.J., Budding P.J., Devantier B.P., Costall D.A. (1999) Effect of cattle and sheep treading on surface configuration of a sedimentary hill soil. Australian Journal of Soil Research 37:743-760.

Betteridge K., Mackay A.D., Pande T.N., Costall D.A., Budding P., Valentine I., Singleton P.L., Drewry J.J., Findlayson J., Boyes M., Judge A. (2002) Cattle treading on wet soils: implications for pasture growth and soil physical condition, in: L. D. Currie and P. Loganathan (Eds.), Dairy Farm Soil Management, Fertilizer and Lime Research Centre, Occasional Report No.15, Massey University, Palmerston North, New Zealand. pp. 79-88.

Birol E., Smale M., Gyovai A. (2006) Using a choice experiment to estimate farmers' valuation of agrobiodiversity on hungarian small farms. Environmental & Resource Economics 34:439 - 469.

Bolan N.S., Horne D.J., Currie L.D. (2004) Growth and chemical composition of legume-based pasture irrigated with dairy farm effluent. New Zealand Journal of Agricultural Research 47:85-93.

Bolan N.S., Khan M.A., Donaldson D.C., Adriano D.C., Matthew C. (2003) Distribution and bioavailability of copper in farm effluent. Science of the Total Environment 309:225-236.

Borren W., Bleuten W., Lapshina E.D. (2004) Holocene peat and carbon accumulation rates in the southern taiga of western Siberia. Quaternary Research 61:42-51.

Boyd J., Banzhaf S. (2007) What are ecosystem services? The need for standardized environmental accounting units. Ecological Economics 63:616-626.

Brauman K.A., Daily G.C., Duarte T.K., Mooney H.A. (2007) The nature and value of ecosystem services: An overview highlighting hydrologic services. Annual Review of Environment and Resources 32:67-98.

Bristow K.L., Marchant S.M., Deurer M., Clothier B.E. (2010) Enhancing the Ecological Infrastructure of Soils, 19th World Congress of Soil Science: Soil Solutions for a Changing World, Brisbane, Australia.

Brown M., Whitehead D., Hunt J.E., Clough T.J., Arnold G.C., Baisden W.T., Sherlock R.R. (2009) Regulation of soil surface respiration in a grazed pasture in New Zealand. Agricultural and Forest Meteorology 149:205-213.

Campbell A. (2009) A strategic approach to sustainable agriculture, in: L. D. Currie and C. L. Lindsay (Eds.), Nutrient Management in a Rapidly Changing World, Fertilizer & Lime Research Centre, Occasional Report No. 22, Massey University, Palmerston North, New Zealand.

Cichota R., Snow V.O. (2009) Estimating nutrient loss to waterways-an overview of models of relevance to New Zealand pastoral farms. New Zealand Journal of Agricultural Research 52:239-260.

Cole K., Cameron J., Edwards C. (1991) Why Economists Disagree: The Political Economy of Economics Longman, Harlow.

Colombo S., Calatrave-Requena J., Gonzalez-Roa M.C. (2005) Testing choice experiment for benefit transfer, 99th seminar of the EAAE (European Association of Agricultural Economists), 24-27 August, Copenhagen, Denmark.

Colombo S., Calatrava-Requena J., Hanley N. (2006) Analysing the social benefits of soil conservation measures using stated preference methods. Ecological Economics 58:850-861.

Cordell H.K., Bergstrom J.C., Bowker J.M. (2005) The multiple values of wilderness Venture Publishing, State College, Pennsylvania.

Costanza R. (1991) Ecological Economics: The Science and Management of Sustainability Columbia University Press, New York.

Costanza R. (2008) Ecosystem services: Multiple classification systems are needed. Biological Conservation 141:350-352.

Costanza R., Hannon B. (1989) Dealing with the mixed units problem in ecosystem network analysis, in: F. Wulff, et al. (Eds.), Network Analysis in Marine Ecology: Methods and Applications, Springer, Berlin. pp. 90-115.

Costanza R., Daly H.E. (1992) Natural Capital and Sustainable Development. Conservation Biology 6:37-46.

Costanza R., Farber S.C., Maxwell J. (1989) Valuation and management of wetland ecosystems. Ecological Economics 1:335-361.

Costanza R., dArge R., deGroot R., Farber S., Grasso M., Hannon B., Limburg K., Naeem S., Oneill R.V., Paruelo J., Raskin R.G., Sutton P., vandenBelt M. (1997) The value of the world's ecosystem services and natural capital. Nature 387:253-260.

Cruickshank M.M., Tomlinson R.W., Bond D., Devine P.M., Edwards C.J.W. (1995) Peat extraction, conservation and the rural economy in Northern Ireland. Applied Geography 15:365-383.

Crush J.R., Woodward S.L., Eerens J.P.J., MacDonald K.A. (2006) Growth and milk solids production in pastures of older and more recent ryegrass and white clover cultivars under dairy grazing. New Zealand Journal of Agricultural Research 49:119-135.

Daily G.C. (1997) Nature's Services: Societal Dependence on Natural Ecosystems Island Press, Washington.

Daily G.C. (1999) Developing a scientific basis for managing Earth's life support systems. Conservation Ecology 3(2): 14.

Daily G.C., Matson P.A., Vitousek P.M. (1997a) Ecosystem services supplied by soils, in: G. C. Daily (Ed.), Nature's Services: Societal Dependence on Natural Ecosystems, Island Press, Washington DC.

Daily G.C., Alexander S., Ehrlich P.R., Goulder L., Lubchenco J., Matson P.A., Mooney H.A., Postel S., Schneider S.H., Tilman D., Woodwell G.M. (1997b) Ecosystem Services: Benefits Supplied to Human Societies by Natural Ecosystems. Issues in Ecology 2.

DairyNZ. (2010) Feed pests, http://www.dairynz.co.nz/page/pageid/2145861168/Pests.

Dale V.H., Polasky S. (2007) Measures of the effects of agricultural practices on ecosystem services. Ecological Economics 64:286-296.

Daly H.E. (1973) Toward a Steady-State Economy. Freeman, W.H., San Francisco, CA.

Daly H.E., Farley J. (2003) Ecological economics : principles and applications Island Press, Washington, D.C.

de Groot R. (1992) Functions of Nature: Evaluation of Nature in Environmental Planning, Management and Decision Making. Wolters-Noordhoff, Groningen

de Groot R. (2006) Function-analysis and valuation as a tool to assess land use conflicts in planning for sustainable, multi-functional landscapes. Landscape and Urban Planning 75:175-186.

de Groot R., Wilson M.A., Boumans R.M.J. (2002) A typology for the classification, description and valuation of ecosystem functions, goods and services. Ecological Economics 41:393–408.

de Groot R., Van der Perk J., Chiesura A., van Vliet A. (2003) Importance and threat as determining factors for criticality of natural capital. Ecological Economics 44:187-204.

de Klein C.A.M. (2001) An analysis of environmental and economic implications of nil and restricted grazing systems designed to reduce nitrate leaching from New Zealand dairy farms. II. Pasture production and cost/benefit analysis. New Zealand Journal of Agricultural Research 44:217-235.

de Klein C.A.M., Ledgard S.F. (2001) An analysis of environmental and economic implications of nil and restricted grazing systems designed to reduce nitrate leaching from New Zealand dairy farms. I. Nitrogen losses. New Zealand Journal of Agricultural Research 44:201-215.

de Klein C.A.M., Ledgard S.F. (2005) Nitrous oxide emissions from New Zealand agriculture - key sources and mitigation strategies. Nutrient Cycling in Agroecosystems 72:77-85.

De Klein C.A.M., Eckard R.J. (2008) Targeted technologies for nitrous oxide abatement from animal agriculture. Australian Journal of experimental agriculture 48:14-20.

de Klein C.A.M., Barton L., Sherlock R.R., Li Z., Littlejohn R.P. (2003) Estimating a nitrous oxide emission factor for animal urine from some New Zealand pastoral soils. Australian Journal of Soil Research 41:381-399.

Defra. (2007) An Introductory Guide to Valuing Ecosystem Services, London.

Degens B.P., Schipper L.A., Claydon J.J., Russell J.M., Yeates G.W. (2000) Irrigation of an allophanic soil with dairy factory effluent for 22 years: responses of nutrient storage and soil biota. Australian Journal of Soil Research 38:25-35.

Dexcel. (2005a) Minimising Muck, Maximising Money - Stand-off and Feed Pads Information Series - Part One: Design and Management Guidelines, Dexcel.

Dexcel. (2005b) Minimising Muck, Maximising Money- Stand-off and Feed Pad Information Series - Part Two: Case Studies, Dexcel.

Dominati E.J., Patterson M.G., Mackay A.D. (2010a) A framework for classifying and quantifying the natural capital and ecosystem services of soils. Ecological Economics 69:1858-1868.

Dominati E.J., Patterson M.G., Mackay A.D. (2010b) Response to Robinson and Lebron - Learning from complementary approaches to soil natural capital and ecosystem services. Ecological Economics 70:139-140.

Donnison A., Ross C. (2009) Survival and retention of Eschenchia coli O157:H7 and Campylobacter in contrasting soils from the Toenepi catchment. New Zealand Journal of Agricultural Research 52:133-144.

Douguet J.-M., O'Connor M. (2003) Maintaining the integrity of the French terroir: a study of critical natural capital in its cultural context. Ecological Economics 44:233-254.

Drewry J.J. (2006) Natural recovery of soil physical properties from treading damage of pastoral soils in New Zealand and Australia: A review. Agriculture Ecosystems & Environment 114:159-169.

Drewry J.J., Paton R.J. (2000) Effects of cattle treading and natural amelioration on soil physical properties and pasture under dairy farming in Southland, New Zealand. . New Zealand Journal of Agricultural Research 43:377-386.

Drewry J.J., Cameron K.C., Buchan G.D. (2001) Effect of simulated dairy cow treading on soil physical properties and ryegrass pasture yield. New Zealand Journal of Agricultural Research 44:181-190.

Drewry J.J., Cameron K.C., Buchan G.D. (2008) Pasture yield and soil physical property responses to soil compaction from treading and grazing - a review. Australian Journal of Soil Research 46:237-256.

Drewry J.J., Singleton P.L., Boyes M., Judge A., Wheeler D. (2003) Short term recovery of soil physical properties after winter grazing in Waikato: inplications for environmental monitoring, in: L. D. Currie and J. A. Hanly (Eds.), Tools for nutrients and pollutant management: Applications to agriculture and environmental quality, Fertilizer and Lime Research Centre, Occasional Report No. 17, Massey University, Palmerston North, New Zealand. pp. 194-204.

Drewry J.J., Littlejohn R.P., Paton R.J., Singleton P.L., Monaghan R.M., Smith L.C. (2004) Dairy pasture responses to soil physical properties. Australian Journal of Soil Research 42:99-105.

Drewry J.J., Littlejohn R.P., Paton R.J., Singleton P.L., Boyes M., Judge A., Monaghan R.M., Smith C.L. (2002) Dairy pasture yield responses to macroporosity and soil physical properties, and variability of large and small samples, in: L. D. Currie and P. Loganathan (Eds.), Dairy Farm Soil Management, Fertilizer and Lime Research Centre, Occasional Report No.15, Massey University, Palmerston North, New Zealand. pp. 61-78.

Dymond J.R., Baisden W.T. (2010) Soil erosion in New Zealand is a net sink of CO2. Earth Surface Processes and Landforms.

Edmeades D.C., Metherell A.K., Waller J.E., Roberts A.H.C., Morton J.D. (2006) Defining the relationships between pasture production and soil P and the development of a dynamic P model for New Zealand pastures: a review of recent developments. New Zealand Journal of Agricultural Research 49:207-222.

Edwards J., Gregg P.E.H., Mackay A.D., Richardson M.A. (1994a) Movement of phosphate in a Wharekohe podzol, Technical workshop on the efficient use of fertilisers in a changing environment, Massey University, Palmerston North, New Zealand. pp. 318-325.

Edwards J., Gre P.E.H., Mackay A.D., Richardson M.A. (1994b) Movement of phosphate on a Wharekohe podzol, in: L. D. Currie and P. Loganathan (Eds.), The Efficient Use of Fertilizers in a Changing Environment: Reconciling Productivity and Sustainability, Fertilizer and Lime Research Centre, Occasional Report No.7, Massey University, Palmerston North, New Zealand.

Eggleston H.S., Buendia L., Miwa K., Ngara T., Tanabe K. (2006) 2006 IPCC Guidelines for National Greenhouse Gas Inventories, Prepared by the National Greenhouse Gas Inventories Programme, in: IPCC (Ed.), Published: IGES, Japan.

Ekins P., Folke C., de Groot R. (2003a) Identifying critical natural capital. Ecological Economics 44:159-163.

Ekins P., Simon S., Deutsch L., Folke C., de Groot R. (2003b) A framework for the practical application of the concepts of critical natural capital and strong sustainability. Ecological Economics 44:165-185.

Ellison R.S. (2002) Major trace elements limiting livestock performance in New Zealand. New Zealand Veterinary Journal 50 (3) Supplement:35-40.

FAO. (1998) World Reference Base for Soil Resources. Food and Agriculture Organization of the United Nations; International Society of Soil Science; International Soil Reference and Information Centre, Rome, Italy.

Farber S., Griner B. (2000) Using conjoint analysis to value ecosystem change. Environmental science and technology 34:1407 - 1412.

Farber S.C., Costanza R., Wilson M.A. (2002) Economic and ecological concepts for valuing ecosystem services. Ecological Economics 41:375-392.

FF. (2010) Farming Facts - Snapshot of New Zealand agriculture, Federated Farmers of New Zealand.

Finlayson J.D., Betteridge K., Mackay A.D., Thorrold B., Singleton P.L., Costall D.A. (2002) A simulation model of the effects of cattle treading on pasture production on North Island, New Zealand, hill land. New Zealand Journal of Agricultural Research 45:255-272.

Fisher B., Turner R.K. (2008) Ecosystem services: Classification for valuation. Biological Conservation 141:1167-1169.

Fisher B., Turner R.K., Morling P. (2009) Defining and classifying ecosystem services for decision making. Ecological Economics 68:643-653.

Fleming P.H. (2003) Farm Technical Manual. Farm management group, Lincoln University, Lincoln, New Zealand.

Fraser P.M., Boag B. (1998) The distribution of lumbricid earthworm communities in relation to flatworms: a comparison between New Zealand and Europe. Pedobiologia 42:542-553.

Fromm O. (2000) Ecological structure and functions of biodiversity as elements of its total economic value. Environmental & Resource Economics 16:303-328.

Garrod G., Willis K.G. (1999) Economic valuation of the environment : methods and case studies. Edward Elgar Pub, Northampton, Mass.

Gasparatos A., El-Haram M., Horner M. (2008) A critical review of reductionist approaches for assessing the progress towards sustainability. Environmental Impact Assessment Review 28:286-311.

Georgescu-Roegen N. (1971) The entropy law and the economic process. Harvard University Press, Cambridge, MA.

Georgescu-Roegen N. (1979) Energy analysis and economic valuation. Southern Economic Journal 45:1023-1058.

Ghani A., Dexter M., Sarathchandra U., Waller J. (2005) Effects of dairy factory effluent application on nutrient transformation in soil. New Zealand Journal of Agricultural Research 48:241-253.

Ghani A., Muller K., Dodd M., Mackay A.D. (2010) Dissolved organic matter leaching in some contrasting New Zealand pasture soils. European Journal of Soil Science 61:525-538.

Gillingham A.G. (2009) Soils and regional land use - Overview http://www.TeAra.govt.nz/en/soils-and-regional-land-use/1 Te Ara - the Encyclopedia of New Zealand.

Gillingham A.G., Morton J.D., Gray M.H. (2007) Pasture responses to phosphorus and nitrogen fertilisers on East Coast hill country: total production from easy slopes. New Zealand Journal of Agricultural Research 50:307-320.

Gillingham A.G., Morton J.D., Gray M.H. (2008) Pasture responses to phosphorus and nitrogen fertilisers on east coast hill country: 2. Clover and grass production from easy slopes. New Zealand Journal of Agricultural Research 51:85-97.

Giltrap D., Saggar S., Li C.S., Wilde H. (2008) Using the NZ-DNDC model to estimate agricultural N2O emissions in the Manawatu-Wanganui region. Plant and Soil 309:191-209.

Grace N. (1994) Managing trace elements deficiencies. AgResearch, New Zealand.

Gradwell M.W. (1965) Soil physical conditions of winter and the growth of ryegrass plants: I. Effects of compaction and puddling. New Zealand Journal of Agricultural Research 8:238-269.

Green S., Clothier B.E. (2009) Nitrate Leaching Under Various Land Uses in Canterbury: A report prepared for Environment Canterbury, Plant & Food Research, Palmerston North.

Green S., van den Dijssel C., Snow V., Clothier B.E., Webb T., Russell J., Ironside N., Davidson P. (2003) SPASMO - A risk assessment model of water, nutrient and chemical fate under agricultural lands, in: L. D. Currie and J. A. Hanly (Eds.), Tools for nutrients and pollutant management: Applications to agriculture and environmental quality, Fertilizer and Lime Research Centre, Occasional Report No. 17, Massey University, Palmerston North, New Zealand. pp. 321-335.

Green S.R. (2008) Modelling the fate of nutrients and pathogens from on-site wastewater systems in the Tararua/Gladstone Road area of Horowhenua District using SPASMO, HortResearch (now Plant and Food Research). pp. 57.

Green S.R., Vogeler I., Clothier B.E., Mills T.M., van den Dijssel C. (2002) Modelling water uptake by a mature apple tree. Australian Journal of Soil Research 41:365-380.

Green S.R., Sivakumaran S., van den Dijssel C., Mills T.M., Blattmann P., Snelgar W.P., Clearwater M.J., Judd M. (2006) A water and nitrogen budget for 'Hort16A' kiwifruit vines, in: A. R. Ferguson, et al. (Eds.), 6th International Symposium on Kiwifruit, Rotorua, New Zealand. pp. 527-534.

Grime J.P. (2001) Plant strategies, vegetation processes, and ecosystem properties. John Wiley & Sons Ltd, Chichester, UK.

Gronvold J. (1979) Possible role of earthworms in the transmission of Ostertagia-ostertagi 3rd-stage larvae from feces to soil. Journal of Parasitology 65:831-832.

GWP. (2011) Global Warming Potentials, http://unfccc.int/ghg_data/items/3825.php, United Nations Framework Convention on Climate Change.

Hanley N., Spash C. (1993) Cost-benefit analysis and the environment. Edward Elgar, Cheltenham.

Hannon B., Costanza R., Herendeen R. (1986) Energy cost and value in ecosystems. Journal of Environmental Economics and Management 13:391-401.

Hawke R.M., Summers S.A. (2006) Effects of land application of farm dairy effluent on soil properties: a literature review. New Zealand Journal of Agricultural Research 49:307-320.

Haygarth P.M., Ritz K. (2009) The future of soils and land use in the UK: Soil systems for the provision of land-based ecosystem services. Land Use Policy 26:S187-S197.

Hedley M., McLaughlin M. (2005) Reaction of phosphate fertilizers and by-products in soils, in: A. Sharpley and T. Sims (Eds.), Phosphorus: Agriculture and the environment. Agronomy Monograph no. 46, American Society of Agronomy, Crop Science Society of America, Soil Science Society of America, Madison, Wisc. pp. 181-252.

Hewitt A., Hedley C., Rosser B. (2010) Evaluation of soil natural capital in two soilscapes, 19th World Congress of Soil Science: Soil Solutions for a Changing World, Published on CDROM, Brisbane, Australia.

Hewitt A.E. (1993) New Zealand soil classification, Science Series No. 1, Landcare Research - Manaaki Whenua Press, Lincoln, NZ.

Hewitt A.E., Shepherd T.G. (1997) Structural vulnerability of New Zealand soils. Australian Journal of Soil Research 35:461-474.

Hillel D. (1980) Fundamentals of soil physics. Academic press, London.

Holmes P. (1998) Investment appraisal International Thomson Business Press, London.

Hoogendoorn C.J., Betteridge K., Costall D.A., Ledgard S.F. (2010) Nitrogen concentration in the urine of cattle, sheep and deer grazing a common ryegrass / cocksfoot / white clover pasture. New Zealand Journal of Agricultural Research 53:235-243.

Horne D., Bretherton M.R., Hanly J.A., Houlbrooke D.J., Roygard J. (2011) The FDE storage calculator – an update, in: L. D. Currie and C. L. Christensen (Eds.), Adding to the Knowledge Base for the Nutrient Manager, Fertilizer & Lime Research Centre, Occasional Report No. 24, Massey University, Palmerston North, New Zealand.

Houlbrooke D.J. (2008) Best practice management of farm dairy effluent in the Manawatu-Wanganui region. AgResearch client report for Horizons Regional Council. February 2008.

Houlbrooke D.J., Monaghan R.M. (2009) The influence of soil drainage characteristics on contaminant leakage risk associated with the land application of farm dairy effluent. AgResearch client report for Environment Southland. May 2009.

Houlbrooke D.J., Horne D.J., Hedley M.J., Hanly J.A., Snow V.O. (2004) A review of literature on the land treatment of farm-dairy effluent in New Zealand and its impact on water quality. New Zealand Journal of Agricultural Research 47:499-511.

Houlbrooke D.J., Horne D.J., Hedley M.J., Snow V.O., Hanly J.A. (2008) Land application of farm dairy effluent to a mole and pipe drained soil: implications for nutrient enrichment of winter-spring drainage. Australian Journal of Soil Research 46:45-52.

Houlbrooke D.J., Drewry J.J., Monaghan R.M., Paton R.J., Smith L.C., Littlejohn R.P. (2009) Grazing strategies to protect soil physical properties and maximise pasture yield on a Southland dairy farm. New Zealand Journal of Agricultural Research 52:323-336.

Howarth R.B., Wilson M.A. (2006) A theoretical approach to deliberative valuation: Aggregation by mutual consent. Land Economics 82:1-16.

Jackson T. (1990) Biological control of grass grub in Canterbury. Proceedings of the New Zealand Grassland Association 52:217-220.

Jackson T.A., Popay A.J., Goldson S.L. (2002) Bioprotection – getting the most out of biological control in pasture pest management. Proceedings of the New Zealand Grassland Association 64:139-142.

James D. (1994) The application of economic techniques in environmental impact assessment. Kluwer Academic Publishers, Dordrecht.

Jenny H. (1941) Factors of soil formation: a system of quantitative pedology. McGraw-Hill, New York.

Kalmakoff J., Archibald R.D., Fleming S.B., Stewart K.M. (1993) The effect of pasture management practises on Wiseana spp populations and disease levels, in: R. A. Prestidge (Ed.), Proceedings of the 6th Australasian grassland invertebrate ecology conference, Hamilton, New Zealand.

Kavamura V.E., Esposito E. (2010) Biotechnological strategies applied to the decontamination of soils polluted with heavy metals. Biotechnology Advances 28:61-69.

Kim H., Dixon J. (1986) Economic valuation of environmental quality aspects of upland agriculture projects in Korea, in: J. Dixon and M. Hufschimdt (Eds.), Economic Valuation Techniques for the Environment: A Case Study Workbook, The John Hopkins University Press, Baltimore, London. pp. 203.

Kladivko E.J., Mackay A.D., Bradford J.M. (1986) Earthworms as a factor reducing surface crusting in soil. Soil Science Society of America Journal 50:191-196.

Kneese A.V., Ayres R.U., d'arge R.C. (1970) Economics and the environment, a materials balance approach. Resources for the future, inc., Washington, D.C.

Kouli M., Soupios P., Vallianatos F. (2009) Soil erosion prediction using the Revised Universal Soil Loss Equation (RUSLE) in a GIS framework, Chania, Northwestern Crete, Greece. Environmental Geology 57:483-497.

Kroeger T., Casey F. (2007) An assessment of market-based approaches to providing ecosystem services on agricultural lands. Ecological Economics 64:321-332.

Lambert G., Clark D., Mackay A.D. (1989) Long term effects of withholding phosphate application on North Island hill country: Ballantrae. Proceedings of the New Zealand Grassland Association 51:25-28.

Lambert M.G., Clark D.A., Litherland A.J. (2004) Advances in pasture management for animal productivity and health. New Zealand Veterinary Journal 52:311-319.

LandcareResearch. (2010) National Soils Database http://soils.landcareresearch.co.nz, New Zealand Soils Portal.

Lavelle P., Decaens T., Aubert M., Barot S., Blouin M., Bureau F., Margerie P., Mora P., Rossi J.P. (2006) Soil invertebrates and ecosystem services. European Journal of Soil Biology 42:S3-S15.

Le Mer J., Roger P. (2001) Production, oxidation, emission and consumption of methane by soils: A review. European Journal of Soil Biology 37:25-50.

Leathwick D., Miller C., Waghorn T. (2010) Development and spatial distribution of the free-living stages of Teladorsagia circumcincta and Trichostrongylus colubriformis on pasture – A pilot study. New Zealand Veterinary Journal.

Ledgard S.F., Penno J.W., Sprosen M.S. (1999) Nitrogen inputs and losses from clover/grass pastures grazed by dairy cows, as affected by nitrogen fertilizer application. Journal of Agricultural Science 132:215-225.

LIC, DairyNZ. (2009) New Zealand Dairy Statistics 2008/09, Livestock Improvement Corporation (LIC) and DairyNZ.

Longhurst R.D., Roberts A.H.C., Waller J.E. (2004) Concentrations of arsenic, cadmium, copper, lead, and zinc in New Zealand pastoral topsoils and herbage. New Zealand Journal of Agricultural Research 47:23-32.

Louviere J.J. (2001) Choice experiments: An overview of concepts and issues, in: J. Bennett and R. Blamey (Eds.), The choice modelling approach to environmental valuation, Edward Elgar, Cheltenham, UK. pp. 13-36.

Lynn I., Manderson A., Page M., Harmsworth G., Eyles G., Douglas G., Mackay A., Newsome P. (2009) Land Use Capability Survey Hand-book - a New Zealand handbook for the classification of land - 3rd ed., Hamilton, Agresearch, Lincoln, Landcare Research, Lower Hutt, GNS Science.

Mackay A.D. (2008) Impacts of intensification of pastoral agriculture on soils: Current and emerging challenges and implications for future land uses. New Zealand Veterinary Journal 56 281-288

Mackay A.D., Stokes S., Penrose M., Clothier B., Goldson S.L., Rowarth J.S. (2011) Land: Competition for future use. Science Review (In press).

Mackay A.D., Gillingham A.G., Smith C.L., Budding P., Phillips P., Clarke-Hill W., Johnstone P.D. (2010) Evaluation of the effects of grass species, irrigation, nitrogen fertiliser and soil compaction on the response of modern dairy pastures to phosphorus fertiliser. Proceedings of the New Zealand Grassland Association 72:153-158.

MAF. (2010) Situation and Outlook for New Zealand Agriculture and Forestry (SONZAF), Ministry for Agriculture and Forestry, Wellington, New Zealand.

Manski C. (1977) The Structure of Random Utility Models. Theory and Decision 8:229-254.

Marshall T.J. (1996) Soil physics. Cambridge University Press, Cambridge, New York.

Martinez-Alier J. (2001) Ecological Economics, in: J. S. Neil and B. B. Paul (Eds.), International Encyclopedia of the Social & Behavioral Sciences, Pergamon, Oxford. pp. 4016-4023.

Martinez-Alier J., Munda G., O'Neill J. (1998) Weak comparability of values as a foundation for ecological economics. Ecological Economics 26:277-286.

Maslow A.H. (1943) A theory of human motivation. Psychological Review 50:370-396.

Max-Neef M. (1992) Development and human needs, in: P. Ekins and M. Max-Neef (Eds.), Real-life Economics: Understanding Wealth Creation, Routledge, London ; New York. pp. 197-213.

McDonald G. (2005) Integrating Economics and Ecology: A Systems Approach to Sustainability in the Auckland Region, Ecological Economics, Massey University, Palmerston North, New Zealand.

McDowell R.W., Monaghan R.M., Wheeler D. (2005) Modelling phosphorus losses from pastoral farming systems in New Zealand. New Zealand Journal of Agricultural Research 48:131-141.

McDowell R.W., Nash D.M., Robertson F. (2007) Sources of phosphorus lost from a grazed pasture receiving simulated rainfall. Journal of Environmental Quality 36:1281-1288.

McDowell R.W., Nash D., George A., Wang Q.J., Duncan R. (2009) Approaches for quantifying and managing diffuse phosphorus exports at the farm/small catchment scale. Journal of Environmental Quality 38:1968-1980.

McLaren R.G., Cameron K.C. (1990) Soil science : an introduction to the properties and management of New Zealand soils. Oxford University Press, Auckland, N.Z.

McLeod M., Aislabie J., Ryburn J., McGill A. (2008) Regionalizing Potential for Microbial Bypass Flow through New Zealand Soils. Journal of Environmental Quality 37:1959-1967.

MEA. (2005) Millennium Ecosystem Assessment: Ecosystems and Human Well-being: Synthesis. Island Press, Washington DC.

Menkhaus S., Lober D.J. (1996) International ecotourism and the valuation of tropical rainforests in Costa Rica. Journal of Environmental Management 47:1-10.

Menneer J.C., Ledgard S., McLay C., Silvester W. (2005) Animal treading stimulates denitrification in soil under pasture. Soil Biology and Biochemistry 37:1625-1629.

MfE. (1991) Resource Management Act, Ministry for the Environment, New Zealand.

MfE. (2005) Land Use and Carbon Analysis System (LUCAS), Ministry for the Environment, Wellington, New Zealand.

MfE. (2009a) Net position report 2009, Ministry for the Environment, Wellington, New Zealand.

MfE. (2009b) New Zealand green house gases inventory 1990-2007, Ministry for the Environment, Wellington, New Zealand. pp. 72-93.

MfE. (2010) New Zealand's soil orders (North Island) http://www.mfe.govt.nz/publications/ser/enz07-dec07/html/chapter9-land/figure-9-1.html.

Ming J., Xian-guo L., Lin-shu X., Li-juan C., Shouzheng T. (2007) Flood mitigation benefit of wetland soil: A case study in Momoge National Nature Reserve in China. Ecological Economics 61:217-223.

Mitchell K.J. (1963) Production potential of New Zealand pasture land. New Zealand Institute of Agricultural Science Proceedings:80-86.

Mitchell R.C. (1989) Using surveys to value public goods : the contingent valuation method. Resources for the Future, Johns Hopkins University Press, Washington DC.

Miyazaki T. (2005) Water flow in soils. 2nd ed. Taylor & Francis, Boca Raton, FL.

Monaghan R.M., de Klein C.A.M., Muirhead R.W. (2008) Prioritisation of farm scale remediation efforts for reducing losses of nutrients and faecal indicator organisms to waterways: A case study of New Zealand dairy farming. Journal of Environmental Management 87:609-622.

Moriarty E.M., Sinton L.W., Mackenzie M.L., Karki N., Wood D.R. (2008) A survey of enteric bacteria and protozoans in fresh bovine faeces on New Zealand dairy farms. Journal of Applied Microbiology 105:2015-2025.

Morton J.D., Roberts A.H. (2001) Pasture response to soil phosphorus levels measured under mowing and dairy grazing. New Zealand Journal of Agricultural Research 44:259-268.

Muirhead R.W. (2009) Soil and faecal material reservoirs of Escherichia coli in a grazed pasture. New Zealand Journal of Agricultural Research 52:1-8.

Müller K., Deurer M., Newton P.C.D. (2010) Is there a link between elevated atmospheric carbon dioxide concentration, soil water repellency and soil carbon mineralization? Agriculture, Ecosystems and Environment 139:98-109.

Munda G., Nijkamp P., Rietveld P. (1994) Qualitative multicriteria evaluation for environmental management. Ecological Economics 10:97-112.

Munda G., Nijkamp P., Rietveld P. (1995) Qualitative multicriteria methods for fuzzy evaluation problems: An illustration of economic-ecological evaluation. European Journal of Operational Research 82:79-97.

Næss A. (1989) Ecology, community, and lifestyle: Outline of an ecosophy. Cambridge University Press, Cambridge, New York.

Niemeyer S., Spash C. (2001) Environmental valuation analysis, public delibaration, and their pragmatic syntheses: a critical appraisal. Environment and Planning: Government and Policy 19:567-585.

Nijkamp P., Vindigni G., Nunes P. (2008) Economic valuation of biodiversity: A comparative study. Ecological Economics 67:217-231.

NIWA. (2010) New Zealand's National Climate Database http://cliflo.niwa.co.nz/.

Noël J.F., O'Connor M. (1998) Strong sustainability: towards indicators for sustainability of critical natural capital, in: S. Faucheux and M. O'Connor (Eds.), Valuation for Sustainable Development: Methods and Policy Indicators, Edward Elgar, Cheltenham. pp. 75-97

Norton B.G. (1986) On the inherent danger of undervaluing species, in: B. G. Norton (Ed.), The Preservation of Species, Princeton University Press, Princeton, N.J. pp. 110-137.

Odum H.T. (1996) Environmental Accounting: EMERGY and Environmental Decision-Making. John Wiley & sons, New York.

Palm C., Sanchez P., Ahamed S., Awiti A. (2007) Soils: A contemporary perspective. Annual Review of Environment and Resources 32:99-129.

Pande T.N., Valentine I., Betteridge K., Mackay A.D., Horne D.J. (2000) Pasture damage and regrowth from cattle treading. Proceedings of the New Zealand Grassland Association 62:155-160.

Pangborn J. (2010) Financial budget manual Dept. of Farm Management and Rural Valuation, Lincoln University, Lincoln, New Zealand.

Parfitt R.L., Baisden W.T., Elliott A.H. (2008a) Phosphorus inputs and outputs for New Zealand in 2001 at national and regional scales. Journal of the Royal Society of New Zealand 38:37-50.

Parfitt R.L., Baisden W.T., Schipper L.A., Mackay A.D. (2008b) Nitrogen inputs and outputs for New Zealand at national and regional scales: past, present and future scenarios. Journal of the Royal Society of New Zealand 38:71-87.

Parfitt R.L., Mackay A.D., Ross D.J., Budding P.J. (2009) Effects of soil fertility on leaching losses of N, P and C in hill country. New Zealand Journal of Agricultural Research 52:69-80.

Parfitt R.L., Ross C., Schipper L.A., Claydon J.J., Baisden W.T., Arnold G. (2010) Correcting bulk density measurements made with driving hammer equipement. Geoderma 157.

Parson A.J., Rowarth J.S., Newton P.C.D. (2009) Managing pasture for animals and soil carbon. Proceedings of the New Zealand Grassland Association 71:77-84.

Parsons A.J., Chapman D.F. (2000) The principles of pasture growth and utilisation, in: A. Hopkins (Ed.), Grass, Blackwells Scientific Publications, British Grassland Society. pp. 31-89.

Patterson M.G. (1983) Estimation of the quality of energy sources and uses. Energy Policy 11:346-359.

Patterson M.G. (1998) Commensuration and theories of value in ecological economics. Ecological Economics 25:105-125.

Patterson M.G. (1999) Assessing the value of New Zealand's biodiversity. School of Resource and Environmental Planning, Massey University, Palmerston North, N.Z.

Patterson M.G. (2002) Ecological production based pricing of biosphere processes. Ecological Economics 41:457-478.

Patterson M.G. (2008) Ecological shadow prices and Contributory value: A biophysical approach to valuing marine ecosystems, in: M. G. Patterson and B. Glavovic (Eds.), Ecological Economics of the Oceans and Coasts, Edward Elgar, Cheltenham, UK - Northampton, MA, USA. pp. 140-165.

Pawson E. (2010) Economy and the environment - An economy based on the environment, http://www.teara.govt.nz/en/economy-and-the-environment/1, Te Ara - the Encyclopedia of New Zealand.

Pearce D.W. (1995) Blueprint 4 : capturing global environmental value. Earthscan, London.

Pearce D.W., Turner R.K. (1990) Economics of natural resources and the environment. Johns Hopkins University Press, Baltimore.

Pearce D.W., Barbier E.B. (2000) Blueprint for a sustainable economy. Earthscan, London.

Pearce D.W., Atkinson G., Mourato S. (2006) Cost-benefit analysis and the environment : recent developments OECD Publishing, Paris.

PGGwrightson. (2010) PGG wrightson seed, Forage Focus, http://www.pggwrightsonseeds.com/publications/, PGG wrightson.

Pidwirny M. (2010) Carbon cycle, http://www.eoearth.org/article/Carbon_cycle, Encyclopedia of Earth, Washington, D.C.

Popay I., Champion P., James T. (2010) An Illustrated Guide to Common Weeds of New Zealand. New Zealand Plant Protection Society.

Porter J., Costanza R., Sandhu H.S., Sigsgaard L., Wratten S.D. (2009) The value of producing food, energy and ES within an Agro-Ecosystem. Ambio 38:186-193.

Robinson D.A., Lebron I. (2010) On the natural capital and ecosystem services of soils. Ecological Economics 70:137-138.

Robinson D.A., Lebron I., Vereecken H. (2009) On the definition of the natural capital of soils: a framework for description, evaluation and monitoring. Soil Science Society of America Journal 73:1904-1911.

Robinson D.A., Lebron I., Ryel R.J., Jones S.B. (2010) Soil Water Repellency: A Method of Soil Moisture Sequestration in Pinyon–Juniper Woodland. Soil Science Society of America Journal 74:624-634.

Robinson D.A., Hockley N., Dominati E.J., Lebron I., Scow K.M., Reynolds B., Emmett B.A., Keith A.M., de Jonge L.W., Schjønning P., Moldrup P., Jones S.B., Tuller M. (2012) Natural Capital, Ecosystem Services and Soil Change: Why Soil Science must Embrace an Ecosystems Approach. Vadose Zone Journal 11.

Rosen M.R., Reeves R.R., Green S.R., Clothier B.E., Ironside N. (2004) Prediction of groundwater nitrate contamination after closure of an unlined sheep feed lot in New Zealand. Vadose Zone Journal 3:990-1006.

Rowarth J.S. (2010) Mackenzie country myths revealed. National Business Review 19/02/2010.

Rutledge D.T., Price R., Ross C., Hewitt A., Webb T., Briggs C. (2010) Thought for food: impacts of urbanisation trends on soil resource availability in New Zealand. Proceedings of the New Zealand Grassland Association 72:241-246.

Ryden J.C., Syers J.K. (1977) Origin of the Labile Phosphate Pool in Soils. Soil Science 123.

Saggar S., Hedley C.B. (2001) Estimating seasonal and annual carbon inputs, and root decomposition rates in a temperate pasture following field C-14 pulse-labelling. Plant and Soil 236:91-103.

Saggar S., Hedley C., Mackay A.D. (1997) Partitioning and translocation of photosynthetically fixed C-14 in grazed hill pastures. Biology and Fertility of Soils 25:152-158.

Saggar S., Hedley C.B., Tate K.R. (2003a) Methane sources and sinks in New zealand grazed pastures. New Zealand Soil News 51:6-7.

Saggar S., Tate K.R., Hedley C.B., Carran A. (2004a) Methane emissions from cattle dung and methane consumption in New Zealand grazed pasture, Proceedings of the Trace Gas Workshop, Wellington, New Zealand. pp. 102-106.

Saggar S., Giltrap D.L., Li C., Tate K.R. (2007a) Modelling nitrous oxide emissions from grazed grasslands in New Zealand. Agriculture, Ecosystems & Environment 119:205-216.

Saggar S., Hedley C.B., Giltrap D.L., Lambie S.M. (2007b) Measured and modelled estimates of nitrous oxide emission and methane consumption from a sheep-grazed pasture. Agriculture, Ecosystems & Environment 122:357-365.

Saggar S., Tate K.R., Giltrap D.L., Singh J. (2008) Soil-atmosphere exchange of nitrous oxide and methane in New Zealand terrestrial ecosystems and their mitigation options: a review. Plant and Soil 309:25-42.

Saggar S., Tate K.R., Hedley C.B., Perrott K., Loganathan P. (2001) Are soil carbon levels in our established pastures at or near steady state? New Zealand Soil News 49:73-78.

Saggar S., Bolan N.S., Bhandral R., Hedley C.B., Luo J. (2004b) A review of emissions of methane, ammonia, and nitrous oxide from animal excreta deposition and farm effluent application in grazed pastures. New Zealand Journal of Agricultural Research 47:513-544.

Saggar S., Clark H., Hedley C.B., Tate K.R., Carran A., Cosgrove G. (2003b) Methane emissions from animal dung and waste management systems, and its contribution to the national methane budget: A report prepared for the Ministry of Agriculture and Forestry, Landcare Research, Wellington.

Saggar S., Andrew R.M., Tate K.R., Hedley C.B., Rodda N.J., Townsend J.A. (2004c) Modelling nitrous oxide emissions from dairy-grazed pastures. Nutrient Cycling in Agroecosystems 68:243-255.

Samarasinghe O., Greenhalgh S. (2009) The value of inherent soil characteristics: a hedonic analysis, 53rd Conference of the Australian Agricultural and Resource Economics Society, Cairns, Australia.

Sandhu H.S., Wratten S.D., Cullen R., Case B. (2008) The future of farming: The value of ecosystem services in conventional and organic arable land: An experimental approach. Ecological Economics 64:835-848.

Sarkar S. (2005) Biodiversity and environmental philosophy: an introduction. Cambridge University Press, Cambridge, New York.

Sarmah A.K., Close M.E., Pang L.P., Lee R., Green S.R. (2005) Field study of pesticide leaching in a Himatangi sand (Manawatu) and a Kiripaka bouldery clay loam (Northland). 2. Simulation using LEACHM, HYDRUS-1D, GLEAMS, and SPASMO models. Australian Journal of Soil Research 43:471-489.

Schipper L.A., Baisden W.T., Parfitt R.L., Ross C., Claydon J.J., Arnold G. (2007) Large losses of soil C and N from soil profiles under pasture in New Zealand during the past 20 years. Global Change Biology 13:1138-1144.

Schipper L.A., Parfitt R.L., Ross C., Baisden W.T., Claydon J.J., Fraser S. (2010) Gains and losses in C and N stocks of New Zealand pasture soils depend on land use. Agriculture, Ecosystems & Environment 139:611-617.

Schloesing T. (1874) Etude sur la terre vegetale (study on vegetable earth). Annales de Chimie et Physique 5.

Schon N.L. (2010) Response of soil invertebrates to pastoral management, and their links to soil services, A thesis presented in partial fulfilment of the requirements of the degree for the Doctor of Philosophy in Ecology, Massey University, Palmerston North, New Zealand.

Schon N.L., Mackay A.D., Minor M.A. (2010a) Effects of dairy cow treading pressures and food resources on invertebrates in two soils. To be published.

Schon N.L., Mackay A.D., Minor M.A. (2010b) Influence of irrigation and fertiliser on soil invertebrates in sheep-grazed pastures. To be published.

Schon N.L., Mackay A.D., Minor M.A. (2010c) Relationships between potential food resources, soil physical conditions and invertebrate abundance and biomass in twelve pastoral soils. To be published.

Schon N.L., Mackay A.D., Yeates G.W., Minor M.A. (2010d) Separating the effects of defoliation and dairy cow treading pressure on the abundance and diversity of soil invertebrates in pastures. Applied Soil Ecology 46:209-221.

Schon N.L., Mackay A.D., Minor M.A., Yeates G.W., Hedley M.J. (2008) Soil fauna in grazed New Zealand hill country pastures at two management intensities. Applied Soil Ecology 40:218-228.

Schon N.L., Minor M.A., Yeates G.W., Mackay A.D., Hedley M.J. (2010e) A soil invertebrate threshold indicator for New Zealand pastoral soils, 19th World Congress of Soil Science, Brisbane, Australia.

Scott D.T., Baisden W.T., Davies-Colley R., Gomez B., Hicks D.M., Page M.J., Preston N.J., Trustrum N.A., Tate K.R., Woods R.A. (2006) Localized erosion affects national carbon budget. Geophysical Research Letters 33:4.

Scott N.A., Tate K.R., Ford-Robertson J., Giltrap D.J., Smith C.T. (1997) Soil carbon storage in plantation forests and pastures: land-use change implications, 5th International CO2 Conference, Cairns, Australia. pp. 326-335.

Scott N.A., Tate K.R., Giltrap D.J., Tattersall Smith C., Wilde H.R., Newsome P.J.F., Davis M.R. (2002) Monitoring land-use change effects on soil carbon in New Zealand: quantifying baseline soil carbon stocks. Environmental Pollution 116:S167-S186.

Scotter D.R., Clothier B.E., Turner M.A. (1979) The soil water balance in a gragiaqualf and its effect on pasture growth in central New Zealand. Australian Journal of Soil Research 17:455-465.

SCS. (1972) Hydrology, Soil Conservation Service National Engineering Handbook, U.S. Govt Printing Office: Washington D.C.

Shepherd T.G. (2000) Visual Soil Assessment. Volume 1. Field guide for cropping and pastoral grazing on flat to rolling country, Horizons & Landcare Research, Palmerston North, New Zealand. pp. 84.

Shiferaw B., Freeman H.A., Navrud S. (2005) Valuation methods and approaches for assessing natural resource management impacts, in: B. Shiferaw, et al. (Eds.), Natural resources management in agriculture : methods for assessing economic and environmental impacts, CAB International, Wallingford, UK. pp. 19-52.

Singleton P.L., Addison B. (1999) Effects of cattle treading on physical properties of three soils used for dairy farming in the Waikato, North Island, New Zealand. Australian Journal of Soil Research 37:891-902.

Singleton P.L., Boyes M., Addison B. (2000) Effect of treading by dairy cattle on topsoil physical conditions for six contrasting soil types in Waikato and Northland, New Zealand, with implications for monitoring. New Zealand Journal of Agricultural Research 43:559-567.

Sparling G., Schipper L. (2004) Soil quality monitoring in New Zealand: trends and issues arising from a broad-scale survey. Agriculture Ecosystems & Environment 104:545-552.

Sparling G.P., Schipper L.A., Russell J.M. (2001) Changes in soil properties after application of dairy factory effluent to New Zealand volcanic ash and pumice soils. Australian Journal of Soil Research 39:505-518.

Sparling G.P., Wheeler D., Vesely E.T., Schipper L.A. (2006) What is soil organic matter worth? Journal of Environmental Quality 35:548-557.

Springett J.A. (1992) Distribution of lumbricid earthworms in New Zealand. Soil Biology and Biochemistry 24:1377-1381.

Springett J.A., Syers J.K. (1984) Effect of pH and calcium content of soil on earthworm cast production in the laboratory. Soil Biology and Biochemistry 16:185-189.

StatisticNZ. (2011) Producer price index output series, http://www.stats.govt.nz/browse_for_stats/economic_indicators/prices_indexes/producers-price-index-info-releases.aspx, Statistic New Zealand.

Stern N. (2007) The Economics of Climate Change: The Stern Review Cambridge University Press, Cambridge.

Stevenson F.J. (1999) Cycles of soil : carbon, nitrogen, phosphorus, sulfur, micronutrients. Wiley, New York.

Straton A. (2006) A complex systems approach to the value of ecological resources. Ecological Economics 56:402-411.

Stromberg B.E. (1997) Environmental factors influencing transmission. Veterinary Parasitology 72:247-256.

Sukias J., Tanner C. (2011) Surface flow constructed wetlands as a drainage management tool – Long term performance, in: L. D. Currie and C. L. Christensen (Eds.), Adding to the Knowledge Base for the Nutrient Manager, Fertilizer & Lime Research Centre, Occasional Report No. 24, Massey University, Palmerston North, New Zealand.

Swift M.J., Heal O.W., Anderson J.M. (1979) Decomposition in Terrestrial Ecosystems. University of California Press, Berkeley, California.

Swift M.J., Izac A.M.N., van Noordwijk M. (2004) Biodiversity and ecosystem services in agricultural landscapes: are we asking the right questions? Agriculture, Ecosystems & Environment 104:113-134.

Swinton S.M., Lupi F., Robertson G.P., Landis D.A. (2006) Ecosystem services from agriculture: looking beyond the usual suspects. American Journal of Agricultural Economics 88:1160-1166.

Swinton S.M., Lupi F., Robertson G.P., Hamilton S.K. (2007) Ecosystem services and agriculture: Cultivating agricultural ecosystems for diverse benefits. Ecological Economics 64:245-252.

Syers J.K., Sharpley A.N., Keeney D.R. (1979) Cycling of nitrogen by surface-casting earthworms in a pasture ecosystem. Soil Biology and Biochemistry 11:181-185

Tate K.R., Saggar S., Baisden T. (2005a) Soils offer a chance to "farm the atmosphere" for carbon credits – a response. New Zealand Soil News 53:120.

Tate K.R., Giltrap D.J., Claydon J.J., Newsome P.F., Atkinson I.A.E., Taylor M.D., Lee R. (1997) Organic carbon stocks in New Zealand's terrestrial ecosystems. Journal of the Royal Society of New Zealand 27:315-335.

Tate K.R., Ross D.J., Saggar S., Hedley C.B., Dando J., Singh B.K., Lambie S.M. (2007) Methane uptake in soils from Pinus radiata plantations, a reverting shrubland and adjacent pastures: Effects of land-use change, and soil texture, water and mineral nitrogen. Soil Biology & Biochemistry 39:1437-1449.

Tate K.R., Scott N.A., Saggar S., Giltrap D.J., Baisden W.T., Newsome P.F., Trotter C.M., Wilde R.H. (2003) Land-use change alters New Zealand's terrestrial carbon budget: Uncertainties associated with estimates of soil carbon change between 1990-2000. Tellus B 55:364-377.

Tate K.R., Wilde R.H., Giltrap D.J., Baisden W.T., Saggar S., Trustrum N.A., Scott N.A., Barton J.R. (2005b) Soil organic carbon stocks and flows in New Zealand: System development, measurement and modelling. Canadian Journal of Soil Science 85:481-489.

TeAra. (2010) New Zealand best soils, http://www.teara.govt.nz/en/soils/9/1, Landcare research.

Tonkin P.J. (2010) Soil investigation, http://www.TeAra.govt.nz/en/soil-investigation/2/4, Te Ara - the Encyclopedia of New Zealand.

TRC. (2006) Managing Dairy Farm Effluent (Dairying and Environment Committee Manual), Taranaki Regional Council.

Trotter C.M., Tate K.R., Saggar S., Scott N.A., Sutherland M.A. (2004) A multi-scale analysis of a national terrestrial carbon budget and the effects of land-use change, in: M. Shiyomi, et al. (Eds.), Global Environmental Change in the Ocean and on Land, TERRAPUB, Tokyo. pp. 311-342.

Turner R.K., Daily G.C. (2008) The ecosystem services framework and natural capital conservation. Environmental & Resource Economics 39:25-35.

Ulanowicz R.E. (1991) Contributory values of ecosystem resources, in: R. Costanza (Ed.), Ecological Economics: The Science and Management of Sustainability, Columbia University Press, New York. pp. 253-268.

USDA. (1975) Soil taxonomy: a basic system of soil classification for making and interpreting soil surveys U.S. Department of Agriculture (USDA), Soil conservation service, Washington DC.

van Genuchten M.T. (1980) A Closed-form Equation for Predicting the Hydraulic Conductivity of Unsaturated Soils. Soil Science Society of America Journal 44:892-898.

Vesely E.T. (2006) Economic valuation of soil quality, Landcare Research Contract Report.

Waghorn T.S., Leathwick D.M., Chen L.Y., Gray R.A.J., Skipp R.A. (2002) Influence of nematophagous fungi, earthworms and dung burial on development of the free-living stages of Ostertagia (Teladorsagia) circumcincta in New Zealand. Veterinary Parasitology 104:119-129.

Wahba M.A., Bridwell L.G. (1976) Maslow reconsidered: A review of research on the need hierarchy theory. Organizational Behavior and Human Performance 15:212-240.

Wall D.H., Bardgett R.D., Covich A.P., Snelgrove P.V.R. (2004) The need for understanding how biodiversity and ecosystem functioning affect ecosystem services in soils and sediments, in: D. H. Wall (Ed.), Sustaining biodiversity and ecosystem services in soils and sediments, Island Press, Washington. pp. 1-12.

Wallace K.J. (2007) Classification of ecosystem services: Problems and solutions. Biological Conservation 139:235-246.

Wang H.L., Magesan G.N., Bolan N.S. (2004) An overview of the environmental effects of land application of farm effluents. New Zealand Journal of Agricultural Research 47:389-403.

Wang X., Bennett J., Xie C., Zhang Z., Liang D. (2007) Estimating non-market environmental benefits of the Conversion of Cropland to Forest and Grassland Program: A choice modeling approach. Ecological Economics 63:114-125.

Way J.T. (1850) On the power of soils to absorb manure. Journal of the Royal Agricultural Society of England 11:311-379.

Webb T.H., Purves A.M. (1983) Influence of soil type on yield of autumn-sown wheat and oats in Canterbury in a dry year. NZ Journal of Experimental Agriculture 11:289-294.

Webb T.H., Wilson A.D. (1994) Classification of land according to its versatility for orchard crop production. Landcare Research Science Series 8:31.

Webb T.H., Wilson A.D. (1995) A manual of land characteristics for evaluation of rural land. Landcare Research Science Series 10:32.

Weber J.L. (2007) Accounting for soil in the SEEA, European Environment Agency, Rome.

Westman W.E. (1977) How much are nature's services worth? Science 197:960-964.

Whitehead J.C. (2006) A comparison of contingent valuation method and random utility model estimates of the value of avoiding reductions in king mackerel bag limits. Applied Economics 38:1725 - 1735

Wilcock R.J., van Assema G.B., Bellingham M.A., Ovenden R., Müller K. (2011) Benefits of wetlands in farming landscapes – Toenepi, in: L. D. Currie and C. L. Christensen (Eds.), Adding to the Knowledge Base for the Nutrient Manager, Fertilizer & Lime Research Centre, Occasional Report No. 24, Massey University, Palmerston North, New Zealand.

Williams J.R. (1991) Runoff and water erosion, in: R. J. Hanks and J. T. Ritchie (Eds.), Modelling plant and soil systems.Agronomy Monograph 31, American Society of Agronomy, Madison, WI, USA. pp. 439-455.

Wilson M., Liu S. (2008) Non-market value of ecosystem services provided by coastal and nearshore marine systems, in: M. G. Patterson and B. Glavovic (Eds.), Ecological Economics of the Oceans and Coasts, Edward Elgar, Cheltenham, UK - Northampton, MA, USA. pp. 119-139.

Wilson M.A., Howarth R.B. (2002) Discourse-based valuation of ecosystem services: establishing fair outcomes through group deliberation. Ecological Economics 41:431-443.

Wischmeier W.H., Smith D.D. (1978) Predicting rainfall erosion losses: a guide to conservation planning, Agriculture Handbook No. 537, US Department of Agriculture Science and Education Administration, Washington, DC, USA. pp. 163

Yaalon D.H. (1997) History of soil science in context: international perspective, in: D. H. Yaalon and S. Berkowicz (Eds.), History of soil science: international perspectives, Catena Verlag, Reiskirchen, Germany. pp. 1-15.

Yeates G.W. (1999) Effects of plants on nematode community structure. Annual Review of Phytopathology 37:127-149.

Yeates G.W., Pattison A.B. (2006) Moving up within the Food Web: Protozoa and Nematodes, in: N. Uphoff, et al. (Eds.), Biological Approaches to Sustainable Soil Systems, CRC Press, Boca Raton. pp. 149-162.

Yu B., Rosewell C.J. (1996) An assessment of a daily rainfall erosivity model for New South Wales. Australian Journal of Soil Research 34:139-152.

Zhang W., Ricketts T.H., Kremen C., Carney K., Swinton S.M. (2007) Ecosystem services and dis-services to agriculture. Ecological Economics 64:253-260.

APPENDICES

Appendix A

Examples of SPASMO Input Files

This appendix presents examples of input files needed in the SPASMO model.

Climate data:

Climate data for all simulations were obtained from NIWA's (National Institute of Water & Atmospheric Research) CLIFLO database (NIWA, 2010) using records for 37 years (1972-2009) from Hamilton City weather station (Latitude: -37.825, Longitude: 175.275). CLIFLO is the web system that provides access to New Zealand's National Climate Database.

The data used in SPASMO includes daily values of incoming global radiation, potential evapotranspiration, maximum and minimum air temperature, wind speed and rainfall (Table A.2).

Table A.1: Notes on interpreting Virtual Climate Station Data (Table A.2).

Measure	Description
MSLPress	Mean sea level pressure at 9am local day (hPa)
PET	24-hour Penman potential evapotranspiration total from 9am local day (mm)
Rain	24-hour rainfall total from 9am local day (mm)
RH	Relative humidity at 9am local day (%)
SoilM	24 hour soil moisture index total from 9am local day (mm, positive = runoff; negative = soil moisture deficit)
TempEarth10cm	Earth temperature at 10cm depth at 9am local day (degC)
Radn	24 hour global solar radiation total from midnight local day (MJ/m^2)
Tmax	Maximum temperature over 24 hours from 9am local day (degC)
Tmin	Minimum temperature over 24 hours to 9am local day (degC)
VapPress	Vapour pressure at 9am local day (hPa)
WindSpeed	Average wind speed at 10m above ground level over 24 hours from midnight local day (m/s)

Table A.2: Climate input file for the Ruakura station in the Waikato, New Zealand.

Station	Date	MSLPress (hPa)	PET (mm)	Rain (mm)	RH (%)	SoilM (mm)	TempEarth10cm (°C)	Radn (MJ/m2)	Tmax (°C)	Tmin (°C)	VapPress (hPa)	WindSpeed (m/s)
30277	1/01/1972	1012	3.3	6.8	86	-88.1	20.8	20.7	25.5	17.9	22.2	-
30277	2/01/1972	1011.3	3.2	38.6	77	-62.7	20.8	13.7	25.1	17.9	19.9	-
30277	3/01/1972	1013.5	4.5	0	91	-63.3	20	22.8	25.3	17.1	21	-
30277	4/01/1972	1016.9	4.6	0	80	-66.9	19.6	29.3	24.5	10.5	16.9	-
30277	5/01/1972	1013.7	5	0	65	-65.3	19	28.8	22	9.2	13.7	-
30277	6/01/1972	1008.3	3.1	9.8	72	-59.4	18.8	20.1	21.3	8.9	14.3	-
30277	7/01/1972	1004.5	3.5	1.4	81	-62	17.6	24.9	19.8	11	13.8	-
30277	8/01/1972	1011.7	4.2	0	56	-66.5	16.6	22.8	19.5	7.7	9.6	-
30277	9/01/1972	1015.6	3.7	0	80	-70.5	16.8	24.6	21.3	8.3	14.8	-
30277	10/01/1972	1016.2	2.5	0	75	-74.1	17.9	19.6	21.7	10	15.4	-
30277	11/01/1972	1014.5	3.3	0	73	-77.5	18.3	19.6	22.1	14.1	15.6	-
30277	12/01/1972	1016.3	3.8	1.5	82	-79.5	19.6	15.9	24.4	16.3	19.3	-
30277	13/01/1972	1016.8	3.1	1.1	81	-81.9	20.1	15	24.1	16.4	20.1	-
30277	14/01/1972	1014.5	2.1	0	80	-85.3	20	13.4	22.2	15.4	18.5	-
30277	15/01/1972	1011.7	5.8	0	66	-88.8	19.2	30.6	20.8	14.1	13.6	-
30277	16/01/1972	1014.8	4.7	0	64	-92.2	18.2	26	22.9	8.8	11.9	-
30277	17/01/1972	1014.1	4.9	0	70	-96.1	18.9	29.3	21.8	10.6	13.7	-
30277	18/01/1972	1014.2	2.9	0	79	-99.3	19.5	21.9	21.9	10.1	16.6	-
30277	19/01/1972	1016	3.4	0	78	-101.4	20.2	20.1	23	12.9	16.7	-
30277	20/01/1972	1014.8	5	0	75	-104.8	19.1	28.8	25.3	9.2	15.7	-
30277	21/01/1972	1015.4	3.5	0	70	-107.4	20.6	17.2	22.6	14.3	14.8	-
...	...	...	...	...	...	...	...	...	...	...	...	...
30277	31/12/2009	1017.5	5.5	0	60	-97.5	17.7	28	20	9.4	10.6	3.8

Soil data:

The soil data is from the National Soils Database from Landcare Research (LandcareResearch, 2010) and includes values for every 10 cm of the profile for inherent properties including stone content, anion storage capacity, hydraulic conductivity (Ksat), clay and sand fractions, parameters to fit Van Genuchten water release curves (van Genuchten, 1980) and manageable properties including bulk density, total C and total N (Table A.3).

Table A.3: Soil input file for a Horotiu silt loam.

Top of the layer	Bottom of the layer	Texture	Bulk density	Stone content	Parameters to fit Van Genuchten water release curves					Total C	Total N	ASC	Ksat	Clay fraction	Sand fraction
(cm)	(cm)		(g/cm^3)	(%)	thetas	thetan	alpha	n	m	(%)	(%)	(%)	(mm/day)	(%)	(%)
0	10	silt loam	0.87	0	66.73	0	0.379	1.153	0.133	8.2	0.77	85	86	17.6	33.8
10	20	silt loam	0.84	0	67.88	0	1.209	1.161	0.139	5.5	0.55	91	86	16.4	33.9
20	30	silt loam	0.81	0	69.04	0	1.269	1.167	0.143	3.3	0.32	96	86	16.4	35.9
30	40	silt loam	0.83	0	68.08	0	7.158	1.109	0.098	1.7	0.15	96	86	29	46.3
40	50	silt loam	0.83	0	68.08	0	7.158	1.109	0.098	1.7	0.15	96	86	29	46.3
50	60	silt loam	0.83	0	68.08	0	7.158	1.109	0.098	1.7	0.15	96	86	29	46.3
60	70	silt loam	0.82	0	68.46	0	13.032	1.092	0.084	1.7	0.14	96	86	18.4	53.5
70	80	silt loam	0.82	0	68.65	0	18.891	1.086	0.079	1.2	0.12	94	86	32.6	46
80	90	silt loam	0.82	0	68.65	0	18.891	1.086	0.079	1.2	0.12	94	86	32.6	46
90	100	sand	0.82	0	68.65	0	14.466	1.09	0.083	0.7	0.07	76	2870	9.5	63.8
100	110	sand	0.82	0	68.65	0	14.466	1.09	0.083	0.7	0.07	76	2870	9.5	63.8
110	120	coarse sand	0.89	11	65.77	0	6.441	1.099	0.09	0.1	0.01	18	2870	1	93.2
120	130	coarse sand	0.89	11	65.77	0	6.441	1.099	0.09	0.1	0.01	18	2870	1	93.2

Appendix B

P sorption Function for P Leaching

Originally, the model parameters describing P partitioning in soil were determined using isotherm experiments performed on soil samples that were previously washed and dried at room temperature. The work was carried out by Landcare Research and the raw data were provided to Plant and Food Research (Hugh Wilde, pers. comm.) in order to calculate the appropriate transport properties for P.

The dataset used to build the isotherms combined soils samples from Gladstone and Tararua Roads near Levin, where the soil is a Kawhatau stony silt loam, and soil samples from Masterton. The soil samples (3 g oven-dry weight) were placed in a 30 ml solution with different quantities of P added (100, 250, 500, 1000 or 2000 mg/kg) and shaken for 16 hours. The final equilibrium P concentration of the solution (C), as well as the P sorbed on the soil (q) was determined. The data from this isotherm studies was fitted to a Langmuir adsorption-isotherm equation of the form:

$(q/Q) = (bC / (1+bC))$

Where C is the equilibrium adsorbate concentration [mg/L], q is the mass of adsorbate per mass of adsorbent at equilibrium [µg/g] (P sorbed on soil particles), Q is the maximum mass adsorbed at saturation conditions per mass unit of adsorbent [µg/g], and b is an empirical constant with units of inverse of concentration [L/mg].

In order to model the P-transport through other soils, the parameters b and Q needed to be determined. Since the New Zealand Soil Database (NZSDB) holds only records of P-retention (an inherent property), a 'pedo-transfer' approach was used to determine appropriate parameter values for the modelling using the combined dataset from Levin and Masterton.

A simple relationship was derived to describe P sorption at saturation (Q, mg/kg) as a function of P retention (P_R, %) (Fig.B.1). The open symbols clustered on the left hand side of the graph (Fig.B.1) represent low P-retention soils found around Masterton. The solid symbols clustered on the right hand side of the graph represent moderate to high P-retention soils from the proposed subdivision site near Levin. The solid line is represented by the equation:

$$Q = Q_N + (Q_m - Q_n) * (1 - \frac{1}{\left(\frac{P_R}{P_H}\right)^X})$$

Where P_H is the value of P-retention such that Q is at half the maximum value of P sorption saturation, X is a fitting parameter that defines the curvature of the 'S-shaped' relationship, and the subscripts m and n represent maximum and minimum values of Q, respectively.

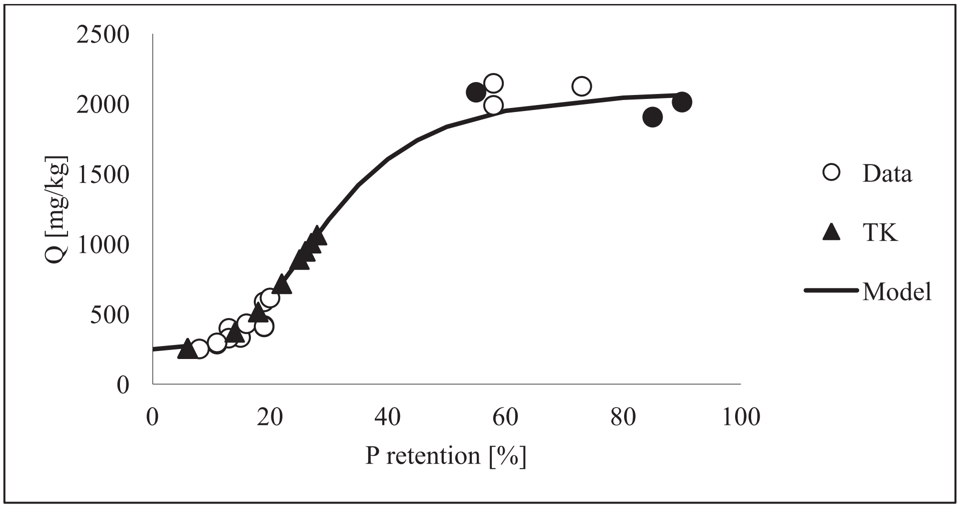

Figure B.1: Simple relationship describing P sorption at saturation (Q, mg/kg) as a function of P retention (%). The triangles are the modelled data for the Te Kowhai silt loam use in this study.

Then, an exponential function is used to represent the relationship between the product Q times d (depth, m) and the fitting parameter b (L mg-1) that describes the Langmuir isotherm for P sorption in soils (Fig.B.2).

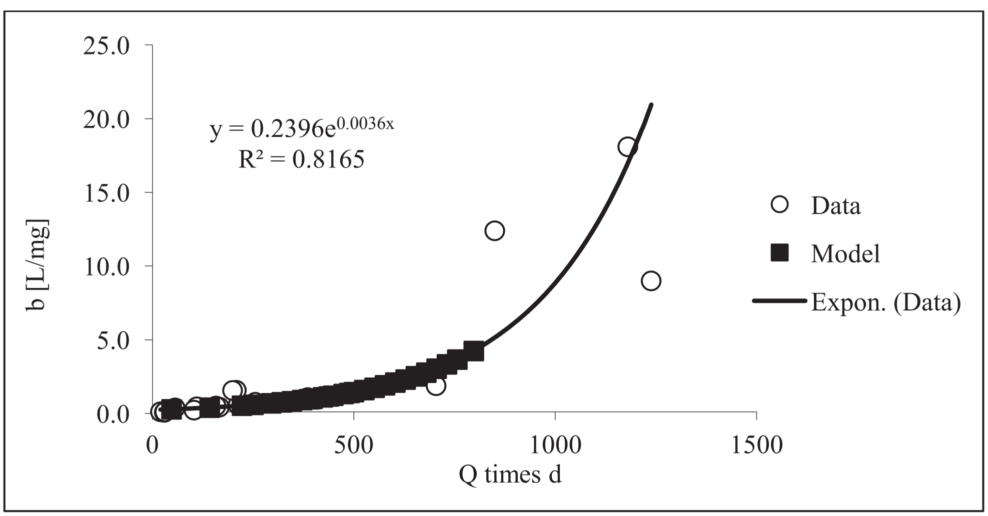

Figure B.2: Exponential function between Qmax (P adsorbed at saturation) times d (depth, m) and the fitting parameter b (L/mg) of the Langmuir isotherm.

Once the parameters Q and b are determined from P solution concentration, P sorbed (q) was calculated from the Langmuir isotherm.

Fig. B.3 presents the data from the isotherm studies for one soil sample from Levin and the fitted Langmuir adsorption-isotherm with the calculated Q and b parameters. The modelling efficiency was 98%.

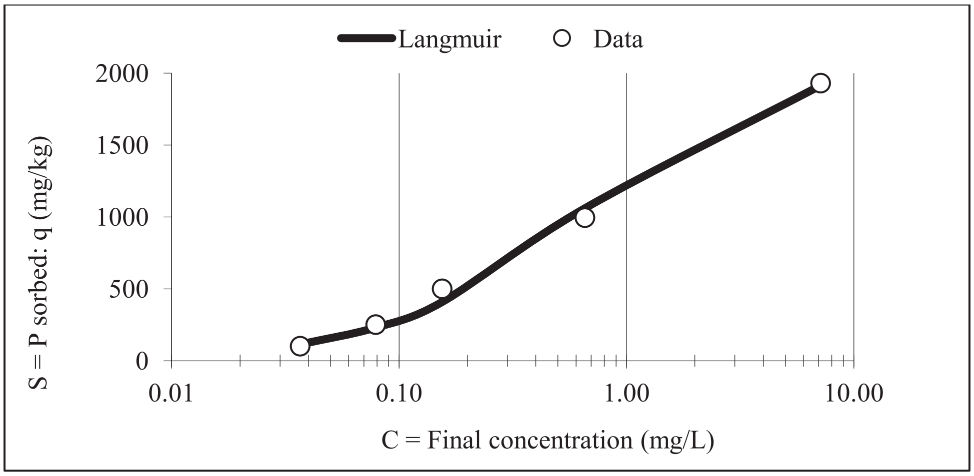

Figure B.3: Langmuir isotherm for P retention from the A horizon (0-20 cm deep) of the soil profile at Tararua Road, Levin.

This routine then enables the SPASMO model to calculate how much P is in solution, how much is sorbed, how much can be released and how much is readily available for plants (supporting processes behind P supply to plants).

Appendix C

Examples of SPASMO Outputs

This appendix presents two examples of the outputs of the SPASMO model used.

<u>Water outputs:</u>

SPASMO outputs daily water related data including rainfall, irrigation, evapotranspiration, drainage, runoff and soil water content at 10cm. These measures are all expressed in mm (Table C.1).

<u>Macroporosity dynamics outputs:</u>

SPASMO also outputs data related to macroporosity dynamics including the time since the last grazing event on this paddock (Tlastgr), the soil water content (SWC) at the time of grazing, the percentage of macropores lost during the grazing event (Mploss) and the resulting macroporosity after the grazing event (Mpore_s) (Table C.2).

Table C.1: Water cycle outputs from SPASMO:

Date	Day	Day Of Year	Rainfall	Irrigation	Evapotranspiration	Drainage	Runoff	Soil Water Content
...	...	...	...	...	...	...	...	...
20/06/1981	3456	171	0	0	0.7	0.16	0	55.91
21/06/1981	3457	172	0	0	0.4	0.33	0	54.26
22/06/1981	3458	173	0	0	0.7	0.58	0	53.28
23/06/1981	3459	174	0	0	0.5	0.91	0	52.85
24/06/1981	3460	175	0	0	0.4	1.32	0	52.52
25/06/1981	3461	176	0	0	0.5	1.79	0	52.08
26/06/1981	3462	177	6.8	0	0.9	2.28	0	55.32
27/06/1981	3463	178	3.3	0	0	2.76	0	55.73
28/06/1981	3464	179	23.4	0	0.3	3.22	3.55	60.21
29/06/1981	3465	180	2.1	0	0.8	3.62	0	60.28
30/06/1981	3466	181	2.8	0	0.4	3.96	0	60.35
1/07/1981	3467	182	0	0	0.6	4.23	0	56.38
2/07/1981	3468	183	7.8	0	0.6	4.43	0	60.49
3/07/1981	3469	184	6.3	0	0	4.58	0	60.56
4/07/1981	3470	185	5.6	0	0.5	4.67	0	60.63
5/07/1981	3471	186	8.6	0	0.6	4.71	0	60.69
6/07/1981	3472	187	4.1	0	0.6	4.73	0	60.76
7/07/1981	3473	188	10.4	0	1.1	4.72	0.03	60.83
8/07/1981	3474	189	25.4	0	1	4.69	4.05	60.89
9/07/1981	3475	190	3.6	0	0.4	4.65	0	60.96
...	...	...	...	...	...	...	...	...
31/12/2009	13870	365	0	0	5.32	-0.05	0	25.3

Table C.2: Macroporosity outputs from SPASMO:

Date	**Day**	**DOY**	**Tlastgr (days)**	**SWC (mm)**	**Mp loss (%)**	**Mp (%)**
...	...	...	...	...	...	...
19/05/2003	11454	139	79	26.63	0	11
20/05/2003	11455	140	80	30.64	0	11.03
21/05/2003	11456	141	0	32.16	11.87	9.72
22/05/2003	11457	142	1	36.42	0	9.77
23/05/2003	11458	143	2	35.27	0	9.81
24/05/2003	11459	144	3	41.26	0	9.86
25/05/2003	11460	145	4	43.06	0	9.9
26/05/2003	11461	146	5	40.9	0	9.94
27/05/2003	11462	147	6	39.55	0	9.98
28/05/2003	11463	148	7	38.65	0	10.02
29/05/2003	11464	149	8	37.66	0	10.06
30/05/2003	11465	150	9	38.41	0	10.1
31/05/2003	11466	151	10	37.77	0	10.14
1/06/2003	11467	152	11	36.8	0	10.18
2/06/2003	11468	153	12	36.03	0	10.22
3/06/2003	11469	154	13	35.37	0	10.26
4/06/2003	11470	155	14	34.67	0	10.3
5/06/2003	11471	156	15	34.27	0	10.34
6/06/2003	11472	157	16	42.6	0	10.38
7/06/2003	11473	158	17	52.79	0	10.41
8/06/2003	11474	159	18	45.46	0	10.45
9/06/2003	11475	160	19	56.99	0	10.49
10/06/2003	11476	161	0	55.45	26.38	7.72
11/06/2003	11477	162	1	53.23	0	7.79
...	...	...	...	...	...	...
31/12/09	13870	365	38	25.3	0	9.4

Appendix D

Determination of Sustainable Yield

To estimate the component of yield derived from the soil natural capital, the influence of P fertilisers need to be subtracted from the total pasture yield modelled using SPASMO.

To assess the contribution of P fertilisers to pasture production, calibration curves are used for recommendation on optimum Olsen P levels for maximum pasture production. These calibration curves (relationship between Olsen P and relative yield (RY)) (Fig.D.1) are based on empirical relationship derived from the >3000+ data entries from field studies into the P data base of AgResearch in the last 40+ years (Morton and Roberts, 2001).

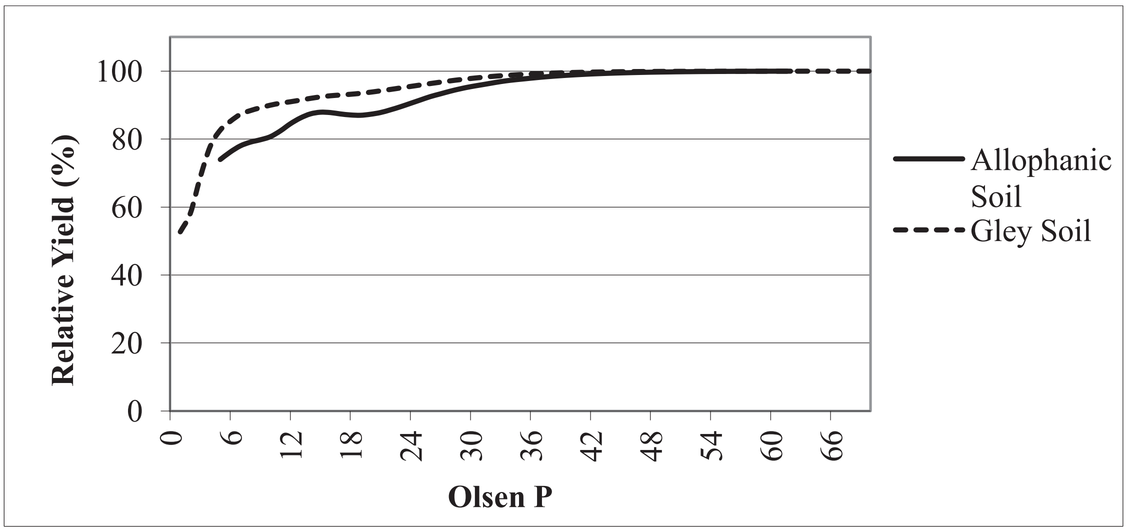

Figure D.1: Calibration curves between Olsen P and relative yield for an Allophanic and a Gley Soil.

The level of pasture production that could be sustained if no fertiliser was applied was considered to correspond to an Olsen P of 4 (Parfitt et al., 2009).

The corresponding relative yield associated with this Olsen P was then calculated from the equations of trend lines applied to the first part of the calibration curves (Fig. D.2 and D.3).

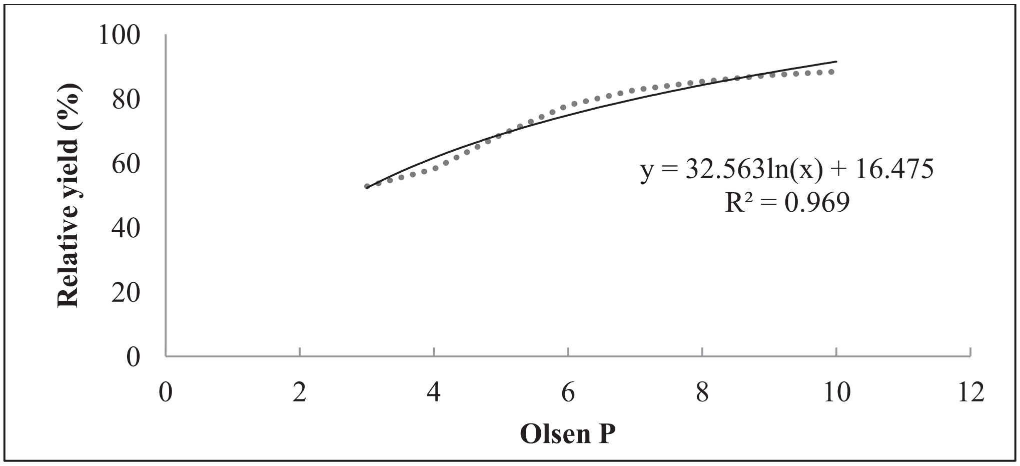

Figure D.2: Calibration curve for an Allophanic Soil.

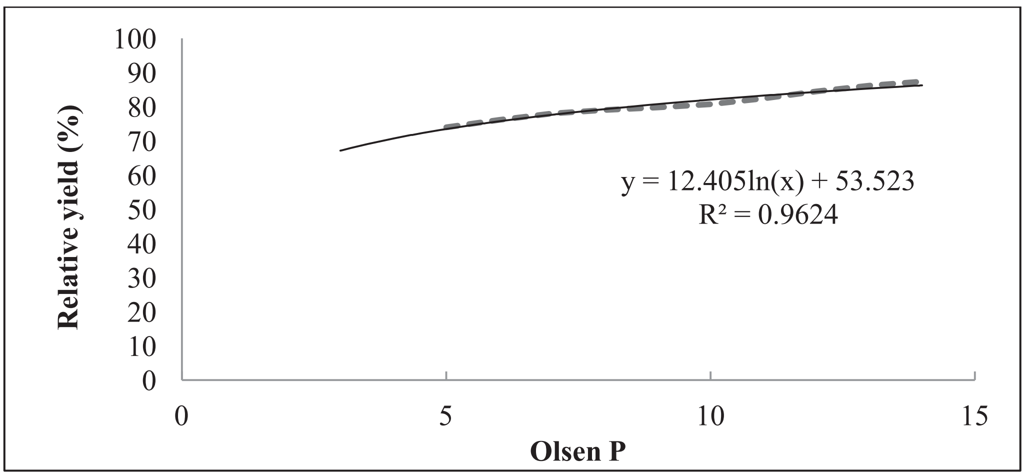

Figure D.3: Calibration curve for a Gley Soil.

The calculation show that a Gley Soil with an Olsen P of 4 can only support a RY of about 60% , whereas an Allophanic Soil with an Olsen P of 4 can support a RY of about 70% (Table D.1).

These values were used to estimate the part of the yield coming from soil natural capital with no influence of P fertilisers.

Table D.1: Calculated minimum relative yield by Olsen P.

Olsen P	Volcanic soil	Sedimentary soil
2	62.1	39.0
4	**70.7**	**61.6**
6	75.7	74.8
8	79.3	84.2
10	82.1	91.5
12	84.3	97.4

Appendix E

Multiplier Effect for the Waikato economy

to determine the impact of the value of the provision of food at the farm scale on the wider Waikato economy, input-output multipliers were used. The input-output multipliers used are for the Waikato Region for 2006-07 (Pers. Comm. Dr Garry McDonald, Market Economics Ltd). For the base case scenario, the value of the provision of food was $4,155/ha/yr (NZ 2011). Since the multipliers used correspond to 2007 NZ$, gross outputs need to be also converted to 2007 NZ$. All data refers to the dairy farming industry. The producer price index between 2007 and march 2010 (latest data available) is 0.6948 (StatisticNZ, 2011). Therefore, $4,155/ha/yr (NZ$ 2011) = 4155 * 0.6948 = $2,887/ha/yr (NZ$ 2007).

To calculate the total impact of the provision of food on the Waikato economy, the backward and forward linkages of the multiplier effect need to be considered:

<u>Backward Linkages</u>: They include direct, and backwards indirect and induced effects.
BL = Gross output in 2007 NZ$ * impact ratio for value added * Backward Linkage Type II value added multipliers = 2887 * 0.44 * 2.17 = $2,782 (NZ 2007)

<u>Forward Linkages</u>: They include direct and forward indirect and induced effects.
FL = Gross output in 2007 NZ$ * impact ratio for value added * Forward Linkage Type II value added multipliers = 2887*0.44*1.50 = $1929 (NZ 2007)

To calculate the total impact of the provision of food on the Waikato economy, the backward and forward linkages of the multiplier effect need to be summed. The direct value added is subtracted from the sum to avoid double accounting.
Total impact = Backward Linkage + Forward Linkage - (Gross output in 2007 NZ$ * impact ratio for value added) = 2782+1929-1284 = $3,426 (NZ 2007).

The production of $2,887/ha/yr (NZ$ 2007) of milk solids from soil natural capital stocks is actually worth $3,426/ha/yr (NZ 2007) to the Waikato economy, which is 18.7 % more than the market value only.

Appendix F

Data and Methods for the Economic Valuation of Soil Services

Data and methods used to calculate the costs of infrastructures and mitigation functions used to value soil services are detailed in this appendix.

F.1 Annualisation of capital costs:

Techniques using revealed preferences were chosen to value soil services. They include market prices, productivity change, defensive expenditure, replacement cost and provision cost. Information on the market value (present value) of human made infrastructures that are used commonly by farmers to deal with a lack of natural capital and low ecosystem services provision was used to implement revealed preferences techniques and value some of the soil services.

When valuing ecosystem services, one wants to put a value on the flows coming from natural capital stocks and not the stocks in themselves. This is why when using the cost of infrastructure to value ecosystem services, one need to annualise these costs in order to determine the annual flows of value that can be attributed to flows of ecosystem services. Such technique is in line with good accounting and economic theory.

A discount rate of 10% was chosen to annualise infrastructure costs because it is the value the most commonly used in the literature. Results using a discount rate of 3%, such as the one used in the Stern report (Stern, 2007), are also presented to allow discussion.

The present value of an annuity (PVA = sum of present values of all annuities) can be calculated with the formula below (Holmes, 1998), where A is the value of the annuity, r is the discount rate and n is the number of years the annuity is received:

$$PVA = A * [\frac{1}{r} - \frac{1}{(1+r)^n}]$$

The annuity (A) to which corresponds the capital cost of an infrastructure (CC), or in other terms the present value of that infrastructure (PVA), is then calculated as follow:

$$A = \frac{CC}{[\frac{1}{r} - \frac{1}{r(1+r)^n}]}$$

Where A is the value of the annuity, CC is the capital cost of the infrastructure, r is the discount rate and n is the number of years the annuity is received.

F.2 Standoff pad costs:

The type of pad considered in this study is a wintering or standoff pad, a specially built area constructed where animals are withheld from grazing during wet periods to minimise damage to pastures. Supplementary feeds are brought to the animals on the pad. As the herd may spend several months on the pad the cows require an area to lie down on, as well as additional space for feeding. These pads are constructed of free-draining material such as sawdust, bark, woodchips, lime or soft metal (rock) mix (Dexcel, 2005a).

The costs of construction and maintenance of a wintering pad were used to value the provision of support to animals and the filtering of nutrients.

These costs were calculated from data gathered in the literature (Dexcel, 2005a; Dexcel, 2005b). Case studies on New Zealand dairy farms provided average data for the pad surface needed per cow in m^2/cow, construction costs in $/$m^2$ and maintenance costs in $/cow/day of use of the pad (Table F.1). Construction costs include earthworks, surface material and equipment (drainage system, gates, feed bins, and lanes). Maintenance costs include surface scrapping and replacement, drainage system maintenance and labour.

For each scenario and each year, the herd size, stocking rate (cows/ha) and number of days of use of the pad were then used to calculate the total costs of construction and maintenance of a wintering pad in $/ha/yr (Table F.2).

The following data was chosen as parameters from the case studies (Table F.1) to calculate the total costs of construction and maintenance of a wintering pad:

- Surface per cow (m^2/cow) = 6
- Cost of construction ($/$m^2$) = 24.6
- Cost of maintenance ($/cow/day) = 0.14

Table F.1: Data on costs of a pad from case studies dairy farms (Dexcel, 2005b)

Farm	Cows number	SR (cows /ha)	Surface (ha)	Pad type	Pad size (m^2)	Nb of cows on pad	Surface per cow (m^2/ cow)	Annual use (days/ yr)	Total cost of pad construction ($)	Cost of construction ($/$m^2$)	Cost of construction ($/cow)	Total cost of pad maintenance ($/yr)	Cost maintenance ($/cow/yr)	Cost maintenance ($/cow/day)
20	326	3	108.7	Standoff pad	1050	120	8.8	90	21285	20.3	65.3	5360	16.4	0.50
13	240	2.7	88.89	Standoff pad	800	215	3.7	98	6000	7.5	25.0	4500	18.8	0.21
4	320	3.4	94.12	Sawdust standoff	2225	320	7.0	100	1400	0.6	4.4	800	2.5	0.03
2	380	2.23	170.4	Concrete feed pad	1300	380	3.4	120	70000	53.8	184.2	500	1.3	0.01
19	490	2.4	204.2	Standoff pad	1980	200	9.9	130	50000	25.3	102.0	8500	17.3	0.33
19	490	2.4	204.2	Standoff pad	3225	220	14.7	130	70000	21.7	142.9	14250	29.1	0.50
8	800	3.8	210.5	Concrete feed pad	2320	594.9	3.9	150	100000	43.1	125.0	500	0.6	0.01
14	750	2.5	300	Concrete feed pad	1100	450	2.4	150	80000	72.7	106.7	2000	2.7	0.03
1	308	2.12	145.3	Concrete feed pad	1508	308	4.9	170	108250	71.8	351.5	2000	6.5	0.04
12	280	3.3	84.85	Concrete feed pad	800	280	2.9	180	24000	30.0	85.7	1500	5.4	0.03
15	360	2	180	Concrete feed pad	1500	360	4.2	180	81000	54.0	225.0	2500	6.9	0.04
15	360	2	180	Standoff pad	1800	360	5.0	180	56000	31.1	155.6	4000	11.1	0.06
16	730	2.5	292	Concrete feed pad	3850	730	5.3	180	222000	57.7	304.1	1000	1.4	0.01

Table F.2: Calculations of wintering pad costs:

Data	Formula	**Value** For a 330 cows herd, 3 cows/ha and 184 days of use of the pad/yr	Parameters	Reference
Pad area (m^2)	Herd size * Surface per cow	330*6 = 1980	Surface per cow = 6 m^2/cow	(Dexcel, 2005a)
Pad construction cost ($)	Pad area * Construction cost	1980*24.6 = 48708	Construction cost = $24.6/$m^2$	(Dexcel, 2005b)
Annualisation of pad construction ($/yr)	$\frac{\text{Pad construction cost}}{[\frac{1}{r} - \frac{1}{r(1+r)^n}]}$	48708/8.51= 5721	r = discount rate = 10% n = life time of asset = 20 years	(Pearce et al., 2006)
Pad maintenance costs ($/yr)	Herd size* Number of days the pad is used per year * Maintenance cost	330*184*0.14 = 8500	Maintenance cost = $0.14/cow/day of use	(Dexcel, 2005b)
Total pad costs ($/ha/yr)	(Annualised construction costs + maintenance costs) / Farm area	(8500+ 5721)/110= 129.3		

F.3 Effluent system costs:

The type of effluent system considered in this study is a holding/storage pond with irrigated treated effluents to land. Effluents from the pad and the milking shed are collected and then treated in a series of sealed ponds.

To calculate the volume of the effluent pond needed on a farm, the volume of effluents produced each year can be calculated using the following data:

- herd size
- time per day spent on the pad
- number of days the pad is used per year
- rainfall
- amount of water needed to clean the pad
- volume of milking shed effluents.

The details of the calculation are presented in Table F.3 (Horne et al., 2011).

The construction costs of an effluent system include excavation and lining. The maintenance costs include stirring and reparations (TRC, 2006). The costs of irrigation material include the initial cost of pumps, irrigators and pipes. The maintenance costs of material include irrigation material replacement, wear and tear, and labour (Dexcel, 2005a; TRC, 2006).

The costs of construction and maintenance of an effluent pond and irrigation material were used to value the decomposition and recycling of wastes. These costs were calculated from data gathered in the literature (Dexcel, 2005a; TRC, 2006). Case studies on New Zealand dairy farms provided average data for the irrigation material costs and maintenance in \$/cow (Table F.4).

The value obtained for total pond costs of \$4/m^3 (Table F.4) is also valid for stocking rates of 4 and 5 cows/ha. This value has been used to value the decomposition and recycling of wastes.

Table F.3: Calculation of the needed volume of an effluent pond.

Data		Formula	Value For a 330 cows herd	Parameters	Reference
Effluent volume per day (L/day)		Herd size * time awake on pad * Effluent per cow	330*10*3.1=10312	Herd size: 330 cows Time awake on pad: 10 hours Effluent per cow: 3.1L/cow/hour	(Dexcel, 2005a)
Effluent volume (m^3/yr)	A	Effluent volume per day * number of days on pad/1000	10312*184/1000=1897	Number of days on pad: 184 days	
Volume of rain added to effluents (m^3/yr)	B	Rainfall * pad area/1000	1200*1980/1000=2376	Rainfall:1200 mm Pad area: 1980 m^2	
Volume of water used to wash the pad (m^3/yr)	C	Volume used/m2 of pad * pad area * number of washed per year /1000	1980*61.3*6/1000=728	Volume used/m^2 of pad:6L/m^2 Number of washed per year: 61.3	(Dexcel, 2005a)
Volume of water used to wash the dairy shed (m^3/yr)	D	Herd size * volume of water used to wash the milking shed/cow/day * number of days in milk / 1000	330*46.4 *270/1000=4136	Volume of water used to wash the milking shed/cow/day: 46.4L/cow/day Number of days in milk: 270 days	(Dexcel, 2005a)
Total volume of effluents (m^3/yr)		A+B+C+D	1897+2376+728+4136=9139		
Effluent pond size (m^3)		Total volume of effluents* Number of days of storage of effluents/ Number of days on pad	9139*60/184=2980	Number of days of storage of effluents: 60 days	(Dexcel, 2005a)

Table F.4: Calculations of effluent system costs:

Data		Formula	Value For a 330 cows herd	Parameters	Reference
Pond volume (m3)		From effluent quantity	2980		
Pond construction costs ($)		Pond volume * Construction cost	2980*15=44700	Construction cost: $15/m^3$	(Pangborn, 2010; TRC, 2006)
Annualised pond construction costs ($/yr)	A	$\frac{\text{Pond construction cost}}{[\frac{1}{r} - \frac{1}{r(1+r)^n}]}$	44700/8.51=5250	r = discount rate = 10% n = life time of asset = 20 years	(Pearce et al., 2006)
Pond maintenance costs ($/yr)	B	Herd size * Maintenance costs	330*10=3300	Maintenance costs: $10/cow	(TRC, 2006)
Material installation costs ($)		Herd size * Material costs	330*45=14850	Material cost: $45/cow	(Dexcel, 2005b)
Annualised material installation costs ($/yr)	C	$\frac{\text{Material installation costs}}{[\frac{1}{r} - \frac{1}{r(1+r)^n}]}$	14850/8.51=1744	r = discount rate = 10% n = life time of asset = 20 years	(Pearce et al., 2006)
Material maintenance costs ($/yr)	D	Herd size * maintenance costs	330*5=1650	Maintenance costs: $5/cow	(Dexcel, 2005b)
Total pond costs ($/ha/yr)		(A+B+C+D)/Farm area	(5250+3300+1744+1650) /110 = 108.6	Farm area = 110 ha	
Total pond costs /volume ($/m3)		Total pond costs / Pond volume	(5250+3300+1744+1650) / 2980 = **4.01$/m^3**		

F.4 Mitigation functions:

The costs of mitigating (i.e. limiting their emissions) of nutrients like N and P on the farm were used to value the filtering of nutrients. The goal was to build a mitigation function for each nutrient to describe the total costs of mitigation depending on the quantity of nutrients to mitigate. Different techniques, their efficiency and their cost were considered in constructing these functions.

F.4.1 N mitigation function:

<u>Choice of the mitigation techniques:</u>

The three techniques most used in New Zealand were chosen to construct the mitigation function:

- Nitrification inhibitors prevent the formation of NO_3^- and N_2O, and increase N use efficiency.
- A standoff pad reduces the total amount of N returned to pasture as urine patches and allows the management of effluent applications according to soil conditions,
- Replacement of fertilisers with low N content supplements, like maize silage, reduces the total amount of N excreted and returned to pasture,

<u>Estimation of the costs of each technique:</u>

Nitrification inhibitors:

A fixed cost for the application of nitrification inhibitors was used: 90$/ha/yr + GST (15%). This includes the cost of the product and the application cost. Nitrification inhibitors are applied twice a year, which represents a total cost of 207$/ha/yr.

Standoff pad:

The cost of using a standoff-pad was determined as explained in Table F.2. The maximum cost of a standoff-pad used here to calculate the mitigation function, is the cost associated with using the pad for 184 days a year, between May and October.

Low N content maize silage:

The amount of maize silage necessary to replace N fertilisers to reduce N losses was calculated for each scenario. It was determined with the OVERSEER® model. The outputs of the SPASMO model were fed into OVERSEER to determine a nutrient budget for the farm considered. OVERSEER scenarios were then used to calculate the efficiency of each mitigation technique. The amounts of maize silage necessary were: with 3 cows/ha, 67 tDM for 110ha, with 4 cows/ha, 120 tDM for 110ha, and with 5 cows/ha, 199 tDM for 110ha. The price of maize silage used was the price of DM on the stack (5% of the price of MS

(6$/kgMS)), that is 0.30 $/kg DM of maize silage. The cost per hectare was calculated by dividing by the number of ha: for example for the base case scenario, the cost of using maize silage is 67 tDM* 1000 * 0.30$/kgDM /110 ha = $182.7/ha. The extra labour costs were not considered because the time saved by not fertilising was assumed to be equivalent to the extra-time spent feeding the cows.

The mitigation costs for each stocking rate are presented in Table F.5.

Table F.5: Mitigation costs to reduce N losses ($/ha/yr)

Mitigation technique	**3cows/ha**	**4 cows/ha**	**5cows/ha**
Nitrification inhibitor (DCD)	207	207	207
Standoff pad	129.3	172.4	215.5
Low N maize silage	182.7	327.3	542.7

<u>Estimation of the efficiency of each technique:</u>

It was assumed that to mitigate N leaching, farmers will first use nitrification inhibitors because it doesn't require a change in the farm management. Then if more N needs to be mitigated, farmers can construct a pad, and, as a third option, switch from N fertilisers to maize silage.

Therefore for each stocking rate studied (3, 4 and 5 cows/ha), the OVERSEER® model was run four times: with no mitigation to determine the original N leaching, with nitrification inhibitors, with nitrification inhibitors and a standoff pad, and with nitrification inhibitors, a standoff pad, and low N feed. The reduction in N leaching for each case was recorded (Table F.6) and associated to the total cost of mitigation.

Table F.6: Efficiency of the three combinations of mitigation techniques to reduce N losses.

Scenario	Mitigation techniques	N mitigated (kg N/ha/yr)	Total cost of mitigation ($/ha/yr)
Horotiu (3cows/ha)	Initial N leached = 25		
	DCD	3	207
	DCD+pad	8	336.3
	DCD+pad+lowN	11	519
Horotiu (4cows/ha)	Initial N leached = 41		
	DCD	5	207
	DCD+pad	11	379.4
	DCD+pad+lowN	22	706.7
Horotiu (5cows/ha)	Initial N leached = 78		
	DCD	12	207
	DCD+pad	24	422.5
	DCD+pad+lowN	53	965.2
Te Kowhai (3cows/ha)	Initial N leached = 14		
	DCD	2	207
	DCD+pad	4	336.3
	DCD+pad+lowN	6	519
Te Kowhai (4cows/ha)	Initial N leached = 23		
	DCD	3	207
	DCD+pad	5	379.4
	DCD+pad+lowN	13	706.7
Te Kowhai (4cows/ha)	Initial N leached = 43		
	DCD	5	207
	DCD+pad	11	422.5
	DCD+pad+lowN	30	965.2

Estimation of the mitigation function:

To estimate the mitigation functions, the total costs of N mitigation, that is the costs of all the mitigation techniques used, was then plotted for all scenarios, for each soil, against the amount of N mitigated (Figure F.1).

Figure F.1 shows that it is more cost efficient to mitigate N on a well drained soil (HR) than on a poorly drained one (TK).

The mitigations functions were then used to value the filtering of N service.

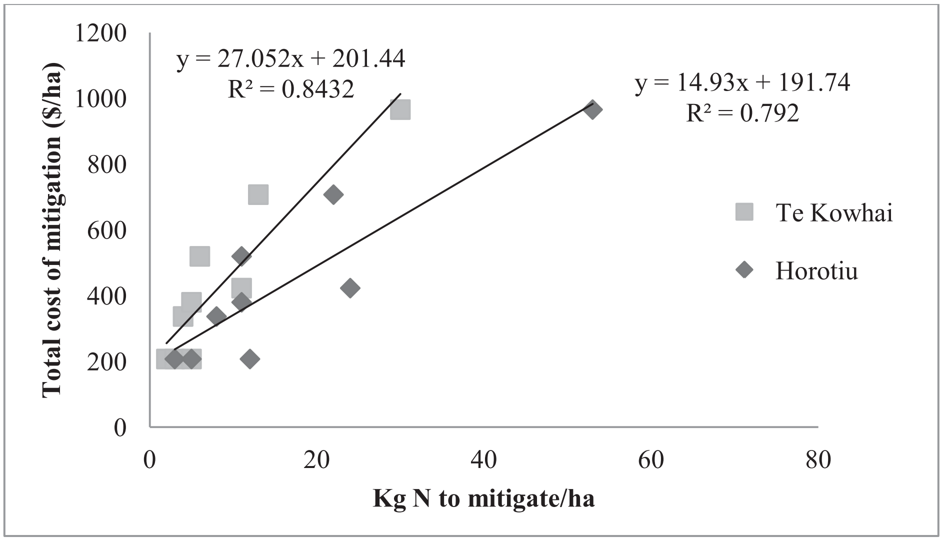

Figure F.1: N mitigation function for Horotiu and Te Kowhai silt loams.

F.4.2 P mitigation functions:

The same method was used to determine the P mitigation functions.

Only one mitigation technique was considered, the use of a standoff pad, because the scenarios defined in this study already assumed best management practises like agronomical optimum for Olsen P and fertiliser application when SWC<FC (Table F.7). the costs of a standoff pad are presented in Table F.5.

Table F.7: Efficiency of a pad to reduce P losses.

Scenario	Mitigation techniques	P mitigated (kg P/ha/yr)	Total cost of mitigation ($/ha/yr)
Horotiu (3cows/ha)	Initial P runoff = 0.37		
	Standoff pad	0.11	129.3
Horotiu (4cows/ha)	Initial P runoff = 0.62		
	Standoff pad	0.22	172.4
Horotiu (5cows/ha)	Initial P runoff = 0.87		
	Standoff pad	0.24	215.5
Te Kowhai (3cows/ha)	Initial P runoff = 0.86		
	Standoff pad	0.27	129.3
Te Kowhai (4cows/ha)	Initial P runoff = 1.4		
	Standoff pad	0.4	172.4
Te Kowhai (4cows/ha)	Initial P runoff = 2.18		
	Standoff pad	0.59	215.5

To estimate the mitigation function, the total costs of P mitigation was plotted against the amount of P runoff mitigated, for all scenarios and each soil (Figure F.2). These P mitigations functions were then used to value the filtering of P service.

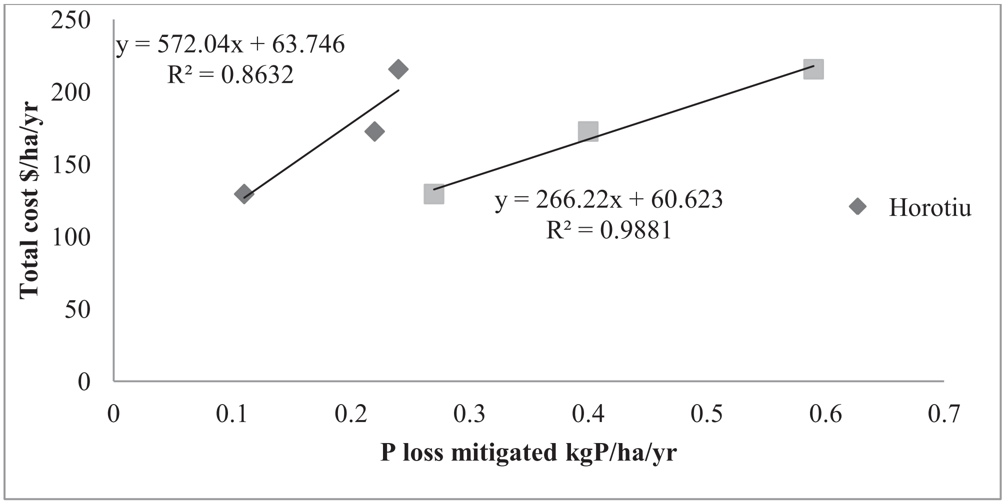

Figure F.2: P mitigation functions for Horotiu and Te Kowhai silt loams.

F.5 Constructed wetlands costs:

The costs of building and maintaining a constructed wetland were used to calculate the value of the filtering capacity of the soil for contaminants. Wetlands are areas including a variety of plant species densely spaced which together with shallow water, provide good wildlife habitat as well as water purification. Constructed wetland systems are designed to simulate and optimise the filtering and organic matter breakdown processes that occur in soils and natural wetlands before discharge to a waterway (Sukias and Tanner, 2011; TRC, 2006; Wilcock et al., 2011). The size of a wetland for a standard New Zealand dairy farm is around 120 m^3. The costs of construction of a wetland was determined from the literature (TRC, 2006) at \$116-\$150/m^3 (for a standard surface flow constructed wetland, 30cm deep) including earthworks, a clay liner, inlet and outlet structures, gravels, plants and additional establishment costs (site survey, design and resource consent processes). The maintenance costs of a constructed wetland were estimated here to be 1% of the construction costs (TRC, 2006) (Table F.8). The total cost of a constructed wetland in \$/yr would be the sum of the annualised constructions costs plus the annual maintenance costs (Table F.8). These costs were used to calculate the value of the filtering capacity of the soil for contaminants.

Table F.8: Calculations of the cost of a constructed wetland.

Data		Formula	Value	Parameters	Reference
Wetland volume (m3)			120		
Wetland construction costs ($)		Wetland volume * Construction cost	120*100= 12000	Wetland construction costs = $100/m^3	(TRC, 2006)
Annualised wetland construction costs ($/yr)	A	$\frac{\text{Wetland construction cost}}{[\frac{1}{r} - \frac{1}{r(1+r)^n}]}$	12000/9.43=1273	r = discount rate = 10% n = life time of asset = 30 years	(Pearce et al., 2006)
Wetland maintenance costs ($/yr)	B	Wetland construction costs*0.001	12000*0.01=120	1% of construction costs	(TRC, 2006)
Total costs of wetland Pond costs /volume ($/yr)	A+B		1273+120= 1393		