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## **On the Value of Soil Resources in the Context of Natural Capital and Ecosystem Service Delivery**

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**ABSTRACT**

The ecosystem services approach endeavours to incorporate the economic value of ecosystems into decision making. This is because many natural resources are subject to market failure. As a result many economic decisions omit the impact that natural resource use has on the earth's resources and the life support system it provides. Hence, one of the objectives of the ecosystem services approach is to employ economic valuation of natural resources in micro- and macro-economic policy design, implementation and evaluation. In this article we examine valuation concepts, and ask why we might attempt to economically value the contribution of soils to the provision of ecosystem services? We go on to examine economic valuation methods, and review economic valuation of soils. By surveying prices of soils on the web we are able to make a first, limited, global assessment of direct market value of topsoil prices. We then consider other research efforts to value soil. Finally, we consider how the valuation of soil can meaningfully be used in the introduction of improved resource management mechanisms such as decision support tools on which valuation can be based, within the UN's System of Environmental and Economic Accounts (SEEA), and policy mechanisms like Payments for Ecosystem Services (PES).

## INTRODUCTION

In recent decades, prominent soil scientists have argued that the soil resource is consistently overlooked or undervalued by society (Bridges and Catizzone, 1996, Bouma, 2005). Yet there appears to be a resurgence of interest in the soil resource, principally in the context of food security, climate change and land stewardship (Koch et al., 2012; Mueller et al., 2012; Jones et al., 2013); especially as it is recognized that an increasing population is stressing our planet's life support systems (Rockstrom et al., 2009). Along with the ecosystem services soils help deliver (Daily et al., 1997; Haygarth and Ritz, 2009; Dominati et al., 2011; Robinson et al., 2013a), soils are increasingly recognized as a key component of the critical zone (Banwart, 2011), the thin layer of the earth's surface from tree-top to bedrock, the biogeochemical engine at the heart of the earth's life support systems, with soil formation underpinning ecosystem services (MEA, 2005). Yet, soil science appears slow in 'refocusing and mobilizing our creative talents' to tackle these broader societal issues that, by its very interdisciplinary nature, is well suited to respond to; why is this?

Bouma (2005) in an article about soil scientists in a changing world, considers that the relationship between soil and society can be considered in the context of, (i) the 'true' soil, explored through scientific investigation, (ii) the 'right' soil, which considers how stakeholders deal with soil in a policy making context, and (iii) the 'real' soil, how individuals and society feel about soils. Bouma makes the point that traditionally soil science has been mostly concerned with the 'true' soil, and perhaps neglected the other two. However, soil science has made some significant contributions to link to policy including the application and development of the Driver-Pressure-State-Impact-Response (DPSIR) framework (Blum et al., 2004).

Within ecology, there has been a rapid development of the ecosystem services approach (Costanza et al., 1997a; Daily, 1997). Ecosystem services, starting out as a metaphor to help us think about nature has now become an integral part of the science-policy debate on the environment (Norgaard, 2010). National and international policy making agencies, such as the United Nations, have been quick to adopt the ecosystem services approach. A growing challenge for soil science is to determine how it fits within this approach as relatively little thought has been given to soils<sup>1</sup> (in relation to science, social science and policy making). The ecosystem services concept goes beyond ecosystem

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<sup>1</sup> This lack of consideration is highlighted by the fact that within the economic analysis conduct as part of the UK NEA there is no consideration of the costs associated with soil erosion; see footnote 92 in Bateman (2012).

function, in that it introduces a subjective/anthropocentric value for ecosystem functions that provide goods and services. The concept that ecosystems and soils provide services of value to society is perhaps a more meaningful way of conveying the importance of soil functions to decision makers and the wider public, who are already familiar with manufactured goods and services in consumer societies.

As a result of the pressure on policy makers to consider soil multi-functionality in their decision making regarding the use of land, it is vital that soil functions are prominent in decision making frameworks. To date, the value of soil has been largely subsumed in the value of land and land use activities, and as such is only implicitly valued. This is one reason why an ecosystem service approach is attractive from a policy makers viewpoint, as it may allow them to see the implications of decisions and trade-offs if soil functions are fully incorporated in decision making frameworks. However, to date, soils are poorly addressed in ecosystem service approaches. In the MEA (2005) soil formation is identified as a vital supporting service. In the follow-up activity to the MEA assessment, suggesting an approach used to assess the economic value of ecosystem services, the TEEB approach, doesn't talk about supporting services anymore following de Groot et al (2002), but identifies supporting processes and functions which underlie the delivery of all ecosystem services. It is therefore incumbent on soil science to contribute to these approaches, by clearly identifying valuable soil functions (Daily et al., 1997; Lavelle et al., 2006; Haygarth and Ritz, 2009; Dominati et al., 2010; Robinson et al., 2012) and developing appropriate approaches, demonstrating the role of soil processes and functions in the maintenance of the final ecosystem service delivery supply chain (Dominati et al., 2010; Robinson et al., 2013a).

We recognise that ecosystem service concepts are not without criticism, with those opposed arguing that ecosystem management cannot, and should not, be reduced to cost-benefit-analysis. However, this article is not about promoting the economic model, it is a critical review of the approach, its drawbacks, and the potential opportunities that such an approach may offer. Valuation must not be confused with price. Economic value seeks to identify all the final use and non-use, market and non-market values, and will often be unrelated to the price that soil commands as a commodity. This is because price only reflects purchase for a single or limited number of uses; whereas economic value tries to identify a combined value for all uses. Definitions of price, cost and value used in this manuscript are; price is the amount of money you pay for something; cost is the price of something that you would be expected to pay. Value is more complex as discussed later on but the sense in which

it is used here is, ‘that quality of an object that permits measurability and therefore comparability’ (Robertson, 2012).

The contribution of this paper is to consider the contexts within which soils are valued and how soils can be valued in the context of the ecosystem services approach. We begin by looking at what value is, why valuing ecosystem services can be useful, the work that has been done on valuing soil ecosystem services to date and the goals of valuation. We then look at valuation in a wider policy context examining developments at the macro-economic national accounting level as well as micro policy mechanisms such as Payments for Ecosystem Services (PES).

## **VALUE, CONCEPTS, DEFINITIONS AND OBJECTIVES IN THE CONTEXT OF SOIL**

Although the mention of value usually brings to mind dollar signs, value is much bigger than simply monetary value. One definition of value is, ‘a framework for identifying positive or negative qualities in events, objects or situations’ (Edwards-Jones et al., 2000). Within the context of valuing nature’s goods and services a useful technical definition of value states that, ‘value is simply that quality of an object that permits measurability and therefore comparability’ (Robertson, 2012). Value is generally divided into two categories, *extrinsic*, also called instrumental, as it is when an object or action serves a recognizable purpose and is thus valued by virtue of function. Conversely, there is *intrinsic* value, which requires no means to an end, but is an end in itself. Intrinsic value can be divided into, aesthetic value, concerned with beauty, and moral value, which are judgements of virtue, rightness of action and justice (Zimmerman, 2010).

The values we hold as humans work within our personal *value system*, defined by Farber et al. (2002) as, ‘the intrapsychic constellations of norms and precepts contained in our world view that guides human judgement and action. They refer to the normative and moral frameworks people use to assign importance and necessity to their beliefs and actions. Our value system determines how we assign rights to things and activities, which implies practical objectives and actions.’ Value is therefore strongly coupled to value system, and ‘valuation’ is the process of expressing one of the qualities of an action or object on a scale. Moreover, valuation is directly linked and inseparable with our decisions about ecosystems and their management (Costanza et al. 1997b).

The value system we adopt, encompassed in our world view, and shaped by society, culture and religion will very much determine our approach to valuing nature and its constituents. Holmes et al. (2011) argue that our value system is important because it motivates us to act. They emphasize the importance of positive messages and avoiding appealing to fear, greed or ego. Turner, (1999) attempts to link our individual value system to our attitude to sustainability. By drawing a diagram with value across the horizontal axis, and the moral standing of biota on the vertical axis we can begin to map out how our world view influences our approach to valuation and sustainability (Fig 1). Anthropocentrism at one end of the vertical axis argues that only humans have moral standing, whereas biocentrism and ecocentrism contend that individual living things, or ecosystems, have moral standing. These dimensions of our world view largely determine the valuation system within which we operate. Economic theory is based largely on an anthropocentric extrinsic view, where as a more biophysical view of the world would argue for the intrinsic value of nature and that it, or parts of it, have moral standing in addition to humans. Hence our societal, cultural and or religious world view will very much influence the way we value nature and the acceptability of general approaches for valuing nature based on economics.

### **The Meaning of Economic Value**

Economic value (neo-classical) is based on a framework for valuation that people are most familiar with as impacting our everyday lives. Total economic value (TEV) is the sum of all relevant use- and non-use values generated now and in the future, i.e. the sum of the producer and consumer surplus under the demand curve, excluding the cost of production (Costanza et al., 1997a). Within this framework TEV is broken down into two categories, i) use and ii) non-use values (Fig 2a).

As shown in Figure 2a use values are typically divided into three categories: direct use values, indirect use values and option values. Direct use values include direct marketable, and direct non-marketable. These are the consumptive and non-consumptive use values for goods and services that are consumed or used locally. Indirect values are associated with the services nature provides that are not directly consumed, often being associated with regulating services. Option value is the value people place on having the option to enjoy something in the future even if they do not currently use it; this can be particularly important in the case of land and soil, passed down through the generations.

Non-use values, also referred to as “passive use” values, are values that are not associated with actual use, or even the option to use a good or a service. For example, existence value is the non-use value that people place on simply knowing that something exists, even if they will never see or use it. Similarly bequest value is the value that people place on knowing that future generations will have the option to enjoy the valued entity in the future and is directly related with the concern of access to resources by future generations (Beaumont et al. 2007).

The valuation typology provided in Figure 2a is in keeping with those in Edwards-Jones et al. (2000) and Bateman et al (2002). Figure 2a neatly illustrates that value is composed of several elements not all of which will be exhibited by all goods and services. It also highlights the fact that market prices only capture a specific aspect of value (ie, direct use) that is frequently too narrow for the effective management and use of soil. For example, Table 1 identifies soil goods and services, recognizing that soils contribute to a range of final services along with other ecosystem components. Moreover, the table shows how value, use and non-use, map onto these goods and services (modified from DEFRA, 2007). The contribution of soils to final goods and services over and above food production shows why they should not always be simply lumped together with land value, but their distinct contribution recognised. For example, soils constitute the largest terrestrial store of carbon (Tipping, 2002) helping regulate climate; moisture, texture and soil structure control the partitioning of precipitation between infiltration and runoff at the land surface, and hence the regulation of surface water flows and flooding. Soil moisture buffers climate extremes such as heat waves (Seneviratne et al., 2006) and fulfils a range of other functions that we could not survive without including nutrient transformation and waste recycling etc. Those regulating services provided by soils have indirect and option use values for society as well as non-use values relating to the use future generations will have of the soil resource, and the responsibility of the current generation to pass on such resources to ensure future well-being.

The economic approach to non-market valuation is however, not without its criticisms and difficulties. For example, it has been noted by Vatn and Bromley (1994) and Gasparatos et al. (2008) that environmental complexity means that when eliciting an individual's willingness to pay (WTP) for non-market goods, preferences are based on imperfect knowledge of ecological processes and functions. There are also long standing disagreements within economics about the meaning of non-market value estimates generated using some of the most popular methods (eg, Contingent Valuation). Vatn (2004) provides a useful summary of the issues, plus more recently there has been a very heated exchange between



Carson (2012) and Hausman (2012). Carson is a strong advocate of non-market valuation whereas Hausman, who is a leading researcher within the wider field of economics, considers efforts at non-market valuation dubious if not plain worthless. Finally, there are whole swathes, of moral, ethical and philosophical criticisms that have been made against non-market valuation (e.g., Sagoff, 1988).

Given the criticisms that exist within the literature, the acceptance of valuation within policy circles means that caution should always be exercised when conducting, interpreting and employing non-market valuation research, in particular valuation based on contingent valuation or choice experiments. Indeed, given the widely discussed limitations, the real merit in conducting this type of exercise is less the “number” that emerges but more the process that is undertaken. This point is neatly expressed by Carson (2012) as follows:

*Much of the usefulness of doing a contingent valuation study has to do with pushing scientists and engineers to summarize what the project would do in terms that the public cares about. Further, the process of developing a contingent valuation survey often encourages earlier involvement by policymakers in thinking more critically about a project's benefits and costs and in considering options with lower costs or greater benefits to the public. (Carson, 2012, page 31).*

## **Economic Valuation Methodologies**

There exists a wide range of economic valuation methodologies (Bateman et al., 2002), with the use of specific approaches dependent on the type of value that is being sought, as well as the costs and time required to undertake the valuation exercise. Figure 2b shows the link between types of value (use and non-use) and valuation methodologies that are currently used in valuation research. The key distinction in the use of economic valuation methodologies is the decision to employ revealed or stated preference methods (Fig 2b). This choice will be informed by the need to include or exclude non-use values in the associated analysis. Revealed preference methods rely on observed behaviour and are commonly used when assessing use values. However, if the decision is to consider non-use values, which can frequently be very important, then stated preference methods must be adopted. Stated preference methods are based on the construction of a hypothetical market which is typically implemented by the use of sophisticated survey instruments and as stated before are the subject of much academic debate. Figure 2b also highlights an alternative approach to

valuation called Benefits Transfer that is popular especially for more applied and policy orientated analysis. This is essentially the use of existing valuation estimates in a new but related context. Benefits transfer can be conducted either in a very simple manner or with the use of advanced econometric methods. The attraction of benefits transfer is that there are a growing number of data bases that allow researchers to undertake this method very rapidly.

The estimates of economic value of goods or services yielded by the various methodologies are usually measured in terms of what resource users or society are willing to pay for the commodity or the service, minus what it costs to supply it; this is revealed by price in markets, but other techniques are required to assess WTP for services without markets.

### **Alternative Valuation Methodologies**

Other approaches to valuation have been proposed but not widely adopted, these include for instance EMERGY, an ‘embodied energy theory of value’ (Hannon et al., 1986), since energy is the fundamental driver of ecological systems and thereby the economy. However, authors like Georgescu-Roegen (1979) rejected a strict energy theory of value, arguing that matter is also important, since it is also subject to the entropy laws. Research in this area has led to theories of value where prices can be determined for biophysical inputs and outputs, leading to a new type of accounting of the economy: a mass/energy accounting or ‘ecological pricing’ (Georgescu-Roegen 1971, Daly 1973).

### **Why value the contribution of soils to the delivery of ecosystem services?**

Valuation in an economic context can be particularly helpful for comparing systems with a complex set of socio-ecological relationships; often the case with ecosystems. Edwards-Jones et al. (2000) argue that documenting ecosystem service values is useful because it:

- Highlights the importance of ecosystem functioning for mankind.
- Highlights the specific importance of unseen, unattractive or unspectacular ecosystems.
- At a local level it can aid in identifying ecosystem services and acting as a help to decision making.

- Can aid in understanding the impacts of change and feeding back to models to improve our understanding of ecosystem function
- Is a way of communicating value by translating to a common reference, e.g. dollars<sup>2</sup>.

All of these are important for the sustainable exploitation and management of soils and other natural resources; something supported by the European Commission Communication COM(2011) 517, “Roadmap to a Resource Efficient Europe”, which highlights the need to value human intervention regarding natural capital, in order to promote a more sustainable use of resources (EC, 2011). Among others, the document proposes actions on the mapping of ecosystem services and assessment of their economic value, together with the development and establishment of instruments and/or mechanisms related to the payment for ecosystem services. The need to secure soil functionality and limit some soil threats are stressed in the document.

### **The Objectives of Valuation**

Common to all valuation is the initial and fundamental question, what is the valuation for? There must be a clearly defined policy objective or management purpose for economic valuation. Thus, the objective could be ex-ante or ex-post policy or project evaluation; alternatively it could be the construction of alternative indicators of resource use that can better help understand the current state of resource quality. Defining the valuation objectives is, therefore, an essential first step.

Different paradigms are used to operationalize environmental policy; a widely used one is management by objectives that sets goals to try and achieve targets. For example, the European Union environmental policy is partly operationalized through the objectives set out in the Sixth Community Environment Action Programme (1600/2002/EC 2002) which addresses biodiversity decline (Edvardsson, 2004). A goal can represent a clear end point to be achieved and is therefore a useful starting point for valuation. However, it is clear that little research has been done on the properties that the management objectives should possess in order to be rational, or functional, and on how to resolve conflict between different goals (Edvardsson, 2004; Edvardsson and Hansson, 2005; Edvardsson, 2007).

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<sup>2</sup> It is worth noting that some ecological economists think that there is too much emphasis on stock and flow within the current application of the ecosystem service approach. For example, Norgaard (2010) argues that the ecosystem service approach has become too micro-orientated when in fact we need a general equilibrium approach.

If we analyze soil science approaches that are used to link to, or inform, policy we can identify some of the problems related to practical application. Regulatory systems are often used, but regulations tend to emphasize technical means rather than focus on environmental processes to define environmental goals for soil, air, and water quality (Bouma, 2005, p75). Objective setting for soil management is often done in the context of improving soil quality or soil health, which is aligned with sustainable soil management. We know that soil quality is important, but in the context of setting policy it is a highly subjective term. Like sustainable, it is problematic because it depends on how we define quality, or sustainable, and ultimately depends on use and intensity. Goals for improving soil quality and health often fall at the first hurdle because they are not specific. Soil science needs to carefully consider better ways to set goals and objectives that can be used in policy and management development, and for valuation.

Some may argue that this is not the job of a soil scientist, but as Bouma (2005) pointed out this is an important aspect of using information collected on the ‘true’ soil to inform those involved in dealing with the ‘real’ soil. It is often easier to articulate and describe the things we don’t want to happen, than try and describe what the ideal soil should be. The EU soil threats paradigm (Table 2) is a good example in this context. For example, carbon decline is not a desirable outcome, since it adds to greenhouse gases and also reduces structural integrity and water holding capacity. Other examples are soil compaction, which reduces oxygen levels, infiltration and enhances runoff; topsoil erosion from agricultural land, leading to loss of organic matter and nutrients; salinization of land, which prevents life from establishing and loss of biodiversity.

Given clearly measurable goals, the change in the measurable property can be monitored and valuation used to assess progress. This is perhaps why there is growing interest in concepts such as natural capital assessment for which measurable change can be determined (Howard et al., 2011). Concepts such as soil health, though laudable, are difficult to legislate for because wanting better soil depends on what better is, for what use, and on which time scale. The benchmark is often the ‘future or attainable’ state, which is hard to determine. Therefore, by identifying threats to soils, and declines in perceived soil value, the thematic strategy offers a helpful starting point in terms of setting goals for sustainable soil management. We must then identify the origin of the threats and their causes, and then design actions targeting the source of the problem in order to achieve our goals.

## **VALUATION OF SOILS TO DATE**

The valuation of soils to date has employed the full range of valuation methodologies to determine the values identified in Figures 2a and 2b. We briefly review examples of various methods to provide the reader with a feel for the magnitude of estimates that have been reported in the literature to date.

### **Direct-use: Market value of soil and soil commodity prices**

The direct use value of soil is what it realises when sold in markets. With regard to value it is perhaps a minimum value. The primary soil products include topsoil, subsoil, peat, and turf-grass. Of these, the turf grass industry, estimated to generate more than \$1 billion annually for the US economy (Christians, 2011) is by far the most visibly valuable. Peat by comparison is only \$13 million in the US (USGS, 2013), with an average price of \$23.0 per short ton in 2012 (USGS, 2013) and 80% sold for horticultural use. There are no readily available figures for topsoil or subsoil commodity prices. In the UK it was recently reported that B&Q, the UK's largest retailer of growth media, sells ~\$7.8 million of topsoil each year (Forster, 2012). Given this figure, annual sales of topsoil in the UK from all retailers are likely to exceed \$10 million. Sales figures for peat are not readily available although England uses ~1.6 million m<sup>3</sup> of peat for gardening each year (Defra, 2011), though it is hoped to phase this out by 2020. Given the US average price for peat of \$24.4 per short ton (\$26.84 per tonne) and assuming a bulk density of 0.2 tonnes/m<sup>3</sup> this would equate to ~\$8.5 million. Less well known, but vital to our technological revolution, is the extraction of rare earth minerals found extensively in laterite iron ore deposits and also in the tropical soils associated with these. China, contributes 90% of the global rare earth output with revenue of \$12.6 billion in 2013 (Els, 2014), but countries in the tropics, for instance Jamaica, are looking to their soils to see if they too contain rare earth deposits (Howe, 2013).

What is not included in the turf and retail topsoil numbers is the market value with regard to soil bought and sold for use in the construction and landscaping industries. There is currently no standard reporting for this economic activity. However, we can get some impression of use from Hooke (1994) who estimated how much earth (soil, sediment and rock) humans moved in 1988 based on US house construction (HC) (0.8 Gtons/yr); mining (3.8 Gtons/yr, of which 0.86 Gtons/yr was sand and gravel (SG)); and road building (RB) (3.0 Gtons/yr), giving a total of 7.6 Gtons/yr. If we consider unconsolidated material (the soil solum, C horizon, and sands and gravels) we might estimate that half the house building and half the road building involved moving this unconsolidated material. This means 0.4 (HC) +

0.86 (SG) + 1.5 (RB) = 2.76 Gtons/yr is activity related to moving unconsolidated material, or about one third of earth material moved. Hooke (1994) also estimated that agriculture moves 1.5 Gtons/yr through tillage but this is turned over rather than transported. Of the 2.76 Gtons sand and gravel is sold in markets and the price recorded; in 2013 this was 6.4US\$ billion for construction and 2.2 US\$ billion for industrial use (USGS, 2013). Of the remaining 1.9 Gtons, if only 1% was sold as top or subsoil, this would equate to US\$ 380 million based on a price of \$20 per ton (\$22.25 tonne<sup>-1</sup>, see Fig 3). The valuable nature of soil in this sense was highlighted following the Tsunami that hit Japan in 2011. Nakamura (2012) reported that, “*A serious shortage of soil and subsequent price increases are delaying efforts to rebuild the disaster-hit Tohoku region and prolonging the misery of survivors who are desperately trying to resume normal lives.*” It was reported that an estimated 40 million cubic meters (~0.05 Gtonnes) of soil was required for reconstruction and defences. According to Hooke (2000) an exponential increase in earth moving has occurred during our industrialised past, so our movement and use of soil will also have increased; however, the economic value is mostly hidden. Businesses have now developed based on soil movement or loss, for example, British Sugar in the UK, obtains 300,000 tonnes of topsoil with their 7.5 million tonnes of sugar beet delivered annually (British Sugar, 2014). British Sugar, through its topsoil division, then turns this soil back into several commercial topsoil products. Furthermore, as a response to needs and a way of recycling estuarine dredged products ‘soil factories’ have begun to emerge. In the 1980’s a soil factory was established by the Scottish Development Agency and the Clyde Port Authority along the river Clyde, Scotland, which produced 2000 tonnes of topsoil per week; feasibility studies have also been conducted in the USA and Republic of Ireland (Sheehan et al., 2010).

### **Direct use: Effect on productivity and replacement cost**

When soil is valued it is frequently linked to non-marketable functions such as nutrient cycling, carbon storage, soil erosion (Adhikari and Nadella, 2011) and soil salinity (Walker et al., 2010). Indirect-use values can account for soil functions such as storing carbon, filtering water, recycling waste etc. A review of the literature indicates that, soil valuation *per se* is uncommon, where it occurs, the cost of soil erosion is the more commonly assessed aspect of soils (Pimentel et al., 1995; Adhikari and Nadella, 2011); Table 3 presents a synthesis of estimated costs regarding soil erosion globally and nationally, demonstrating that this represents a major economic loss, moreover, a major environmental loss. These estimates only account for the onsite loss of production from the soil; consideration of offsite

costs, such as silting of water ways and pollution would significantly increase the economic loss (e.g. Repetto et al, 1997; Pretty et al., 2000; Nanere et al., 2007). These numbers are not insignificant, so why would a private landowner allow this economic loss? The answer is complex. For example, land tenure in developing nations is often insecure so there is no incentive to deploy soil conservation measures (Yirga and Hassan, 2010). In developed countries, the costs of soil conservation often falls onto the farmer, who might or might not be able to cope with it, depending on financial aids or the state of the farm finances, whereas the beneficiaries of soil conservation extend to the whole of society.

Estimates of soil erosion have been used to modify estimates of Total Factor productivity (Repetto et al, 1997). The methods used to conduct this type of analysis are based on adjustments to either productivity decline or the replacement cost of maintaining the level of soil quality. There have also been efforts to assess the off-site costs of soil erosion. For example, Nanere et al. (2007) estimated by how much Australian agricultural productivity needs to be changed when off-site costs of soil erosion are taken into account. There have also been a few studies estimating the national economic cost of soil erosion and sedimentation in New Zealand. See for example Barry et al. (2011) who looked at the cost of both on and off site effects.

Other studies at the micro level examine how specific forms of agricultural practice have induced the emergence of negative externalities such as salinity which in turn affects productivity (Ali and Byerlee, 2002). This research (and more recent work) shows that technology adoption can increase productivity but at the same time have an impact on the resource base (ie soil quality) that has a negative impact on productivity.

One study, Dominati and Mackay (2013), looked at soil ecosystem services *per se*. The study implemented an ecosystem services approach at the farm scale for New Zealand hill country sheep and beef farms looking at the quantification of land degradation by erosion and the value of soil conservation practices. The study focused on how an erosion event or the implementation of soil conservation policies affected soil change and therefore the provision of ecosystem services long term. Economic valuation methods were used in a cost benefit analysis including the economic value of the whole range of soil services.

In more developed nations it has been more cost effective to replace lost nutrients with cheap fertilizer produced from cheap energy supplies. Moreover, the subsequent damage to rivers and streams has generally not been borne by the land manager. This over exploitation of the soil resource, largely to produce food, is now attracting greater attention (Mueller et al., 2012) and is being checked as soils reach the lower limits of fertility, with the

spectrum of nutrients and micro-nutrients in need of replacement (Jones et al., 2013). Concurrently, the cost of energy and fertilizer production is increasing, and the environmental damage, such as dead zones in rivers such as the Mississippi and Yangtze is becoming more socially unacceptable. Moreover, the importance of soils in terms of their multi-functional use, e.g. carbon storage, waste recycling, water filtration, climate buffering, rather than just their food production function is being recognized by policy makers (Blum et al., 2004). The soil thematic strategy is the response of policy makers in the European Union who commissioned a valuation exercise to scope the scale of threats to soil function. The findings of the Impact Assessment (SEC(2006) 620, 2006) are presented in Table 2 and clearly show that the economic costs of allowing our soils to be degraded are sizeable. Moreover, soils also present a major economic natural hazard in the form of shrink-swell this can be regarded as a degradation process leading to negative outcomes. According to Jones and Jefferson (2012), the Association of British Insurers has estimated that the average cost of shrink–swell related subsidence to the insurance industry stands at over £400 million a year (Driscoll & Crilly, 2000). In the US the estimated damage to buildings and infrastructure exceeds \$15 billion annually.

### **Indirect values: Stated preference research**

There are a much smaller number of stated preference studies that estimate the value of agricultural soil conservation programs (eg, Colombo et al., 2005 and 2006; Almansa et al., 2012; Rosario-Diaz et al., 2013). It is these methods that cause so much tension and debate in relation to non-market valuation. This in part might explain why there have been so few applications. However, it is also the case that the majority of on-site externalities that arise from land use management can be reasonably well captured by the methods already discussed. But, when research turns to off-site externalities or on-site effects that relate to biodiversity and conservation it is the case that there are more obvious costs to society not captured in output prices or land values and it is, therefore, more meaningful to employ stated preference research methods.

In general all these studies set out to examine the preferences of farmers to adopt specific farm level soil management practices and the costs associated with adoption and implementation, with a view to reducing off-site externalities from soil erosion. In particular, Almansa et al. (2012) give an overview of valuation techniques applied to soil erosion, noting that replacement valuation methods are most widely used, but that newer stated preference techniques offer some advantages when dealing with specific issues. The authors indicate



their scepticism when initially applying contingent valuation methods, but conclude that stated preference methods can provide useful information for decision makers, providing a more accurate assessment of the socio-economic returns. In many ways these observations are in keeping with those made by Carson (2012) about the process of undertaking a contingent valuation is as informative as the value estimates generated.

### **Global web survey of soil price**

As part of our review of direct use value, we conducted what we believe to be a first, limited, web survey of topsoil prices from around the globe (Fig 3). Prices were collected from English and Spanish speaking countries, and from partners in Crete and Iceland, using web search engines to find topsoil prices. Searches were conducted in 2013 using the key words, soil, topsoil, price and specific countries. The search was limited to topsoil being sold in large quantities, e.g. 1 tonne plus for landscaping, as price is highly variable for small quantities sold in shops. Values were calculated for 1 tonne of topsoil in \$US after removing taxes from the prices; these were then plotted as soil value adjusted according to purchasing power parity (For more information see: Common and Stagl, 2005) which is a technique that can be used to determine a 'relative value' for monetary values that are in different currencies. Figure. 3 shows that across the western world soil prices show some variability, with the median price being ~\$22 per tonne in the USA and Canada, and \$47 per tonne in the UK, perhaps a reflection of energy prices.

### **Replacement costs**

In conjunction with this it is insightful to examine some back of the envelope calculations with regard to soil replacement costs. This is done by determining the components of soil that contribute most to its market price based on replacement costs for major constituents. Table 4 considers market retail prices of stocks from the UK (£) that could be used to create basic topsoil, not accounting for the transport, mixing, or time required to create genuine soil. Examining the costs of the constituents discloses some revealing numbers, for instance, simply replacing the mineral component (Sand, silt and clay) is expensive because of the large amounts required, so when we see mineral soil blowing away, or being washed off a field into a water course, there's potentially a sizeable equivalent replacement cost. The price used for carbon (£150) reflects the approximate current abatement cost for a tonne of carbon based on the numbers in the Stern review (Stern, 2006). Keeping carbon in soils constitutes a major component of the topsoil value for combating

climate change; a 1% loss of soil carbon would be equivalent to the UK's annual fossil fuel emissions (Defra, 2009). Finally, we considered adding 2 tonnes of worms as a surrogate for soil biota. Worms are not grown in mass production, so the retail cost for composting worms is relatively high. However, it makes the point that small amounts of soil biota add high value to the soil. Conserving and encouraging soil biota represents a major investment in maintaining and building soil ecological infrastructure and the soils natural capital (Robinson et al., 2013b; Dominati et al., 2014). Farmers are often concerned with nutrients, as fertiliser inputs are the major input they buy, but although the cost per tonne is relatively high, the amount per ha is relatively low and thus not a major contributor to the soils value above what is already there. Although this is a simplistic analysis of the price of topsoil it does reveal some insight into the relative replacement costs of the stocks constituting soil natural capital (Robinson et al., 2009) and shows the very high economic price of such *critical natural capital* (Ekins 2003). This is before the externalities associated with soil loss are accounted for; these increase the costs associated with improved soil management. The analysis in Table 4 illustrates that replacing soil is expensive and should encourage those managing the land to conserve and invest in building their soils.

## **SOIL AND ITS INCLUSION IN THE DESIGN, IMPLEMENTATION AND EVALUATION OF POLICY**

### **Decision support tools for assessing ecosystems services on which valuation can be based**

Valuation requires information about what it is that we seek to value. This can be based on data alone, but increasingly output from models is being used, with an array of decision support tools (DSTs), both spatial and non-spatial being developed to assess ecosystem services. The output from these models can then serve as the basis of an economic valuation and decision making.

Life Cycle Assessment (LCA) is being increasingly used as a DST in environmental impact assessment, adapted from commodity production, for use in policy intervention scenarios. LCA consists of a tool to quantitatively evaluate, environmental impacts resulting from a product or service life cycle, from material extraction to waste management. By means of environmental indicators, associated with specific impact categories (e.g., 'climate change', 'land use', 'acidification'), resource flows are associated with different impacts

(midpoints) and damages (so-called ‘endpoints’) on the environment. The EC COM((2011)571) emphasizes the need to look at resources over their whole life cycle, taking into account not only the impacts generated from cradle-to-grave, but also their value chain, in order to reach a more efficient use and sustainable consumption and production patterns, avoiding burden shifting along the life cycle. Several methodologies have been applied, from qualitative to quantitative methods, based on monetization, expert panels, proxy approaches, technology abatement or distance-to-target. Regarding the monetisation methods, damages resulting from a specific production system may be evaluated in monetary terms, with values associated with the WTP for the potential reduction or avoidance of these damages. No consensus exists on the use of specific methodologies nor the values, or weights, given to specific impacts, and little differentiation is done between average and marginal effects. Despite the important role of ecosystem services and goods in human well-being and activities, some challenges exist for their accounting in LCA (Bakshi and Small, 2011). First, some services, such as regulating, are difficult to quantify in physical terms. Second, aggregation (by means, for example, monetary valuation), which is used to ease interpretation of data, may hide important information on individual resources. Finally, not all methods that account for ecosystem services are well suited to a life cycle evaluation. As to what concerns soil quality, current modelling still neglects the complexity and interaction of soil characteristics and value of functions, such as cycling of nutrients, mainly due to the difficulty in relating the impacts on soil quality to specific flows (Garrigues et al., 2012); a necessary step in LCA. Moreover, no direct valuation of ecosystem services supplied by soil is yet made operational in current life cycle assessments.

An alternative suite of DSTs seeks to make a fuller assessment of ecosystem services through greater biophysical assessment and modelling, using either mechanistic or statistical models. There are no spatially explicit DSTs designed for soils or soil management that we are aware of. However, within the wider context of managing land for multiple uses and particularly in the context of ecosystem services, there are a number of tools developing (Vigerstol and Aukema, 2011; Bagstad et al. 2013a). The majority of these utilise soils data and predict soil change to some extent, e.g. erosion. The global unified metamodel of the biosphere (GUMBO) was perhaps one of the first of these assessment tools containing predictions for soil formation, and nutrient cycling, alongside social and economic information (Boumans et al., 2002). InVEST (Nelson et al., 2009) is perhaps the best known, or more widely applied of the ecosystem service assessment tools, and uses a mechanistic modelling approach to predict ecosystem service dynamics, whilst tools such as the

ARtificial Intelligence for Ecosystem Services (ARIES) tool takes a more statistical approach, set within a conceptual framework which encompasses both the biophysical supply and the spatial delivery of service to the beneficiaries (Bagstad et al., 2011 and 2013b). At the regional to national level the Land Utilization and Capability Indicator model (LUCI) is another emerging tool optimised to quickly use nationally available datasets to determine ecosystem services (Jackson et al., 2013). LUCI models a number of soil-mediated processes including infiltration, flood control, carbon storage and sequestration and soil fertility. These tools link to valuation in different ways. InVEST for example includes a full economic valuation tool allowing the user to obtain monetary values, whilst LUCI uses biophysical levels as part of a trade-off evaluation component. The user can specify biophysical thresholds resulting in five categories, and high existing value, existing value, marginal value, opportunity to improve a service, and high opportunity to improve a service.

In most spatial DSTs to date, soils information has been incorporated purely as a GIS input layer, on which to base other derivations (e.g., soil C, agricultural productivity), and rarely incorporated for their own sake. With an increasing focus on the essential role of soils in the delivery of final services, such as carbon sequestration, or crop production, there is a need to address these aspects within DSTs. Moreover, there is the need to recognize the soil as a valuable ecosystem in itself and protect the variety of diversity within it.

If this is to be achieved, there are a number of issues which must be overcome. One relates to the spatial resolution of existing soil survey data and land-cover or land-use data, which while comprehensively surveyed at a national scale in many countries, does not provide resolution down to the farm scale. There are often other data available from a wider range of sources e.g. extensive farm surveys, soil quality consulting and scientific survey data which could be released and collated centrally (after a suitable period), even exploiting crowd-sourcing of data (Shelley et al., 2013). Soil temporal change is also rarely monitored, but is important for assessing the impact of policy and management as for example highlighted by the findings of the Countryside Survey (Reynolds et al., 2013). Another issue is that response functions or models linking the contribution of different soil types to many ecosystem services and other functions are currently lacking, e.g. infiltration, or above- and below-ground biodiversity. Nor do we have a good understanding of the impact of soil depth on ecosystem service delivery, but we know from studies that deep soils (>2m) make important contributions to carbon cycling (Jobbagy and Jackson, 2000; Richter and Markewitz, 1995). Within the context of ecosystem services it is vital that models consider

soils to depths beyond the solum, and that appropriate soil data is obtained and linked to land-cover and land-use data to support this effort.

### **Macro-economic performance, indicators and soil**

As we have already explained, societal economic activity impacts the environment, however, it is widely recognized that current measures of economic activity such as Gross Domestic Product (GDP) and Net National Product (NNP), generated by the system of national accounts (SNA) are inadequate at accurately measuring the contribution of, and impact on the environment. Basically, the costs of environmental degradation, natural resource depletion and non-market values are either not included because the SNA only considers goods and services transacted in markets or accounted for as a benefit, as loss often incurs additional economic activity (Harris and Fraser, 2002). Thus, the current macro-economic measures of performance that inform policy and debate can provide misleading information with respect to sustainable use of resources. This point has been articulated by Robert Repetto (1988) as “*steering by the wrong compass*”.

Despite shortcomings the SNA and associated measures of economic activity such as GDP remain central to policy making. This can in part be traced to the extent to which the SNA are embedded in economic decision making. Introduced by the United Nations Statistical Division (UNSD) it provides an internationally agreed national accounting framework (ie principles, concepts and classifications) providing a consistent description of market based economic activity within, and between, all economies.

The limitations of the SNA in relation to the environment and depletion of natural resources have led to the development of the 2003 System of Environmental and Economic Accounts (SEEA, 2012). The approach articulated within the SEEA is not to explicitly include monetary estimates of environmental damage (such as soil erosion) and resource use in accounts. Instead the SEEA advocates disaggregated, issue specific “satellite” accounts that sit beside the existing SNA that captures resource use and environmental degradation.

Within the SEEA report soil is dealt with in two main areas, as a *physical asset*, and in the *physical supply and use tables* (PSUT) (SEEA, 2012). As a physical asset, assessment is based on area and volume. In terms of area it states, ‘*the focus is on the area of different soil types at the beginning and end of an accounting period and on changes in the availability of different soil types used for agriculture and forestry.*’ (SEEA, 2012, page 174). In terms of volume, ‘*since the intent of the soil resources account is to recorded changes in the volume of soil resources that can operate as a biological system, the loss of the top layers of soil*

*resource due to this extraction should be recorded as permanent reductions in soil resources unless the purpose is to create new biological soil systems in other locations.’* (SEEA, 2012, page 175); and as we’ve seen in the previous sections the amount of soil moved annually is substantial. The implications of this for soil science are that soils must be viewed in a much more dynamic way, and assessed more often, to capture this. Furthermore, if the emphasis is on soil as a biological system, then the current soil survey lower boundary depth of 1-2m, depending on system, may be inadequate to capture this. As previously stated, many soils, especially where forests are located, have biological activity going deeper than this (Richter and Markewitz, 1995), which will be important for carbon accounting etc (Jobbagy and Jackson, 2000). The report makes it clear that,

*‘the accounting framework presented in the Central Framework does not fully describe the overall state or condition of soil resources, changes in the health of soil resources, or their capacity to continue to provide the benefits that soil resources generate.’* (SEEA, 2012, page 176).

Nor is this captured in terms of value where it states, “in the Central Framework the value of soil resources is tied directly to the value of land. In this context connections may be made between changes in the combined value of land and soil and changes in the associated income earned from use of the soil resources.” This means the accounts focus on changes in quantity but not quality or functionality, which underpins the delivery of ecosystem services. Hence, quantity is a useful start to capture the value of soil as an extracted good but fails to capture the value of soil in support of the delivery of ecosystem services.

Ecosystem services literature has changed the focus of research from just flows to include stocks of environmental resources, and in turn has produced new thinking about adjustments to economic measures of economic performance, as well as the type of environmental data we need to collect. For example, Walker et al. (2010) undertook a case study in South East Australia in relation to agricultural land use and soil salinity. They focused on stock resilience (defined in this case as water table depth) and showed how it had changed (fallen) between 1991 and 2001. However, this practical application is illustrative and it highlights the demands for scientific data as well as the associated uncertainties. But despite the obvious limitations of this approach, which is a long way removed from green GDP it does offer an approach to address the question of land use and sustainability.

There is also a gradual change in thinking about sustainability and how we assess it. For example, in the UK there is now the Natural Capital Committee (<http://www.defra.gov.uk/naturalcapitalcommittee/>). This group, which reports directly to

government in the form of the Economic Affairs Committee, provides government with better information about Natural Capital and as a result helps set priorities for policy actions. This committee has started to examine what is referred to as a Natural Asset Check (NAC). A NAC is in many ways an extension of the green GDP research agenda and the development of satellite accounts, but with a stronger emphasis on how the information can be used to inform policy. The key issue with the NAC is that it will monitor key environmental indicators over time and it will be the changes in these indicators that will help inform policy choice. In terms of how best to implement the NAC the work undertaken by the European Environment Agency (EEA) and its development of Eco-system accounts has been highlighted. In many ways the various activities and research agendas are linked, albeit not always explicitly. But if we wish to pursue a natural asset check then this requires not only more effort to augment and extend existing national accounts but it will require the comprehensive collection and collation of far more bio-physical data to allow for the construction of more comprehensive biophysical ecosystem accounts.

### **Valuation for payments for ecosystem services (PES)**

Traditionally farms have been managed for the single function of production. Increasingly growers are being asked to manage land for a number of different functions and services. Agricultural policies are changing, reflecting the need to make payments to land owners for the provision of services that are important for the common good. Payments for ecosystem services offer incentives to farmers or landowners in exchange for managing their land to provide some sort of ecological service. The concept of PES can perhaps be traced to the Dust Bowl era and the initiation of the United States' 'Conservation Reserve Program'. The US Federal government 'rents' ~140,000 km<sup>2</sup> of land annually to reduce soil erosion, improve water quality, enhance water supply through groundwater recharge, increase wildlife habitat, and reduce damage caused by floods and other natural disasters. This is achieved by payment of approximately ~\$1.8 billion a year to farmers and landowners to plant long-term ground cover. More recently programs such as REDD (Reduced Emissions from Deforestation and Degradation; <http://www.un-redd.org/>) are being promoted as ways to raise the viability of sustainable forest management (SFM) through the use of PES. The promotion of conservation and SFM in the tropics faces a range of market, policy and governance failures that encourage alternative land uses, often resulting in high social and environmental externalities (Richards and Jenkins, 2007).

In terms of carbon in soil the focus of research efforts relates to climate change. In particular, economic analysis has examined the role of agricultural land use and the associated implications for soil management as a means to offset, by sequestration, other forms of carbon emissions (eg, Gonzalez-Ramirez et al., 2012; Antle et al., 2001; Post et al., 2004; Lal, 2011). There is also a great deal of interest in soil carbon management in relation to developing countries via REDD which is at the forefront of implementation of (PES) in developing countries.

Farley and Costanza (2010) recognize two distinct approaches to PES in the literature. I) defined by Wunder (2005), where an ideal PES scheme should integrate ecosystem services into markets, and should be like any other market transaction; and II) defining, “*PES as a transfer of resources between social actors, which aims to create incentives to align individual and/or collective land use decisions with the social interest in the management of natural resources*” (Muradian et al., 2010, page 1205). According to Farley and Costanza (2010) the second approach is more closely aligned with ecological economics. One of the debates concerning PES is whether payments should be conditional on doing something or reciprocal, where payments are seen as a fair share of the costs of undertaking a desired activity, such that the recipients feel an intrinsic obligation to reciprocate (Vatn, 2010).

With regard to soils the new European Union common agricultural policy (CAP) contains mechanisms that provide PES. Traditionally focused more on production (Axis 1), reforms were phased in between 2004 and 2012 that increasingly transferred more payment to land stewardship rather than specific crop production (Axis II). In June 2003, EU farm ministers adopted a fundamental reform of the CAP which "decoupled" subsidies from particular crops. It introduced a new 'single farm payment' which is subject to 'cross-compliance' conditions relating to environment, food safety and animal welfare standards. Soil is now explicitly captured under good agricultural and environmental conditions (GAEC) and the water framework directive (WFD). The GAEC are the cross-compliance – you do and then we pay.

## **SUMMARY**

‘Value is simply that quality of an object that permits measurability and therefore comparability’ (Robertson, 2012), and should be seen as helpful in this context. But, understanding what constitutes economic value (Fig 2) is necessary if efficient and effective



resource management is to occur. Furthermore, understanding value yields key insights into the methods required to undertake valuation activities. Valuation (and valuation activities) offers an important mechanism to highlight the specific importance of often unseen, contributions of soil to benefit humanity, and that of the earth system. Valuation must not be confused with price, which is a lower bound to economic value.

Our review highlights that soils make critical and essential contributions to the economy, e.g. through waste processing, climate and water regulation and production of soil products such as turf grass, and that soil loss represents a major environmental and economic loss. A survey of soil commodity prices on the web indicates that the median direct market value of topsoil in terms of price per tonne is ~\$22 in the USA and Canada, and ~\$47 in the UK. Most direct value assessment in the literature is based on replacement costs and relates to erosion, whilst relatively little indirect valuation using stated preference methods has been undertaken with regard to soil. It is difficult to find studies dealing with soil *per se* as it is usually included in assessments of land or production, making it difficult to assess how the soil resource itself is changing.

Soils are increasingly recognised as a valuable economic resource in their own right, for example in the UN SEEA. However, SEEA currently deals more with soil quantity than quality or functionality, perhaps as it is easier to assess. In the SEEA it is the ability of soil to act as a biological system that is considered, which may challenge how soil survey traditionally defines soil depth and spatial extent. Moreover, the accounts require ‘change’ in volume and spatial extent to be reported on annual time scales, something not captured in traditional soil surveys.

Yet, and this is a fundamental limitation, soil is valued as a component of land, which is insufficient for capturing changes in the value of soil associated with alteration of soil quality or functionality as is clearly stated. It is important to capture changes to the soil ecosystem and its functionality, and methods should be developed to capture soil value under various uses, for **both quantity and functionality**. This could be achieved by accounting for the amount of soil, above and below key biophysical thresholds, e.g. carbon levels, or salinity levels, etc. In these situations, economic assessments would require more frequent soil functional monitoring on which to base valuation. In order to work well, economists and soil scientists must work together to develop indicators that can be used to assess the state of ‘soil function,’ if a soil ‘quality’ aspect is to be incorporated into approaches such as the SEEA. Economists and soil scientists will benefit from this relationship by developing a more

informative soil quantity and functionality accounting framework, with a fuller recognition of soils from an economic point of view.

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Table 1. Soil goods and services and the types of value associated with them that make up the total economic value.

Goods or Services	Total Economic Value (TEV)					Non-use value Bequest
	Use value	Indirect	Option value	Existence/ Altruism		
	Direct and marketable	Direct and non-marketable				
<b>Provisioning services</b>	<b>Topsoil</b>	X				
	<b>Subsoil</b>	X				
	<b>Peat</b>	X				
	<b>Sand/Clay minerals</b>	X				
	<b>Soil for rare earth extraction</b>	X				
	<b>Soil organisms, earth worms</b>	X				
	<b>Biomedical resources, antibiotics &amp; new organisms used in medicine</b>	X				
	<b>Provision of physical support</b>	X				
<b>Regulating services</b>	<b>Provision of food wood and fibre</b>		X			
	<b>Waste processing</b>					
	• Detoxification			X	X	X
	• Nutrient recycling			X	X	X
	<b>Nutrient/contaminant Filtering</b>			X	X	X
	• Water filtration					
	<b>Hydrological regulation</b>					
	• River flows mitigation/water levels			X	X	X
	• Flood peak regulation			X	X	X
	<b>Climate regulation</b>					
	• Carbon storage			X	X	X
	• Soil moisture buffering of heat and cold waves			X	X	X
	• Greenhouse gases mitigation					
	<b>Hazard regulation</b>					
	• Structural support shrink swell			X	X	X
	• Dust emissions			X	X	X
	• Liquefaction			X	X	X
	• Landsliding and slumping			X	X	X
<b>Pests and Disease regulation</b>						
• Human and animal pathogens		X		X	X	
• Disease transmission and vector control		X		X	X	
<b>Cultural services</b>	<b>Burial ground</b>	X				X
	<b>Scenery</b>		X		X	X
	<b>Recreation</b>		X		X	X
	<b>Preservation of artefacts</b>			X	X	
	Total direct and marketable	Total direct and non-marketable	Total indirect	Total option	Total non-use	

Table 2. EU soil threats, based on the Impact Assessment (SEC(2006) 620, 2006). Conversions to 2013 USD use an exchange rate for the given year (1.3, 2006) and inflation using a CPI index calculator (Areppim, 2014).

Soil Threat	Estimated annual cost
1) Erosion	€0.7 – 14.0 billion USD 1.05-21.03 billion, 2013
2) Organic matter	€3.4 – 5.6 billion USD 5.11-8.41 billion, 2013
3) Compaction	no estimate possible,
4) Salinisation	€0.158 – 0.321 billion USD 0.237-0.482 billion, 2013 (1.3)
5) Landslides	up to €1.2 billion per event USD 1.80 billion, 2013
6) Contamination	€2.4 – 17.3 billion USD 3.61-25.99 billion, 2013
7) Sealing	no estimate possible
8) Biodiversity decline	no estimate possible

Table 3. Estimated annual cost of soil degradation at different administrative scales, for detailed references see (1, Adhikari and Nadella, 2011; 2, Kuhlman et al., 2010; 3, Cohen et al., 2006; 4, EA, 2002; 5, Jones et al., 2008). Conversions to 2013 USD use an exchange rate for the given year and inflation using a CPI index calculator (Areppim, 2014)

Country	Source	Annual Cost
World	Dregne and Chou, 1	42 billion (1990 US\$) - ~75 billion, 2013
EU	Crosson modified by 2	370 million (2004 €) - 575 million, 2013 (1.26)
EU	Gorlach et al., 2	532 million (2004 €) - 827 million, 2013 (1.26)
EU	van den Born et al, modified in 2	1700 million (2004 €) - 2641 million, 2013 (1.26)
EU	Kuhlman et al., 2	500 million (2004 €) - 777 million, 2013 (1.26)
Rwanda	Berry <i>et al.</i> , 1	23 million (2003 US\$) - 29 million, 2013
Ethiopia	Berry <i>et al.</i> , 1	139 million (2003 US\$) - 176 million, 2013
Ethiopia	Bojo and Cassels, 1	130 million (1994 US\$) - 204 million, 2013
Ethiopia	Sutcliffe, 1	155 million (1994 US\$) - 244 million, 2013
Ethiopia	FAO, 1	14.8 million (1994 US\$) - 23 million, 2013
Zimbabwe	Grohs, 1	0.6 million (1994 US\$) - 0.9 million, 2013
Zimbabwe	Norse and Saigal, 1	99.5 million (1994 US\$) - 156 million, 2013
Zimbabwe	Stocking, 1	117 million (1994 US\$) - 184 million, 2013
Lesotho	Bojo, 1	0.3 million (1994 US\$) - 0.5 million, 2013
Mali	Bishop and Allen, 1	2.9–11.6 million (1994 US\$) - 4.5-18 million, 2013
Malawi	World Bank, 1	6.6–19 million (1994 US\$) - 10-30 million, 2013
Ghana	Convery and Tutu, 1	166.4 million (1994 US\$) - 262 million, 2013
Kenya	Cohen et al., 3	390 million (2006 US\$) - 451 million, 2013
England and Wales	EA, 4	205 million (2002 £) - 398 million, 2013 (1.5)
New Zealand	Jones et al., 2008, 5	159 million (2008 NZ\$) - 112 million, 2013 (0.65)

Table 4. Back of the envelope calculations to determine the value of soil components based on replacement costs using materials bought in bulk in the UK unless otherwise stated. Soil bulk density assumed to be  $\sim 1.36\text{g/cm}^3$  (Loam: 40% sand 60% clay & silt), prices exclude taxes, conversion to USD uses exchange rate of 1.55 for 2013.

	Commodity price per tonne	T/ ha to 30cm	Cost, 30cm topsoil / ha
Sand Wanlip sand & gravel, Leicester, UK	£ 17.38	1560	£ 27,113 \$ 42,025
Silt/Clay mix Cardigan sand and gravel, Cardigan, UK	£ 7.33	2340	£ 17,152 \$ 26,586
Carbon Stern review	£ 150 .00	107.25	£ 16,088 \$ 24,936
Nutrients (NPK) Representative price Feb 2013 DairyCo market information	£ 350	2	£ 700 \$ 1,085
Water ( $25\text{m}^3\text{m}^{-3}$ ) Utility retail price metered $\text{m}^3$	£ 1.57	750	£ 1,178 \$ 1,826
Worms (USA) Red worm composting blog Lowest retail price (\$15/lb) Range (\$15-40)	£ 4300	2	£ 8,600 \$ 13,300
Reconstituted topsoil		Total	£ 70,830 \$ 109,787
Bulk recycled screened topsoil, Wanlip sand & gravel, Leicester, UK	£ 10	3900	£ 39,000 \$ 60,450
Bulk topsoil Median UK price Fig 3	£ 30.38	3900	£ 118,482 \$ 183,647
Retail topsoil premium grade $1\text{m}^3 / \sim 1$ tonne, Rolawn loam topsoil, Tesco.com	£ 100	3900	£ 390,000 \$ 604,500

Figure 1.

Figure 1, Dimensions for value frameworks based on value type on the horizontal axis and moral standing of humans and biota on the vertical axis. The dashed line represents the sustainability axis indicating where within the value dimensions different sustainability world views tend to be located. Economic valuation is for example anthropocentric and extrinsic, and often classified as very weak sustainability.

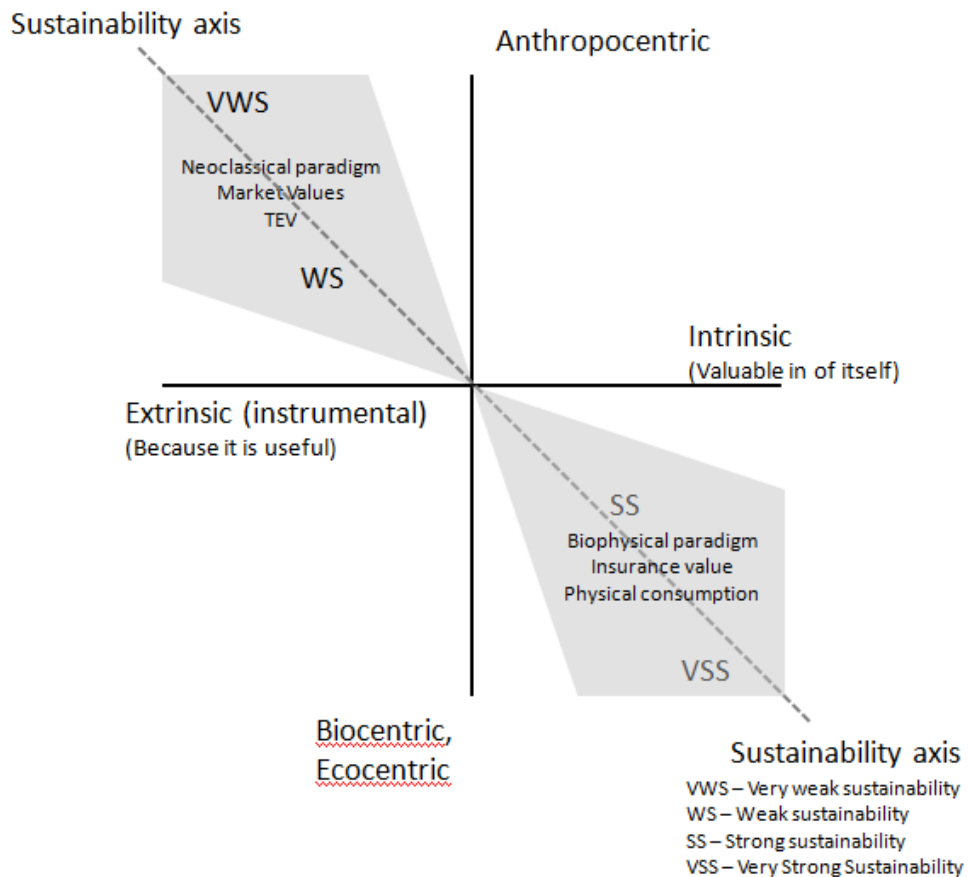


Fig. 2a The total economic value framework (TEV) showing different types of economic value. Note price comes under direct use. 2b. Economic methods used to estimate different types of value.

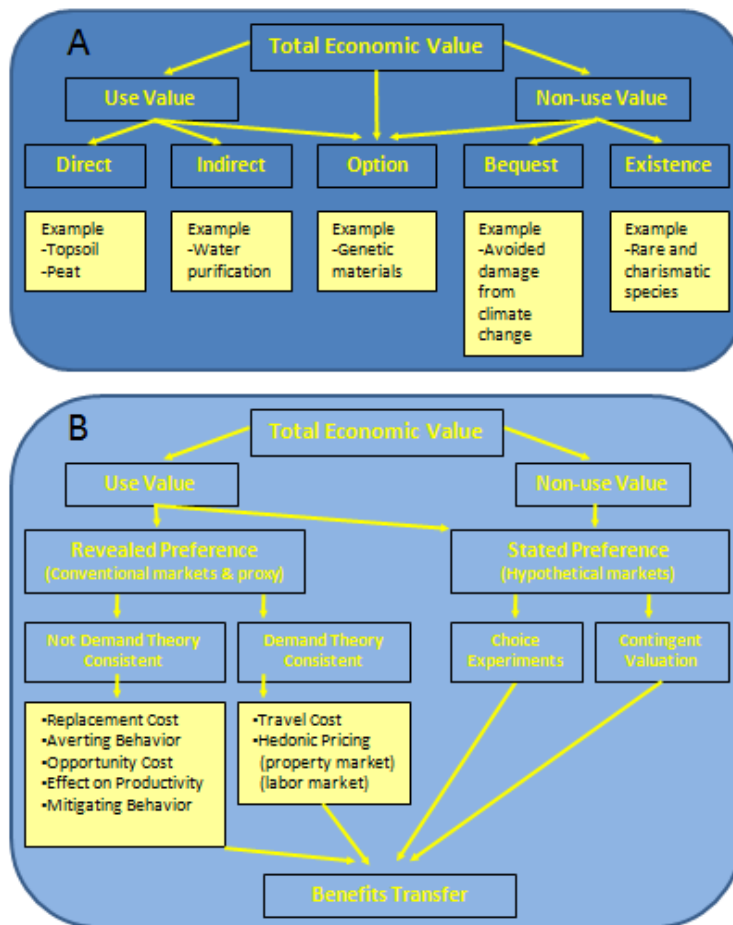


Fig 3. Geospatial assessment of soil prices around the globe based on a web survey of sites selling bulk topsoil. Median price in the USA and Canada \$22.25 per tonne, Median price in the UK is \$47.09 per tonne. Red dots indicate where there was no bulk soil data available and prices reflect small quantities sold in supermarkets where prices might be as much as \$1000 per tonne. The soil price data collected for the different countries is expressed in power purchasing parity (PPP). PPPs are the rates of currency conversion that equalize the purchasing power of different currencies by eliminating the differences in price levels between countries. All soil prices are adjusted to the US\$ which has the ratio of 1.0.

