LOOKING AT SOILS THROUGH THE NATURAL CAPITAL AND ECOSYSTEM SERVICES LENS

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EXECUTIVE SUMMARY

Land degradation impairs the soil's ability to support our well-being through loss of actual or potential productivity or utility; and further deterioration is expected (Eswaran et al 2001). Land degradation has been identified as one of the most insidious and under-acknowledged global challenge of this century (Montgomery 2010). Conveying the importance and value of soil to society is therefore critical. The framing of soil as a natural capital stock that yields a flow of valuable goods and services into the future, together with other forms of capital, is expected to resonate with those who have a human centric utilitarian view.

Soil natural capital has been defined as the stocks of mass and energy in the soil and their organisation/entropy. Soil natural capital can be characterised as having inherent and dynamic properties. Inherent soil properties do not change considerably with management (e.g., stoniness, clay type) and are used to determine soil/land capability or suitability. Dynamic soil properties may, however, change with management (e.g., organic matter content, soil moisture) and are used to assess soil quality for specific land uses. Monitoring the condition of soil stocks is necessary to ensure that increases in the intensity of our use of ecosystems (e.g., for food and fibre production) do not compromise soil natural capital stocks which in turn compromise the flow of all ecosystem services in the long term.

Soil contributes to many of the services derived from ecosystems, and the role of soil changes depending on both the ecosystem and its use. The Millennium Ecosystem Assessment framework, which differentiates between the provisioning, regulating, cultural, and supporting services provided by ecosystems, has been applied to soils in this report highlighting the soil's multi-faceted contribution to human well-being.

By valuing changes in ecosystem services it is anticipated that the contribution of ecosystems, including soils, to human well-being will be better recognised and incorporated in societal decision making. Most commonly, valuing ecosystems (including soils) has focused on economic valuation. Economic valuation is premised on the assumption that people have preferences for or against changes in ecosystem services and that these preferences can be expressed in monetary terms. A series of economic valuation techniques have emerged in an attempt to address the diverse nature of goods and services provided by ecosystems, and the various motivations for assigning them an economic value. The techniques rely on actual, surrogate or hypothetical markets to observe revealed or stated preferences for changes in ecosystem services (Table E1).

Table E1: **Valuation techniques as they relate to market types and how human preferences are elicited.**

Type of Market	Revealed Preference Methods	Stated Preference Methods
Actual markets	Productivity change approach; replacement cost (provision cost); defensive expenditures; benefits transfer	
Surrogate markets	Hedonic pricing; travel cost; benefits transfer	
Hypothetical markets		Contingent valuation; choice modelling; deliberative monetary valuation; benefits transfer

Some examples of how these techniques have been applied to soils are outlined in Box E1. However, there are a number of challenges to using economic valuation to value changes in ecosystem services, and the role of soils in providing these services. Economic valuation is often hindered by the paucity of both bio-physical and economic data, the inability to place a monetary value on all services or the double-counting of some services, the time-bound character of value estimates, and the lack of agreement on many aspects of how to aggregate the values of different ecosystem services provided by the same ecosystem. The latter can be quite challenging as aggregating values tends to mask the trade-offs between services, especially where a change results in both costs (negative impacts) and benefits (positive impacts). A number of decision supporting frameworks (such as cost-benefit analysis, multi-criteria assessment and modelling tools) are available and can be used to weigh the costs and benefits of changes in ecosystems (and the subsequent flow of ecosystem services).

Box E1: Examples of economic valuation techniques applied to soils.

- Sparling et al. (2006) used the productivity change approach to value the food provisioning and regulating services generated through soil organic matter recovery in three contrasting New Zealand soil orders.
- Drechsel et al. (2004) used the replacement cost approach to value soil fertility by looking at the cost of fertilizers needed to replace the soil nutrients to maintain a certain level of productivity.
- Chichilnisky and Heal (1998) used the provision cost approach to measure the value of clean drinking water provided by the Catskill watershed in New York City by estimating the cost to construct and maintain a water filtration plant.
- Hansen et al. (2002) used the defensive expenditures method to estimate the cost of soil erosion in a watershed in the United Stated by estimating dredging costs.
- Samarasinghe and Greenhalgh (2013) used hedonic pricing and examined the relationship between soil characteristics and rural farmland values in the Manawatu catchment of New Zealand.
- Feather et al. (1999) used the travel cost method to value the water-based recreational benefit of soil conservation programmes in the United States that were aimed to reduce soil erosion and improve water quality.
- Scrimgeour and Shepherd (1998) used contingent valuation surveys of the wider community and of farmers to estimate both use and non-use values associated with loss of soil structure (compaction) in the Manawatu region of New Zealand.
- Colombo et al. (2005) used choice modelling to identify preferences for reducing the off-farm effects of soil erosion in the Alto Genil watershed in Southern Spain.

With or without monetary values the application of an ecosystem service approach is proposed for both government and business to help evaluate, in a structural manner, the impact and dependency of a decision on ecosystems, including the soils and the services they provide. The practical application of this ecosystem services approach can encompass:

- establishing the link between the decision and ecosystem services
- assessing the associated risks and opportunities of any decision
- exploring the future, and
- choosing policy, planning, and reporting approaches to sustain ecosystem services.

Some key policy areas that contribute to soil protection and therefore effect the provision of ecosystem services include: planning, water quality and quantity, agriculture, forestry, climate change, air quality, rural development, biodiversity, contaminated land, cultural heritage, and policy to supports research and education. Decision-makers will likely need to evaluate and implement a mix of policy instruments such as outreach and education, regulatory instruments, economic instruments, and protected areas, land retirement and stewardship agreements to achieve a desired outcome for ecosystems and ecosystem services. Information on soil characteristics and functions may also be important components for assessing the effectiveness and efficiency of the policy instruments used to achieve any desired outcome.

Given the developments in the conceptualisation of soils as natural capital, in the assessment and valuation of the role soils play in the provision of ecosystem services, and in the application of an ecosystem services approach to informing decision-making, there is an opportunity for an enriched perspective of the value of our soils and how we can articulate those values.

1. INTRODUCTION

"Soil is the very source of civilisation – Culture begins with cultivation. It is the beginning and end place for land-borne life, a vital link in the cycles of life: the digester of the dead and the birthplace of the new" (Buchan 2010), thus "a nation that destroys its soils destroys itself" (Roosevelt 1934).

There is soil under our pastures and kiwi orchards. There is soil under our magnificent kauri trees. There is soil in our community gardens in which we grow food and in our rain-gardens that treat the storm-water runoff we generate in cities. There is mud from washed away soil in our waterways and harbours. But we tend to see only the grass, the kiwifruit, the kauri, the pumpkin or the mud; the soil is less visible.[1] External forces – the drought, the rain, and the wind – impact on our soils. So do our choices. We seal soils over for roading and housing, strip them off to mine the minerals below, irrigate and fertilise them to grow crops and pasture, compact and erode them when we apply inappropriate land-management practices. Some soils are resilient which enables us to correct the damage from our earlier choices; others are not sufficiently resilient. As a consequence of what we do we can lose the soils in a short time or build them up slowly with generations of hard work. What we do to our soils matters.

In this report we use the concepts of soil natural capital and ecosystem services to look at our soils through a new lens. The framing of soil as a natural capital stock that yields a flow of valuable goods and services into the future, together with other forms of capital, is expected to help convey the soil's importance and its value to those who have a human centric utilitarian view. We focus on the functionality of soil natural capital and provide greater clarity about the soil's multi-faceted contribution to ecosystem services so that land-use and soil-management decisions can be informed by the monetary and non-monetary values attached to these services.

This report gives a brief introduction to placing monetary values on soil ecosystem services as one way of demonstrating the value of our soils. An ecosystem services approach to decision making is used to describe a structure in which the impact and dependency of a decision on ecosystems, including the soils and the services they provide, can be evaluated. Through this approaches we hope that when we see the grass, the kiwifruit, the kauri, the pumpkin and the mud, we also see the soil.

The report is structured as follows: chapter 2 discusses soil natural capital and ecosystems services; chapter 3 is dedicated to economic valuation in a soil context and its use in decision making; chapter 4 covers the policy instruments for soils; chapter 5 investigates the practical application of the ecosystem service approach; and chapter 6 contains the conclusions.

We tend to see only the grass, the kiwifruit, the kauri …

… the soil is less visible.

[1] These insights were gained from a series of semi-structured interviews conducted in 2010 in New Zealand. Interviewees were experienced and knowledgeable, with various interests in soils and from a range of agencies encompassing regional and national government, businesses, research, education and the non-profit sector. We would like to acknowledge their participation.

2. SOIL NATURAL CAPITAL AND ECOSYSTEM SERVICES

2.1 Soil as a form of natural capital

"Natural capital is the soil and atmospheric structure, plant and animal biomass, etc., that, taken together, forms the basis of all ecosystems. This natural capital stock uses primary inputs (sunlight) to produce the range of ecosystem services and physical natural resource flows." (Costanza et al. 1991:8)

The term 'capital' was first used in economics to describe assets that enable future economic production, such as buildings and machinery. In sustainable development literature this notion of capital has been broadened to include four types of capital – financial and produced capital, natural capital, human capital, and social capital. Capital assets in this broader sense can be defined as resources that generate a flow of goods and services that enhance well-being over time (Ekins et al. 2003; Statistics New Zealand 2008). Therefore, natural capital, as commonly defined, is any stock of natural resources or environmental assets that yields a sustainable flow of useful goods and services into the future (Pearce & Turner 1990; Costanza & Daly 1992; Costanza et al. 1997; Wackernagel & Rees 1997; Macdonald et al. 1999; Olewiler 2004).

At the time of its introduction in the end of the 1980s, the concept of natural capital represented new, more ecologically aware thinking in economics (Akerman 2003). It was used to reflect both the accountant's view of nature with emphasis on environmental assets (e.g. Pearce 1988), and the ecosystem modeller's view of nature with emphasis on ecosystem processes and ecological knowledge (e.g. Costanza et al. 1991). Ultimately, the concept of natural capital became a means to affect the rules according to which claims concerning sustainable development could be made (Akerman 2003). 'Weak sustainability' was seen to require the maintenance of the aggregate stock of capital allowing the depletion of natural capital if compensated with rising stocks of manufactured or human capital (Pearce & Turner 1990). 'Strong sustainability', on the other hand, was based on the complementarity of man-made and natural capital in economic production and was seen to require keeping 'critical' natural capital intact, where criticality was linked to lack of substitutes, high importance and high threat (Daly 1995; Ekins 2003).

The soil – this complex natural resource produced by the fragmentation and weathering of rock or volcanic emissions with an extremely slow rate of regeneration (Buchan 2010; Toth et al. 2007) – was part of early definitions of natural capital. However, only recently have there been efforts to define soil natural capital per se (Vesely 2006; Palm et al. 2007; Robinson et al. 2009; Dominati 2010). Palm et al. (2007) define soil natural capital as texture, mineralogy, and soil organic matter, whereas Robinson et al. (2009) broadens the concept to define soil natural capital as the stocks of mass and energy and their organisation/entropy, which can be quantified and evaluated in terms of their quantity and quality.

Soil natural capital can be categorised as having 'inherent' and 'dynamic' properties (Robinson et al. 2009, Dominati et al. 2010). Inherent soil properties are user invariant properties that do not change considerably with management (e.g. soil texture, stoniness, clay type, and mineralogy). In contrast, dynamic properties are those that may change with management or environmental conditions, such as organic matter content, and soil moisture. While the inherent properties are used to determine soil/land capability or soil/land suitability, the dynamic properties are used to assess soil quality and health, for example, by identifying target ranges for soil parameters given specific land uses. Evaluating soil stocks and determining how they change with time is challenging (Robinson et al. 2012). Soil indicators have been used for monitoring the physical, chemical and biological condition of soils in New Zealand under different contexts (Schipper & Sparling 2000; Lilburne et al. 2004; SNZ 2009; ARC 2010; see Appendix A).

The framing of soil as natural capital stock yielding a flow of valuable goods and services into the future is expected to help convey the soil's importance and its value to a wider society with a predominantly human centric utilitarian viewpoint. In addition, the treatment of soil as capital stock highlights the importance of monitoring its condition and managing it so that it does not depreciate in value. When focusing on the flow of ecosystem services, it is important to acknowledge the role of the stock of natural capital from which the ecosystem services are derived (Robinson et al. 2012). The provision of services should not happen at the expense of such changes in the soil natural capital stock that would compromise the flow of services in the long term. Erosion, organic matter decline, salinization, nutrient saturation, compaction and landslides, contamination, and sealing have been identified as common threats leading to the soil degradation (Toth et al. 2007; Townsend & Howarth 2010). Land degradation – the process by which the ability of soil to maintain the flow of ecosystem goods and services is deteriorated – has been identified as the most insidious and under-acknowledged global challenge of this century (Montgomery 2010).The most commonly used economic indicator of human well-being, GDP, does not deduct the depreciation of natural capital (Dasgupta 2010). Also the current framework for national income accounts does not provide the information necessary to monitor either the value of natural capital or its transformation into other forms of capital (Auty 2007). These shortcomings are increasingly recognized (TEEB 2009), and an important step toward overcoming them is the development and adoption in some countries of the System of Economic Environmental Accounting (SEEA), which covers land, water, environmental expenditures, and social issues in monetary and physical terms (United Nations et al. 2003, TEEB 2009).

2.2 Soil and ecosystem services

"From the fleeting meeting of a bee and a flowering plant in a summer meadow to the great and continuous interactions of air, water and soil — ecosystems embody the foundations of life on earth." (EEA 2010)

Ecosystems are a dynamic complex of natural resources or environmental assets (plant, animal, and microorganism communities and their non-living environment) interacting as a functional unit (e.g. rainforest, coral reef, desert) (United Nations 1992, Article 2). Soil is a key component of terrestrial ecosystems (Figure 1) and can be considered the most complex biomaterial on the planet (Young and Crawford 2004). These ecosystems comprise the aggregation of natural capital stocks and from these stocks come the flow of ecosystem services.

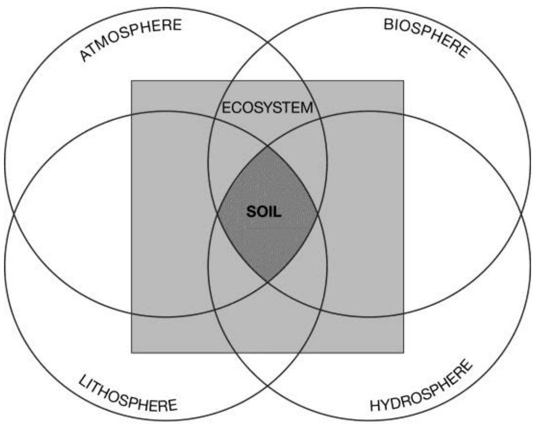

Figure 1: **Soil, a key component of ecosystems and a critical interface (Source: Szabolcs 1994).**

The processes (transformation of input into outputs) that take place in an ecosystem as a result of interaction among the plants, animals, and other microorganisms are known as ecosystem functions. The benefits people obtain from these ecosystem functions are called ecosystem services (MEA 2003). The well-being of every human population in the world is fundamentally and directly dependent on these services (TEEB 2008). From a human perspective, what matters about the environment are not particular natural capital stocks per se, but their ability as a whole, to perform those functions important for human well-being (Figure 2). Ecosystem functions are not necessarily uniquely performed by particular natural capital stocks but result from the interaction of one or more such stocks.

Given its importance to life on earth, soil has been referred to as 'Earth's living skin' (NRC 2009) and 'the biological engine of the Earth' (Haygarth & Ritz 2009). It physically supports plants, retains and delivers nutrients to plants, disposes wastes, renews soil fertility, buffers and moderates the hydrological cycle, and regulates major element cycles (e.g. carbon and nitrogen) (Daily et al. 1997a). Soil not only underpins the production of food, feed, fibre, and fuels, but also plays a central role in determining the quality of our environment (Pathak et al. 2005; Palm et al. 2007; Blanco & Lal 2008; UNEP 2010; Banwart 2011; Robinson et al. 2012). Ultimately, soil contributes to the many services derived from ecosystems, and the role of soil changes depending on the ecosystem and its use. It should be remembered that soils are only one component of an ecosystem and that the flow of services from an ecosystem depends also on the vegetative stock that exists as well as the geologic and climatic conditions. Thus trying to isolate soil ecosystem services is challenging.

Many attempts have been made to classify ecosystem services (De Groot 2002; MEA 2003;, Ekins et al. 2003; Boyd & Banzhaf 2007; Costanza 2008; Fisher et al. 2009), and the classifications are often context specific (Fisher et al. 2009; Turner et al. 2010). In this report, we use the Millennium Ecosystem Assessment (MEA) (2003) classification framework, which is the most widely used framework today. The MEA framework and its variations have been applied to soils (Figure 3) by numerous authors, including Robinson et al. (2009), Palm et al. (2007), Swinton et al. (2007), Barrios (2007), Lavelle et al. (2006), Zhang et al. (2007), Dominati et al. (2010), Jeffery et al. (2010), and Government Office for Science (2010).

The MEA framework divides ecosystem services into:

- provisioning services – products obtained from ecosystems
- regulating services – benefits obtained from the regulation of ecosystem processes
- cultural services – nonmaterial benefits obtained from ecosystems through spiritual enrichment, cognitive development, reflection, recreation, and aesthetic experiences, and
- supporting services – services that are necessary for the production of all other ecosystem services (e.g. soil formation, primary production, nutrient cycling).

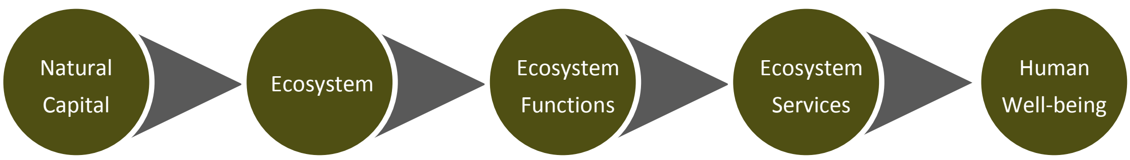

Figure 2: **Relationship between natural capital and human well-being.**

Provisioning, regulating, and cultural services provide relatively direct short-term benefits to humans, while the impacts of supporting services are either indirect or occur over a long period of time (MEA 2003). For example, soil formation services are not directly used by people yet changes in these services impact the provision of food production. No classification of ecosystem services is likely to be all-inclusive and appropriate for all purposes. However, the MEA provides a sound framework from which to discuss ecosystem services. The MEA recognises the overlap between some of the categories, e.g. freshwater can be categorised as a provisioning service and also as a regulating service. These overlaps may mean there is a danger of double counting should this framework be used for accounting purposes (e.g. in a GDP context) (Boyd & Banzaf 2007; Fisher et al. 2009).

For accounting purposes a precise unit of account is required to measure the ecosystem services (Boyd & Banzhaf 2007). Identifying services as intermediate and final based on the directness in contribution to human well-being (Costanza 2008; Fisher & Turner 2008; Fisher et al. 2009) can help avoid double counting in economic valuation and national environmental accounting. However, the classification of the service as final or intermediate will change depending on the beneficiaries (Fisher et al. 2009) and some services can be intermediate as well as final (Costanza 2008). For example, if a landowner is interested in managing the fertility of soil on his/her land, then soil nutrient cycling service would be a final service, whereas, if the land owner is interested in the provision of a better crop yield, then soil nutrient cycling service would move from being a final service to an intermediate one. Boyd and Banzaf (2007) define ecosystem services with the specific purpose of being used for environmental accounting and argue that ecosystem services are components of nature, directly consumed, enjoyed or used to produce human well-being. In their view ecosystem services only include the final services.

A further categorization, that is useful to consider, is based on the spatial and temporal characteristics of ecosystem services (Costanza 2008). This characterisation can complement the MEA's classification to identify how impacts and dependencies are separated in time and space. Temporal characterisation of ecosystem services can help identify when services are delivered, while spatial characterisation can help identify where the services are delivered.

The following categories can be used to spatially characterise ecosystem services:

- global non-proximal: when the service can be used anywhere irrespective of the point of production (e.g. climate change mitigation by soil carbon sequestration)
- local proximal: when service use takes place within a buffer area surrounding the point of production (e.g. pollination, food and fibre production)
- directional flow related (short/long distance): when service use takes place only in a given direction (e.g. storm protection, provision of downstream freshwater)
- in situ: when the point of service production is the same as the point of use (e.g. soil formation)
- user movement related: when the service produced is accessed by user movement (e.g. recreation).

The spatial characterisation of ecosystem services can help in situations where the focus is on managing a given landscape for the provision of ecosystem services across scale, while the temporal characterisation is helpful for valuation and accounting purposes (Fisher et al. 2009). By valuing ecosystem services in common units – usually, but not always, monetary – it is anticipated that the contribution of ecosystems, including soils, to human well-being will be recognized in societal decision-making (Pearce et al. 2006). Otherwise, there is a tendency to consider only those goods and services that are currently traded in markets (Edwards-Jones et al. 2000). This, however, is not without its challenges. Negative values are masked when values are aggregated, and derived monetary values may have large uncertainties and be treated with scepticism. Chapter 3 of this report is dedicated to the valuation of ecosystem services in a soil context, while chapter 5 describes how an ecosystem service approach can be applied to inform decision-making.

Soil natural capital in interaction with other forms of capital

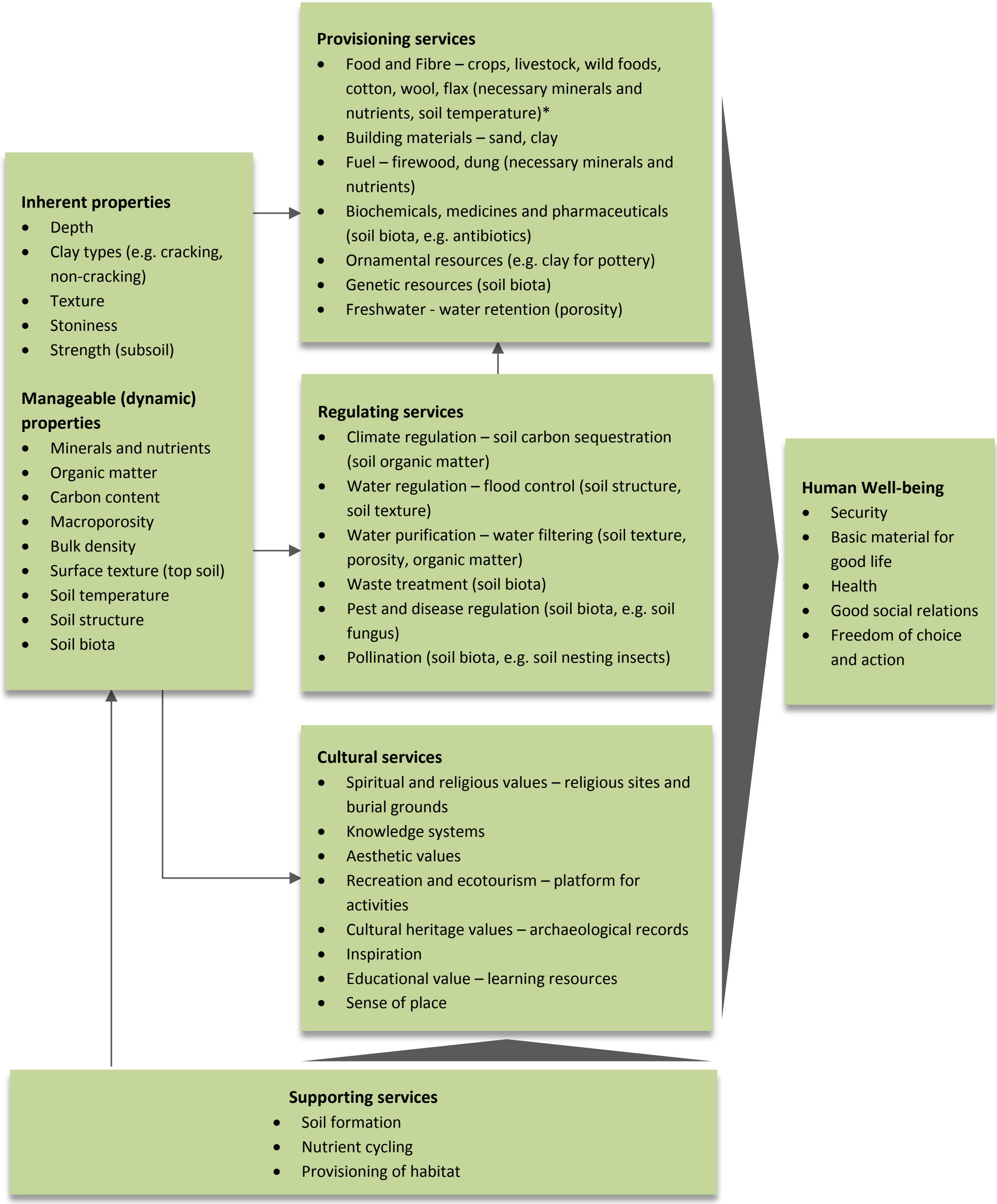

Figure 3: **Provisioning, regulating, cultural and supporting ecosystem services provided by soil natural capital with the interaction of other forms of capital (based on MEA 2003; Lavelle et al. 2006; Zhang et al. 2007; Dominati et al. 2010; Robinson et al. 2009).**

Note: The supporting services indirectly impact on human well-being and occur over a very long time (MEA 2003). * The brackets provide examples of soil elements and properties important for the delivery of a particular ecosystem service.

3 ECONOMIC VALUATION IN A SOIL CONTEXT

3.1 Introduction to economic valuation

Economic valuation attempts to elicit public preferences for changes in the state of the environment in monetary terms (DEFRA 2007). Ecosystems and their associated services have economic value for society because people derive utility from their actual or potential use and also value services for reasons not connected with use (i.e. non-use values) such as altruistic, bequest and stewardship motivations. These human-centred utilitarian values are often considered within the framework of Total Economic Value (TEV, Figure 4) and are measured in monetary terms through their impact on human welfare (MEA 2003; DEFRA 2007). Non-utilitarian values, such as ecological, socio-cultural, and intrinsic values, typically have separate non-monetary metrics (MEA 2003).

The economic valuation of ecosystem services serves a number of purposes (MEA 2003; EFTEC 2006; DEFRA 2007; Ranganathan et al. 2008; TEEB 2009) including:

- choosing between competing uses, e.g. of land use and setting priorities within a sector plan or across different sectors
- demonstrating the importance of an ecosystem service
- providing information for decision-making that takes into account the costs and benefits to the natural environment of a decision to determine whether a policy intervention that alters an ecosystem condition delivers net benefits to society
- providing evidence on which to base decisions on 'value for money' and prioritising funding
- informing the identification of risks and opportunities of a decision
- providing evidence to underpin the development of markets for services
- assessing liability for damage to the environment and determining compensation in environmental litigation
- determining an appropriate level for environmental pricing and taxation
- helping the construction of environmental accounts
- improving indicators of changes in wealth and well-being.

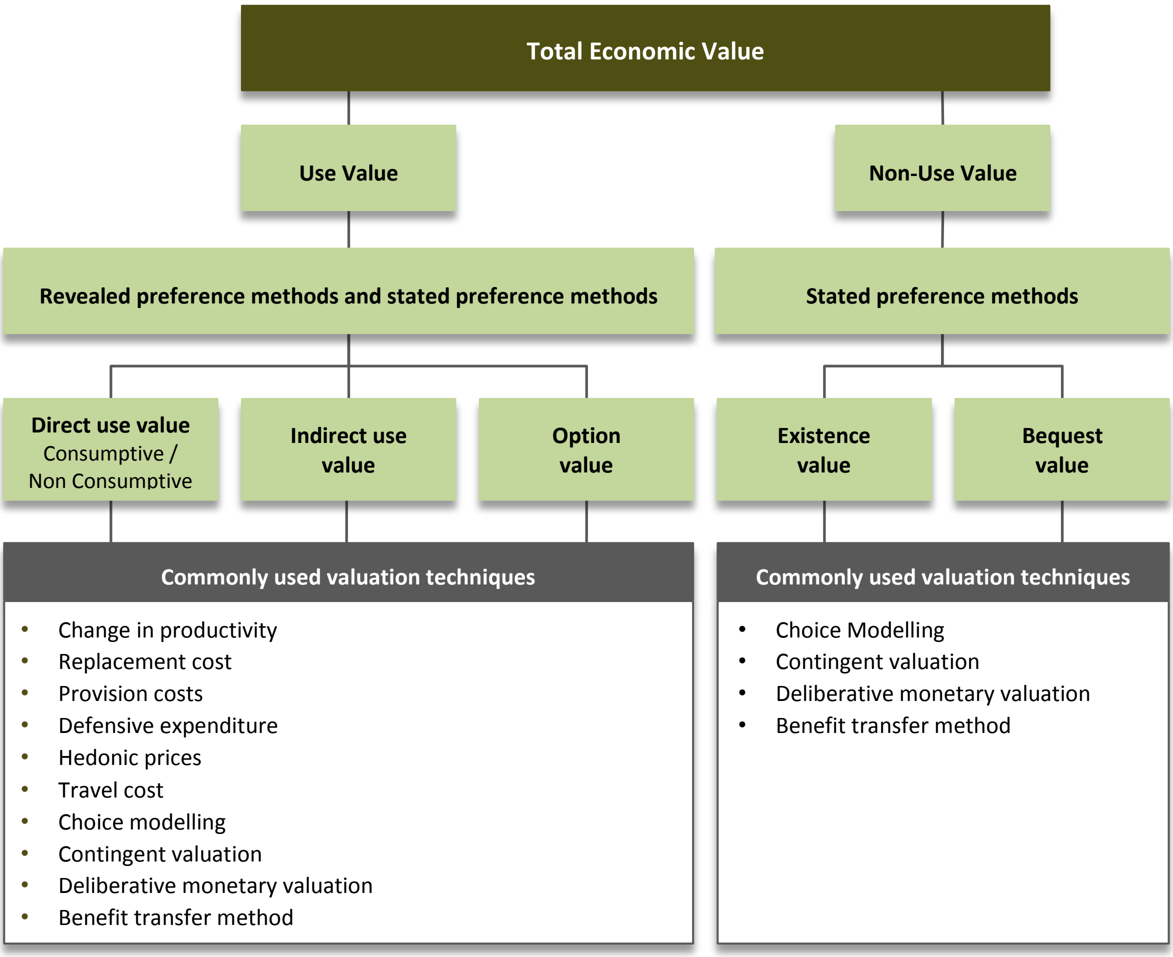

Figure 4: The total economic value framework and the valuation techniques (adapted from TEEB 2009).

Economic valuation methods are premised on the assumption that people have preferences for or against changes in ecosystem services and that these preferences can be expressed in monetary terms (EFTEC 2006). Most economic valuation techniques value the change in ecosystem services or the ecosystem but not their total value. These marginal changes are the most relevant for decision-making and the many types of value attached to these marginal changes are highlighted in the TEV framework. The valuation question can be framed in terms of two alternative measures of value: willingness to pay (WTP) and willingness to accept (compensation) (WTA). These two approaches imply different presumptions about the distribution of property rights and the values derived can differ substantially, depending on the availability of substitutes and income limitations. In many contexts, methodological limitations necessitate the use of WTP rather than WTA (NRC 2004). WTP depends on variables such as the relative scarcity of the benefit, the income of the users, and the extent to which the benefit may be replaceable or not (Balmford et al. 2008; TEEB 2008). WTA depends on variables such as the perception of the benefit, opportunity cost, and relative scarcity of the benefit.
A range of economic valuation techniques have emerged that take into account the nature of different goods and services. Goods and services are ''excludable'' to the degree that individuals can be excluded from benefiting from them. Goods and services are ''rival'' to the degree that one person's benefiting from them interferes with or rivals another's benefiting from them. Excludability is largely a function of supply (to what extent can producers exclude users) and is related to the cultural and institutional mechanisms available to enforce exclusion. Rivalry is a function of demand (how do benefits depend on other users) and is more a characteristic of the good or service itself (Costanza 2008). Ecosystem services that are excludable and rival can be privately owned and traded and can have a market price, like fruits grown in a farmland or timber yield of a private forest (Table 1). The services for which it is difficult or impossible to exclude others from benefiting and that are non-rival are public goods, like a well-regulated climate, erosion control, or the aesthetic benefits of a forest. As these goods and services are not traded in markets and therefore remain unpriced, it is necessary to assess their relative economic worth using non-market valuation techniques.

Economic valuation techniques can be distinguished by the type of market they use – actual, surrogate or hypothetical. Such techniques can also be distinguished as stated preference methods and revealed preference methods, depending on how the preferences are obtained. Revealed preference (RP) method obtains an individual's preferences through observed behaviour, and relies on activities in the market. Stated preference method (SP), on the other hand, uses carefully constructed surveys to ask individuals what their preferences are.

Table 1: Characterisation of ecosystem services based on excludability and rivalry (Cowen 1985; table adapted from Costanza 2008)

	Excludable	Non-excludable
Rival	Most provisioning services (i.e. market goods and services), e.g. fruits grown in a farmland; timber yield of a private forest	Some provisioning services such as open access resources, e.g. fruits grown in a public domain; flax grown on an open space
Non-rival	Some cultural services such as certain recreation services, e.g. hiking in a fee paying national park	Most cultural and regulating services such as public goods and services, e.g. soil carbon sequestration

Table 2: Valuation techniques as they relate to market types and preference elicitation methods

Type of Market	Revealed Preference Methods	Stated Preference Methods
Actual Markets	Productivity Change Approach; Replacement Cost (Provision Cost); Defensive Expenditures; Benefits Transfer	
Surrogate Markets	Hedonic Pricing; Travel Cost; Benefits Transfer	
Hypothetical Markets		Contingent Valuation; Choice Modelling; Deliberative Monetary Valuation; Benefits Transfer

Table 2 shows how the different valuation techniques relate to market types and preference elicitation methods.

3.2 Economic valuation techniques and their application to soils

Soils are an important form of natural capital and contributor to ecosystem services but their importance and value is frequently overlooked (Daily et al. 1997b; EEA 2010). The importance of soils beyond the impact on farming and forestry is often not well understood by the wider society (Robinson et al. 2009; Mogren 2010; Robinson et al. 2012), with the connection between human well-being, ecosystem services and soils being more subtle (NRC 2009; Robinson et al. 2012). While the total value of soils is likely incalculable as it includes the value of human society and the millions of other species contained within the world's ecosystem (Daily et al. 1997a), there have been many attempts to value marginal changes in the contribution of soils to ecosystem services. This section focuses on the utilitarian value of soil and the economic valuation techniques that can be used to measure it. Many of the examples used to demonstrate these economic valuation methods as applied to soils show how soils affect or contribute to the flow of ecosystem services, e.g. erosion or climate regulation or provision of freshwater.

3.2.1 Actual markets

Productivity change approach

Description
The productivity change approach (PCA) focuses on the relationship between a particular ecosystem service and the production (yield) of a marketed good. The ecosystem service in question is considered as an input to the production process (EFTEC 2006). PCA can be used to measure actual change or, when coupled with production simulations, the likely impacts of possible interventions.

Pros and Cons
PCA can be used to measure provisioning services and some regulating services (e.g. water purification). PCA is a straightforward approach if data are available. It is easy to build into cost-benefit analysis, and the results are directly applicable and understood by end users. However, there are high data requirements on the change in soil condition, the subsequent change in ecosystem services, and the resulting impact on production. Furthermore, it may also be difficult to link changes in production to changes in ecosystem services and then to changes in soil condition. PCA cannot be used to estimate non-use values.

Application to soils
PCA usually values the change in soil productivity as expressed through changes in crop yield, multiplied by the unit price of the crop, less the differential in production costs. The PCA's ability to cost alternative crop production practices can easily be built into cost-benefit analyses with direct applicability to farmers (see Walpole & Sinden 1997). Involving farmers in the research is an advantage as it helps ensure critical socio-economic components are not ignored, and the results are relevant to farmers.

Sparling et al. (2006) used PCA to value the food provisioning and regulating services generated through soil organic matter recovery in three contrasting New Zealand soil orders (Recent, Melanic, and Granular). Soil chemical and physical characteristics of the three pairs of matched soils with low organic matter content (after long-term continuous cropping for vegetables or maize) or high organic matter content (continuous pasture) were used as input data for a pasture (grass-clover) production model. The differences in pasture dry matter yields (non-irrigated) were calculated for three climate scenarios (wet, dry, and average years). The difference in carbon (C) accumulation in the soils was estimated using the CENTURY model, while the difference in the amount of nitrogen (N) stored in soils was calculated assuming that the recovery curve for N follows the same pattern as for C, and had a C to N ratio of 11:1.

The productivity benefit of soil organic matter was estimated from the difference in monetary value of the yields from the low and high organic matter content soils. First, the annual pasture dry matter yield under the various scenarios was converted to milk solids from dairy production; then the yield of milk solids was converted to a monetary value using a national milk solids price obtained by averaging inflation-adjusted dairy company pay-outs to farmers over a 10-year period. The value of the increase in climate regulation through carbon sequestration during the soil organic matter recovery was estimated by multiplying the amount of additional sequestered C by the 2007 voluntary and regulated global carbon market price, NZ$ 25.36 and NZ$ 108.83 Mg^{-1} C respectively. The improvement in water purification services was estimated using the increase in N storage due to soil organic matter recovery. Its value is based on the assumption that a nutrient trading scheme exists; such schemes are operational in the United States, Australia, and New Zealand for managing water quality. The price of NZ$ 13.08 per kg of N was based on the price set by the Connecticut Nitrogen Credit Exchange Program for 2007.

For all three soil orders, and for the three climate scenarios, pasture dry matter yields were lower in the soils with lower organic matter contents. The extra organic matter in the high C soils was estimated to be worth NZ$ 42-236 per hectare per year in terms of increased milk solids production.[2] The lower yields from the previously cropped soils were predicted to persist for 36 to 125 years, but with declining effect as organic matter gradually recovered, giving an accumulated loss in pastoral production worth around NZ$811–1,938 per hectare. The hypothetical value of the organic matter recovery as a mechanism to store C and N varied between NZ$2,849 and NZ$55,210 per hectare depending on the soil, discount rates (3.5% and 10%) and values used for carbon and nitrogen credits. When other impacted services such as erosion control and flood prevention are also included, the value of soil organic matter recovery is likely to be much greater than calculated here.

Further readings:

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Singh J, Pal J 1995. Land degradation and economic sustainability. Ecological Economics 15(1): 77–86.

Atis E 2006. Economic impacts on cotton production due to land degradation in the Gediz Delta, Turkey. Land Use Policy 23(2): 181–186.

Vesely E-T 2009. Natural capital restoration and economic efficiency. PhD thesis, The University of Auckland.

Replacement cost approach

Description

The replacement cost approach (RCA) estimates the monetary value of ecosystem services based on the costs of substitutes that can be used for replacing or restoring damaged ecosystem services to their original productivity levels. For example, RCA can be used to estimate a value of soil fertility by looking at the cost of fertilizers needed to replace the soil nutrients to maintain a certain level of productivity (Drechsel et al. 2004). The validity of the RCA is based on three conditions: (1) the chosen substitute provides functions that are equivalent in quality and magnitude to the ecosystem service; (2) the chosen substitute is the least cost alternative way of replacing the ecosystem service; and (3) individuals in aggregate would be willing to incur these costs if the ecosystem service was no longer available (Shabman & Batie 1978).

The variant of the RCA that refers to the cost of providing/replacing the damaged service through other means is called the provision cost approach, e.g. the cost of replacing the water filtration service provided by a watershed with the construction and maintenance of a water-filtration plant.

Pros and Cons

This approach is relatively easy to apply, and uses actual cost outlays. However, the RCA can only be used to place a value on ecosystem services that are easily substitutable. Thus it could be used to measure the value of the services provided or affected by the dynamic soil properties but not the inherent properties. Moreover, RCA does not consider social preferences for ecosystem services. This approach cannot be used to estimate non-use values, and tends to overestimate the actual value as individuals' WTP for the substitutes are hypothetical (TEEB 2009).

Application to soils

The RCA has been used in several studies to measure the cost of soil erosion and the value of soil nutrients (Kim & Dixon 1986; Scott et al. 2000; Drechsel et al. 2004; Hansen & Hellerstein 2007). Kim and Dixon (1986) used replacement cost method to measure the cost of upland soil erosion in Korea. The costs of maintaining a given level of crop production by physically replacing lost soil and nutrients and by adopting soil management techniques were compared. The estimated cost of soil erosion (or the value of soils' contribution to the food provisioning service) may be an overestimation as other substitutes were not considered. Drechsel et al. (2004) used RCA to estimate the costs of soil nutrient depletion in farming systems in Ghana. They used the local price of poultry manure to calculate the replacement cost of soil nutrients as this was used by farmers to compensate for the lost nutrients.

The provision cost method has been used to value the water retention and water filtering capacity of soils. DOC (2006) used a provision cost approach to value the provision of water in the Te Papanui catchment in the Otago region. The 22 000 ha of tussock land that occupy the area trap condensation from the mists and deliver the water into the soil structure, where it is retained. They found that the cost to provide the water from the catchment free of charge for drinking, hydro-electricity generation, and irrigation from somewhere else would be substantial. The provision cost approach was also used to measure the value of clean drinking water provided by the Catskill watershed in New York City by estimating the cost to construct and maintain a water filtration plant (Chichilnisky & Heal 1998).

[2] The reported results are from Vesely (2009), which contains new price assumptions and revised calculations.

Further readings:

Drechsel P, Giordano M, Gyiele LA 2004. Valuing nutrients in soil and water: concepts and techniques with examples from IWMI studies in the developing world. IWMI Research Report 82. Colombo, International Water Management Institute.

Scott CA, Zarazua JA, Levine G 2000. Urban wastewater reuse for crop production in the water-short Guanajuato river basin, Mexico. IWMI Research Report 41. Colombo, International Water Management Institute.

Kim H, Dixon J 1986. Economic valuation of environmental quality aspects of upland agriculture projects in Korea. In: Dixon J, Hufschimdt M eds Economic valuation techniques for the environment: a case study workbook. London, BA, The John Hopkins University Press. 203 p.

Hansen L, Hellerstein D 2007. The value of the reservoir services gained with soil conservation. Land Economics 83(3): 285–301.

Chichilnisky G, Heal G 1998. Economic returns from the biosphere. Nature 391: 629–630.

DOC (New Zealand Department of Conservation) 2006. The value of conservation: what does conservation contribute to the economy? URL: http://www.doc.govt.nz/upload/documents/conservation/value-of-conservation.pdf (accessed 3 May 2013).

Defensive expenditures (Averting expenditures)

The defensive expenditures approach uses the costs incurred in mitigating the adverse effects of a reduction in ecosystem service quantity or quality. For example, an individual may bear the cost of soil conservation to avoid damages from soil erosion. When the extent and potential effect of soil degradation or improvement are difficult to assess, actual preventive or defensive expenditures may be used to assess a rough value of the change.

Pros and Cons

This approach uses actual preventive or defensive costs. However, this approach faces issues relating to degree of substitutability and it can only be used for situations where it is possible to avoid or prevent adverse effects. The estimates from the defensive expenditures approach are limited by income. They usually present a lower bound measure of the WTP to avoid adverse effects of reduced ecosystem services as the averted expenditures are estimated before adverse effects are incurred. Therefore, the estimate will not necessarily capture the true value of a reduced supply of a resource, i.e. where the resource is scarce, and the effect is no longer hypothetical. The defensive expenditures approach cannot be used to estimate non-use values.

Application to soils

Hansen et al. (2002) used dredging costs in a watershed in the United States to estimate the cost of soil erosion. Without this dredging the eroded soil that entered the shipping channels and harbours would hinder navigation, posing a cost to the industry. These estimates did not take into account the productivity changes that might have resulted from the loss of soils from farm land and, therefore, might have underestimated the cost of soil erosion.

Further readings:

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Hansen L, Ribaudo M 2008. Economic measures of soil conservation benefits: regional values for policy assessment. Technical Bulletin No. (TB-1922). Washington, DC, United States Department of Agriculture.

Hansen L, Breneman VE, Davison CW, Dicken CW 2002. The cost of soil erosion to downstream navigation. Journal of Soil and Water Conservation 57(4): 205–212.

3.2.2 Surrogate markets

Hedonic Pricing Method

The hedonic pricing method (HPM) uses property or land prices to estimate the economic value of associated attributes that affect property or land prices, e.g. size of the land area, number of rooms, distance to amenities, soil quality. The sale or rental price of land with different attribute qualities is assessed using regression analysis. The basic assumption is that, all other attributes being equal, higher quality attributes translate into higher property values. For this to be true there needs to be an open and competitive market for property or land. The hedonic pricing method, for example, can be used to value the changes in potential rooting depth by analysing the farmland price differences, given that all other farmland attributes are held constant.

Pros and Cons

The hedonic pricing method uses revealed preference data to estimate relationships between market price and property attributes where resulting estimates are marginal values of the attributes. The strength of the hedonic approach depends on the accuracy of price data, the speed of market adjustment, and buyer expectations. It requires large amounts of data and is sensitive to the functional specification of regression analysis. Moreover, market prices may not fully reflect quality differentials of the property attributes. Caution is needed when the attribute levels are expected to change rapidly. Hedonic pricing method cannot be used to estimate non-use values.

Application to soils

In an attempt to value soil natural capital, Samarasinghe and Greenhalgh (2013) used the inherent characteristics of soil (i.e. natural capital) and land valuation data to examine the relationship between soil characteristics and rural farmland values in the Manawatu catchment of New Zealand. Using the HPM, they found that the characteristics used to describe soil natural capital stock, e.g. gravel class, drainage class, potential rooting depth, and profile available water, are reflected in rural farmland values and are already implicitly valued in the market. Moreover, they found that these characteristics of soil stocks do not behave simply as independent variables but that there are complex relationships between them that influence their value.

Further readings

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King DA, Sinden JA. 1988. Influence of soil conservation on farm land values. Land Economics 64(3): 242–255.

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Miranowski JA, Hammes BD 1984. Implicit prices of soil characteristics for farmland in Iowa. American Journal of Agricultural Economics 66: 745–749.

Palmquist RB, Danielson LE 1989. A hedonic study of the effects of erosion control and drainage on farmland values. American Journal of Agricultural Economics 71(1): 55–62.

Rosen S 1974. Hedonic prices and implicit markets: product differentiation in pure competition. Journal of Political Economy 82: 34–55.

Samarasinghe O, Greenhalgh S 2013. Valuing the soil natural capital – a New Zealand case study. Soil Research 51(4) 278-287. http://dx.doi.org/10.1071/SR12246.

Travel Cost Method

The travel cost method (TCM) examines the trade-off between the satisfaction gained from participating in an activity at a site and the value of money and time given up. The fundamental assumption is that people may weigh the money and time costs of travel to a site in the same way as an admission fee. The values of ecosystem services are captured by TCM to the extent they can be represented as factors that influence a person's decision about where to travel or how often to travel to a given site (EPA 2009). For example, the quality of water and the state of river banks would influence a person's decisions about whether or how often to visit a river site for recreation.

Pros and Cons

Travel cost method uses revealed preference data and is a relatively straightforward method. Estimated results from this method are relatively easy to interpret and explain. However, this method is limited to estimating impacts on recreational values of soils and requires a large amount of data. There are a number of issues that can arise during the estimation of the travel costs such as how to treat the opportunity cost of travel time and on-site time, and how to estimate visitation costs. The method is hard to use when travel encompasses multiple destinations or is for multiple purposes. As with the HPM, sensitivity to the functional specification can also be an issue. The TCM cannot be used to estimate non-use values.

Application to soils

Suspended sediment in waterways decreases water's aesthetic appeal, which lowers the quality of fishing, swimming, and other water-contact activities. TCM has been used to value the water-based recreational benefit of soil conservation programmes in the United States that were aimed to reduce soil erosion and improve water quality (Feather & Hellerstein 1997; Feather et al. 1999).

Further readings

Kerr GN, Sharp BMH 1987. Valuing the environment: economic theory and applications. Lincoln College, Canterbury, New Zealand, Centre for Resource Management.

Feather P, Hellerstein D 1997. Calibrating benefit function transfer to assess the conservation reserve program. American Journal of Agricultural Economics 79(1): 151–162.

Feather P, Hellerstein D, Hansen L 1999. Economic valuation of environmental benefits and the targeting of conservation programs: the case of the CRP. Agricultural Economic Report 778. Washington DC,US Department of Agriculture, Economic Research Service.

3.2.3 Hypothetical markets

Contingent valuation

Where actual market data are lacking, contingent valuation (CV) can be used to discover how people value changes in certain ecosystem services by directly questioning a sample of the population. These changes, and the markets in which they are to be valued, are hypothetical, hence the name of the technique. The most common CV method uses questionnaire surveys and the results are scaled to represent a value for the total population.

The quality of the contingent valuation estimates is assessed based on content, construct, and criterion validity (USEPA 2000). Content validity is how well the environmental change being valued is described. Questions added to the survey to probe for the respondents' comprehension can offer an indication of the potential reliability of the study. Construct validity is determined by variables expected in theory to be important determinants of preferences that are statistically significant with the correct sign. Finally, the criterion validity of the contingent valuation estimates can be determined by comparing the results with those of other valuation techniques.

Pros and Cons

Contingent valuation is potentially widely applicable and can be used to estimate use and non-use values. However, this method is based on stated preference and the resulting preferences might be biased, e.g. the respondents might be answering a different question from the one the surveyor had intended (such as expressing positive WTP not because they value something but because it feels 'right' or 'good' to say so). Contingent valuation is sensitive to factors such as information provision (how informed the respondents are about ecosystem services), the framing of the questions, and the payment vehicle (e.g. a tax as opposed to a contribution or donation). The CV method is also time consuming and costly.

Application to soils

Scrimgeour and Shepherd (1998) used CV surveys of the wider community and of farmers to estimate both use and non-use values associated with loss of soil structure (compaction) in the Manawatu region of New Zealand. A postal survey was carried out on a random sample of 240 people from the Manawatu region and on 70 farmers selected on the basis of information from satellite imagery. The farmers' willingness to pay to avoid soil compaction was estimated by asking them to state their preparedness to pay for uncompacted and compacted land, respectively. The difference between the two represents the present value of the costs of compaction and this was transformed into an annual willingness to pay by multiplying it by a discount rate.

Further readings

Scrimgeour FG, Shepherd TG 1998. The economics of soil structural degradation under cropping: some empirical estimates from New Zealand. Australian Journal of Soil Resources 36: 831–40.

Kerr GN, Sharp BMH 1987. Valuing the environment: economic theory and applications. Lincoln, College, Canterbury, New Zealand, Centre for Resource Management.

Choice modelling

Choice modelling is an indirect stated preference method in which individuals are asked (via survey techniques) to choose from alternative bundles of attributes. Each alternative is described by a number of attributes including the state of different ecosystem services. One of the attributes is generally monetary. The survey design is based on *a priori* assumptions regarding the interaction between attributes. These are identified using focus groups and pilot studies. The preferred alternative is assumed to have higher expected utility for the respondent than any other alternative presented to them. If sufficient information is available about people's choices, it becomes possible to use statistical methods to derive estimates of coefficients in a utility function that describe the partial values ascribed to each attribute, including the different ecosystem services (Kerr & Sharp 2003).

Pros and Cons

Choice modelling can be used to value both use and non-use values. It can provide multiple ecosystem service value estimates. Asking the respondents to choose from a series of alternative options with different ecosystem service conditions allows them to consider trade-offs between ecosystem services. Choice modelling, being a stated preference method, may extract preferences in the form of attitudes instead of actual behavioural intentions. There is also a possibility of potential bias based on learning and fatigue effects due to lengthy interview times. This method also requires complex techniques for design and analysis.

Application to soils

Colombo et al. (2005) used choice modelling to identify preferences for reducing the off-farm effects of soil erosion in the Alto Genil watershed in Southern Spain. Using focus groups and informal interviews, the authors identified the subset of soil erosion effects to be used as attributes for the choice modelling. The selected attributes included the area to be covered by the soil conservation plan, landscape impacts, wildlife impacts, effects on water quality, and effects on rural employment. The implicit prices for the attributes considered were found to be positive, implying respondents have a positive WTP for increases in the quality or quantity of each attribute.

Further readings

Bennett J, Blamey R eds 2001. The choice modelling approach to environmental valuation. , Cheltenham, UK. Edward Elgar.

Colombo SN, Hanley N, Calatrava-Requena J 2005. Designing policy for reducing the off-farm effects of soil erosion using choice experiments. Journal of Agricultural Economics 56(1): 81–95.

Hasler B, Lundhede T, Bille T 2006. Valuation of nature restoration and protection of archaeological artefacts in Great Aamose in western Zealand, Denmark. Paper presented at the ENVECON Conference, 24 March, 2006.

Deliberative Monetary Valuation method

Deliberative monetary valuation (DMV) is a combination of a stated preference method and a participatory method. Small representative groups are selected to discuss and deliberate in an open and fair environment to express the value of ecosystem services in monetary terms (Spash 2007; Turner et al. 2010). Spash (2007) described four approaches to DMV:

- Charitable contribution – individual (disaggregated) values are provided by individuals in a group setting
- Fair price – individual (disaggregated) value is provided by a group
- Expressed social WTP / WTA – social values are provided by individuals in a group setting
- Arbitrated social WTP / WTA – social value is provided by a group.

Pros and Cons

The DMV method can be used to value both use and non-use values. It improves the knowledge of the issue at hand and allows individuals to look beyond immediate self-interest. It increases the likelihood of stakeholder engagement and encourages community participation in decision making. The use of small participatory groups may, however, lead to a lack of representation, selection bias, and may be subject to group norms (e.g. dominant effects).

Moreover, this method can be costly to implement and may not result in monetary estimates.

Application to soils
This method has been used in valuing improvements in water quality (Alvarez-Farizo & Hanley 2006; Robinson et al. 2008), but to our knowledge has not yet been used in a soil context.

Further readings

Spash CL 2007. Deliberative monetary valuation (DMV): issues in combining economic and political processes to value environmental change. Ecological Economics 63(4): 690–699.

Alvarez-Farizo B, Hanley N 2006. Improving the process of valuing non-market benefits: combining citizens' juries with choice modelling. Land Economics 82(3): 465–478.

Robinson J, Clouston B, Suh J, Chaioupka M 2008. Are citizens' juries a useful tool for assessing environmental value? Environmental Conservation 35(04): 351–360.

3.2.4 Benefits (or environmental value) transfer

Benefits transfer is often used when budget and/or time constraints limit the ability of conducting an original valuation study. When performing benefits transfer, results are taken from the context of one or several previously undertaken studies and transferred to a similar context specifically relevant for the project or policy of interest (Smith et al. 2002). The comparison of the two contexts can involve concordance in factors like the affected resource, the magnitude of damages or improvements, the existence of substitute resources, and the economic and demographic characteristics of the affected population.

There is a range of approaches available for benefits transfer. Frequently used approaches include transferring a single point estimate or a value range (referred to as unit value transfer) or transferring a value function. The transfer of a value function accounts for differences in various characteristics of the site and population. Value functions may be derived from single studies or methods that combine information from two or more studies (Johnston & Rosenberger 2010). For international value transfers, adjustments are recommended for purchasing power parity and income differentials. The more frequent the cultural and historical differences associated with values derived in other countries mean caution needs to be taken with transferring values between countries.

Bateman et al. (2010) suggests validation of the benefits transfer approach by ensuring:

- the source valuation studies are of sufficient quality
- similarity of good/service in the source studies to the new context (including the nature of the good/services and its quality and quantity)
- similarity of the contexts (e.g. characteristics of the site and the population, accessibility of the good/service, availability of substitutes and capacity, income constraints of the population)
- relevance of the source study explanatory variables and their value range to the new context and
- the relationships embodied within the value function reflect economic theory.

Benefits transfer methods are subject to both measurement errors of the primary study and transfer errors related to the transfer process. The acceptable level of error will depend on how risk averse those using the information from the new study area (e.g. policy decision-makers), the relative uncertainty of other data used in the economic analysis, and the costs of a primary study (Johnston & Rosenberger 2010).

Pros and Cons
This method is less costly and less time consuming than conducting original valuation studies. It can be used as a screening technique to determine the need for a primary valuation study. However, it is often hard to find suitable previous studies from which to transfer values. Resulting estimates are only as accurate as the initial value estimates. Furthermore, many factors can vary even when the contexts seem similar. Benefits transfer usually involves transfer not only across geographical space but also across time, introducing more variability.

Application to soils
Colombo et al. (2007) compared valuation estimates from two choice experiment applications to the benefit of reducing soil erosion in similar watersheds located in the southeast of Spain and found that for their study the unit value transfer approach was more suitable than the value function transfer approach.

Further readings

Bateman I, Brouwer R, Cranford M, Hime S, Ozdemiroglu E, Provins A 2010. Valuing environmental impacts: practical guidelines for the use of value transfer in policy and project appraisal. Available online at http://archive.defra.gov.uk/environment/policy/natural-environ/using/valuation/documents/technical-report.pdf (accessed 11 June 2013).

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Colombo S, Calatrava-Requena J, Hanley N 2007. Testing choice experiment for benefit transfer with preference heterogeneity. American Journal of Agricultural Economics 89(1): 135–151.

Johnston RJ, Rosenberger RS 2010. Methods, trends and controversies in contemporary benefit transfer. Journal of Economic Surveys 24(3): 479–510.

3.3 Challenges with economic valuation

While economic valuation can be an important component of decision-making, a number of challenges need to be acknowledged and considered when determining whether the monetary valuation of ecosystem services should be undertaken. The challenges include (but are not limited to):

1. Paucity of data. On the bio-physical side there are gaps in the underpinning science relating ecological processes to ecosystem services and to the production of goods and estimating how management decisions and external drivers will affect these services. On the economic side, there are limited valuation studies on different types of decisions in different contexts. While benefit transfer and other similar methodologies (Johnston & Rosenberger 2010) are used to overcome the absence of an original valuation study, their usefulness still depends on the availability of high quality valuation studies that compare well in terms of the affected resource, the magnitude of damages or improvements, the existence of substitute resources, and the economic, demographic and cultural characteristics of the affected population (Bateman 2011).

2. Inability to place an economic value on all ecosystem services. Some economists argue that economic principles do not necessarily hold for some shared social values such as spiritual values (Bateman et al. 2011). Hence, the use of stated-preference methods is not applicable. In New Zealand, the indigenous Māori peoples do not succumb to the belief that it is possible to place a monetary value on their spiritual relationships with the environment (Garth Harmsworth, Landcare Research, pers. comm., February 2011; Awatere 2008). Some alternatives offered have included the adoption of ecological standards (or a 'safe minimum standard') approach to preserve natural capital (Farmer & Randall 1998).

3. Double counting. The issue of double counting is now widely recognized and frameworks and alternative classification systems are evolving to address this. Boyd and Banzhaf (2007) suggest valuing only the final services. Fisher et al. (2009) note that where ecological processes are valued (e.g. nutrient cycling) this increases the potential for double counting as these processes may support many ecosystem services. When both ecological processes and ecosystem services are valued there is danger of overestimating the total values. Of course, a focus only on final services and their utility to humans may place the underlying ecological assets at risk if the unsustainable use of ecological assets or natural capital is not considered (Gren et al. 1994; Turner et al. 1999; Bateman 2011).

4. Values are time-bound. Both market and non-market values change over time. Market prices respond to changes in global, regional and/or local supply and demand. Non-market values change in response to changes in societal preferences or the external circumstances and drivers that may affect choices. The use of inflators or deflators to adjust for time does not address the underlying causes that may change the perception of value or prices over time.

5. Aggregation of values. As it is not unusual for valuations to have large uncertainty ranges, not incorporating them into an assessment is likely to provide inaccurate comparisons and assessments. Societal valuation methods focus on external value judgments about how individual values should be aggregated to determine the social welfare implications of a decision (EFTEC 2006; Slootweg & van Beukering 2008). When aggregating values, differing spatial and temporal scales need to be accounted for, and this can be challenging. Aggregating values can also mask the trade-offs that may exist between the costs and benefits of changes in the condition of ecosystems. Given these challenges, our ability to assess in monetary terms the benefits provided by ecosystems, or the costs generated by their degradation and loss is frequently limited (TEEB 2008). The adequacy and cost of ecosystem service valuations, together with their context of use, might hinder their practical applications, leading to a gap between the ambitions of ecosystem service valuation and the concrete achievements in terms of influencing decision-making (Laurans et al. 2013).

Consequently, it is important not to limit assessments to monetary values, but to include both qualitative analysis and quantitative assessment in other metrics than monetary value (TEEB 2008, Figure 5). Even simply listing the services derived from an ecosystem, using the best available knowledge, can help ensure the recognition of the full range of potential ecosystem services impacted by a given policy or decision and the consequent effect on human well-being (EPA 2009).

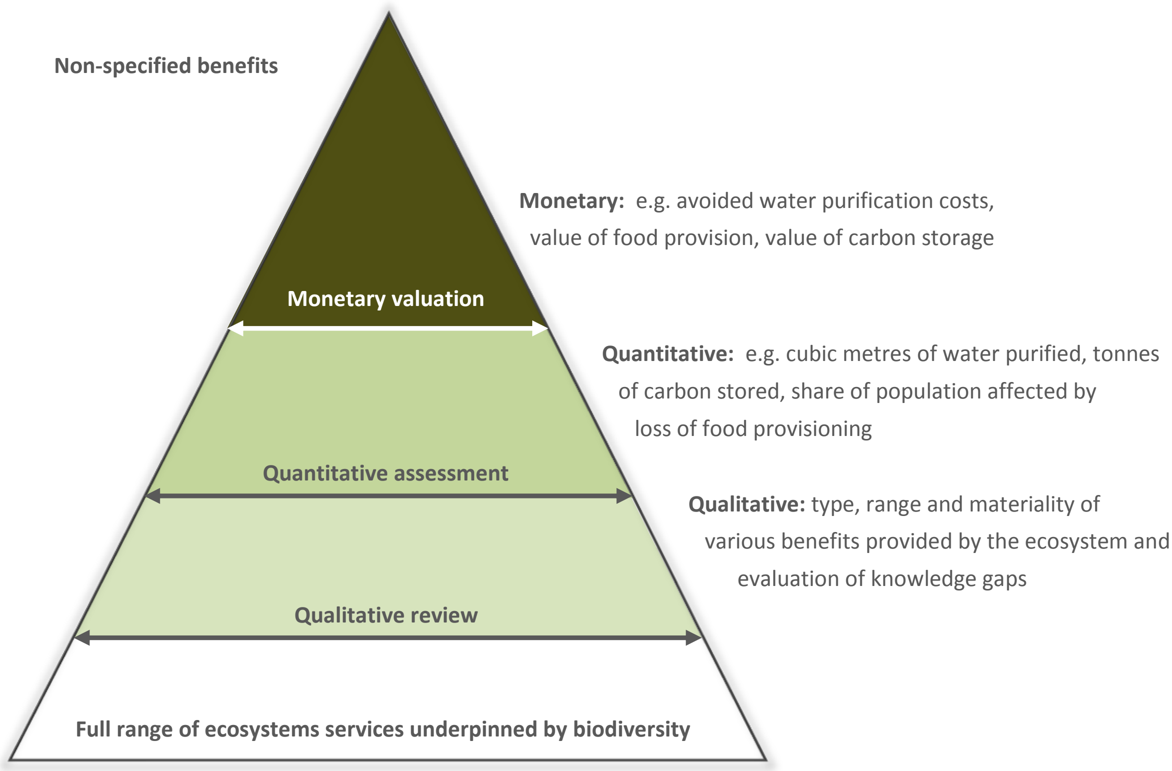

Figure 5: Valuing ecosystem services (Source: TEEB 2008).

3.4 Using economic valuation in decision-making

One aim of economic valuation is for the information to improve decisions by taking into account the costs and benefits of those decisions on the natural environment (EFTEC 2006; DEFRA 2007; TEEB 2009).

There is a range of frameworks, tools and approaches to support decision-making, a number of which are described in Table 3. These frameworks vary in the extent to which all relevant costs and benefits are incorporated, the metric used for their measurement, and the way time is treated. Multi-criteria analysis (MCA), which can use monetary and non-monetary values, can be as comprehensive as cost benefit analysis (CBA) in the inclusion of costs and benefits, and may even be more comprehensive once goals beyond efficiency and distributional incidence are considered.

All the remaining frameworks, tools, and approaches either deliberately narrow the focus on benefits alone, or ignore costs. Comprehensive CBAs rely on the monetisation of all relevant ecosystem services. However, this is not straightforward as obtaining the bio-physical, social and economic data to support the monetisation of ecosystem services may be expensive and challenging. Therefore, actual decisions are often made using a subset of the required information, compromising the theoretical analytical rigour of a CBA. It is argued that theoretical economists need a far better understanding of the pressures that affect actual decisions, while those who make actual decisions need a far better understanding of economics (Pearce et al. 2006). New frameworks are evolving to support an ecosystem service approach to decision-making (see Chapter 5) that does not rely solely on economic data.

Table 3 Decision supporting frameworks

Decision support frameworks	Approach	Advantages	Limitations	Application
Cost-benefit analysis (CBA)	CBA compares the benefits and costs of an option (e.g., project, policy), where benefits and costs are typically described in economic or financial terms.	Can be used to: • filter or rank options • find the most efficient option • indicate whether an objective is worth attaining	Only compares identified benefits and costs valued in monetary terms. Potentially very sensitive to discount rates. Does not consider equity implications (in distribution of benefit and costs). Trade-offs are hidden when the benefits and costs are reduced to a single value, the net present value.	Comparing investment options in soil remediation (Van Wezel et al. 2008). Comparing treatment of erosion (Walpole & Sinden 1997).
Cost-effectiveness analysis (CEA)	CEA compares the costs of alternative ways of producing the same or similar outcomes.	Often used when policy is constrained by existing environmental targets or objectives Can be used to: • find the most efficient (least cost) way of achieving a given objective • evaluating a single option for its ability to attain a standard / threshold given a fixed budget	Only compares identified costs valued in monetary terms. Does not indicate whether an objective is worth attaining. Difficulty in discounting of non-monetary units, especially any benefits. It is challenging to apply when the outcome is multidimensional.	Assessing conservation practices in reducing water erosion induced sedimentation (Zhou et al. 2009; Yang et al. 2010).
Multi-criteria assessment (MCA)	MCA evaluates the performance of each option against a set of criteria by weighting each criterion according to its relative importance.	Includes monetary as well as non-monetised societal and ecological values. Can be used to: • find a single most preferred option • rank options • evaluate and choose among alternatives • short-list a limited number of options for subsequent detailed appraisal • distinguish acceptable from unacceptable possibilities	Does not indicate whether an objective is worth attaining. Difficulties in selecting the set of criteria and weighting.	Assessment of contaminated sediment management technologies (Linkov et al. 2006; Oen et al. 2010). Evaluating alternative options for cleaning polluted soil (Hokkanen et al. 2000).
Optimisation model	Optimisation models find the most efficient combination of measures or options to meet a specified set of objectives given a specified set of constraints.	Data, information, theories, and empirical findings from various contributing disciplines are handled in a systematic and consistent way. Assumptions, theories and facts are made explicit.	Models tend to simplify complex systems Misinterpretation or arbitrary choice of disciplinary perspectives by the model. Complex models are difficult to calibrate and validate to real situations, and can lack transparency.	Comparing abatement policies for a combined reduction of soil acidification (Schmieman & van Ierland 1999). Comparing options for reducing soil erosion (Schuler & Sattler 2010) and salinity management options (Greiner & Cacho 2001).

Decision support frameworks	Approach	Advantages	Limitations	Application
Total factor productivity (TFP)	TFP investigates the impact of technological, environmental or organisational change on productivity. It reflects all factors that influence the relationship between inputs and outputs in a production system.	Total Factor Productivity is often seen as the real driver of growth within an economy.	High data needs- value and quantity time series data for each output and input. Requires a means of adding diverse output and input quantities into measures of total output and total input quantity. Criticised for not having meaningful units of measurement.	Land/ soil degradation (Jayasuriya 2003; Ali & Byerlee 2000).
Computable general equilibrium (CGE) models	CGE models portray the simultaneous operation of many markets. They are economy-wide models and are solved computationally using solution algorithms.	Models are comprehensive and flexible. Based on explicit micro-economic behavioural assumptions. Can be applied at different levels of aggregation, from local to district, region, country, group of countries, to global. Can model policy changes, shocks (e.g. droughts, floods), technology improvements, population growth or environmental degradation.	High data needs. Difficulty with parameterisation. Involves complex modelling. Sensitive to initial base conditions or assumptions.	Assess the impacts of erosion induced loss of soil productivity (Alfsen et al. 1996). Assess the impacts of soil degradation (Coxhead & Jayasuriya 1994; Wiig et al. 2001; Holden & Lofgren 2005).
Partial equilibrium (PE) models	PE models are similar to CGE models except they are not economy wide but focus on a single sector, e.g., agricultural sector, energy sector.	Models are comprehensive and flexible for the sector they represent. Based on explicit micro-economic behavioural assumptions. Can be applied at different levels of aggregation, from local to district, region, country, group of countries, to global. Can model policy changes, shocks (e.g. droughts, floods), technology improvements, population growth or environmental degradation.	High data needs. Difficulty with parameterisation. Involves complex modelling. Sensitive to initial base conditions or assumptions.	Assessing the cost of soil erosion to downstream navigation (Hansen et al. 2002). Assessing the impact of agricultural policy on soil erosion, water quality and greenhouse gas emissions (Greenhalgh & Faeth 2002; Greenhalgh & Sauer 2003; Daigneault et al. 2012).

Decision support frameworks	Approach	Advantages	Limitations	Application
Simulation models	Simulation models link economic behaviour to biophysical resource use and management changes by using mathematical relationships.	Can be used to simulate future scenarios. Allows experimentation.	Simplifies complex systems. They are complex models that may be difficult to calibrate and validate, and lack transparency.	Comparing impacts of land-use/land-management options (Coiner et al. 2001).
Life cycle analysis/ assessment (LCA)	LCA is an analytical method used to quantify the specified impact (e.g., energy use, greenhouse gas emissions) related to the production of a product/service, including all upstream inputs and downstream uses.	Comprehensive Standards (ISO) exist to standardize how they are developed and applied. Weighting can be used to reflect local environmental priorities.	High data needs Sensitive to system boundaries. Weighting can be difficult.	Assessing erosion induced potential desertification impacts of different human activities (Nunez et al. 2010). Soil salinisation/ acidification (Feitz & Lundie 2002).

4 POLICY INSTRUMENTS FOR SOILS

Soil degradation has serious consequences detrimental, for example, to water quality and quantity, food safety, climate change, and human health. To reduce soil degradation from human activity, and to safeguard the contribution of soils to the flow of ecosystem services, soils need protection. When considering how to maintain or enhance the suite of services provided by ecosystems, decision makers such as governments, businesses, and non-profit organizations may undertake actions that are influenced by policy or that create policy. In a soil context, policies may target soil erosion, the decline in soil organic matter and soil biodiversity, soil contamination and soil nutrient saturation, soil sealing, soil compaction, and salinisation. Some key policy areas that contribute to soil protection include: planning, agriculture, forestry, climate change, air quality, water quality and quantity, rural development, biodiversity, cultural heritage, contaminated land, and policy that supports research and education. This section outlines some commonly used policy instruments and where they have been used to protect soils or ecosystem services that are affected by soil condition.

Outreach and education

Outreach and education are important for highlighting the role of ecosystem services and the interactions between decisions and ecosystem services. Policies that create opportunities for outreach and education include ensuring easy access to up-to-date and credible information on ecosystem services and the role soils play in the provision of ecosystem services, raising awareness of soils and ecosystem services, promoting environmental education, and providing technical assistance on improving the quality or reducing the degradation of ecosystem services. Some examples include:

- S-Map Online, which provides fast, easy access to New Zealand soil data. It allows users to explore interactive soil maps, view detailed information about a soil class or attribute, create custom soil maps and download soil factsheets for specific locations.[3] Additional information on the properties of soils for growing pasture, leaching nutrients, etc. (which relate to ecosystem service provision) will be provided in future additions of S-map.
- The soil health environmental snapshot prepared by the New Zealand Ministry for the Environment. This environmental snapshot reports soil health from approximately 740 sites in 12 regions, sampled by regional councils between 1995 and 2009. Close to 300 of these sites were re-sampled to determine changes over time. The samples represent soils under indigenous land cover and five productive land uses: drystock, dairy, forestry, cropping, and horticulture. Seven soil measures were monitored to provide information about the organic reserves, fertility, acidity, and physical status of the soils. Collectively, these measures allow changes in soil health due to land management to be detected,[4] and could be used to understand how the flow of ecosystem services may have changed over time.
- Public lectures, e.g. 'Soil science and the challenge of agricultural production and environmental protection' organised by the Canterbury Branch of the New Zealand Royal Society.[5]
- Technical assistance provided by regional councils. For example, Horizons Regional Council's Sustainable Land Use Initiative (SLUI) provides support for the development of whole farm management plans and suggestions for improved management through good management practices for landholders on highly erodible lands.[6]

Regulatory instruments

Regulatory instruments are commonly known as command-and-control approaches. Environmental bans and restrictions, standards, and environmental caps (also known as quotas) are some of the commonly used regulatory instruments. Some examples of the application of regulatory instruments include:

- New Zealand's Environmental Risk Management Authority's (ERMA) ban on the use of endosulfan, an insecticide used on a wide range of fruit and vegetables and also on sports turf in New Zealand. Endosulfan has triggered international action because of its toxicity, its persistence in the environment, and its ability to accumulate up the food chain.[7]
- The National Environmental Standard (NES) for Assessing and Managing Contaminants in Soil to Protect Human Health was developed to deal with legacy soil contamination from the use of hazardous substances in industry, agriculture and horticulture. The standard provides a nationally consistent set of planning controls and numerical values for soil contaminant concentrations to protect human health, and ensures that contaminated land is appropriately identified and assessed before it is developed. If necessary the land is remediated or the contaminants contained to make the land safe for human use.[8]

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[3] http://smap.landcareresearch.co.nz (accessed 15 April 2013)
[4] http://www.mfe.govt.nz/environmental-reporting/report-cards/soil-health/2010/ (accessed 15 April 2013)
[5] http://canterbury.rsnzbranch.org.nz/pipermail/meetings_canterbury.rsnzbranch.org.nz/2012-June/000050.html (accessed 15 April 2013)
[6] http://www.horizons.govt.nz/managing-environment/sustainable-land-use-initiative-slui/ (accessed 14 April 2013)
[7] http://www.safefood.org.nz/endosulfanmedia08.php (accessed 15 April 2013)
[8] http://www.mfe.govt.nz/laws/standards/contaminants-in-soil/ (accessed 14 April 2013)

- Horizons Regional Council proposed a water quality regulation that allocated nutrient discharge allowances based on natural capital. Natural capital, for this context, was defined using Land Use Capacity (LUC) classes. LUC classes are a way to categorise land based on soil properties, climate, slope, and productive potential. In this instance, soil quality is used to underpin a regulation aimed at water purification services.

Economic instruments

Economic instruments provide incentives for behavioural changes to reduce the negative impacts of people's actions on ecosystem services. Price-based instruments and market-based instruments are two categories of economic instruments. Price-based instruments include taxes (e.g. polluter pays taxes), subsidies (e.g. payment for ecosystem services), tax credits, and low-interest loans. Market-based instruments include eco-labelling, environmental markets and auctions and tenders. Some examples include:

- Greenhouse gas markets that include carbon sequestered in soils. New Zealand does not currently account for changes in soil carbon under the Kyoto Protocol, and so soil carbon is not included in the NZ Emissions Trading Scheme. However, a number of voluntary markets are beginning to work on ways to measure and verify such carbon build up, e.g. Carbon Farmers of Australia.[9] The Chicago Climate Exchange (CCX) has also trialled the voluntary trading of soil carbon credits derived from better soil management (Chicago Climate Exchange 2009).
- The ecolabel, BioGro, promotes the highest level of organic integrity by certifying producers as organic. The standards used to audit and certify the processes carried out by organic producers influence soil condition, for example, by specifying allowable soil amendments.[10]
- The East Coast Forestry Project established in 1992 is a tender/auction programme to control erosion and stabilise hill sides in the erosion prone areas of Gisborne and the East Cape. The project provides grants, through a tender process, to land holders for planting radiata pine or other species, gully planting and actively managing the reversion of pastoral land to indigenous scrub/forest.[11]

Protected areas, land retirement and stewardship agreements
Protected areas, ecosystem restoration, land purchase for retirement and stewardship agreements are specifically aimed at ecosystem preservation and restoration. An example includes:

- Covenants established by the QEII National Trust are stewardship agreements that encourage and promote, for the benefit of New Zealand, 'the provision, protection, preservation and enhancement of open space'. The covenants place permanent restrictions on the use and management of open space with implications for the local soils and ecosystem services.[12]

Each policy instrument has strengths and weaknesses with regards political acceptability, cost of implementation, suitability for multiple ecosystem services, cost burden, innovation potential, and level of certainty in the environmental outcome. The appropriateness of a policy instrument depends on a number of issues such as the type of the target ecosystem service(s), the existing ecosystem conditions, external drivers influencing decisions, existing institutional and policy arrangements, the policy target (e.g. landowners, businesses), political will and available management or technology options for improving the ecosystem service(s). No single policy instrument is likely to provide the solution to an environmental problem. Rather, decision-makers will likely need to evaluate and implement a mix of policy instruments. The effective design of policy instruments is conditional on monitoring, research, and evaluation, while the implementation of policy instruments relies on the availability of suitable institutions, their capacity, and governance structures (Greenhalgh & Selman forthcoming).

Information on soil characteristics and functions may be important for the effectiveness and efficiency of environmental policy instruments. For example, for policies targeting the protection of lakes from eutrophication soil characteristics will influence nutrient leaching potential and the attenuation of nutrients as the nutrients move from their source to the lake or waterway. Often, however, such information is not readily available or there is not enough attention paid during the design of the policy instruments to the opportunities associated with such information. Where policy or legislation does relate to soil, it is generally limited to the protection of a specific impact or function of that soil. Policies relevant to soil protection tend to be fragmented, potentially confusing, and lacking coordination. Often policy instruments specifically developed with the protection of soil in mind are lacking.

[9] http://www.carbonfarming.org.nz/wp-content/uploads/InfoSheet_6_10-06-09web.pdf (accessed 15 April 2013)
[10] http://www.biogro.co.nz/index.php/ (accessed 15 April 2013)
[11] http://maxa.maf.govt.nz/forestry/publications/ecfp/ (accessed 15 April 2013)
[12] www.openspace.org.nz/ (accessed 15 April 2013)

5 APPLYING AN ECOSYSTEM SERVICES APPROACH

Paramount to the concept of ecosystem services is its use to inform decision-making processes, both government (e.g. policy, development) and business (e.g. investment and operational decisions). The ecosystem services approach can help evaluate the effect of a project or policy on ecosystems, including their soils, and the services they provide. The figure on the following page (Figure 6) is one decision-making framework that supports the practical application of an ecosystem services approach. In this chapter we describe this framework and through a series of case study boxes hypothetically illustrate how it may be used to highlight the role of soils in providing ecosystem services and their influence on decision-making.

The case study example refers to the hypothetical decision of whether to develop dairy and intensive pastoral farming in South Island high country. The context of the case study is outlined in Box 1, and is based on information gathered for a 2010 consent application to convert tussock grassland to dairy in the MacKenzie Basin of the South Island[13]. The case study example, however, is not based on the actual decision. Rather, it is to demonstrate how an ecosystem services approach can be applied in decisions where soil natural capital is a key consideration.




Photos: (left) Peter Scott, (top right) Warwick Scott, (centre right) Nicholas Head and (bottom right) Larry Burrows.

[13] A decision has since been made by the relevant government agencies to decline the consent applications and a recent agreement between the key stakeholders would allow both ecological restoration and intensive dairy farming to occur in the MacKenzie Basin (http://www.nbr.co.nz/article/deadlock-breaks-plans-mackenzie-country-intensive-dairying-bd-140029). However we only refer to the context of this application in our case study to hypothetically illustrate how the decision-making framework could be applied.

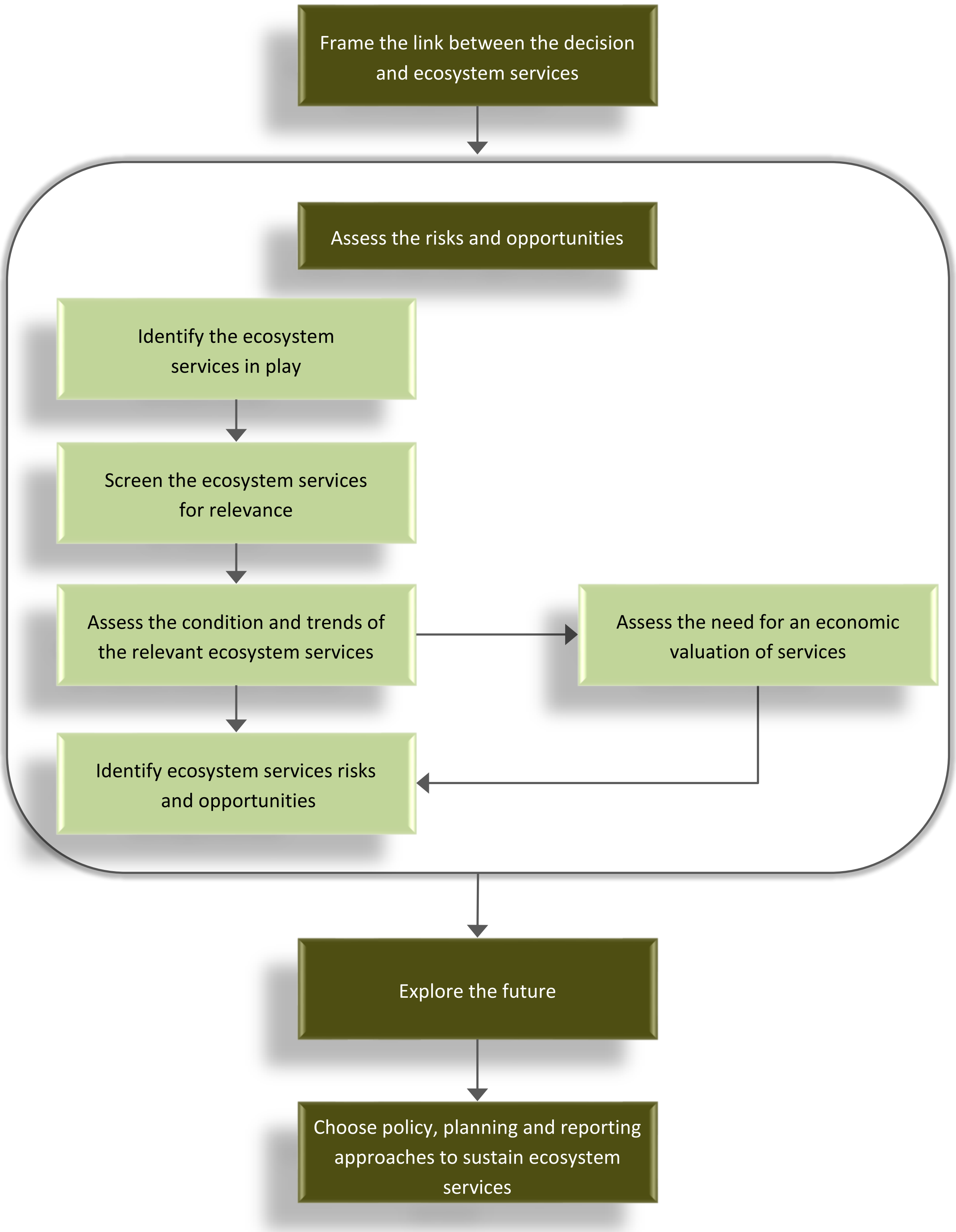

Figure 6: Ecosystem decision making framework; adapted from Ranganathan et al. (2008).

Box 1. South Island high country dairy intensification case study: context

Landscape: The high country is located near the centre of South Island in an intermontane basin.[14] The area has a distinctive physical environment with glacially derived landforms, extremes of cold, drought, wind, and shallow, stony, porous and infertile soils.[15] It comprises dry tussock grassland, large glacial lakes and snow-capped mountains that provide habitat to threatened plant, bird and fish species.[19] This landscape is treasured because of its biogeographical distinctiveness, the openness, naturalness, vastness of the huge expanses of tawny tussock grassland and underlying visible landforms.[19] It is also part of the biggest dark-sky reserve in the world – Aoraki-Mackenzie International dark-sky reserve.[16] Most of the land area has been under crown ownership and has been leased to farmers to be used for dryland extensive pastoral grazing.[19] Tussock grasslands contain large amounts of vegetative carbon and contributes to water capture and retention. The high country is a popular destination for tourists, tens of thousands of people visit the region each year, and international visitors to the region are forecasted to increase in the future.[17]

Decision: The decision relates to a conversion of tussock grassland to intensive pastoral farming. There have been resource consents lodged with the relevant District councils and Regional council for land-use change, effluent discharge, and irrigation in an area slated for conservation. The consents involve approximately 20 000 cows to be run on about 9000 ha. The estimated amount of effluent is close to 20 million litres per day (i.e. more than that of a city of 250 000 people[18]) and just over 5000 ha is slated for irrigation.[19] Farmers, industries (irrigation and power generation), recreational users, and environmentalists are all interested in the decision because of their conflicting interests.

Economic trends: Regarding current trends, New Zealand dairy production has risen 77% over the past 20 years. New Zealand dairy exports went to 151 countries (year ending 2009), key markets being China, the US, Japan, and the EU. The milk solids production reached 1555 million kg in 2012. Milk price increased from 472 cents per kg milk solids in 2009 to 687 cents in 2012 and it is forecasted to increase to 864 cents in 2015.[20]

Conversion Impacts: Globally, the Millennium Ecosystem Assessment (2005) found habitat change and pollution impacts on temperate grassland are increasing. In New Zealand, there were about 560 000 ha of tussock grassland remaining in 2008. Approximately 15 000 ha of indigenous grassland were lost between 2001 and 2008 (Weeks et al. 2013), mostly from conversions to exotic pasture involving over-sowing of the tussock grassland with legume species and exotic grass forage species. Often conversions were accompanied by irrigation infrastructure and increased application of fertiliser to overcome the productivity limitations of the otherwise infertile soils.

Step 1: Frame the link between ecosystem services and a decision

The first step in identifying the ecosystem services impacted by a decision and on which a decision depends is to identify the list of potential ecosystem services relevant to the decision. This will take account of both location and use of the ecosystem. The ecosystem services listed in the MEA (2005) can be used as a first pass to identify the potential ecosystem services and the impacts and dependencies of the decision on these ecosystem services. When identifying relevant ecosystem services, current and future use of the ecosystem is important as this will affect how soils contribute to ecosystem services. Box 2 illustrates how Step 1 may apply to the hypothetical case study.

Box 2. South Island high country dairy intensification: Establishing links between the decision and ecosystem services

The tussock grassland ecosystem in the South Island high country is providing provisioning services such as food (sheep meat) and fibre (wool), regulating services such as erosion control, water filtration and flow regulation as well as a series of cultural services, especially aesthetic, recreational and spiritual. Given the high country's extreme climate conditions and infertile soils, irrigation, soil cultivation and added nutrients will be needed to successfully establish intensive pastoral farming.[19] The native plants and animals that evolved over millions of years to survive the extreme climate would not survive in irrigated improved pastures.[21] Thus, native tussock grassland and its biodiversity will be lost transforming the landscape for intensive farming.[19] The change in the ecosystem will lead to changes in the ecosystem services provided. The irrigated pasture is associated with provisioning services such as food (dairy milk solids). The loss of native grassland is expected to impact negatively on provisioning services such as fibre (wool), regulating services such as water flow regulation, and cultural services such as recreation.

[14] http://www.edsconference.com/content/docs/2010_papers/Maturin,%20S%20paper.pdf
[15] http://www.landcareresearch.co.nz/research/obi/public/WalkerMackenzieSymposium_FINAL.pdf
[16] http://www.stuff.co.nz/the-press/news/7074544/Southern-skies-get-starlight-reserve-status
[17] http://www.forestandbird.org.nz/files/file/HighCountry.pdf
[18] http://www.beehive.govt.nz/release/minister-calls-mackenzie-basin-dairy-discharge-consents
[19] http://www.stuff.co.nz/timaru-herald/news/6023682/Dairy-water-plans-denied
[20] http://www.mpi.govt.nz/agriculture/pastoral/dairy.aspx
[21] http://www.stuff.co.nz/the-press/opinion/perspective/4385643/Plotting-Mackenzie-Countrys-future

Step 2: Assess risks and opportunities related to ecosystem services

The assessment of risks and opportunities related to ecosystem services starts with the identification of ecosystem service dependencies and impacts and their screening for relevance in the decision making context. A decision depends on an ecosystem service if the service serves as an input or enables, enhances, or influences the conditions necessary for a successful outcome in relation to the decision; while a decision impacts an ecosystem service if actions associated with the decision alter the quantity or quality of a service (Ranganathan et al. 2008). Many assessments account for the impacts of a decision but fail to acknowledge the services that a decision depends upon. Dependencies and impacts can be direct or indirect and have various spatial and temporal patterns.

Once the relevant ecosystem service dependencies and impacts are identified, the current condition and future trends of those ecosystem services can be assessed. This will provide the baseline for investigating how the decision will affect the condition and trend of those relevant ecosystem services. Remote sensing, Geographical Information Systems (GIS), inventories, ecological models, participatory approaches and expert opinion are some of the methods that can be used to assess the condition and trend of ecosystem services (Ranganathan et al. 2008).

An optional tool to support decision-making is economic valuation. This can be used to estimate the magnitude of the anticipated changes in the flow of ecosystem services in monetary terms. In this context the characterisation of ecosystem services based on ownership status will reveal who stands to gain and who stands to lose from the changes.

Thinking of ecosystem service changes in terms of trade-offs can be helpful in identifying risks and opportunities. Trade-offs are caused by management choices or actions that intentionally or otherwise alter the quantity or quality of an ecosystem service to achieve a goal where some win and others lose. The following questions can be used to identify risks and opportunities associated with ecosystem service dependencies and impacts:[22]

- Does the decision depend on ecosystem services that were either previously unrecognized or in poorer condition than previously known?
- How do soils contribute to the provision of that service and does that soil contribution rely on the dynamic or inherent properties of soil?
- Could the goals of the decision be jeopardized because users are competing for an ecosystem service in limited supply and do these services rely on the inherent properties of soils? If so, are cost-effective substitutes available?
- Are there any unforeseen impacts of the decision on ecosystem services that others depend on for their well-being?

Box 3 continues the case study using an example that highlights the importance of soils.

Box 3. South Island high country dairy intensification: Assessing the ecosystem service risks and opportunities – an example highlighting the importance of soils

A highly relevant ecosystem service in the context of this decision is the provisioning service of dairy milk solids. Although the decision aims to increase this service in the South Island high country area (opportunity), its success depends on the ability to establish exotic pastures on tussock grassland. This, in turn, depends on high nutrient supplies (it could be argued that in this managed pasture and livestock production context the soil's nitrogen concentration, and supply to plants, is a final ecosystem service). Due to local soils being naturally low in nutrients, they will not be able to supply the necessary quantity of nutrients. The low availability of this soil service means fertiliser will need to be applied. Adding fertiliser will have upstream and downstream effects on ecosystem services. Upstream effects are linked to the extraction of raw materials, manufacturing, and transportation of fertilisers. Downstream effects could include the loss of excess nutrients into water bodies which would affect services such as the freshwater provisioning service of adequate quality for recreation and other purposes (risk); these effects are expected to be local and regional. If required, value of the deterioration in water quality can be estimated in monetary terms, for example, by using the provision costs or defensive expenditures approaches. The value of change in recreational services can be measured with the travel cost method.

Step 3: Explore the future

Future scenarios are assumptions about a variety of possible future events, changes in societal preferences or composition of ecosystems, where existing conditions are not expected to continue and current trends are not expected to extend into the future. Exploring future scenarios is important to assess the effectiveness of a decision and to think about possible changes in relevant ecosystem services in the future, especially when uncertainty exists. Scenarios are formulated using potential indirect and direct drivers of ecosystem services (Table 4). Future scenarios can help understand linkages between policy options and the impacts and dependencies on ecosystem services. Analysing and comparing the outcomes of different scenarios can reveal and help policy makers avoid unintended consequences and gain a better understanding of ecosystem service trade-offs. Moreover, it can help resolve conflicts among stakeholders in relation to ecosystem services and the choice of policies for sustaining those services (Ranganathan et al. 2008). The outcome of scenarios can be analysed using several methods, such as computer-based simulation models, biophysical models, ecological models, participatory methods involving stakeholders, and expert opinions. Box 4 demonstrates how scenarios can be used to assess potential futures in the context of the intensification case study.

[22] Adopted from Ranganathan et al. (2008)

Table 4: **Indirect and direct drivers of ecosystem change (MEA 2005)**

Indirect drivers of ecosystem change	Direct drivers of ecosystem change
• Demographic • Economic (e.g. globalization, trade, markets, policy) • Socio-political (e.g. governance, institutional and legal framework) • Science and technology • Cultural and religious (e.g. beliefs, consumption choices)	• Changes in local land use and cover • Species introduction and removal • Technology adaptation and use • External inputs (e.g. fertilizer use, pest control, and irrigation) • Harvest and resource consumption • Climate change • Natural, physical, and biophysical drivers (e.g. evolution, volcanoes)

Box 4. South Island high country dairy intensification: Exploring futures

Derived scenarios can be used to explore how the opportunities and risks associated with ecosystem service dependencies and impacts identified for the South Island high country dairy intensification might play out in the future. Some drivers that could be considered as part of the scenarios are:

- increased international demand for milk products (current trend)
- tourism grows substantially with increased growth in wealth of Asian countries
- climate change may result in more frequent and more prolonged droughts
- limits setting process for freshwater management results in high/low limits for water extraction and/or high/low water quality limits
- projected fertiliser prices
- break-through technologies discovered for pollution control or pasture utilisation

The soil's role will be important for the impacts of some of the drivers. For example, if dairy intensification proceeds, additional pasture biomass will be needed to provide feed for the cows (i.e. to increase food provisioning services).

However, the soils are relatively infertile in the area. Increase in pasture growth will require both additional fertiliser and also additional water as soils in the area to retain nitrogen, whether from urine or manure spreading or fertiliser application is low. This has adverse effects on water quality, as water purification services are not sufficiently able to filter the additional nutrients. Therefore, soil properties and fertility will impact on the ability to grow sufficient pasture to increase milk supply (without supplementing fertility with substitutes like fertiliser), limit setting processes, the need to respond to changing fertiliser prices and the importance of break-through technologies. Similarly, many soils in the area have low soil water holding capacity, which increases the need for irrigation to grow pasture. Again, soil properties will impact on milk supply, limit setting processes and the risk of climate change induced drought.

When assessing how to address the impact of drivers, the dynamic soil properties become important, as does the cost of substitutes. Poor soil fertility may be cost-effectively addressed through fertilisers, poor soil water holding capacity through irrigation and poor nutrient retention through good management practices. However, the aggregate cost of these substitutes may overwhelm the benefits from additional milk production.

Step 4: Choose policy, planning and reporting approaches to sustain ecosystem services

After the impacts and dependencies are identified, the resulting risks, opportunities, and scenario exploration can help understand the trade-offs that may need to be made. In reality, every decision is likely to have a mix of positive and negative impacts on ecosystem services, and will involve trade-offs between ecosystem services. There are a number of methodologies that can help decision makers make trade-offs between ecosystem services (see some of the decision support tools in Chapter 3), and policy instruments (see Chapter 4) can be chosen and designed to minimise trade-offs. The intensification case study example is continued in Box 5.

Box 5. South Island high country dairy intensification: Policy, planning and reporting

The decision to intensify dairy should be made in the context of any existing relevant policy. Some existing national, regional or district legislation to consider would be the National Policy Statement on Freshwater Management, Canterbury Regional Policy Statement, The Canterbury Water Management Strategy, any relevant district or regional plans,[23] as well as initiatives like the Dairying and Clean Streams Accord.

The risks and opportunities with the proposal to intensify dairy production and the assessment of potential future scenarios indicate there will be trade-offs between different ecosystem services. If it is decided to intensify dairy production then the food provisioning services will increase with the increase in dairy production. However, many of the regulating and cultural ecosystem services are likely to degrade. If a decision is made, instead, to create a dryland conservation park, the food provisioning services will decrease while many regulating and cultural services will increase. In this instance, there are trade-offs between the provisioning services and regulating and cultural services. Any policy instrument chosen and its design would need to minimise negative impacts.

For instance, if a dryland conservation park is established (protected area) then even though lost food provisioning services may not be able to be compensated for, the lost revenue from this service not being provided could be compensated by efforts to increase the tourism income stemming from the creation of the park.

[23] http://ecan.govt.nz/our-responsibilities/regional-plans/Pages/rps-regional-plans.aspx

6. CONCLUSION

We explored soil natural capital and ecosystem services as a new lens to look at our soils. The framing of soil as a natural capital stock yielding a flow of valuable goods and services into the future is expected to help convey the importance and value of soil to a wider society with a predominantly human centric utilitarian viewpoint.

With a series of recent attempts that build on and complement each other, there has been progress in the definition of the soil natural capital concept. This is proposed to refer, for example, to the stocks of mass and energy in the soil and their organisation/entropy. While there is more clarity on what soil natural capital is, evaluating soil stocks and determining how they change with time remain challenging. However, monitoring soil stock condition is critical to assure the provision of certain ecosystem services does not happen at the expense of changes in the soil natural capital stock that would compromise the flow of services in the long term.

Trying to isolate soil ecosystem services is also challenging because soils are only one component of an ecosystem and their role changes depending on the ecosystem and its use. The Millennium Ecosystem Assessment framework, arguably the most widely used framework for the classification of ecosystem services, has been applied to soils. This framework differentiates between provisioning, regulating, cultural, and supporting services. Such a classification can be complemented by the spatial and temporal characterisation of ecosystem services to identify where and when the services are delivered.

It is anticipated that by valuing changes in ecosystem services in common units – usually, but not always monetary – the contribution of ecosystems, including soils, to human well-being will be recognized and incorporated in societal decision making.

A series of economic valuation techniques have merged to address the diverse nature of goods and services provided by ecosystems, and the various motivations for assigning them economic value. The techniques rely on actual, surrogate or hypothetical markets to observe revealed or stated preferences for changes in ecosystem services. There are examples of these techniques being applied in a soil context. The economic valuation of changes in ecosystem services is hindered by the paucity of data on both the bio-physical and economic sides, the inability to place an economic value on all services or the double-counting of some services, the time-bound character of value estimates, and the lack of agreement on many aspects of value aggregation.

With or without monetary valuation, the application of an ecosystem service approach is proposed for both government and business to help evaluate the impact and dependency of a decision on ecosystems, including the soils and the services they provide. The case study of the hypothetical South Island high country dairy intensification showcased the practical application of the ecosystem approach by establishing the link between the decision and ecosystem services, assessing the associated risks and opportunities, exploring the future, and choosing policy, planning, and reporting approaches to sustain ecosystem services.

Given these developments in the conceptualisation of soils as natural capital, in the assessment and valuation of the ecosystem services soils provide, and in the application of an ecosystem services approach to informing decision-making, there is an opportunity for an enriched perspective on the value of our soils and how we can articulate those values.

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APPENDIX A – SOIL INDICATORS

Study	Context	Scale	Indicators
Schipper LA, Sparling GP 2000. Performance of soil condition indicators across taxonomic groups and land uses. Soil Science Society of America Journal 64: 300–311.	Assessing soil condition under different land uses	National scale	• Total Carbon • Total Nitrogen • Cation exchange capacity • Olsen phosphate • pH • CO2 production • Microbial biomass • Potentially mineralisable N • Bulk density • Moisture release • Hydraulic conductivity • Particle size distribution and particle density
Lilburne L, Sparling G, Schipper L 2004. Soil quality monitoring in New Zealand: development of an interpretative framework. Agriculture, Ecosystems & Environment 104(3): 535–544. Sparling G. P. and Schipper, L. A 2004. Soil quality monitoring in New Zealand: Trends and issues arising from a broad-scale survey. Agriculture Ecosystem & Environment 104: 545–552. Sparling G. 2005. Environmental indicators for land: overall soil quality in the Waikato Region 1998–2004. Environment Waikato Technical Report 2005/47. Available online at http://www.waikatoregion.govt.nz/Services/Publications/Technical-Reports/Environmental-indicators-for-land-Overall-soil-quality-in-the-Waikato-region-1998–2004/ (accessed 11 June 2013)	Assessing the 'life supporting capacity of soil' and whether current practices will meet the 'foreseeable needs of future generations' under five different land uses	Regional and national scale	• Total Carbon • Total Nitrogen • Anaerobic mineralisable nitrogen • pH • Olsen phosphate • Bulk Density • Macroporosity
Beare MH, Lawrence EJ, Tregurtha CS, Harrison-Kirk T, Pearson A, Meenken ED 2005. Crop & Food Research Confidential Report No. 1408 New Zealand Institute for Crop & Food Research Limited. Available online at http://maxa.maf.govt.nz/sff/about-projects/search/02-125/02125-finalreport.pdf (accessed 07 June 2013).	Developing the Land Management Index (LMI) to promote economic and environmental sustainability of agricultural soils under six different land uses	Paddock scale	• Total Carbon • Total Nitrogen • Hot water extractable carbon • pH • Olsen phosphate • Bulk Density • Aggregate stability • Penetration resistance • Soil texture • Soil organic matter • Percentage of potentially erodible aggregates • Percentage of large dense aggregates • Aggregate stability

Study	Context	Scale	Indicators
Statistics New Zealand 2009. Measuring New Zealand's Progress Using a Sustainable Development Approach: 2008. Wellington, Statistics New Zealand.	Measuring sustainable use of land and soils	National scale	• Area of land used for farming • Soil health - ○ phosphate content ○ pH in water ○ total carbon ○ total nitrogen ○ bulk density ○ ability to absorb water • phosphorus content in soil • Contaminated soil sites • Versatile soil extinction • Hill country erosion
ARC (Auckland Regional Council) 2010. State of the Environment and Biodiversity. Available online at http://www.aucklandcity.govt.nz/council/documents/technicalpublications/Chapter%204_2%20-%20Land.pdf (accessed 7 June 2013)	Measuring the stability of the land to understand how well the land resource is remaining in place so that it continues to be available for urban use, farming, forestry and conservation across the Auckland region	Regional scale	• Land stability • Soil disturbance • Bare soil
MfE (Ministry for the Environment) 2010. Soil Health Environmental Snapshot. Wellington, Ministry for the Environment. Available online at http://www.mfe.govt.nz/environmental-reporting/report-cards/soil-health/2010/#_edn3 (accessed 07 June 2013)	Measuring the health of soils for environmental snapshot reports - soils under indigenous land cover and five productive land uses.	National scale based on data from regional monitoring sites	• Organic reserves - ○ Total carbon ○ Total nitrogen ○ Mineralisable nitrogen • Fertility - Olsen phosphorus • Acidity - pH • Physical status ○ Bulk density ○ Macroporosity