Soil Ecosystem Services: Sustaining Returns on Investment into Natural Capital.

by

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Submitted as:
Chapter 10

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Abstract
Sustaining soil productivity in response to climate change is critical for two reasons: feeding the world under straitened circumstances, plus adapting to and mitigating climate change itself. The supporting, provisioning and regulating ecosystem services provided by soil are critical for food provision and for meeting the challenges of climate change. We outline how an ecosystem services approach that recognises and values the soil’s natural capital stocks of carbon, air, water, and water, might offer a means to ensure maximised use of natural capital and minimised use of added capital resources. We present three examples of securing sustainable returns on investment into the soil’s natural capital: we link soil processes controlled by carbon to soil ecosystem services; we value supporting and regulating soil services in the provision of terroir value for wine; and we use a valuation of the soil’s natural capital to allocate a nutrient-loss right for policy to limit non-point source pollution by nutrients from farms.

Firstly, we found for two similar soils that had undergone 12 years of apple growing using different carbon-management practices, phenoforms had developed with different levels of soil carbon. Thus the natural capital value of the soil with the higher carbon content we found to be greater as a result of its enhanced supporting and regulating ecosystem services. Conversely for vineyard soil we found that natural capital value declined with increasing soil carbon throughout the profile, for the terroir lessened as a result of excessive vegetative vigour resulting from greater nutrient provision and water storage. However, if a mulch just raised the carbon content of only the surface soil, then the natural capital value of the soil would be enhanced through better supporting, regulating and provisioning services. Finally we outline how soil scientists can become involved with policy analysts to develop a nutrient management policy that is based on the concept of natural capital. This policy does not focus on capping inputs, but rather it seeks to engage with land-users to maximise their return on investment into their natural capital assets and to encourage their sustainable management of the landscape’s biophysical resources without compromising the ecosystem services of receiving environments.

Keywords: F.H. King, natural capital, ecosystem services, virtual water, soil macropores, soil carbon, terroir, nutrient leaching, water quality, resource management policy.
Introduction

Sustaining soil productivity in response to climate change is critical for two reasons: feeding the world under straitened circumstances, plus adapting to and mitigating climate change itself.

The supporting, provisioning and regulating ecosystem services provided by soil are critical for food provision and for meeting the challenges of climate change. In many cases, however, the supporting services of soils are inadequate for sufficient provisioning of food, fibre and fuel. Water and nutrients are often needed to ensure economic levels of plant productivity. Irrigation and fertilisers are frequently used to overcome inadequate natural stocks of water and nutrients in the soil. It is estimated that irrigated agriculture covers some 260 million hectares of the earth’s surface, such that this 17% of the world’s cultivated lands can thankfully provide 40% of the global production of food and fibre (Fereres and Evans, 2006). Irrigated agriculture consumes about three quarters of the world’s fresh water that is abstracted for human use. There are pressures on the quantity of water available for feeding the world. Globally, some 175 million tons of nitrogen are taken up by crops, and synthetic fertilisers account for about 40% of this. Takahashi (2006) calculated that some 2 billion people, one third of the world’s population are dependent up synthetic nitrogenous fertilizers. These fertilizers, along with manures, through the leakage of nitrates from the rootzone, are diminishing the quality of our stocks of ground and surface waters. There are scientific matters, policy imperatives and ethical issues that we need to address so that soils, which are a finite resource, can continue to produce food, fibre and fuel in the face of climate change.

Soils play a key role in climate regulation, so there are direct feedforward and feedback links between soil productivity and climate change. Soils, through their regulating services in relation to the two greenhouse gases (GHG) of carbon dioxide and nitrous oxide are critical for climate-change mitigation and adaptation. There are about 750 Gt of carbon (C) currently in the atmosphere, whereas there are about 1700 Gt-C in the soil, and about 550 Gt-C stored in vegetation. Small changes in soil C, or land-use impacts on vegetation, can thus have a significant impact on the carbon dioxide in the atmosphere. Sarmiento and Gruber (2002) noted that since pre-industrial times, soils and vegetation have provided a sink for 65 Gt-C, thereby buffering the rise in atmospheric CO$_2$ in response to anthropogenic emissions. Yet through land-cover change the terrestrial stock of carbon has been reduced by 124 Gt-C (Sarmiento and Gruber 2002). The carbon and nitrogen cycles in soil are intimately linked. Soil productivity and fertilizer use is therefore an integral part of strategies for climate-change mitigation and adaptation. Nitrous oxide is a potent GHG and the fourth largest GHG emission in terms of radiative forcing (Intergovernmental Panel on Climate Change (IPCC) 2007). Soil is the dominant source of atmospheric N$_2$O, contributing about 9 Tg y$^{-1}$, or about 60% of the total annual global emission (IPCC, 1977).

It is therefore imperative that policies, based on sound scientific understanding, be developed to ensure that soil productivity can be sustained to feed the world and to mitigate and adapt to climate change. In this chapter, we describe the political and ethical issues associated with balancing the need to produce food with greenhouse gas mitigation. Then we outline how an ecosystem services approach that recognises and
values the soil’s natural capital stocks of carbon, air, water, and water, can offer a means to ensure maximised use of natural capital and minimised use of added capital resources. We then provide three examples of:

- how land management practices in relation to soil C can be adopted to improve soil functioning and health,
- how natural capital can be valued and used to enhance soil and water productivity, and
- how resource management policy based on the natural capital value of the soil enables best use of the land and water resources within a catchment to realise a desired environmental outcome.

This chapter derives from a presentation made at the Organisation for Economic Development (OECD) workshop on “Sustaining Soil Productivity in Response to Global Climate Change – Science, Policy and Ethics” held in Madison, Wisconsin in July 2009. Before proceeding to outline how we can sustain returns on our investment into the natural capital of our soils, it is apposite to reflect on the seminal work on sustainability by Franklin Hiram King – a professor of agricultural physics at the University of Wisconsin, Madison, between 1888 and 1902 (Tanner and Simonson 1993).

**F.H. King – “Farmers of Forty Centuries”**

What is a temporal metric of sustainable soil productivity?

[ Insert Figure 10.1 about here ]

In 1909, F.H. King (Figure 10.1) visited Japan, Korea and China, countries where farmers had been farming productively on the same plot of land for forty centuries (King, 1911). King set out to “… consider the practices of some five hundred millions of people who have an unimpaired inheritance acquired through four thousand years”. He sought “… to learn how it is possible after even forty centuries for their soils to be made to produce sufficiently for the maintenance of such dense populations”. This challenge still remains a clarion call for today’s researchers.

King found that “… almost every foot of land is made to contribute material for food, fuel and fabric”. He noted that “… which ever way we turned we were amazed at how these nations have been and are conserving and utilising their natural resources and surprised at the magnitude of the returns they were getting from their fields. It is evident that these people, centuries ago, came to appreciate the value of [natural resources] in crop production”. What is amazing to us now is that a century ago F.H. King was using the language we currently associate with contemporary ecological economics and natural capital valuation.

In delving deeper, King found that “… judicious and rational methods of fertilisation are everywhere practiced. Lumber and herbage for green manure and compost, and ash of the fuel and lumber used at home finds its way ultimately to the field as fertilizer. Manure of all kinds, human and animal, is religiously saved and applied to the fields in a manner which secures an efficiency far above our own practices”. Figure 113 from King (1991) is reproduced here as Figure 10.2, where King has
added the legend that sustained soil productivity “… is the product of brain, brawn and utilized waste”.

There is an interesting connection made by King in relation to his observation of the Asian farmers’ utilisation of organic wastes to sustain production through forty centuries. He found that “… the drinking of boiled water has been universally adopted as an individually available, thoroughly efficient safeguard against that class of deadly disease germs in the drinking water of a densely peopled country [where] the wastes of the body, of fuel, and of fabric beyond use are taken back to the field”. King thus keenly concluded that “… there is little reason to doubt that the tea industry had its foundation in the need of something to render boiled water palatable for drinking purposes”.

Thus one return on investment into the natural capital of recycled waste in soil has been the cultural service of tea drinking.

**Soil: Valuable Natural Capital**

The natural capital concept attempts to integrate thinking about economics and ecology by conceiving ‘nature’ as ‘capital’ (Fenech et al. 2003). Fenech et al. (1999) had noted that a skeletal conception of natural capital emerged around 1994; however, the notion can even be seen to be inherent in the earlier thinking of F.H. King and others. In observing the consequences of the losses of soil assets during the Dust Bowl era, Franklin Delano Roosevelt, US President from 1933 to 1945, noted that “… the nation that destroys its soil destroys itself”. In the formation in New Zealand of a multi-institute research programme called the Sustainable Land Use Research Initiative (SLURI), we calculated that 17% of our nation’s GDP was reliant on the top 150 mm of soil (Kirkham and Clothier 2007). Soil is indeed valuable natural capital.

Akin to the returns on financial capital, we also benefit from returns on investment into natural capital through the ecosystem goods and services that flow from natural capital stocks. This recent notion of ecosystem services (Costanza and Daly, 1992) can indeed be seen latent in the thinking of early ecologists. Aldo Leopold in his “A Sand County Almanac” wrote in 1949 that “… land then, is not merely soil, it is a fountain of energy flowing through a circuit of soils, plants and animals”.

Here we discuss soil as valuable natural capital, and in the next section we discuss the ecosystem goods and services that flow from the soil’s natural capital. We begin by defining natural capital and ecosystem services. Following Dominati et al. (2009b) we take natural capital to be the stocks of natural materials and energy. Ecosystem services, which we discuss later, are the beneficial flows of goods between natural capital stocks, or between stocks and humans.

Fenech et al. (1999) noted that “… by bringing economic science and environmental science to an objective common ground, a natural capital model has the potential to provide a concrete means of comparing the economic and ecological costs and benefits of particular policies and programmes.” They opined that “… existing
microeconomic theory may be ‘ungreenable’, if it is not reformulated”. An inconvenient truth for economists.

Costanza (2009) has revisited this in light of the recent economic crisis, and he asserts that “… the current financial meltdown is the result of under-regulated markets built on an ideology of free market capitalism and unlimited economic growth”. The basis for this was that “… the mainstream vision of the economy is based on a period when the world was still relatively empty of humans and their built infrastructure [and] in this ‘empty world’ context, built capital was the limiting factor, while natural capital and social capital were abundant. But the world has changed dramatically [as we recognize] that natural capital and social capital are not infinitely substitutable for built and human capital and that real biophysical limits exist to the expansion of the market economy”. He calls for a “… a more sustainable and desirable future that focuses on quality of life rather than merely quantity of consumption”.

Intensification of land-use can, in the short-term, meet our demands for increased quantities of consumption of food, fiber and fuel and so achieve returns on investment into the soil’s natural capital. However, through intensification we can, as Hawken et al. (1999) warn “… temporarily exceed the carrying capacity of the earth, but put our natural capital into decline”. Then Hawken et al. (1999) wryly added “…put another way, the ability to accelerate a car that is low on gasoline does not prove the tank is full”.

The World Wide Fund for Nature (WWF) has just released its Living Planet Report 2008. They commence by stating that “… we only have one planet [and] when human demand on this capacity exceeds what is available we surpass ecological limits”. Indeed they assert that humanity’s ecological footprint exceeded the world’s biocapacity of our single planet around 1986. They calculated that our ecological footprint now exceeds the planet’s regenerative capacity by about 30%. To describe this better, they refer to the footprint exceedance in financial terms as being ‘ecological debt’, and they call for an ecosystems approach to enable a return to sustainability by reducing the ecological debt, and re-establishing a global biocapacity reserve.

A prime natural-capital stock is that of the waters in our rivers, lakes, groundwaters, and soil storages. Soil provides the supporting and provisioning ecosystem services for growing plants and raising animals that enables the world to attempt to feed itself. There are, and increasingly there will be, critical challenges to ensure that we can produce enough food to feed ourselves. Sustaining soil productivity will be critical. Yet, as The Economist (September 18, 2008) reported “… the world is not facing so much a food crisis as a water crisis”. The Economist then added that presently some 70% of the world’s water consumption is used in farming and that there are pressing needs to make these natural capital stocks go further by developing knowledge and tools to increase water-use efficiency - that is, to reduce the water footprint of food, fiber and fuel production. It concluded that indeed “… farming tends to offers the best potential for thrift”. This will nonetheless be a challenge. Robert Glennon in his book “Unquenchable – America’s Water Crisis” notes that “California growers consume 80% of the state’s water yet contribute only 2% to the gross state product [and] while cynics accuse farmers of milking the government, farmers are suffering from their own remarkable productivity … the United States has the cheapest food
supply in the world. Better equipment, mechanization, hybrid seeds, fertilisers, pesticides and irrigation explain the so-called green revolution”. Yet Glennon (2009) concluded optimistically for the case of agriculture, as “… farmers are responding nimbly to these [thrifty] challenges by engaging in vanguard agriculture [through] identifying new ways of growing and marketing produce”.

This thrifty imperative for vanguard agriculture will be ever more challenging and will require a nimble responses to climate change.

The Intergovernmental Panel on Climate Change (IPCC, 2007) has released its fourth assessment report on the causes and impacts of climate change. Most IPCC climate-change scenarios point to a reduction in the natural capital stocks of water in the world’s major food-producing regions. The IPCC Special Report on Emissions Scenarios (IPCC 2000) considers four storylines. Storyline A1 is for a market-oriented world, with strong economic growth and strong governance. The B scenario of this storyline is for a balance of energy production from both fossil and non-fossil sources, such that greenhouse gas emissions peak in 2050 and then decline. Scenario A1B is therefore a moderate scenario in terms of the global rise in temperature. In Figure 10.3 are shown the projections by the IPCC (2007) for the change in annual water runoff, which is a metric of the availability of water stocks, in terms of percentages for 2090-2099, relative to 1980-1999. The predicted diminishments in the natural capital stocks of available water are large in the food-producing areas of the western US, Mexico, Chile, Argentina, South Africa, Australia, and especially the so-called MENA countries of the Middle East and North Africa. In contrast, the emerging economies of the BRIC countries of Brazil, Russia, India and China will be blessed with largely unchanging, or enhanced stocks of water availability to sustain soil productivity

[ Insert Figure 10.3 about here ]

The global challenge for all countries, however, is to sustain productivity and returns on investment into their natural capital stocks of our soils and waters. Two international trends are working in diametrically opposing directions in relation to reducing the ecological footprint of humanity and sustaining global productivity. Water footprinting and ethical-eating initiatives are working towards reducing ecological debt. On the other hand, trans-national land-for-food deals and food protectionism are, we consider, threatening our global ability to feed ourselves.

In the United Kingdom on August 20, 2008 the newspaper *The Guardian* cited the WWF report by Chapagain and Orr (2008) which “… revealed the massive scale of UK’s water consumption”. Whereas household water use in the UK is about 150 litres per person per day, Britons consume nearly 30 times as much in the ‘virtual water’ that is used in the production of, or embedded in the imported food and textiles. Chapagain and Orr (2008) therefore estimated, in sum, that each Briton consumes 4,645 litres per day. Of that, only 38% of the water comes from Britain’s own resources, and the rest depends on the water systems of other countries, many of whom are already, or likely to be adversely affected by climate change. For example, Spain exports some 1.4 million cubic metres of virtual water to the UK each year, and the top three ‘containers’ of this virtual water are olives (344 Mm³/y), grapes (180) and oranges (91). Spain uses 70% of its available ‘blue’ water from rivers, lakes and
groundwater for agriculture. Thus there is an imperative to manage soils and irrigation to use this as efficiently as possible and to establish procedures to reduce the reliance of horticultural production on blue water, and to develop strategies to maximise the natural capital stocks of the ‘green’ water stored in the soil from rainfall. In The Guardian article, the technical director of the supermarket Marks & Spencer’s, David Gregory, said “… water was already a key part of the company’s strategic decisions about where to source food for its stores … and where to grow crops in the future”. There are moves to develop water footprinting labels on products, especially food, so that consumers can see how their purchasing choices affect the country-of-origin’s natural capital stocks of water. The Food Ethics Council of the UK has just released its recommendations on water labels for food (Segal and MacMillan 2009). This initiative is aimed at consumers so that they can make ethical purchase choices that will act to reduce the consumption of virtual water associated with products and that will enable us to move towards meeting the WWF’s challenge of 2008 “… to reduce our footprint and get better at managing the ecosystems that provide services”. Ethical eating might help us sustain soil productivity. There is, however, a contrary side to this story – a new form of food protectionism.

The Economist of May 21, 2009 reported on a powerful and contentious trend sweeping the poor world that might, we consider, imperil the sustaining of global soil productivity: the outsourcing of food production by sovereign states to foreign lands. The Economist found many examples of where countries that export built and financial capital, but import food, are outsourcing farm production to countries that need financial capital, but have the natural capital of land to spare. In their report, The Economist cites Peter Brabeck-Letmathe, the Chairman of Nestlé who asserts that “… the purchases weren’t about land, but water. For with the land comes the right to withdraw the water linked to it, in most countries essentially a freebie that increasingly could be the most valuable part of the deal – the great water-grab”. The Fortune magazine (June 16, 2009) even reported on ‘reaping reward from farmland’, and they noted that “… the biggest investors in farmland over the next decade will probably be sovereign wealth funds and governments of crop-starved countries eager to secure food supplies for their rapidly growing populations”.

This contrary side to the food production stories raises another ethical issue. What value has the land ethic by companies or sovereign states that outsource food production to foreign lands? Aldo Leopold defined his land ethic as being simply an enlargement of “… the boundaries of the community to include soils, waters, plants, and animals, or collectively: the land. … [A] land ethic changes the role of Homo sapiens from conqueror of the land-community to plain member and citizen of it” (Leopold 1949). Will foreign-based companies, or foreign states, sustain a land ethic when they are neither a member of the community in which their food is grown, nor a citizen of the country from which their food is sourced? We doubt it, and it concerns us for it seems to us that this variant form of neo-colonialism is unfortunately returning “… Homo sapiens [again to a] conqueror of the land-community” and worse, conqueror of someone else’s land-community!

The outsourcing of food production to foreign and often fragile soils will, we consider, lead to further degradation of global natural capital stocks, and more reductions in the number and vitality of global ecosystem services. What global value have such services?
Valuing Ecosystem Services

In a landmark paper, Costanza et al. (1997) quantified how valuable ecosystem services are. They determined how the flow of goods and services from natural capital stocks provide value over and above the simple rent, or producer surplus, that is received from them, for there is also value in a consumer surplus that comes from non-marketed benefits. Costanza et al. (1997) considered 17 ecosystem services across 16 biomes covering both marine and terrestrial systems. They based their valuation on estimates of ‘willingness-to-pay’ and concluded that on average the global value of these services would be US$ 33 trillion, being some 1.8 times their estimate of global gross national product.

Despite the massive economic value of these natural capital stocks and these ecosystem services, it was detected early in the twenty-first century that human actions were still resulting in losses of the inventory value of natural capital assets and reductions in the number and vitality of ecosystem services. In 2001, the Millennium Ecosystem Assessment (MA) programme was established by the United Nations to assess the consequences of ecosystem change for human well-being and to provide the scientific basis for the actions needed to enhance the conservation and sustainable use of those systems and their contribution to human well-being. This assessment involved the work of more than 1,360 experts worldwide. Their findings (MA, 2005) provided a state-of-the-art scientific appraisal of the condition and trends in the world’s ecosystems and the services they provide and the options to restore, conserve or enhance the sustainable use of ecosystems. The Millennium Ecosystems Assessment categorised ecosystem services into four broad, often overlapping groups:

- Provisioning services,
- Regulating services,
- Cultural services, and
- Supporting services.

As shown in Figure 10.4, the MA linked these four ecosystem services to five constituents of human well-being: security, basic materials, health, social relations and freedom of choice. Their ecosystem-services approach is based on considering many interacting ecosystem services, and not just looking at the benefits from a single service. This holistic approach allows integrated assessment of the overall benefits that an ecosystem provides, and enables assessment of how these benefits can change when different pressure are exerted on the ecosystem.

In 2007, the World Resources Institute (Irwin and Ranganathan 2007) published a report on restoring nature’s capital, which was an action agenda to sustain ecosystem services. They lamented that national accounting systems had failed to keep track of the inventory value of natural capital assets, and they called for a fundamentally new approach to managing the natural capital assets upon which all life depends. A year later they produced a guide for decision makers on ecosystem services (Ranganathan et al. 2008). They suggested how decision makers could put into operation the
concept of ecosystem services by identifying how a decision depends on nature’s flows, or ecosystem services, and how a decision will in turn affect these flows. This procedure would increase the ability to understand and make trade-offs across ecosystem services, in space and time, and “… in doing so win more and lose less”. The guidance scheme seeks to improve the overall outcome of these trade-offs by building on knowledge gained from multiple-use ecosystem management, and through identifying ecosystem services more explicitly.

Nonetheless, Daily and Matson (2008) note that “… transformations will be required to move from conceptual frameworks and theory to practical integration of ecosystem services into decision-making, in a way that is credible, replicable, scalable, and sustainable”. In relation to soil ecosystem services, Daily (1997) argued that “research to better characterize [than the MA classification] the ecosystem services supplied by soils, was needed, along with a better understanding of the relationships between the services supplied by soils and other systems”. Dominati et al. (2009a) noted that in the MA scheme the processes of soil functioning, that is the means of production of goods and services, are mixed up with the benefits. They have provided a draft framework that makes explicit distinction between soil processes and the benefits derived from soil. Their draft framework relates the inherent and manageable properties of the soil’s natural capital to natural capital stocks and inventories, as well as the underlying supporting processes. Dominati et al. (2009b) then link the soil’s natural capital through provisioning processes, regulating processes, and socio-cultural processes to the servicing of human needs and creating benefits.

Daily and Matson (2008) conclude that “… there remain many highly nuanced scientific challenges for ecologists, economists, and other social scientists to understand how human actions affect ecosystems, the provision of ecosystem services, and the value of those services.” This is an exciting area of active research, which is not without challenges.

We now present three examples of securing sustainable returns on investment into the soil’s natural capital: we link soil processes controlled by carbon to soil ecosystem services; we value supporting and regulating soil services in the provision of terroir value for wine; and we use a valuation of the soil’s natural capital to allocate a nutrient-loss right for policy to limit non-point source pollution by nutrients from farms.

**Valuing Carbon and Soil Ecosystem Services**

Soil carbon is critical for the biophysical processes operating in the soil and it determines the value of the ecosystem services provided by soil. Carbon breathes life into soil. Soil carbon confers fertility to the soil beyond that provided by the weathered parent material. Carbon acts to sustain the soil functioning, which enables soil to provide the ecosystem services that are essential for human well-being. Soil without organic carbon, we call rock. The natural capital stock of soil carbon is, in response, strongly controlled by land management practices. There are good returns on investment from enhancing the natural capital stocks of carbon. Lal (2009) has outlined land management strategies for enhancing soil carbon stocks: growing more biomass, recycling biomass and biosolids, reducing soil erosion, decreasing leaching, and minimising decomposition. Here we discuss the changing value of the soil’s
ecosystem services as a result of different land management practices, which have resulted in neighbouring apple orchards ending up with different values of soil carbon.

Deurer et al. (2008) studied an organic apple orchard, and a neighboring integrated apple production system in the Hawke’s Bay, New Zealand. Both orchards had the same general soil characteristics. The genoforms of the soils (Droogers and Bouma, 1997) are Fluvisols and have a silt-loam texture. The organic orchard system had been under organic management since 1997. The apple trees in the orchard were 13 years old. The apple cultivar was ‘Braeburn’, and the rootstock cultivar was MM.106. Green-waste compost was applied to the topsoil of the tree rows once a year at a rate of 5 to 10 t/ha, and lime was added at a rate of 300 kg/ha every 4 years. The apple trees in the adjacent integrated orchard system were 12 years old. The apple cultivar was ‘Sciros’/Pacific Rose™, and the rootstock variety was MM.106. A 0.5-m wide strip under the trees was kept bare by regular herbicide applications. The apple trees were drip-irrigated during the vegetative period. The irrigation, nutrient, and pest management followed the guidelines of integrated fruit production.

Spatially averaged, the total carbon stock in the integrated orchard is now 2.6 kg-C m$^{-2}$, whereas it is 3.8 kg-C m$^{-2}$ in the organic orchard. Deurer et al. (2009) found by substrate-induced respiration that the microbial biomass carbon in top 100 mm of the soil of integrated orchard was 73 g-C m$^{-2}$, whereas it was two times higher at 143 in the organic orchard. They also found a near-twofold difference in anecic worm fresh weight in the soil between the integrated orchard (85 g m$^{-2}$) and the organic orchard soil (154 g m$^{-2}$). Thus after 12 years, these two similar soils now have different phenoforms (Droogers and Bouma 1997) as a result of the different carbon strategies and orchard-management practices. Not surprisingly then, the structures of the soils are now also quite distinct, as the two X-ray images in Figure 10.5 reveal. Deurer et al. (2009) have used tomographic analysis of these images to determine the impact of the different carbon contents on the macroporous structure of these soils.

Here, by selecting tomographic analyses from two cores, one each from the different orchards, we reveal the magnitude of, and reasons for the different macroporous structure. Using the methodology of Deurer et al. (2009) we present in Figure 10.6 the detailed profile in macroporosity of two cores from that study. Here, we take a macropore to be a pore whose diameter is greater than 0.3 mm.

For core 1 from the integrated orchard, the average volumetric macroporosity is 3.1±1.4 %, whereas it is 9.4±1.7 % for the organic orchard. Deurer et al. (2009) present the results from the additional two cores taken from each orchard. On average the macroporosity found for soil of the integrated orchard was 2.4±0.5 %, and for the organic orchard soil it was 7.5±2.1 %. This difference was despite the mean macropore radii being similar between the integrated orchard (0.41±0.02 mm) and the organic orchard (0.39±0.01 mm). Different carbon management practices have,
therefore, led to these soils now having different phenoforms, especially in relation to macroporosity.

These different phenoforms will also have different functioning, and so the ecosystems services they provide will also be different. Clothier et al. (2008c) identified that 12 of the 17 terrestrial ecosystem services valued by Costanza et al. (1997) would be enhanced by soil macropores, and they estimated the global contribution of macropore functioning to be worth US$304 billion per year. Thus the natural capital value of the soil in the organic orchard is now considered to be greater than that of the integrated orchard, as a result of the different carbon management practices.

We detail the contrasting value of the soil ecosystem services differentially provided by these soils. We consider the different value of different service types: two are regulating services and one is a supporting service.

Deurer et al. (2009) simulated gaseous diffusion through the macropore systems of the two different soils (Figure 10.5). The relative diffusion coefficients, that is, relative to free-air diffusion, at the aggregate scale in the organic orchard soil was 0.024, whereas it was an order of magnitude less, at 0.0056 in the integrated orchard. This difference provides, we consider, a regulating service, for we hypothesise that this different gaseous functioning at the aggregate scale would reduce nitrous oxide production and emissions should the soil be wet. So carbon management in the organic orchard has provided some mitigation of climate change firstly by sequestering carbon in the soil, and then by providing a regulating service through reducing the emissions of nitrous oxide, a potent greenhouse gas.

Aslam et al. (2009) asked whether this increase in organic carbon would improve the filtering functioning of soil for organic pesticides – a regulating ecosystem service. Not only did they consider these two orchard soils, they also considered phenoforms of a pastoral soil whose carbon contents now differ: the soil from a stock camp had a total carbon content of 8.5%, whereas the same soil type of the grazed pasture outside the stock camp was 4.8%. They found that for both land uses, the increase in soil carbon significantly increased the values of indicators of the pesticide filtering functioning for sorption and degradation of pesticides. This pesticide regulation is a valuable service, for Pretty et al. (2000) have assessed that the total external costs of pesticide use in agriculture in the United Kingdom is £33 ha\(^{-1}\) of farmland receiving pesticides -which on average amounted to 3.84 kg-active ingredient ha\(^{-1}\). The total external cost to the UK of agricultural pesticide-use was found to equal £193 m. So, if by enhancing levels of soil carbon some of this could be filtered and rendered harmless in the soil, then the value of the external cost saving from that regulating service would be substantial.

However, in the pastoral system the higher level of soil carbon was found by Aslam et al. (2009) to increase the degree of soil hydrophobicity. This at first glance might suggest an ecosystem dis-service (Zhang et al. 2007), for hydrophobicity could probably lead to increased run-off of rainwater, which would be considered a regulating dis-service by wasting water and potentially leading to soil degradation through sheet erosion. However, as noted by Clothier et al. (2008c), if a hydrophobic soil also has an open-vented macroporous system, then any surface water-film created
by water repellency would quickly be captured by the macropores and routed to the subsurface soil where water contents are likely to be higher, and the degree of hydrophobicity less severe to enable absorption there. Thus macropores can provide a regulating service of water capture to overcome the dis-service created by the functioning of hydrophobicity.

The prime reason the organic growers use composts is to provide the supporting ecosystem service of nutrient generation from the composted organic matter. Over and above the provision of nitrogen directly from the compost, the resulting change in the carbon content of the soil was found by Kim et al. (2009) to enhance the supporting soil service of nutrient creation through nitrogen (N) mineralisation. They found using undisturbed soil cores that the average net N-mineralisation, across three temperatures, in the integrated orchard was 0.41±0.25 µg·N·g⁻¹·d⁻¹, and some five-fold higher in the organic orchard soil at 2.28±0.50 µg·N·g⁻¹·d⁻¹. This difference they found to be correlated with relative sizes of the labile pool of soil carbon as measured using the hot-water extractable content of the soil’s C (Sparling et al. 1998). The hot-water carbon content of carbon in the integrated orchard was found to be 0.06 kg·C·m⁻² (cf. total C at 2.6), and for the organic orchard it was 0.11 kg·C·m⁻² (cf. total C at 3.8). Thus through different orchard practices in relation to organic matter, the soil of the organic orchard now has a higher level of total soil carbon and a higher fraction of labile soil carbon than the integrated orchard. As a result of different soil functioning, the supporting ecosystem service of nutrient generation is greater in the soil phenoform with the higher soil carbon content.

Soil productivity in response to climate change can be sustained if good carbon management practices are adopted. Furthermore, these same carbon management practices can actually assist with the mitigation of, and adaptation to climate change. Sustaining soil productivity thus enables our landscapes to provide the valuable provisioning service of food production.

Valuing Terroir

The ecosystem services provided by soil, climate, and local landscape, coupled with grape variety and the skills of the viticulturalists and oenologists confer on wine the valuable notion of terroir. Studies have shown that soil and weather ecosystem services explain most of the terroir effect (Conradie et al. 2002; Morlat 1998), with vine water supply, soil depth and potential vine vigour being the major contributing factors (Morlat 2001, Bodin and Morlat 2006). Here we quantify the role of the supporting ecosystem services of soil water storage and nitrogen mineralisation on the natural capital value of terroir. We have developed a natural capital valuation model for terroir that contains four sub-models of a plant-growth meta-model based on weather and soil, soil and water management practices, environmental impacts, and economic valuations. Here we have applied this to examine two aspects of natural capital: the terroir return on investment into water for irrigation, and the return on investment of changing the carbon content of the soil. Salient details are described below.

Plant growth model A simplified meta-model of SPASMO (Soil Plant Atmosphere System Model) (Green et al. 2008) is used to grow the vine through capturing carbon and allocating it to the plant parts of shoot, root, leaf and berry in relation to the
prevailing weather conditions and the status of soil-water and nitrogen in the rootzone. For simplicity the soil is considered as a single layer, and for each run we output the results for soil depths from 200 mm through to 1800 mm in steps of 200 mm. The model runs on a weekly time step and used a 30-year weather record for the viticultural region of Marlborough, New Zealand.

**Soil and water management practices** Irrigation is applied whenever the soil water content in the rootzone drops to a trigger value whereupon the deficit is replaced. Should the water content of the root zone exceed field capacity then drainage and leaching occurs. Two dressings of nitrogen (N) in 10 kg-N ha\(^{-1}\) applications on 1 November and 1 January are applied. Meanwhile nitrogen is mineralised by the soil in relation to the labile carbon fraction of the soil’s humus pool, and in response to soil temperature. Nitrogen is also mineralised from the litter pool of prunings, and there is transfer of carbon between the litter and humus pools. Nitrogen is leached depending on the concentration of nitrate in the soil solution, and nitrous oxide is generated according to IPCC rules in relation to nitrogen fertilisers and prunings. Vine growth is enhanced according to the water and nitrogen status of the root zone. The vines are pruned every time the canopy biomass reaches a trigger value, and the prunings left on the soil surface as decomposing litter.

**Environmental impacts** At present for environmental costs we have only considered nitrate leaching and nitrous oxide emissions by assigning an environmental cost to nitrate leaching of NZ$10 per kg-N ha\(^{-1}\) (after Pretty et al. 2000) and a cost due to nitrous oxide emissions using IPCC rules with the price of carbon set at NZ$50 t\(^{-1}\). It would be possible to add in easily the costs of other externalities, such as pesticides, by using such information as presented by Pretty et al. (2000).

**Economic costs and valuation.** We define a temporal trend from bud break to harvest in both the soil water content of the rootzone (Figure 10.7a) and soil nitrogen storage (Fig 10.7b) that we consider would confer the maximum value to terroir. Freely available water is considered best right up until flowering to ensure maximum fruit set and optimum nitrogen storage in the roots, vine, shoots and leaves. After flowering through until veraison, we consider it best if there is a reduction in soil water and nitrogen, so as to limit vegetative vigour (Green et al. 2008). Following veraison, lower levels of water and nitrogen are considered ideal to limit vegetative vigour, enhance light penetration to the bunches and encourage a rise in sugar (ºBrix) levels in the berries. Our economic modelling penalises any deviation from these ideal time courses.

The cost of any daily deviation from these ideal paths is subtracted, in weekly blocks, from the maximum terroir value. A bottle of Sauvignon blanc from Marlborough can fetch over £20 in London, and so we take here the maximum terroir value to be NZ$25,000 ha\(^{-1}\). The daily deviation cost for when the soil water lies to either side of the optimum is taken to be NZ$20,000 per (m\(^3\)/m\(^3\)) of water content. For nitrogen, because of its greater impact on vine vigour, any deviation from the optimum is penalised by the square of the distance from the ideal value, and a positive deviation is penalised more than a deficit. The daily penalty cost for N excess is taken to be NZ$0.40 per (kg-N ha\(^{-1}\))^2, and NZ$0.20 per (kg-N ha\(^{-1}\))^2 for an N deficit. The weekly
penalty costs for deviating from the ideal water and nitrogen trajectories are summed, and then subtracted from the maximum terroir value, as are the environmental costs associated with nitrate leaching and nitrous oxide emissions.

The first application of the terroir calculator we consider here is for the case where the natural capital stock of blue water available for irrigation (Rockström et al. 1999) is considered to be free. The model irrigates ‘perfectly’ by meeting green-water demands as required, and the only cost associated with irrigation is thus operational, say because of pumping costs, which we set at NZ$20 ha$^{-1}$. The value of terroir in this case is shown in Figure 11.8 as a function of soil depth. For shallow soils, there is a need to irrigate more frequently, and the operational cost of doing so reduces net terroir value. For deeper soils, the larger water holding capacity and greater provision of nitrogen through mineralisation result in greater deviations away from the sought-after decline in water and nitrogen stocks. There are additional costs associated with leaf plucking and pruning. Morlat and Bodin (2006) found that Chenin vines growing on deeper soils had larger berries that were lower in sugars and anthocyanins, higher acidity, and with a lower phenolic index than those growing on shallower soils. In our case, a soil depth of 800 mm would maximize terroir value in relation to water and nitrogen regimes.

[Insert Figure 10.8 about here]

If water were not taken to be free because a value had been assigned to the natural capital stocks of blue-water used for irrigation, then we can assess what impact this might have on terroir value. If we now re-run our meta-model with the price for water at NZ$4 m$^{-3}$, being the cost of domestic water in some American and European cities, we obtain quite a different pattern for terroir value. The return on investment into the soil’s natural capital changes (Figure 10.8). It is now less because of the additional costs associated with using irrigation to meet water demands over and above those that cannot be supplied as green water from the supporting services from the soil and climate. For the shallowest soil of 200 mm, the inability to store sufficient water, and the now the additional costs of supplying the vine’s needs, mean that terroir is almost valueless. In this case where water is a valuable commodity, deeper soils will have a much higher natural capital value.

[Insert Figure 10.9 about here]

Soil carbon is an inherent property of the natural capital value of soil through its role in controlling soil functioning. This inherent property would relate to the genoform of the soil (Droogers and Bouma, 1997). But, in addition, soil carbon can be changed through land management practices (Figure 10.5), and so it is also a manageable property of the soil’s natural capital value (Dominati et al. 2009b). Thus land management by changing the soil’s carbon content can affect its regulating and provisioning ecosystem services, and therefore alter its natural capital value. The soil with a changed soil-carbon content due to land management practices is referred to as a phenoform (Droogers and Bouma 1997). In Figure 10.9 we compare the terroir value of the genoform with its soil carbon content of 1%, with that of the phenoform having a soil carbon content of 1.75%. We altered the water and nitrogen provisioning services of the soil in relation to C using the soil water content changes suggested by Rawls et al. (2003), and for nitrogen mineralisation by Kim et al. (2009).
The 0.75% rise in soil carbon would, for such a sandy soil, result in a 3.75% increase in the field capacity value of the soil according to Rawls et al. (2003). The rise in soil carbon would, according to Kim et al. (2009) change the maximum mineralisation rate from 0.2 mg-N l\(^{-1}\) d\(^{-1}\) to 0.39 mg-N l\(^{-1}\) d\(^{-1}\). The increase in the water and nutrient supporting services provides a marginal increase in terroir value for very shallow soils, but once the soil depth increases beyond about 500 mm, the additional water and nitrogen results in vegetative vigour as a result of not being able to track down the sought-after declining pattern of soil water content and nitrogen storage. The prescribed penalty of deviating from the nitrogen line of NZ$0.40 per (kg-N ha\(^{-1}\))\(^2\) means that ‘pruning’ costs go from NZ$294 ha\(^{-1}\) for a 200-mm deep soil to NZ$12,795 ha\(^{-1}\) for an 1800-mm deep soil.

Yet the comparative pattern of the two curves in Figure 10.9 does suggest how carbon management can be used in viticulture to enhance soil health and the provision of supporting, regulating and provisioning services. Mulches of composted prunings and marc (the crushed grape skins) can be used to raise the carbon content of just the surface soil. Such a shallow depth of higher carbon contents does not provide excessive vigour through either enhanced water holding capacity, or nitrogen mineralisation (Figure 10.9). Indeed, there are many other associated soil and vineyard-health benefits from using mulches in vineyards, as detailed by Agnew et al. (2002). These include soil microbial diversity, weed suppression and reduced herbicide use, plus increases in the yeast available nitrogen in the grape juice.

Thus if nimble land management practices are used to manage better carbon in vineyards, the grower will be able to secure good returns on investment into the natural capital of the soil through enhanced supporting, regulating and provisioning services.

**Land use policy, nutrient management and natural capital**

To maintain food supplies and enhance the productive capacity of the landscape, there has been land-use intensification through the increasing use of fertilisers and irrigation to overcome shortfalls in the soil’s natural capital (Mackay 2008). Soils have pores and are leaky. So there is a hydrologic connection between the soil of the rootzone and both surface and underlying groundwaters. The intensification of land use through fertiliser use and irrigation to realise levels of supporting services well beyond that supported by the natural capital value of the soil is putting at risk both our terrestrial and marine water resources. The Guardian Weekly noted in 2002 that “water is now known as ‘blue gold’ … and ‘blue gold’ is this century’s most urgent environmental issue” (Vidal 2002). Land management determines water quality. There is increasing urgency to manage our lands sustainably so that the ‘gold mine’ of our waters is protected and enhanced. It is imperative for our productive and ecological futures that we develop policy for sustainable management of land to protect the natural capital of our ground and surface waters.

Nutrient management policies have been developed over the last 20-30 years to limit non-point source pollution by fertilizers. In 1980 the European Union issued an EC Directive on “The Quality of Water Intended for Human Consumption”. In 1991 they issued an EU Nitrates Directives. Agriculture was increasingly becoming regulated by environmental policies. The Dutch introduced a nutrient accounting system called
MINAS in an attempt to improve water quality. Levies would be charged if farms exceeded certain nutrient loadings on the environment. However this had limited success because the levies were not sufficiently prohibitive, relative to the productive value of fertiliser. The Dutch have now simply resorted to focusing their controls by limiting applications of nutrients. In New Zealand, some Regional Councils have developed policy for nutrient caps based on a benchmarking of current fertiliser and farm practices, so called grand-parenting. Trading and demanding changed farm practices is then intended to be used to bring down the loadings on water bodies from farms. This is the so-called ‘cap and trade’ process. Such policy is a blunt instrument focused on inputs, and there are equity issues involved that inadvertently favour ‘today’s polluter’. In Australia, environmental auctions are being used to procure changes that will improve ecosystem services, and through bundling these it is hoped to realise water quality improvement, greenhouse gas reduction and habitat provision. However, because of the small financial incentive in the value of the auction, relative to the required magnitude of the changes required to address widespread non-point source pollution by nutrients, these auctions are unlikely to be successful.

Reconsidering how we view nature’s bounty by acknowledging natural capital stocks would, we consider, provide a better foundation upon which to base nutrient management policy. As we pointed out earlier, Costanza (2009) concluded that if we are to move toward a new sustainable economy that we need to recognise “… that natural capital [is] not infinitely substitutable for built capital, and that real biophysical limits exist”. Hawken et al. (1999) questions as to “… how it is that we have created an economic system that tells us it is cheaper to destroy the earth and exhaust its people than to nurture them?” They answer that this will be “… when natural capital is no longer treated as free, unlimited and inconsequential, but as an integral and indispensable part of production [and] our system of accounting will change”. One natural capital asset they highlighted was soil, noting that “… we can no more manufacture a soil with a tank of chemicals than we can invent a rainforest”, and they concluded that “… understanding soil, the ultimate natural capital is the key to changing agriculture from part of the climate problem into part of the solution”.

We now outline how we have been involved with policy analysts in developing a nutrient management policy for Horizons Regional Council in New Zealand that is based on the concept of natural capital. Horizons’ new omnibus resource management policy (http://www.horizons.govt.nz/default.aspx?pageid=307 - valid 6 August 2009) the One Plan has in Rule 13-1, classified dairy farming and other intensive land uses as a controlled activity, rather than a permitted activity as it is now. Farming as a controlled activity will, if the policy becomes law, require a resource consent with nitrogen leaching and run-off values calculated in accordance with a Farmer Applied Resource Management Strategy (FARM Strategy). The hearings for the One Plan take place in late 2009, with policy intended to be passed into law in 2010. Here, we outline how the FARM strategy, which is based on natural capital, was developed.

Current nitrogen (N) loadings in the Upper Manawatu River are more than twice (744,000 kg-N/year) the N limits (341,000 kg-N/year) based on recommended standards in the One Plan. Horizons Regional Council have good data sets on the contribution of the major point-source N loadings to the river. Remedial actions have been successful. We sought to determine the contributions of non-point source N
loadings from dairy and sheep-beef farms in the Upper Manawatu catchment. By looking into the N loadings in the river from two linked catchments with different land-use patterns, we inversely inferred that the N lost to the river from the average dairy farm would amount to 15.4 kg/ha/year and for sheep-beef farm the N loss would on average be 3.9 kg/ha/year (Clothier et al., 2007). Over 90% of the total N in the river was found to be from these two non-point sources, with dairy contributing about half the N loading in the river, despite only representing 16% of the land use in the catchment (Clothier et al. 2008a)

The N loss from the rootzone within the average dairy farm was calculated using the OVERSEER® farm nutrient-model and found to be 31 kg-N/ha/year, and for the average sheep and beef farm, 7 kg-N/ha/year. Obviously then not all the N lost from farms makes it into the river, for en route there are losses and attenuations. But a link between farm practice and river water quality had been established, with the N transmission coefficient being 0.5 for both dairying and sheep and beef operations (Clothier et al. 2008b). From this transfer function linking of land-management decisions, to farm N losses and the nutrient loadings in the river we could explore resource management policy options that would protect river water quality.

Mackay et al. (2008) noted that there are a number of policy approaches that could be used to manage nutrients on farms in order to protect receiving waters. These include:

- **Capping** current production systems and allocate nutrient losses on the basis of present performance by a process that has been called grand-parenting. Then there would be a managing downwards of these caps by ensuring improved farm management, and possibly by trading under what is termed a ‘cap & trade’ market

- **Limiting** the losses of nutrients from intensive land uses. This focussed approach on the N loss ‘hot spots’ would place restrictions on any further intensification and would require mitigation practices for any further land development, even including practices for the currently less intensively farmed lands

- **Equalising** nutrient loss limit across the catchment. This democratic approach would achieve water quality standards by sharing the loss between all land owners. For the Upper Manawatu catchment this would be a river-sensed loss of 6.5 kg-N/ha for each farm, or a reduction of 60% from current values

- **Allocating** N-loss rights based on the biophysical potential of the inherent natural capital value of the soils across the farm. If all land in the catchment had the same inherent natural capital, then equalising the nutrient-loss limit across whole catchment would be simple. The reality is, however, that inherent natural capital varies enormously across the catchment from elite and versatile soils, through to soil with substantial limitations for production agriculture.

Horizons Regional Council opted for the last of these in their One Plan.
How then can the value of the soil’s natural capital be quantified for use in determining an N-leaching loss right? Presently direct methods for calculating a soil’s natural capital value are still in development. Dominati et al. (2009b) have proposed a draft framework for classifying and measuring the value of the natural capital and ecosystem services of soil. This is based on our understanding of soil-forming processes, soil taxonomy and classification, plus soil processes and the links to land use.

In the absence of a method for calculation of a soil’s natural capital, a proxy that serves as a useful alternative is the ability of the soil to sustain a legume-based pasture that is fixing N biologically and is under optimum management, and this is before the introduction of additional technologies. A legume-based pasture is a self-regulating biological system with an upper limit on the amount of N that can be fixed, retained, cycled and made available for plant growth. It reflects the underlying capacity of soil to retain and supply nutrients and water, as well as the capacity of the soil to provide an environment to sustain legume and grass growth under the pressure of the grazing animal. Potential production therefore reflects the underlying biophysical capacity of the soil’s natural capital value and the ecosystem services of the climate to allow production with resilience and durability. To calculate the N-loss limit for a given landscape unit, the potential animal stocking rate that can be sustained by this legume-based pasture fixing N biological, under optimum management, before the introduction of additional technologies, is listed in the extended legend of the Land Use Capability (LUC) worksheets “Attainable potential livestock carrying capacity”.

The LUC system (Lynn et al. 2009) has two components. Firstly, the land resource inventory is compiled, which is an assessment of physical factors critical for long-term land use and management. Secondly, the inventory is then used for LUC classification, whereby land is categorised into eight classes according to its long-term ability to sustain one, or more productive uses. The LUC classification ranges from Class I through to Class VIII, and this is in essence a classification of natural capital value, from high value land (Class I) to land whose value is less because of various limitations as a result of erosion, wetness, soil or climate (Class VIII). The language of the LUC system is that of natural capital and ecosystem services.

From the attainable potential livestock-carrying capacity in the LUC, namely a measure of the supporting ecosystem services of that LUC class, a value for pasture production can be determined. This can then be used in OVERSEER® to calculate the N-leaching loss under that grazed pastoral use. The N-loss values that Mackay et al. (2008) calculated for each of the LUC classes were then used by Horizons Regional Council to establish Table 13-2 of their One Plan (Figure 10.10), and this forms the basis of the FARMS Strategy that was incorporated into the One Plan.

[Insert Figure 10.10 about here]

It can be seen that soils of higher natural capital value are provided with a higher N-leaching loss right in terms of kg-N ha⁻¹ y⁻¹. This approach recognises the value in the natural capital of the most productive soils, and this acknowledges the reduced need to substitute for natural capital ‘failings’ by using fertilisers as added capital. The unit of calculation for the FARMS strategy is the farm. The loss right for each of the LUC units within the enterprise is really summed to provide a loss right for the farm in kg-
This provides the farmer with flexibility to manage the enterprise as a unit to develop the most productive configuration of the farm, albeit within the constraint of the farm’s total loss right. This approach enables change away from N-leaching losses based on resource-use efficiency to one that recognises the flexibility of landscapes in relation to their natural capital values and their versatility for productive land uses and their mitigation options.

As time progresses under the One Plan, through its policy imperatives, improved farm practices are sought so that there is a continuous reduction in N losses from the landscape. The reduction in N loss is focused on the soils of higher natural capital value, LUC I-IV, for these more versatile soils are better able to be managed to mitigate N-losses. Over time, water quality will improve, and the improvement in river water quality is predicted by the transfer function that we established (Clothier et al. 2008b). This policy approach recognises the underlying natural biophysical resources of the landscape, irrespective of current land use, or even of future patterns of land use. It provides all land users in the catchment with certainty by defining a nutrient loss limit based on the biophysical assets of the suite of soils across their farms. The policy does not focus on capping inputs, but rather it seeks to engage with land-users to maximise their return on investment into their natural capital assets and to encourage sustainable management of the landscape’s biophysical resources without compromising the ecosystem services of receiving environments.

**Conclusion**

Sustaining soil productivity in response to climate change is imperative. The supporting, provisioning and regulating ecosystem services provided by soil are critical for food provision, and for meeting the challenges of climate change. In many cases, however, the supporting services of soils are inadequate for sufficient provisioning of food, fibre and fuel. Water and nutrients are often needed to ensure economic levels of plant and animal productivity. Thus, irrigation and fertilisers are frequently used to overcome inadequate natural stocks of water and nutrients in the soil. Intensification of land-use by this substitution of built capital for shortfalls in natural capital can, however, degrade the soil’s regulating ecosystem services and imperil the asset values of the natural capital stocks of our soils and receiving waters. In developing sustainable farm-management practices and for drafting good resource management policy, if we adopt an ecosystems services approach this will enable us ‘win more and lose less’ by providing us with an assessment of the ecosystem impacts of our decisions, so that we might continue to be better able to feed the world and mitigate climate change.
References


Dominati, E., Patterson, M., Mackay, A. 2009a. A framework for the ecosystem services provided by soils. Paper, 8th International Conference of the European Society for Ecological Economics, University of Ljubljana, Slovenia, 29 June - 2 July 2009.

Dominati, E., Patterson, M., Mackay, A. 2009b. A draft framework for classifying and measuring soil nature capital the ecosystem services. Ecological Economics [in draft form]


Green, S.R., B.E. Clothier, C. van den Dijsssel, M. Deurer and P. Davidson, 2008. Measurement and modelling the stress response of grapevines to soil-water deficits in their rootzones. Chapter 15, In Soil Science Society America Monograph “Modeling the response of crops to limited water: Recent advances in understanding and
modeling water stress effects on plant growth processes”, L. Ahuja et al.[Eds] Chapter 12, pp 357-386.


Figure Captions

Figure 10.1. Franklin Hiram King (1848-1911), Professor of Agricultural Physics, University of Wisconsin – Madison between 1888 and 1901. King travelled to Asia in 1909 and wrote the book “Farmers of Forty Centuries”, which was published by his wife soon after his death in 1911 (reproduced with the kind permission of Dover Publications, who republished his book in 2004).

Figure 10.2. Where yield is the product of brain, brawn and utilized waste. This is a reproduction of Figure 113 from F.H. King’s book “Farmers of Forty Centuries (reproduced with the kind permission of Dover Publications, who republished his book in 2004).

Figure 10.3 The Intergovernmental Panel on Climate Change projection for the global change in the availability of stocks of water at the end of the 21st century under scenario A1B (reproduced with permission from Climate Change 2007: Synthesis Report. Contribution of Working Groups I, II and III to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change. Figure 3.5. IPCC, Geneva, Switzerland).

Figure 10.4 The four-way classification of ecosystem services of the Millennium Ecosystem Assessment (2005) and the links between these services and the five constituents of human well-being.

Figure 10.5 X-ray tomographic images of soil using a Metris X-tek Benchtop CT system at SIMBIOS Centre of the University of Abertay (Dundee, Scotland) with the 160 kV X-ray source and a 12-bit CCD camera (adapted from Deurer et al. 2009)

a. (Left) An undisturbed soil core (70 mm diameter, 100 mm length) taken from row of the integrated apple orchard where the top of the column is the soil surface.

b. (Right) As for 11.5a, but for an undisturbed soil core from the row of the integrated apple orchard.

Figure 10.6. The macroporosity (pores > 0.3 mm diameter) as a function of depth below the soil surface in the tree row of an organic (core 9) and integrated (core 1) apple-orchard system. On average the volumetric macroporosity of core 1 was 3.1% ± 1.4 and of core 9 it was 9.4% ± 1.7. For each depth, and core, the average and standard deviation of three sub-columns is shown, where the dimension of the sub-columns was 43 x 20 x 17 mm, such that the volume of the sub-columns was 14620 mm³.

Figure 10.7 a. The time course of water content in a viticultural soil in Marlborough, New Zealand, that would provide for optimum conditions for vine and berry growth and maximise the quality of the grapes to confer maximum terroir value.
b. The time course of soil nitrogen storage in a viticultural soil in Marlborough, New Zealand, that would provide for optimum conditions for vine and berry growth and maximise the quality of the grapes to confer maximum terroir value.

Figure 10.8 The pattern of terroir value as a function of soil depth calculated for a viticultural soil in Marlborough, New Zealand. The maximum terroir value was set at NZ$25,000 ha\(^{-1}\) and penalties were accrued throughout the season as rootzone conditions deviated from the ideal time courses for water and nitrogen stocks outlined in Figure 10.7. For the upper curve (solid line) there was no cost for the water, whereas for the lower curve (dotted line) water was priced at NZ$4 m\(^{-3}\).

Figure 10.9 The pattern of terroir value as a function of soil depth calculated for a viticultural soil in Marlborough, New Zealand. For the upper curve the soil had a soil carbon content of 1%, whereas for the lower curve (dotted line) the carbon content of soil was increased to 1.75% and there was enhanced soil water storage and nitrogen mineralisation.

Figure 10.10. The maximum nitrogen loss values for each Land Use Capability (LUC) class adopted as Table 13-2 by Horizons Regional Council in their notified One Plan. LUC is a proxy for natural capital value, and for LUC classes I through V, the One Plan seeks reductions in losses so that receiving-water quality is improved.
Figure 10.1

Figure 10.2
Predicted changes in water availability at the end of the 21st century

Scenario A1B, 2090-2099

Figure 10.3

CONSTITUENTS OF WELL-BEING

Security
- Personal safety
- Secure resource access
- Security from disasters

Basic material for good life
- Adequate livelihoods
- Sufficient nutritious food
- Shelter
- Access to goods

Health
- Strength
- Feeling well
- Access to clean air and water

Good social relations
- Social cohesion
- Mutual respect
- Ability to help others

Source: Millennium Ecosystem Assessment

Figure 10.4
Figure 10.5

Figure 10.6
Figure 10.7

Figure 10.8
Figure 10.9

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<td>29</td>
<td>22</td>
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Figure 10.10