

NATURAL CAPITAL AND ECOSYSTEM SERVICES OF SOILS

Estelle J. Dominati

AgResearch, Grasslands Research Centre, Tennent Drive, Private Bag 11008, Palmerston North 4442, New Zealand

ABSTRACT: The concepts of natural capital, ecological infrastructure, and ecosystem services are reviewed for soils and managed landscapes. Translating theoretical frameworks and insights addressing soil functioning into operational models and tools for resource management remains a challenge. Six general principles (steps) of a new methodology for quantifying soil ecosystem services are presented: differentiate soil services from the supporting processes; identify the key soil properties and processes behind each soil service; distinguish natural capital from added or built capital; identify where and how external drivers affect natural capital stocks; analyse the impact of degradation processes on soil natural capital; base any economic valuation on measured proxies. The methodology is then demonstrated for a pastoral soil under dairy use. The methodology and examples presented here comprise a work in progress but represent an advance in defining and quantifying soil ecosystem services. Finally, applications of an ecosystems approach to resource management are discussed, including challenges and future options. The development of technologies for land use should, for example, switch from overcoming limitations to investing in ecological infrastructure that will increase the natural capital of soil and enhance the provision of ecosystem services.

Key words: ecological infrastructure, ecosystems approach, ecosystem services, managed ecosystems, natural capital, resource management, soil.

INTRODUCTION

The finite capacity of Planet Earth is rapidly being exceeded. A recent report to the United Nations (Costanza et al. 2012) emphasises the need for new economic models and agricultural systems ‘that respect planetary boundaries and recognize that the ultimate goal is sustainable human well-being and not growth of material consumption’ (Rockström et al. 2009).

If, as Lester R. Brown argues in his latest book (Brown 2012), ‘food is the new oil and land is the new gold’, then we should be alarmed at how increased soil erosion, desertification, salinity, and the expansion of urban areas over our best and most versatile soils are rapidly making the most productive land scarcer. Awareness of these problems has led to increasing interest over the last decade in an ecosystem services approach to resource management (Banwart 2011). Despite ongoing debate on the nature of ecosystem services and on methodologies of assessing them (Boyd and Banzhaf 2007; Fisher and Turner 2008; Braat and de Groot 2012), the ecosystem services approach is very attractive for land management and decision making, because of its integrative nature (Banwart 2011; Faber and van Wensem 2012). There is a general agreement that standardised methods for quantifying ecosystem services are overdue (Haygarth and Ritz 2009; Robinson et al. 2009; Dominati et al. 2010a; Robinson and Lebron 2010; Rutgers et al. 2011; Faber and van Wensem 2012).

The failure to fully appreciate the contributions of soils to human welfare beyond food production can be traced to the fact that the full range of ecosystem services is usually not adequately quantified, and therefore not included in financial balance sheets alongside commercial services and built capital (Costanza et al. 1997; Braat and de Groot 2012). Only in the last 15 years have attempts been made to place economic values on ecosystem services (Costanza et al. 1997) and agro-ecosystems (Sandhu et al. 2008; Porter et al. 2009; Breure et al. 2012). To satisfy the growing appeal of an ecosystem services approach for resource managers and decision-makers (Braat and de Groot 2012; Robinson et al. 2012a), methodologies and operational models are required that can quantify the whole range of services.

This chapter presents the concepts of natural capital, ecological infrastructure, and ecosystem services, and examines framework development and the ecosystem services supply chain in

the context of managed ecosystems including soils. The chapter also presents a new methodology to quantify ecosystem services provided by soils, and finally discusses the challenges associated with such concepts, and their use for resource management.

CONCEPTS AND FRAMEWORKS

To find a less destructive and more sustainable way forward, we must protect some ecosystems from development and better manage those we use for production. The ‘ecosystems approach’ to resource management focuses on how to better manage our natural resources (Convention on Biological Diversity principles of the ecosystem approach, <http://www.cbd.int/ecosystem/principles.shtml>), and recognises the wide range of benefits from the harvested goods and ecosystem services they deliver. This requires better representation of natural ecosystems in decision-making frameworks (Robinson et al. 2012b).

The ecosystem services approach offers the ability to explore the influence of land use and practices on natural capital stocks, on the processes that build and degrade these stocks, and on the flow of ecosystem services from the use of these stocks (Dominati et al. 2010a). However, translating theoretical frameworks and insights into operational models and tools remains a challenge.

The quantification of ecosystem services also faces an ongoing challenge: the absence of standardised definitions (Dominati et al. 2010a; Rutgers et al. 2011). Boyd and Banzhaf (2007) and Wallace (2007) used ecosystem components (i.e. natural capital stocks) instead of processes as proxies for services because the structure and composition of ecosystems are better known than the processes involved in soil functioning. Robinson et al. (2012b) also prefer soil stocks for quantifying soil ecosystem services, for two reasons: flows can be inferred from stocks, and soil stocks are either available from existing soil surveys and land resource inventories or can be readily measured (Robinson et al. 2009; Dominati et al. 2010a; Robinson and Lebron 2010; Balmford et al. 2011; Rutgers et al. 2011). When assessing if resources are being sustained, it is important to separate the contribution of soil natural capital (soil stocks) from the contribution of the added capital (infrastructures, inputs such as fertilisers or irrigation water) in the provision of each service.

Soil natural capital

For soil science and agronomy, natural capital is perhaps the most intuitive concept because it focuses on stocks, which are routinely measured and inventoried in soil surveys. Soil stocks are the building blocks of the soil's infrastructure so maintaining and developing them is key to delivering ecosystem services. Costanza and Daly (1992) used a generic approach, defining natural capital as a stock of natural assets yielding a flow of either natural resources or ecosystem services. More recently, Palm et al. (2007) defined soil natural capital as texture, mineralogy and soil organic matter, and Robinson et al. (2009) (Figure 1), recognising the importance of connections and organisation of the stocks, added 'matter, energy and organization'. In a further refinement, Dominati et al. (2010a) (Figure 2) differentiated between *inherent* and *manageable* soil properties, similar to the *inherent* and *dynamic* notions used by Robinson et al. (2009). These concepts attempt to differentiate between stocks that change slowly through pedological processes and those that can be changed by management. Inherent soil properties typically include soil depth, texture, and mineralogy, which cannot readily be changed without significantly modifying the soil or its environment (Dominati et al. 2010a), while manageable or dynamic soil properties typically include nutrient content, organic matter, macroporosity and soil moisture, all of which can be influenced by land use. Robinson et al. (2012b) synthesised these concepts with soil biology concepts (Barrios 2007) by splitting the capital stocks into abiotic and biotic pools (Figure 1) and recognising the constant fluxes of materials between pools. It is these fluxes that contribute to soil formation and development: Dominati et al. (2010a) call them 'supporting processes' (Figure 2) as they are not services but underpin them.

The state of soil natural capital stocks determine soil quality. Farmers are familiar with this concept and continually explore ways to supplement stocks or compensate for a lack of soil natural capital. Most commonly, they supplement soil natural capital with added capital or built capital, which is associated with technologies that replenish and lift the productive capacity of soils; for example, they use fertilisers or animal wastes to replace depleted

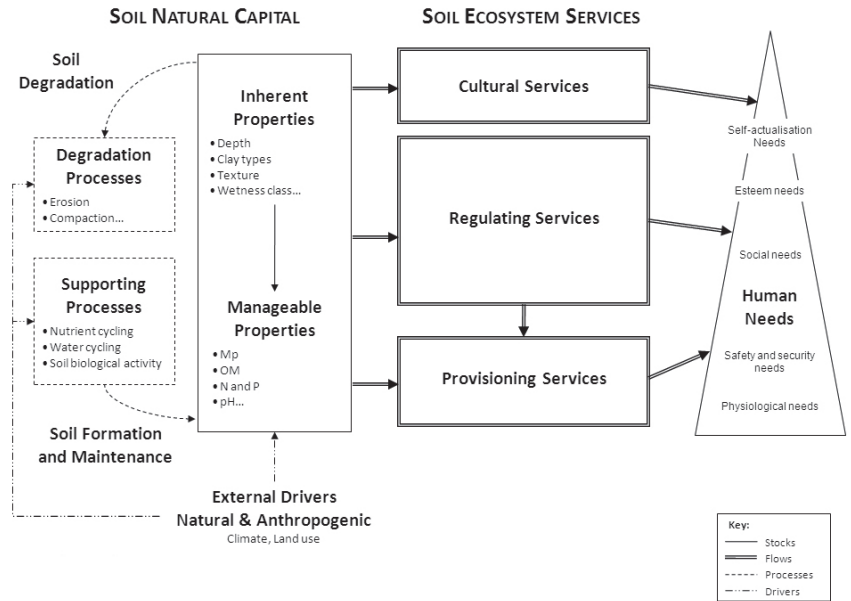


FIGURE 2 Conceptual framework linking soil natural capital and functioning to ecosystem services provision and human needs (Dominati et al. 2010).

nutrients, and irrigation to overcome limited water supplies or water holding capacity. However, a critical precondition for assessing the sustainability of land uses is the need to identify where soil natural capital stocks are limiting and how they can be improved (Mackay 2008; Dominati 2011).

Ecosystem services frameworks and soils

Existing frameworks for ecosystem services (Costanza et al. 1997; de Groot et al. 2002; MEA 2005; TEEB 2010; Balmford et al. 2011) fall short in their interpretation of how soils supply ecosystem services (Dominati et al. 2010a, b; Robinson and Lebron 2010; Robinson et al. 2012a), and this limits their use for progressive land management within ecological boundaries.

The Millennium Ecosystem Assessment (MEA 2005) (Figure 3) demonstrated the link between ecosystems and human well-being, while introducing the concept of ecosystem services. Most work on ecosystem services in agro-ecosystems uses the MEA (2005) framework (Swinton et al. 2007; Zhang et al. 2007; Porter et al. 2009; Sandhu et al. 2008). However, the MEA framework consigns the contribution of soils to 'supporting services'; for example, it mentions 'soil formation' as a supporting service and states simply that 'many provisioning services depend on soil fertility' (MEA 2005). Moreover, while it mentions the role of soils in the provision of regulating services like flood mitigation, filtering of nutrients and waste treatment, it does not explicitly identify the role of soils in providing these services, nor, more generally, in providing services from above-ground ecosystems.

There is now international agreement (CICES 2011) on the status of these 'supporting services'. The Economics of Ecosystems and Biodiversity initiative (TEEB 2010) removed supporting services from their framework for ecosystem services because these do not directly benefit society; TEEB now refers to them as 'biophysical structure, processes and functions' (Figure 4). Similarly, Dominati et al. (2010a, b) preferred the term 'supporting processes' because this emphasises the important differences between processes and ecosystem services. These differences must be considered very carefully when assessing the provision of ecosystem services, because the role of soils in providing services could

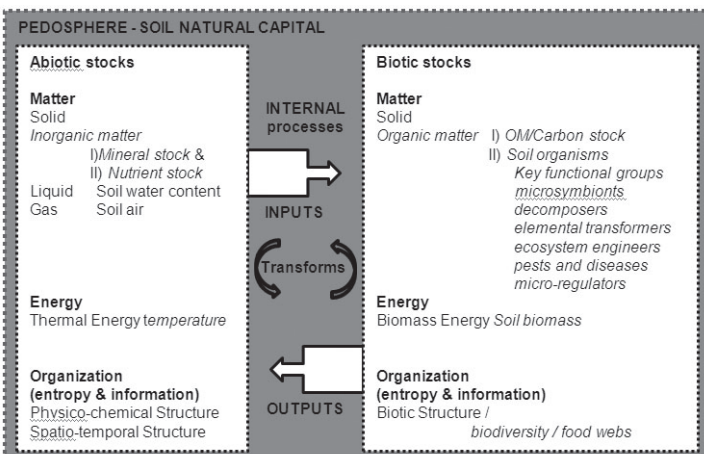


FIGURE 1 Soil natural capital stocks using a matter, energy and organisation framework (Robinson et al. 2009) divided between abiotic and biotic components (Robinson et al. 2012b).

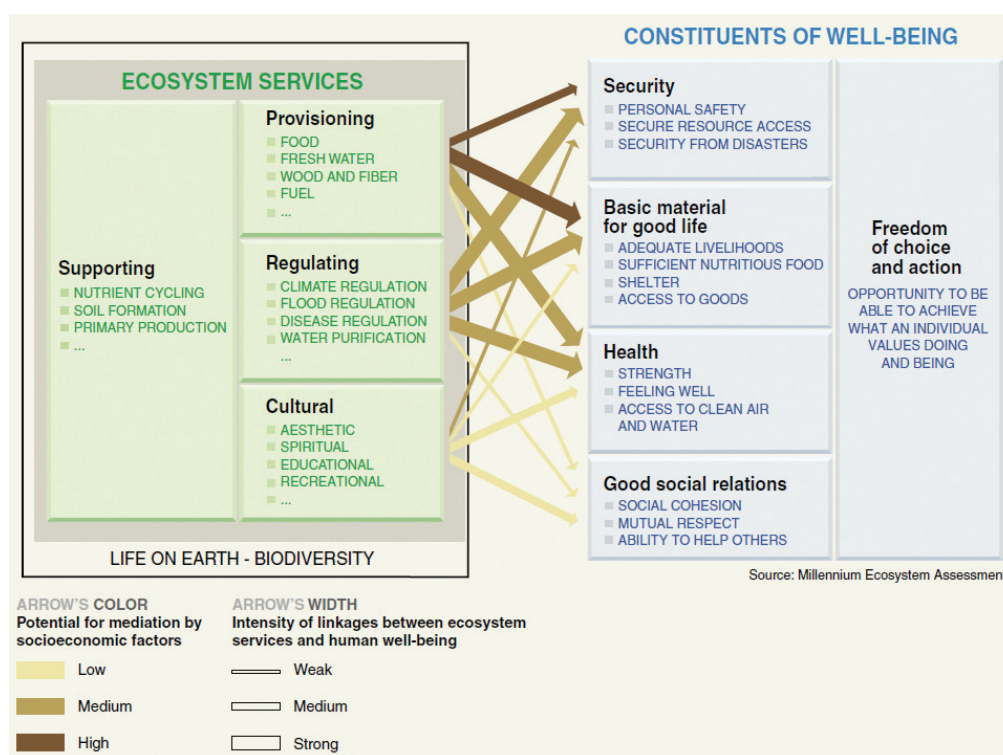


FIGURE 3 Millennium Ecosystem Assessment framework for ecosystem services (MEA 2005) (<http://www.unep.org/maweb/en/GraphicResources.aspx>).

be overlooked; therefore, general ecosystem services frameworks need to be extended to explicitly detail the relationships between soil stocks, soil processes and services, because all these factors contribute to the ecosystem services supply chain.

Soils are essential for the provision of services to human society, including buffering of floods, being a substrate for plant growth, and recycling wastes. These services have been described

in detail by Andrews et al. (2004) and Wall et al. (2004), and also by Daily et al. (1997), who noted that soils are a valuable asset that take ‘hundreds to hundreds of thousands of years to build and very few to be wasted away’ (p. 113). The crucial role of soil biology in the functioning of soils has prompted increasing interest in how below-ground biota and microbial communities support these processes and thereby provide services (Wall et al. 2004; Bell et al. 2005; Barrios 2007; de Bello et al. 2010; Gianinazzi et al. 2010; Guimarães et al. 2010; Smukler et al. 2010; van Eekeren et al. 2010; Hedlund and Harris 2012; Keith and Robinson 2012; Wall 2012).

Until recently, frameworks detailing the ecosystem services provided specifically by soils (Daily et al. 1997; Wall et al. 2004) did not distinguish stocks from flows. Dominati et al. (2010a, b) (Figure 2) addressed this by developing a framework that shows how some ecosystem services flow from soil natural capital stocks; this framework links changes in the status of a soil resource (also known as natural capital) under a use, to the provision of ecosystem services. More generally, the idea of a ‘service cascade’ leading from the ecological infrastructure to human well-being (Figure 4; Haines-Young

TABLE 1 Ecosystem services provided by soils (Dominati et al. 2010)

Service	Definition	
Provisioning services	Provision of food, wood and fibre	Agro-ecosystems’ first purpose is to produce food and grow crops for a diversity of purposes. Soils physically support plants and supply them with nutrients and water.
	Provision of raw materials	Soils and vegetation can be source of raw materials, e.g. topsoil, peat, turf, sand, clay minerals, biomedical and medicinal resources, genetic resources, ornamental resources. However, the renewability of these stocks is sometimes questionable.
	Provision of support for human infrastructures and animals.	Soils represent the physical base on which human infrastructures and animals (e.g. livestock) stand.
Regulating services	Flood mitigation	Soils have the capacity to absorb and store water, thereby regulating water flows (freshwater levels) and mitigating flooding.
	Filtering of nutrients and contaminants	Soils can absorb and retain nutrients (N, P) and contaminants (E. coli, pesticides) and avoid their release in water bodies.
	Carbon storage and greenhouse gases regulation	Soils have the ability to store carbon and regulate their production of greenhouse gases such as nitrous oxide and methane.
	Detoxification and the recycling of wastes	Soils can absorb (physically) or destroy harmful compounds. Soil biota degrade and decompose dead organic matter thereby recycling wastes.
	Regulation of pests and diseases populations	By providing habitat to beneficial species, soils and vegetation of agro-ecosystems can control the proliferation of pests (crops, animals or humans) and harmful disease vectors (viruses, bacteria) and provide biological control.
Cultural services	Recreation / Ecotourism	Natural and managed landscapes can be used for pleasure and relaxation (walking, angling, mountain biking).
	Aesthetics	Appreciation of the beauty of natural and managed landscapes (wildlife viewing, scenic driving)
	Heritage values	Memories in the landscape from past cultural ties (landscape associated with an important event of regional or national significance)
	Spiritual values	Sacred places
	Cultural identity / inspiration	Natural and cultivated landscapes provide a sense of cultural identity. This establishes a strong cultural linkage between humans and their environment.

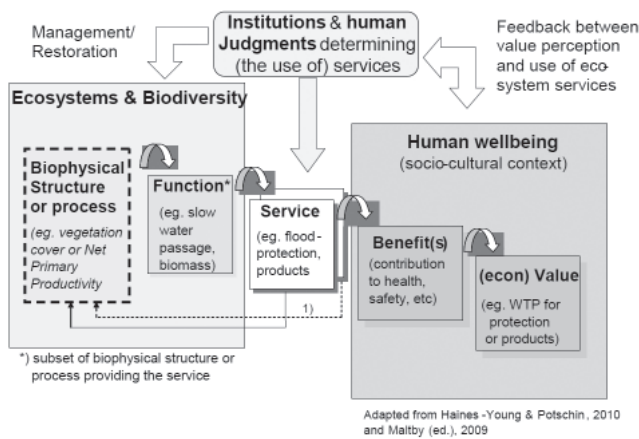


FIGURE 4 The Economics of Ecosystems and Biodiversity initiative framework for ecosystem services: the pathway from ecosystem structure and processes to human well-being (TEEB 2010). (See fig. 1.4, <http://www.teebweb.org/ecological-and-economic-foundations-graphs/>).

and Potschin 2010) formed the theoretical base of the TEEB study (TEEB 2010), while Robinson et al. (2012b) argued that the ‘ecosystems framework should incorporate stocks (natural capital) showing their contribution to stock-flows and emergent fund-services as part of the supply chain’.

Research on agro-ecosystems generally addresses several main ecosystem services: provision of food, wood, and fibre; regulating services including filtering of nutrients and contaminants, pollination, and carbon storage and regulation of greenhouse gases; and cultural services including recreation and aesthetics. Many other ecosystem services provided by managed ecosystems (Table 1) have not been studied; consequently, the value of agricultural land is currently based on two factors, location and productive capacity, and little else.

Holistic framework

Ecological infrastructure (EI) has been defined as the underlying framework of natural elements, ecosystems, and functions and processes that are spatially and temporally connected to supply ecosystem services (Figure 5); it is how natural capital stocks are organised to provide ecosystem goods and services (Bristow et al. 2010). Recent work has recognised the importance of understanding the ecosystem services supply chain

(Mooney 2010; Robinson et al. 2012b). The first of these studies brought the concepts of natural capital, ecological processes, and ecosystem services together in overarching frameworks (Figures 2 and 4) (Dominati et al. 2010a; Haines-Young and Potschin 2010). In those frameworks, the EI component is represented by soil natural capital and supporting processes in Figure 2, and biophysical structure, processes and functions of ecosystems in Figure 4. Following the same goal of building holistic frameworks to show how ecosystems provide services, Robinson et al. (2012b) fitted these concepts within the stock-flow, fund-service framework used in ecological economics (Georgescu-Roegen 1971; Daly and Farley 2010; Farley and Costanza 2010).

Dominati et al. (in press) incorporated concepts developed by Bristow et al. (2010), Dominati et al. (2010a) and Robinson et al. (2012b) within a framework that includes not just the pedosphere but also the biosphere, geosphere, atmosphere, hydrosphere, and anthroposphere (Figure 6). The earth-system approach on the left-hand side of Figure 6 recognises all the earth’s resources, including the atmosphere; the hydrosphere, including oceans, lakes, and surface and ground water; the biosphere with its plants and animals, including humans; the pedosphere, comprising the thin skin of soil around the earth; and the geosphere, containing rocks and minerals. The pedosphere is expanded to give an example of the biotic and abiotic stocks (soil natural capital) each compartment contains. The processes that build up or degrade stocks and result in the cycling, flow, and transformation of materials are represented by the arrows within each sphere. Supporting processes comprise soil formation, nutrient cycling, and water cycling while degradation processes comprise carbon loss, erosion, salinisation, biodiversity loss, and compaction; both are influenced by external drivers, embodied by flows coming from and going to the other spheres. Connectivity among and within spheres – namely, flows of matter, energy and information – is the core of the ecological infrastructure (MEA 2005; Dominati et al. 2010a; TEEB 2011).

The right-hand side of the framework represents the anthroposphere, so the framework illustrates how ecosystem services fulfil human needs by flowing to the anthroposphere from the EI. The anthroposphere is contained within the biosphere and includes various types of anthropocentric capital: built capital, human capital, and social capital (Figure 6). Ecosystem services, classified here according to the Millennium Ecosystem Assessment typology (MEA 2005), are flows that come from the EI and are directly useful to humans. Many of these services are intangible, so they cannot be stockpiled but can be measured as rates (per unit time) (Robinson et al. 2012b). Anthropogenic drivers, like land use and management, alter natural capital stocks levels and thereby modify the provision of ecosystem services.

The integrity, connectivity and health of the ecological infrastructure ensures ecosystems continue to provide services, meaning the whole system, including soil and managed ecosystems, needs to be considered when identifying and determining the impacts of land management on the different spheres.

The ecosystem services approach highlights the holistic value of managed ecosystems. When managing agro-ecosystems two main purposes need to be kept in mind: not only do they produce socio-economic goods such as food, fuel, and fibre, but they also help maintain the integrity of the ecological infrastructure that underpins continued provision of essential ecosystem services. Holistic frameworks provide a means to integrate the management of agro-ecosystems, especially their soil, with management of other ecological elements.

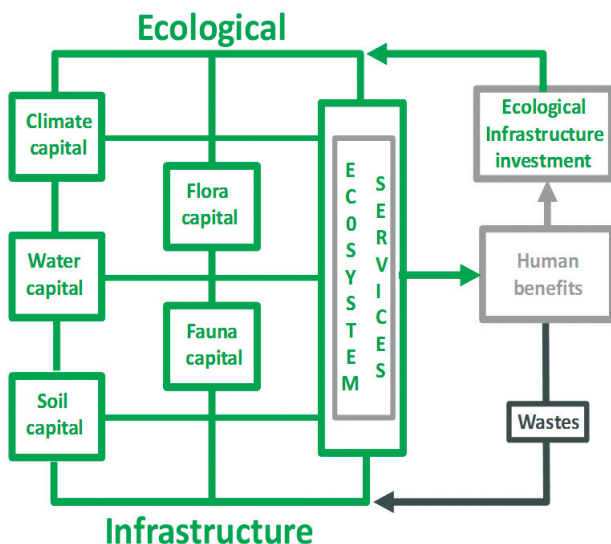


FIGURE 5 Relationships between natural capital, ecological infrastructure and ecosystem services (Bristow et al. 2010).

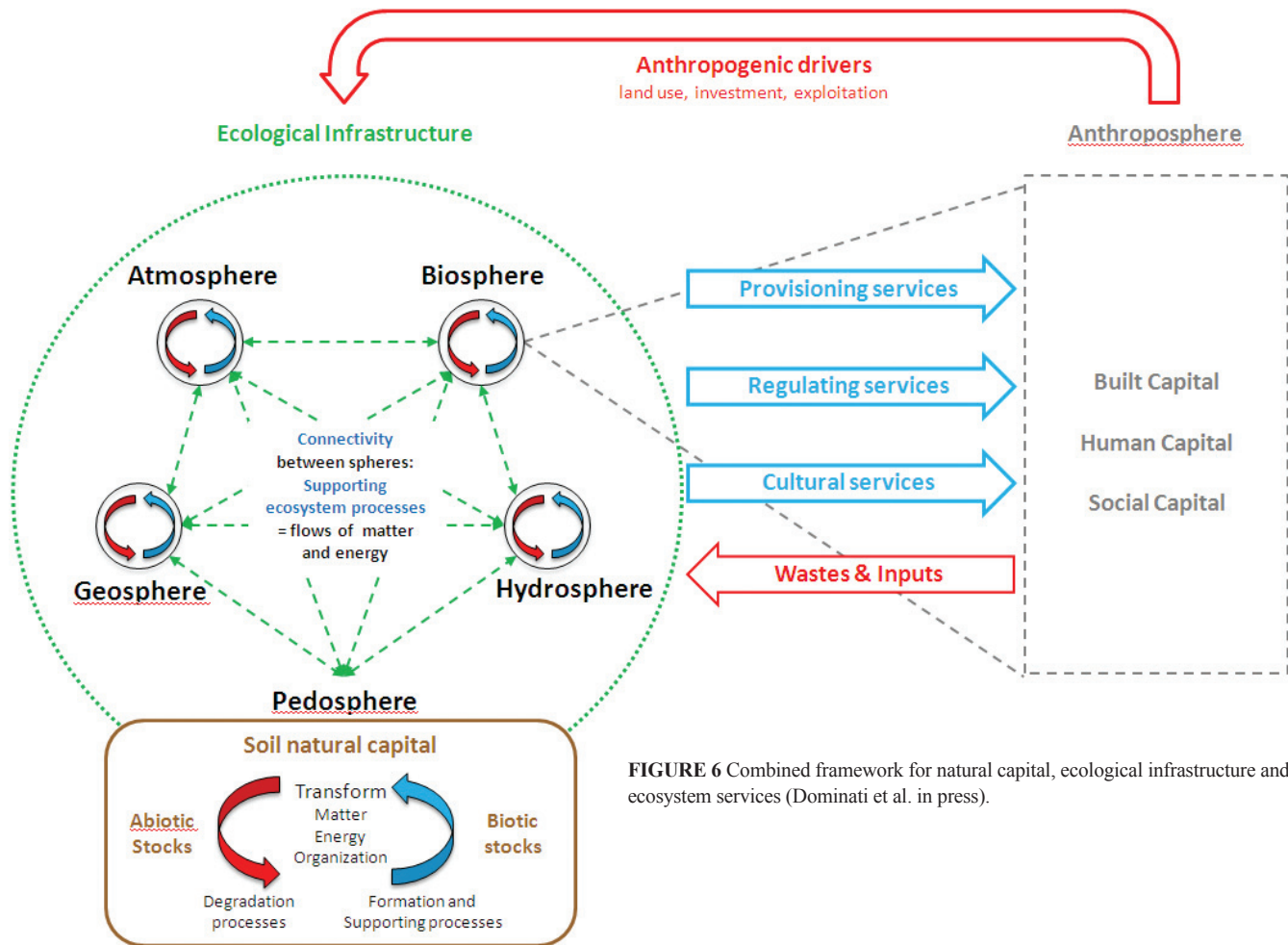


FIGURE 6 Combined framework for natural capital, ecological infrastructure and ecosystem services (Dominati et al. in press).

METHODOLOGY FOR QUANTIFYING SOIL ECOSYSTEM SERVICES

The concepts discussed above, coupled with an ecosystem approach, have been used to develop a series of principles that can capture a flow of ecosystem services. Here the methodology focuses on soil in agro-ecosystems but the principles are applicable to any combination of land use and ecosystem.

Soil services can be quantified and valued in economic terms using the following six steps:

1. Differentiate soil services from the supporting processes that form and maintain soil natural capital stocks. Quantification of ecosystem services needs to focus specifically on benefits directly useful to humans rather than on processes underlying ecosystem functioning (Fisher et al. 2009; Balmford et al. 2011). As already discussed, ecosystem services are by nature flows, and therefore should be measured as rates.

2. Identify the key soil properties and processes behind each soil service. To determine how soils provide ecosystem services, the soil properties (natural capital stocks) and processes that underpin each soil service need to be investigated in detail. This is the role of soil science, where most of the information on soil functioning resides. Changes to soil natural capital stocks and the processes driving these changes must be understood first in order to shed light on incidental changes to the delivery of ecosystem services.

3. Distinguish natural capital from added or built capital when defining proxies for quantifying soil ecosystem services. To determine the correct proxy, the definition of each service is crucial. The proxies must capture the dynamics of soil natural

capital stocks to inform the provision of the service. These proxies must not only be rigorously identified and defined, but should differentiate the part of the service coming from soil natural capital from that coming from added or built capital (e.g. infrastructures, inputs such as fertilisers or irrigation water). Differentiating these will enable the contribution of each to be calculated. Proxies constructed from dynamic soil properties should be based on the part played by the soil in the provision of the service.

4. Identify where and how external drivers affect natural capital stocks and thereby the provision of soil services. External drivers like climate and land use affect soil properties (e.g. natural capital stocks) and processes, and thereby influence the flows of soil services. Identifying these impacts is important for determining whether natural capital stocks are being sustained or degraded, and therefore if the flow of ecosystem services they provide is sustainable.

5. Analyse the impact of degradation processes on soil natural capital and thereby ecosystem services. Many processes degrade soil natural capital stocks and thereby affect the flows of soil ecosystem services. Knowing where and how degradation processes affect soil natural capital is essential for determining their impact on the flow of soil services.

6. Base the economic valuation on measured proxies. When the aim is economic valuation of ecosystem services, the techniques used to value each service should be based on the biophysical measures of the services and be relevant for the chosen scale and land use.

These principles represent an advance in defining and

quantifying the soil's ecosystem services. Previously, quantifying these has been largely confined to determining the status of soil natural capital stocks, without seeking information on the delivery of actual services or actual quantification of ecosystem services flows. This new methodology bridges the gap between the concept of ecosystem services and its application at different scales.

QUANTIFYING SOIL ECOSYSTEM SERVICES FOR A PASTORAL SOIL UNDER DAIRY USE

Here, the example of a dairy-grazed system in New Zealand shows how soil ecosystem services can be quantified using this new methodology. For each soil service (Table 1), I discuss the natural capital stocks that underpin the service, the measurements needed to quantify the service, and how proxies based on dynamic soil properties are defined.

Provision of food quantity and quality

In a dairy-grazed system, food is embodied as pasture growth and quality because pasture is consumed *in situ* by grazing animals. The amount and quality of pasture and its utilisation determine animal growth, health, and milk production. Pasture growth, and thereby the provision of food, is supported by natural capital stocks.

- Soil physical structure — the distribution of pore sizes and conductivity influence the supply of gases, water, and nutrients to plant roots, thus regulating plant growth.
- Available water capacity — the total amount of water a soil can store and provide is crucial for plant development, as is the ability of a soil to remove excess water by drainage. The pores volume and size distribution determine the amount of water the soil can store and move. Soil texture and structure determine a soil's permanent wilting point and field capacity.
- Nutrient status — soil fertility, or the nutrient status of a soil, determines the provision of nutrients to plants. The two macronutrients that most limit plant growth when they are deficient are nitrogen (N) and phosphorus (P). Trace elements are also important; their provision to plants and thereby to grazing animals affects both plant growth and animal health.

To quantify the provision of food from soils, two aspects of the service need to be considered: the amount of food grown, and its quality.

Food quantity — to quantify the amount of food provided from soils, the contribution of soil natural capital stocks must be distinguished from added capital inputs including fertilisers, irrigation, and drainage. To calculate the combined contribution of soil natural capital to the supply of nutrients and water and to physical support for pasture growth, the influence of P and N fertiliser inputs, irrigation water, and drainage must be subtracted from measured or modelled total pasture production. To do this, response curves to inputs must first be determined. The service can then be quantified, for example as kilograms of pasture dry matter per hectare per year. Pasture utilisation by animals must then be used to convert kilograms of pasture dry matter into kilograms of milk solids. In contrast to this approach, previous studies that quantified ecosystem services from agro-ecosystems (Sandhu et al. 2008; Porter et al. 2009) considered total yields as the service without subtracting the influence of fertilisers, but this is not a true reflection of the sustainable flows from natural capital stocks.

Food quality — some soils can be deficient in one or several trace elements (e.g. pumice soils are usually cobalt deficient) (Grace 1994). For optimum milk production, dairy cows need adequate levels of macronutrients and trace elements (selenium, cobalt, copper, and iodine). A measure of the service is the amount of each trace element currently provided by the soil to the pasture per hectare per year. To determine if the soil needs supplementation, this measure must then be compared to the levels of each trace element needed for the desired level of milk production.

Provision of support

Soils form the surface of the earth and represent the physical base on which plants grow, and on which infrastructure, animals, and humans stand. Soil properties that characterise the provision of support include:

- Soil strength — one determinant of soil strength is soil texture, because clay content influences the soil's cohesive strength and silt and sand content affect internal friction; another determinant is organic matter, which helps bond soil aggregates. Soil strength also tends to increase with increasing bulk density and decreasing water content (Marshall 1996). Bulk density takes into account the pore space of the soil so it indicates the level of compaction.
- Soil water content — this determines the soil's sensitivity to treading damage: the wetter the soil, the more easily it can be compacted or deformed.
- Soil intactness and particle cohesion — cohesion within and between soil horizons, and between soil and bedrock, influences movement of soils at the landscape scale.
- Geomorphology — slope and orientation, in combination with climatic factors, partially determine erosion patterns.

Physical support is important at different scales. At the farm scale, the soil's capacity to support animals in paddocks depends on the bulk density and compaction of the upper horizon, whereas its capacity to support buildings and farm tracks depends more on the strength of the deeper horizons and the subsoil. At the landscape level, geomorphology (slope, orientation) and the soil's sensitivity to landslides and other erosion processes also affect the provision of support.

Provision of support for human infrastructures — for human infrastructure, the most important soil property behind the service is soil strength. For building purposes, the most valuable soils are those that are compacted, very stable, and do not sink, deform or erode beneath a building or a road. Light soils must be compacted before building, and to measure compaction, bulk density and macroporosity are good indicators. Because the top 10 cm of soil is usually removed, bulk density below 10 cm represents the already available compaction provided by the soil and can therefore be used as a proxy to measure the service. The higher the bulk density, the better the provision of support to infrastructure. Parfitt et al. (2010) showed that New Zealand soils have bulk density between 0.42 and 1.84 g cm⁻³. Since the provision of this service depends on inherent soil properties that are relatively stable over time, e.g. structure of the soil profile, it is acceptable to express its provision as a non-time-related measure.

Provision of support for farm animals — this depends on the interaction between soil texture, structure and moisture, which determines the soil's sensitivity to treading damage. To avoid soil deformation and consequent production losses, New Zealand farmers increasingly remove animals from pastures to standoff-pads when wet soils fail to provide support. These wet soils

are most common during winter and spring (May to October), which have been identified as critical periods for soil damage on New Zealand dairy farms (Houlbrooke et al. 2009). As a rule, soils are most at risk of structural damage when water content is above field capacity during grazing. Soil water content depends on the soil's drainage class and the dynamics of soil macroporosity; a poorly drained soil will stay saturated longer than a well-drained soil, and therefore will be able to support animals for a shorter period.

A proxy to measure this service can be defined as the number of days per year between May and October when soil water content is less than halfway between field capacity and saturation. This measure represents the days when the soil is not sensitive to treading and provides adequate support to animals.

Provision of raw materials

Raw materials provided by ecosystems comprise renewable biotic resources (Costanza et al. 1997; de Groot et al. 2002; MEA 2005), also called natural capital stocks (wood, fibres, biochemicals), and energy resources (fuel wood, organic matter, gene pool) directly of use for humans. De Groot et al. (2002) specified that abiotic resources like minerals and fossil fuels should not be considered ecosystem services because these resources 'are usually non-renewable and/or cannot be attributed to specific ecosystems'. Consequently, in examining the capacity of soils to provide raw materials, renewable resources must be distinguished from non-renewable resources. Here, I discuss raw materials within the soil profile only, not in the bedrock. Examples of non-renewable materials in soils include peat and clays; their provision should not be considered an ecosystem service (de Groot et al. 2002). At the farm level, these materials are often not present or not exploited, but they could be in some situations, for example, at a regional or nationwide scale. However, the harvesting rates of these materials are usually not sustainable. Therefore, for New Zealand dairy farms, provision of raw materials from soils should usually not be included.

Mitigation of water flows

The ability of soils to store and release water is a service to humans because buffering excessive rainfall reduces flood risk, and releasing water slowly regulates river levels and thereby sustains minimum flows. Flood mitigation does not remove the risk of flooding, but makes it less likely and reduces the need for man-made flood-protection structures. Soils absorb and store important amounts of rainfall water, and start draining before runoff begins; this reduces peak flow by decreasing runoff intensity, which delays the flood peak. Thus, the flood mitigation potential of a soil depends on the drainage class of the soil, and how much water the soil can absorb and store before becoming saturated. In turn, the amount of water a soil can store depends on both inherent and manageable soil properties:

- Soil structure affects water storage in two ways: soils with a high macroporosity can store a greater volume of water before becoming saturated (Marshall 1996), and surface aggregate stability and pore size distribution affect infiltration rate and soil water recharge.
- The depth of the soil profile (an inherent property) affects the total volume of water that can be stored.
- A pan (an impermeable layer) within the profile can slow down or prevent the infiltration of water through the profile (e.g. drainage).
- The stone content of the soil affects water storage because

stones reduce the volume of soil available for water storage.

- The depth of the water table also limits storage because any soil within the water table is no longer available for water storage.
- Slope influences infiltration and runoff.

To quantify flood mitigation, annual rainfall must be considered in relation to the permeability of the soil. For example, on an impermeable surface like concrete, all rainfall might be lost through runoff. Thus, a measure of the service can be defined as the difference between annual rainfall and the amount of water that runs off the land per hectare per year. This is an integrative measure which represents the amount of water absorbed by the soil.

Filtering of nutrients and contaminants

Soils receive rainfall and are the substrate through which water passes before entering rivers, lakes, groundwater, oceans and other water bodies. Soils act as filtering agents. In dairy-grazed systems, materials like animal dung and urine, dairy farm effluents, fertilisers and pesticides are applied to pastures and soils. These materials contain nutrients (including N and P in different forms), organic matter, pathogens, endocrine-disrupting chemicals, and heavy metals. The filtering capacity of a soil refers to its ability to retain nutrients and contaminants by weakly to strongly bonding them to organic or mineral soil constituents, and thereby preventing their release into water passing through the soil profile.

To refer to soil nutrient retention capacity, soil scientists talk about cation exchange capacity (CEC), and anion storage capacity (ASC). A soil's nutrient retention capacity is an inherent property with several dimensions. First, soil properties determine the number and type of sites that can retain nutrients (Stevenson 1999; Hedley and McLaughlin 2005); these properties include the nature and quantity of clay minerals, organic matter content, pH, soil depth and the level of saturation of the soil's exchange sites. Second, nutrients and contaminants can differ in form, stability, and solubility, all of which influence the probability of their being retained or released. Third, soil processes such as ion exchange, adsorption, occlusion or precipitation transform nutrients from soluble to labile or non-labile forms and vice versa, and they affect the saturation of the soil's exchange sites and the soil's nutrient retention capacity (McLaren and Cameron 1990).

Phosphorus in runoff and drainage waters threatens New Zealand surface waters; similarly, nitrogen lost in this way threatens groundwaters. In grazing systems N is lost primarily by nitrate (NO_3^-) leaching from urine patches through the soil to below the roots. The amount of N deposited on a urine patch can reach the equivalent of 200–1000 kg ha⁻¹ (Hoogendoorn et al. 2010). In contrast, P is a specifically sorbed anion tightly held by the soil, so P is lost largely through surface runoff unless the soil demonstrates preferential flow (e.g. cracking clays) or has very low sorption capacity (e.g. podzols) (Edwards et al. 1994). P is lost in two forms, soil-bound P and dissolved-P, with the former often the dominant (60–90%) mechanism in less intensively farmed hill catchments (Parfitt et al. 2009).

Filtering of nutrients and contaminants can be quantified as the amount of nutrient the soil does not lose. A measure of the service can be defined as the difference between maximum and actual loss, where maximum loss depends not just on the soil's absorption capacity, but also the amount of nutrients entering the soil, the amount of nutrients used by soil microfauna and plants, and the soil's drainage class. Maximum loss is the amount of

nutrient that could potentially leach but does not, due to the soil's nutrient retention capacity. To quantify this potential maximum loss, leaching due to the soil's nutrient retention capacity must be artificially isolated from inevitable losses from plant turnover and mineralisation. To do so, potential maximum nutrient loss can be determined by modelling losses for a soil with almost no ability to retain the nutrient; in other words, with an ASC close to zero.

Directly quantifying a soil's ability to filter contaminants such as pathogens (e.g. *E. coli*), pesticides or endocrine-disrupting chemicals by looking at contaminant loads and leaching is usually difficult because of a lack of data. However, it can be quantified indirectly. In a dairy-grazed system, dung is deposited on pasture during grazing. The risk that dung pads will contaminate runoff water during grazing can be considered as a proxy for the filtering of contaminants if information on soil water content, runoff and the timing of grazing events is available from either a simulation model or field data. Assuming that dung can still significantly contaminate runoff water up to 5 days after grazing (Aarons et al. 2004), a measure of the service can be defined as the difference between rainfall and runoff within 5 days after grazing. This measure of the service, in millimetres per hectare per year, represents the amount of water that would be contaminated during those 5 days if the soil was not absorbing and filtering it.

Detoxification and recycling of wastes

Increasing amounts of wastes are applied to New Zealand soils each year. These materials include dung and urine from farm animals, effluent from dairy sheds and standoff areas, sludge from effluent ponds, and composts. They contain two types of threats: organic or other chemical compounds potentially harmful to the environment, and living organisms (pathogens such as viruses, bacteria, or parasites). The ability of soils to deactivate non-organic contaminants (detoxification) and biologically degrade organic wastes constitutes an ecosystem service linked directly to human health. It is a service in itself, separate from the filtering of nutrients and contaminants or the provision of nutrients to plants.

Two main processes support the detoxification and recycling of wastes: sorption of compounds on clays and organic matter surfaces, and biological degradation of organic and chemical materials. Macrofauna like earthworms first incorporate materials into the soil, then mesofauna and microfauna break down the organic compounds in the residues, releasing other compounds and carbon dioxide (CO₂). Biodegradation of contaminants is controlled by the availability of nutrients in the soil (C:N ratio).

Because the detoxification and recycling of wastes is complex, the service must be quantified indirectly. In a dairy-grazed system, decomposition of dung can be used as a proxy for this service. Soil conditions affect microbial activity in several ways: soil moisture, soil aeration (macroporosity) and nutrient levels are the key soil properties (i.e. natural capital stocks) controlling invertebrate and microorganism populations, which in turn are the main agents that detoxify and recycle wastes. Soil water content can be recorded at grazing when dung is deposited on the pasture. Ideal conditions for decomposition of wastes by soil fauna are associated with soil water content being between stress point and field capacity. To quantify this service the amount of dung deposited in ideal conditions can be used as a proxy; it represents the amount of waste successfully decomposed, measured as kilograms of dung dry matter per hectare per year.

Carbon storage and greenhouse gas regulations

Carbon storage and greenhouse gas regulation are soil services

increasingly acknowledged by the general public. Soils emit and consume CO₂ and can store C, which is of interest for signatory countries of the Kyoto Protocol including New Zealand. Soils can also regulate their emissions of greenhouse gases like nitrous oxide (N₂O) and methane (CH₄). These services are supported by soil natural capital stocks, including: soil structure and macroporosity, which determine soil water content; clay content, which determines carbon sorption and organic matter stabilisation; nutrient status (C:N ratio and nutrient availability); and soil microfauna diversity including denitrifiers and methanotrophs.

Carbon flows — when investigating soil C stocks, it is essential to consider net flows of C from soils because these flows determine the stability of C stocks. A measure of the service can be defined as the annual net C flows to the soil. Processes involved in the C cycle include net primary production, the return of dead organic matter to the soil (e.g. dung, dead plant material, effluent), heterotrophic respiration, and C losses such as organic matter degradation, erosion and dissolved organic C leaching. The net balance among these processes will determine if the soil is losing or accumulating C. If the net balance is positive, soil C accumulation occurs, which is the service. However, if the net balance is negative, the soil is losing C, which is not a service but a degradation process (Dominati et al. 2010a). Such a degradation process impacts on a range of natural capital stocks and soil supporting processes and so can affect the provision of all ecosystem services.

Nitrous oxide (N₂O) regulation — the production of N₂O by soils is a major concern for New Zealand. In 2007, N₂O emissions from agricultural soils comprised 33.8% of agricultural emissions and 16.3% of New Zealand's total greenhouse gas emissions (MfE 2009). Gaseous N losses are a result of denitrification: biological denitrification, carried out by nitrobacteria in anaerobic conditions, producing N₂O, and chemical denitrification producing N₂. Nitrous oxide emissions are influenced by the soil's ability to deal with all anaerobic conditions, including waterlogging and poor drainage; these emissions are regulated by natural capital stocks including soil structure, soil water content, and nutrient status. The service can be defined as the difference between the maximum potential N₂O emission if the soil was always waterlogged and the actual N₂O emission calculated for each year. The measure of the service represents the N₂O that could potentially be emitted from the soil, but is not due to soil water content regulation, in, for example, kilograms of CO₂ equivalent per hectare per year regulated. The maximum potential N₂O emission every year can be obtained by simulating dung and urine deposition systematically on wet soils.

Methane (CH₄) oxidation — methane is a powerful greenhouse gas, so its degradation by soil biota is an ecosystem service. However, the amount of CH₄ oxidised by pastoral soils at the farm scale is very small: between 0.3 and 2 g CH₄-C ha⁻¹ day⁻¹ (Saggar et al. 2008); in other words, about 0.9 kg CH₄ ha⁻¹ year⁻¹ or 19 kg CO₂ eq ha⁻¹ year⁻¹ (using the global warming potential of CH₄ as 21 for a 100-year time period). Methane oxidation depends on soil natural capital stocks including soil water content and organic matter content. Any CH₄ oxidation from soil represents a service independent of net C storage.

Regulation of pest and disease populations

In dairy farm systems, soils play a major role in regulating some pest and disease populations. This biological regulation is supported by natural capital stocks including macroporosity, soil water content and food sources (e.g. organic matter inputs to the

soil), all of which influence soil biodiversity. In New Zealand dairy systems, two important pasture pests are porina caterpillars (*Wiseana* spp.) and grass grub (*Costelytra zealandica*). Eggs and young larvae of both pests are very sensitive to extremes of soil water content between October and December, while older larvae are sensitive to cattle treading and low macroporosity between January and March. Therefore, to quantify the service, the dynamics of soil water content and macroporosity must be linked to pest development and level of infestation. The service can be measured indirectly by determining the number of days unfavourable to pest development between October and March, when populations of these pests are regulated by soil properties. From October to December, unfavourable conditions can be defined as a soil too dry (soil water content below stress point) or too wet (soil water content above field capacity), and from January to March as a soil with a macroporosity below 9%. The total number of unfavourable days between October and March can then be linked to a level of infestation to serve as a proxy for pest regulation. The age of the pasture also needs to be considered when developing this proxy for infestation levels, because biological control agents of these pests increase over time and often reach substantial levels in pastures older than 5 years (Jackson 1990; Kalmakoff et al. 1993). A measure of the service can then be the number of well-regulated days per year.

The need for large amounts of high quality data and the absence of a set of standardised definitions make quantifying soil ecosystem services a challenge. The methodology and examples presented here comprise a work in progress but represent an advance in defining and quantifying soil ecosystem services. Such methodology bridges the gap between the concept of ecosystem services and its application at different scales.

APPLICATION TO RESOURCE MANAGEMENT

Challenges of the ecosystems approach for resource management

To fully appreciate the contributions of soils to human welfare beyond food production, resource management needs powerful tools that link on-site changes to off-site impacts. The ecosystems approach and the concepts of natural capital, ecological infrastructure, and ecosystem services can provide these tools. They also foster a holistic approach to the place of managed ecosystems within the greater ecological infrastructure.

As a multidisciplinary approach, the ecosystems approach presents challenges when applied to the management of soils and agro-ecosystems. To develop the soils component of the ecosystems approach (Robinson et al. 2012a), guidelines for applying the ecosystems approach (TEEB 2010; Braat and de Groot 2012) can be combined with key research areas identified as needing attention, resulting in a set of steps for action:

- Keep quantifying changes to natural capital stocks under the impact of natural and anthropogenic drivers. This can be achieved by soil science and through monitoring and modelling of stocks, fluxes, and transformations, within and between spheres. This requires more and better quality information on the functioning of ecological infrastructure, in turn requiring better methods for generating or collecting data, analysis, validation, reporting, monitoring, and integration with other disciplines.
- Harmonise methods, measurements, and indicators for the sustainable management and protection of soil resources.
- Better assess the spatial and temporal dynamics of service provision, especially in relation to beneficiaries.

- Develop management strategies and decision-support tools including models, and maps of natural resources.
- Develop standardised ways to value ecosystem services and incorporate these values into decision-making about alternative management options.

A new holistic approach to resource management

The ecosystems approach offers several options for the future of resource management. In New Zealand, regional councils base their state-of-environment monitoring and reporting for soil quality on target ranges for soil quality indicators (Sparling and Schipper 2004). A project that began in June 2010, 'Soil quality indicators: New generation' linked these soil quality indicators to outcomes at the paddock, farm and catchment scales using a soil natural capital and ecosystem services framework. Linking these indicators to the provision of ecosystem services increases their value to managers and policymakers because it enables changes in indicators to be linked to outcomes at the farm or catchment scale. The project has the potential to offer a nationally consistent approach for regional and national managers to assess whether land use and land use changes align with regional policy statements.

The ecosystems approach can also provide new insights to inform the debate about land use change and how best to use New Zealand's land resources. The frameworks presented here could serve as a basis for a national framework of interest on land, with associated national standards; this would help regions and districts in New Zealand provide guidelines and limits for policy development on land management and land use changes at local and regional levels (Mackay et al. 2011). The ecosystems approach, as an integrated approach, can be used to assess the wider implications of ongoing land-use change on society.

Improving the quantification and economic valuation of soil ecosystem services can also promote discussions about investment in ecological infrastructure and how such investment can improve the yield of ecosystem services from land (Bristow et al. 2010) and make land uses more sustainable. Like built infrastructure, ecological infrastructure needs public investment to maintain its integrity; however, while investment in built infrastructure has been increasing continuously, we have not been investing sufficiently in ecological infrastructure (Bristow et al. 2010). Indeed, this lack of adequate investment in ecological infrastructure has led to a worsening environmental crisis in which critical ecosystem services continue to be lost across the globe (MEA 2005).

The ecosystems approach can also provide new insights into land development. Over the last 100 years, science has been at the forefront of the development of production technologies aimed at overcoming soil limitations like low nutrient status (e.g. fertilisers, legumes), wetness (e.g. drainage, flipping), low water holding capacity (e.g. irrigation) and stoniness (removal or burial). In future, however, land development must increasingly focus on the efficiency of use of natural resources, like land and climate (e.g. rainfall), and scarce inputs like nutrients. Thus, the development of technologies for land use need to switch from overcoming limitations to investing in ecological infrastructure that will increase natural capital and enhance the provision of ecosystem services (Mackay et al. 2011).

For example, soil conservation policies aim to reduce soil erosion on vulnerable land, downstream costs associated with nutrient losses and sediment loadings to waterways, and damage to productive farmland and towns. Currently, evaluating soil

conservation policy and justifying its associated expenditure are limited to assessing the reduction in losses of productive capacity, soil, and sediment, and downstream impacts on communities of flooding and sedimentation, but until the full range of above- and below-ground ecosystem services is considered in the analysis, the full cost of erosion (beyond productivity loss) and full value of soil conservation (investment in ecological infrastructure) are not available for informed decisions about land use.

In future, resource management should focus on three strategies that must be carried out concurrently:

Restoring degraded ecological infrastructure;

Maintaining and enhancing the capacity of current ecological infrastructure to continue providing ecosystem services;

Providing solutions based on sound science to minimise potential damage instead of looking only for solutions to overcome limitations.

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