Soil: Ecosystem Services, Economic Analysis and Sustainability Indicators

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Dissertation submitted in partial fulfilment of a *Philosophiae Doctor* degree in Environment and Natural Resources

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Abstract

Soils are an important natural capital that provides multiple services to humans including provisioning food, feed and fibre, water filtration and stabilizing climate. Soil security and soil sustainability are the basis for obtaining many of the UN 2030 Sustainable Development Goals and are needed to support the coming agricultural intensification in this century. Increased resource use due to population increase, technological advances and changing consumption patterns have led to increasing pressures on this fundamental type of natural capital. Exacerbating this problem, leading to over-exploitation and degradation, the economic value of soil services, despite their importance, has all but been omitted from land-use decision making. If the current degradation rate continues humanity might suffer the loss of most of the world's arable topsoil within the 21st century. To promote soil security and sustainable soil management, we need appropriate conceptual frameworks, methods, tools, and policies to monitor, promote and implement the sustainable management of soils.

The aims of this PhD thesis were twofold A) to develop a methodological framework and carry out an evaluation of the economic value of soil ecosystem services (ES), and B) to develop indicators which could guide decision-makers on sustainable soil management. The resulting work consists of four main parts: 1) A review research of methods and economic valuation of soil ecosystem services (paper I); 2) A framework for the economic valuation of soil ES (papers I and II); 3) Pilot studies using the soil ES framework on a watershed scale and on a plot scale (papers II and IV); and 4) A set of indicators for sustainable soil management developed in partnership with expert stakeholders (paper III).

The review study of soil ES economic value (paper I) is the first in the ES literature, illustrating the immense value of soil natural capital and associated ES. The case studies (paper II and IV) use the soil ES framework proposed in papers I and II to bring forth the value of soil on a watershed and plot scale illustrating how soil ES classification and economic valuation can be carried out. In addition, paper IV introduces a novel combination of methods and tools (Soil ES, Energy Return on Investment and CBA) are used to offer deeper, multi-dimensional and holistic insights into sustainable soil management and agricultural

operations. Besides the value of soil ES, information on soil conditions, soil use, and other important soil factors are also needed to assess whether soils are managed sustainably. Deciding which important soil factors to focus on can be complex as there are many stakeholders at different levels with various priorities. Paper III in the thesis puts forward a transdisciplinary approach to develop soil indicators for sustainable development and reports on the outcome of a Delphi survey among expert stakeholder groups, where scientists, policy makers and soil practitioners reached a consensus on a core and satellite sets of indicators deemed most important for sustainable soil management.

Overall the thesis illustrates the significant value of soil natural capital and associated soil ES and puts forward a framework on how to carry out valuations of soil ES. It shows how to establish soil sustainability indicators with active stakeholder participation. And finally, how to use the tools and methods to estimate the sustainability of an agricultural operation.

Útdráttur

Jarðvegur er náttúruauður sem gefur af sér margvísleg náttúrugæði sem mannkynið nýtur góðs af. Það má jafnvel halda því fram að samfélag manna byggi tilveru sína á honum. Jarðvegur hreinsar og miðlar vatni, bindur kolefni, er uppspretta byggingarefna og nær öll matvælaframleiðsla heimsins tengist nýtingu á jarðvegi. Jarðvegur er grundvallarþáttur í allri fæðuöflun, fóður- og trefjaframleiðslu og einnig er mikilvægur stöðugleika loftslagsins. Vegna þessa er sjálfbær nýting jarðvegs samofin fjölmörgum markmiðum Sameinuðu Þjóðanna um Sjálfbæra Þróun til ársins 2030.

Jarðvegur og þá einkum hin ræktanlegi hluti jarðvegs, gróðurmoldin, er takmörkuð auðlind og endurnýjast hægt. Þrátt fyrir það hefur maðurinn gengið á hana eins og hún væri óendanleg. Þessi aukna auðlindanotkun er tilkomin vegna fólksfjölgunar, tækniframfara og breytts neyslumynsturs sem veldur sívaxandi álagi á þessa mikilvægu auðlind. Til að auka á vandann, hefur hingað til hið hagræna virði og fjölþætta hlutverk jarðvegs ekki verið tekið nægilega til greina við ákvarðanatöku í landnýtingu og hefur það leitt af sér ákveðið skeytingarleysi og ákvarðanir sem ýtt hafa undir frekari hnignun jarðvegsauðlindarinnar. Ef ofnýting hennar heldur áfram þá gætum við tapað mestum hluta gróðurmoldarinnar á þessari öld. Til þess viðhalda jarðvegsauðlindinni og stuðla að sjálfbærri nýtingu á henni er þörf á viðeigandi aðferðafræðilegum nálgunum, tólum og tækjum og stefnumótun til þess að fylgjast með, ýta undir og innleiða sjálfbæra nýtingu á jarðvegsauðlindinni.

Markmið þessarar doktorsritgerðar var tvíþætt A) að þróa aðferðafræðiramma fyrir hagrænt mat á gæðum jarðvegs (jarðvegsþjónustu) og nota aðferðafræðirammann til að gera slíkt mat B) þróa vísa sem gætu nýst við ákvarðanatöku á sjálfbærri nýtingu jarðvegs. Ritgerðinni má skipta upp í fjóra meginhluta: 1) Yfirlitsgrein um aðferðir og hagrænt mat á gæðum jarðvegs (grein I); 2) Aðferðarfræðiramma fyrir gerð hagræns mat á gæðum jarðvegs (greinar I og II); 3) Rannsóknir þar sem aðferðafræðiramminn var notaður á vatnasviði og á tilraunareit (greinar II og IV); og vísasett fyrir sjálfbæra nýtingu jarðvegs sem var þróað í samstarfi við sérfróða hagsmunaðila (grein III).

Yfirlitsgreinin um hagrænt virði gæða jarðvegs (grein I) er ein fyrsta greinin um slíkt efni sem birtist í fagritum um gæði vistkerfa (vistkerfaþjónustu). Í henni er varpað ljósi á mikilvægi og hið mikla virði jarðvegsauðsins og þeirra gæða sem hann veitir. Í rannsóknunum sem birtast í greinum II og IV er settur fram rammi til að flokka og meta virði gæða jarðvegs og þeim ramma beitt bæði á vatnasvið og tilraunareiti. Í grein IV er að auki kynnt til sögunnar nýstárleg blanda af þremur aðferðum (hagrænt mat á gæðum jarðvegs, endurgreiðslutími orku, og kostnaðar- og ábatagreining) til þess að gefa ítarlegri og heildstæðari innsýn varðandi sjálfbæra jarðvegsnýtingu og sjálfbær landbúnaðarkerfi. Til viðbótar við hagrænt virði gæða jarðvegs er nauðsynlegt að hafa upplýsingar um ástand jarðvegs, jarðvegsnýtingu og aðra mikilvæga jarðvegsþætti til þess að hægt sé að meta heildstætt hvort jarðvegsnýting sé sjálfbær. Það að ákveða hvaða jarðvegsþættir eru mikilvægastir getur verið snúið þar sem margir ólíkir hagmunaaðilar koma að nýtingu jarðvegs og mörg mismunandi sjónarmið. Í grein III er sett fram þverfagleg nálgun við þróun á jarðvegsvísum fyrir sjálfbæra þróun og greint frá niðurstöðum úr Delphi könnun á meðal sérfræðinga þar sem vísindamenn, stefnumótendur og jarðvegsnotendur völdu sett af sjálfbærnivísum sem þeir töldu vera þá mikilvægustu er varðar sjálfbæra jarðvegsnýtingu.

Þessi doktorsritgerð dregur því fram mikilvægi jarðvegs og leiðir í ljós hvernig hægt er nota aðferðafræði hagfræðinnar til þess að meta það mikilvægi. Ritgerðin sýnir einnig hvernig hægt er að þróa sjálfbærnivísa fyrir jarðveg með virkri þátttöku hagsmunaaðila. Að endingu er sýnt fram á hvernig hægt er að nota þessar aðferðir til þess að meta sjálfbærni landbúnaðarkerfa á heildstæðan máta.

Dedication

To my three beautiful daughters

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Paper II: Chapter 4

Jónsson, J.Ö.G., Davíðsdóttir, B., Nikolaidis, N.P., 2017. Valuation of soil ecosystem services. Advances in Agronomy 142. Elsevier.

Paper III: Chapter 5

Jónsson, J.Ö.G., Davíðsdóttir, B., Jónsdóttir, E.M., Kristinsdóttir, S.M., Ragnarsdóttir, K.V., 2016. Soil indicators for sustainable development: A transdisciplinary approach for indicator development using expert stakeholders. Agriculture, Ecosystems & Environment 232, 179-189. Elsevier.

Paper IV: Chapter 6

Jónsson, J.Ö.G., Davíðsdóttir, B., Nikolaidis, N.P., Giannakis, G.V., 2019. Tools for Sustainable Soil Management: Soil Ecosystem Services, EROI and Economic Analysis. Ecological Economics 157, 109-119. Elsevier.

Paper V: Chapter 7

Robinson, D.A., Fraser, I., Dominati, E.J., Davidsdottir, B., Jonsson, J.O.G., Jones, L., Jones, S.B., Tuller, M., Lebron, I., Bristow, K.L., Souza, D.M., Banwart, S., Clothier, B.E., 2014. On the Value of Soil Resources in the Context of Natural Capital and Ecosystem Service Delivery. Soil Science Society of America Journal 78, 685-700. Soil Science Society of America, Inc.

Book chapter: Chapter 8

Jónsson, J.Ö.G., Davíðsdóttir, B., Ragnarsdottir, K.V., 2015. The value of soil ecosystem services. In: Ragnarsdottir, K.V., Banwart, S.A. (Eds.), Soil: The Life Supporting Skin of Earth. University of Sheffield. University of Iceland, Sheffield, United Kingdom
Reykjavík, Iceland, pp. 39-49.

Abbreviations

AMSL	Above Mean Sea Level
BRNS	Biogeochemical Reaction Network Simulator
CAST	Carbon Aggregation and Structure Turnover
CEC	Cation Exchange Capacity
CORINE	Coordinated Information on the European Environment
CSD	Commission on Sustainable Development
CV	Contingent Valuation
CZO	Critical Zone Observatory
ENR	Environmental and Natural Resources
EROI	Energy Returns of Investment
ES	Ecosystem Services
EU	European Union
FAO	Food and Agricultural Organization of the United Nations
GHG	Greenhouse gases
HRU	Hydrological Response Unit
ICZ	Integrated Critical Zone
MEA	Millennium Ecosystem Assessment
NGO	Non-Governmental Organization
OECD	Organization for Economic Cooperation and Development
POM	Particulate Organic matter
PPP	Purchasing Power Parity
SD	Standard Deviation
SES	Soil Ecosystem Services
SIFSD	Soil Indicators for Sustainable Development
SoilTrEC	Soil Transformation in European Catchments
SOM	Soil Organic Matter
SWAT	Soil and Water Assessment Tool
TEEB	The Economics of Ecosystems and Biodiversity
TSOM	Total Soil Organic Matter
UN	United Nations
US	United States
WTA	Willingness to Accept
WTP	Willingness to Pay
TUC	Technical University of Crete
Yr.	Year

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I wish to sincerely thank the SoilTrEC project for giving me the chance to study the multiple aspects of soil. This was a real opportunity to go into depth on topics that I'm interested in, i.e. the relationship between humans and the natural world. Soil is fundamental to our existence and its health and vitality is closely linked to our own.

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1 Introduction

1.1 The importance of soil

Life on Earth is supported by soil and human civilisation would hardly exist if it were not for these resources. Soil has been referred to as the living skin of the Earth (Ragnarsdottir and Banwart 2015), forming arguably the most complex and diverse ecosystem on the planet and home to a vast number of different species (Science, 2004). In one spoonful of soil, there are millions of bacteria and thousands of fungi (Fierer et al., 2007). It is sobering to note that almost all of our food, 95-99%, comes from the land and the soil (FAO, 2015d; Pimentel, 2006) and that just 30 cm of topsoil is sufficient to keep humans from starvation. This important ecosystem beneath our feet creates the basis for human well-being by providing multiple benefits, besides food, like filtering our drinking water, cycling nutrients and making these available to plants, providing source materials for different industries and sequestering carbon from the atmosphere (Jonsson and Davidsdottir, 2016). Soils' economic importance cannot be overstated. Just the gross production value of world agriculture, a soil-based activity, was 7.1 trillion USD in 2016 (FAO, 2016). Their importance for species and other ecosystems is fundamental, as the land and soil form the medium that underpins terrestrial ecosystems which account for 80% of macroscopic species (Grosberg et al., 2012) and countless micro-species.

Due to the many roles that soils have they are used extensively by humans and our actions heavily influence their condition. Despite their importance, their value has been largely ignored in land use decision making (Dominati, 2011), leading to overuse and exploitation. History offers many examples of where humans have exhausted the soil to their own peril; from the demise of the Fertile Crescent to the collapse of the Easter Island, to the US dust bowl of the 1930s to the current situation concerning worldwide soil degradation (Diamond, 2005; FAO, 2015d; Montgomery, 2012). Increased pressure on soil from a growing world population and changes in land use patterns in the last 50 years has detrimentally affected soil ecosystems on a global scale (FAO, 2015d). Soils are under constant degradation pressure, undermining their capacity to function properly. International agencies, like the United Nations Food and Agricultural Organisation (FAO) and United Nations Environmental Program (UNEP), have over the last decades assessed the rate of soil degradation. Initiatives like the Global Assessment of Soil Degradation (GLASOD) in the late 1980s made an inventory of soil degradation and brought to light its extent. Later, the Soil Thematic Strategy (European Commission, 2006) of the European Union (EU) formalised the concept of soil threats and many essential soil functions (FAO, 2015d). According to the Thematic Strategy, the key environmental, economic and social functions essential for life are: food and other biomass production; storing, filtering and transformation of minerals, organic matter, water and energy, and diverse chemical substances; habitat and gene pool maintenance, enabling essential ecological functions; provision of a beneficial physical and cultural environment for mankind; and a source of raw materials (European Commission, 2002, 2006). The threats to these essential soil functions include: erosion; decline in organic matter; local and diffuse contamination; sealing; compaction; decline in biodiversity; salinization; floods and landslides (European Commission, 2006). One especially nefarious threat is soil erosion as it destroys soil if unabated. Soil erosion is a serious problem in many areas around the world, and soils are being lost with incredible speed, on average 10 - 100 times faster than soil formation rates, and in some locations up to 1,000 times faster (Brantley et al., 2007; Lal, 1990; Montgomery, 2007; Pimentel and Kounang, 1998).

These erosion rates are much higher than known soil formation rates, typically well below one t/ ha^{-1}/yr^{-1} with median values of approx. 0.15 t/ ha^{-1}/yr^{-1} (FAO, 2015d). Due to these slow formation rates soils are a non-renewable resource (FAO, 2015a) and should be considered as such. It can take many decades to reverse degradation trends as the technical challenges are often underestimated and it can be costly to build up soil natural capital again (Lal, 2009). If these rates of soil deterioration continues, humanity might end up losing a critical proportion of the arable soil within this century (Russel, 2014), the part of the soil which provides us with 90-95% of our food. FAO officials have even put forward an “End of the World Scenario” talking about the prospect of there only being around 60 harvests left (Russel, 2014) and rich nations like the United Kingdom might be only 30-40 years away from eradicating their soil fertility (Van der Zee, 2017).

Soil erosion and the threats mentioned above are often induced and exacerbated by human activities that frequently involve unsustainable land use. To make matters even worse, climate change is likely to further amplify these threats by putting even more stress on soils (Lal, 2013). Scholars (Lal, 2013; Sverdrup et al., 2013; Sverdrup and Ragnarsdottir, 2014) have even argued that “Peak” soil has already been reached, a state where the maximum available arable soil has been used and the resource is now in decline. This prospect is alarming as it is estimated that in the next 50 years we have to produce more food than humanity has produced since the advent of agriculture (Potter, 2009).

Growing world population, with increased affluence and changing dietary consumption patterns, are fuelling increased demands on land and soil. A 50% increase in food production before 2030 is forecasted and a 100% increase by 2050 (Baulcombe et al., 2009). This increase will inevitably increase water use whilst usable water resources are diminishing (Lal, 2013). Providing food and water to additional millions of people relies heavily on having arable soils and associated ES intact but trends are going in the opposite direction (FAO, 2015d). There is a real doubt as to whether soils can cope with the increased food and water demand expected in the 21st century (Banwart, 2011).

One of the main policy questions for this century is how we humans are going to feed a larger future population with less soil and less water available. In recent years, the importance of viable soils has been gaining more attention on the international policy level. The European Union proposed a Soil Thematic Framework as a way of combatting the multiple soil threats on an EU level; 2015 was The United Nations International Year of Soil declared by the General Assembly; a Global Soil Partnership and the Intergovernmental Technical Panel on Soils was recently formed[1] by the Intergovernmental Science — Policy Platform on Biodiversity and Ecosystem Services (IPBES) which covers several aspects of soils, the Food and Agricultural Organisation of the United Nations published the first report on the Status of the World's Soil Resources (FAO, 2015b), and, more recently, The Economics of Ecosystems and Biodiversity (TEEB) report on Agriculture and Food (TEEB, 2018), where soil is at the centre. These international bodies and others are calling for a policy agenda on soil security, framing soil use as a security issue, similar to the concept of food security (Yawson et al., 2016). Soil security has been defined by Bouma and McBratney (2013, p. 131) as "the maintenance or improvement of the world's soil resources so they can provide sufficient food, fibre and fresh water, contribute to energy sustainability and climate stability, maintain biodiversity, and deliver overall environmental protection and ecosystem services".

[1] http://www.fao.org/global-soil-partnership/en/

Sustainable soil management, one of the most pressing issues of our times, is central to soil security. It is defined by the Wold Soil Charter (FAO, 2015d, p. 8) as “activities that maintain or enhance the supporting, provisioning, regulating and cultural services provided by soils without significantly impairing either the soil functions that enable those services or biodiversity”. Soil, being underneath our feet most of the time and covered by vegetation, is usually ignored by most people and is sometimes even referred to as the forgotten ecosystem (Field et al., 2016). Concomitantly, soil has been ignored by most when it comes to land use decision-making, and its value is ignored or not factored into associated protocols and regulation. Its absence from decision-making frameworks further exacerbates the likelihood of its loss, heart-breaking as sustainable soil management holds the key to many of the most pressing current socio-economic and environmental problems (Bouma and McBratney, 2013; Koch et al., 2013; Lal, 2011; Lal and Pimentel, 2008). Even high profile scientific research projects, looking at the human use of natural resources and global ecosystems, have left out soil (Kumar, 2010; Rockstrom et al., 2009). In 2015, the United Nations published the Sustainable Development Goals (SDG) to be reached by 2030 and soils have direct relevance for 7 of the 17 SDG goals (Jonsson et al., 2016a).

To promote soil security and sustainable soil management, we need appropriate conceptual frameworks, methods, tools, and policies to monitor, promote and implement the sustainable management of soils.

1.2 Ecosystem services, natural capital and soil

In the last few decades, a new form of environmental and resource management system has been emerging which offers some hope that humans might be able to better manage the multiple benefits to human beings provided by soils. This approach is called Ecosystem Services (ES) and has been used extensively for the last two decades to classify and value various components of ecosystems worldwide (Costanza et al., 1997; Gomez-Baggethun et al., 2010; Kumar, 2010; MEA, 2005). The approach is based on the notion that the environment is a form of natural capital that can be considered a stock that yields a flow of goods and services into the future (Costanza et al., 1997). This approach has highlighted the value of many otherwise under-valued components of ecosystems and changed the dialogue on the value that humans derive from ecosystems. It has shifted the discussion about the value that the natural world provides as international bodies (e.g. the EU, FAO, UNEP, World Bank) are now talking ES and the benefits they supply. ES are usually split into four broad categories:

supporting, provisioning, regulating and cultural, with some variance between the many frameworks delineating their typologies. The distinction between the categories is as follows: supporting functions are services that are necessary to produce other services, such a primary production and soil formation; provisioning services are usually products that people obtain from ecosystems, such as food, fuel and fibre; regulating services are the benefits that people get from the regulation of ecosystem processes, such as climate regulation, erosion control and water purification; and cultural services are the non-material benefits that people obtain from experience in and from ecosystems.

Soil has until recently been mostly left out of these kinds of large-scale ecosystem evaluation analysis, at least as a vital ecosystem, as it was omitted from some of the most fundamental reports on natural resource management in the last two decades, such as the Millenium Ecosystem Assessment (MEA) (MEA, 2005), The Economics of Ecosystem and Biodiversity (TEEB) report (de Groot, 2010), and the Planetary Boundary evaluations by Rockstrom et al. (2009). This omission is peculiar as it might be argued that soil, as natural capital, is one of the most important ecosystems to human wellbeing. Soils serve as the fundamental bases for biodiversity on Earth, as they contain more species, both in terms of number and quantity, than all other above ground biota put together. Soils play an important role in Earth´s water cycle as the first filters of world's water, as they absorb, filter and store water, regulating flow rates and making it available to terrestrial ecosystems. They can control water quality by filtering contaminants and making nutrient solutes available to plants when they are needed. Soils play an important role in climate regulation, affecting global temperature and precipitation (Haygarth and Ritz, 2009; Lavelle et al., 2006; Turbé et al., 2010; Wall, 2004). They provide a physical environment for human infrastructure, plants and animal species, and they provide a suitable living and reproduction space for different flora and fauna. Soils cycle nutrients, maintaining fertility that supports plant growth and by providing nutrients, water and physical environment, soils provide the conditions for terrestrial biomass production. They can be a source of raw materials, for example topsoil, clay and peat. Soils have played a major part in the worldview and religious belief of different societies (Gould et al., 2014; Wells and Mihok, 2009), and soils and sediments act as geological archives, giving insight into past climate and environmental conditions. Given these multiple roles, collectively called soil ecosystem services (ES), it is evident that soil needs to be maintained and the only way to do this is to protect and better manage this form of natural capital.

Ecosystem services are fundamentally important for economic prosperity and human well-being. In the market economy, a dominant form of an economic system in the western world, decision making is largely based on signals provided by the market through prices, costs and quantities, and thereby focuses on market-based goods and services. Soils as natural capital and many soil ES are considered non-market goods and services. Their nature does not easily lend itself to be traded in markets and thus they do not have a market price, but regardless they are important. Economic values derived from natural capital, like soils, are broken into several types as shown in figure 1.

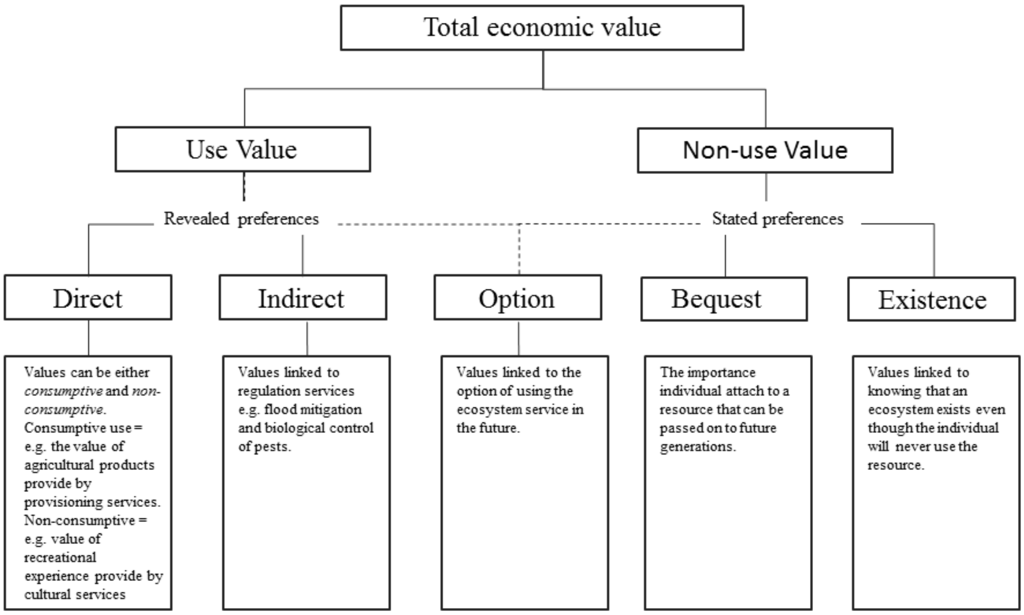

Figure 1 Total economic value (Jonsson and Davidsdottir, 2016)

The two main types of values are called use value and non-use value. Use values are further broken down into direct and indirect use values. Direct use values include consumptive uses like food (collection of berries, mushrooms, herbs and plants) and fibre, while non-consumptive uses include recreation, photography and the view from a dwelling. Indirect use values include values that are not consumed such as carbon sequestration, hydrological buffering, filtering of nutrients and contaminants, and biological control of pests and diseases. Non-use values are the values that people assign to economic goods even if they never have and will never use them, e.g. paying for the protection of species they will probably never encounter or consume. The value types, use (direct and indirect) and non-use, are selected based on what is appropriate for the respective ES and the purpose of the valuation exercise.

Economists choose between various valuation tools they have at their disposal to place an estimated monetary value on the identified services, Total economic value is the sum of all the relevant use and non-use values for all the various services a particular ecosystem provides (Freeman, 2003; Hanley et al., 2006).

These valuation types are used in various ES frameworks, where they link to the categories of ES and appropriate valuation techniques, market and non-market. The most commonly used ES frameworks are from the Millenium Ecosystem Assessment (MEA) (MEA, 2005), The Economics of Ecosystems and Biodiversity (TEEB) (Kumar, 2010), Mapping and Assessment of Ecosystem Services (MAES) (EEA, 2016), and The Common International Classification of Ecosystem Services (CICES) (Haines-Young and Potschin, 2012).

The ES approach is a relatively new approach to ecosystem and natural resources management and can be used along with other tools for sustainable soil management, as shown in this thesis. By using the ES approach, we can systemically categorise and value the services from soils and incorporate them into land use decision making. This is one way of promoting sustainable soil management, increasing soil security and the protection of soil natural capital.

Several authors have been calling for a framework for soil economic valuation to use for soil policy (Breure et al., 2012), and an ES approach to sustainable soil management has been gaining support in the soil research community (Robinson et al., 2012) and can be found in the FAO (2015d, p. 4) World Soil Charter where it is stated in Principle three that "*Soil management is sustainable if the supporting, provisioning, regulating, and cultural services provided by soil are maintained or enhanced without significantly impairing either the soil functions that enable those services or biodiversity.*"

To date, no comprehensive agreed upon framework for soil ES exists, although there have been many proposed which have contributed to a better understanding on how to represent and value soil ES (Dominati et al., 2010a; Jonsson and Davidsdottir, 2016). What has also been lacking thus far is a systematic way of connecting biophysical data relating soil to economic valuation, with use of economic methods (Baveye et al., 2016). Papers one and two discuss those issues, first by providing a systematic overview of methods and valuation for soil ES, second, by providing a framework (figure 2) on how to connect biophysical data with economic value, and, third, providing a valuation of three important soil services using the framework.

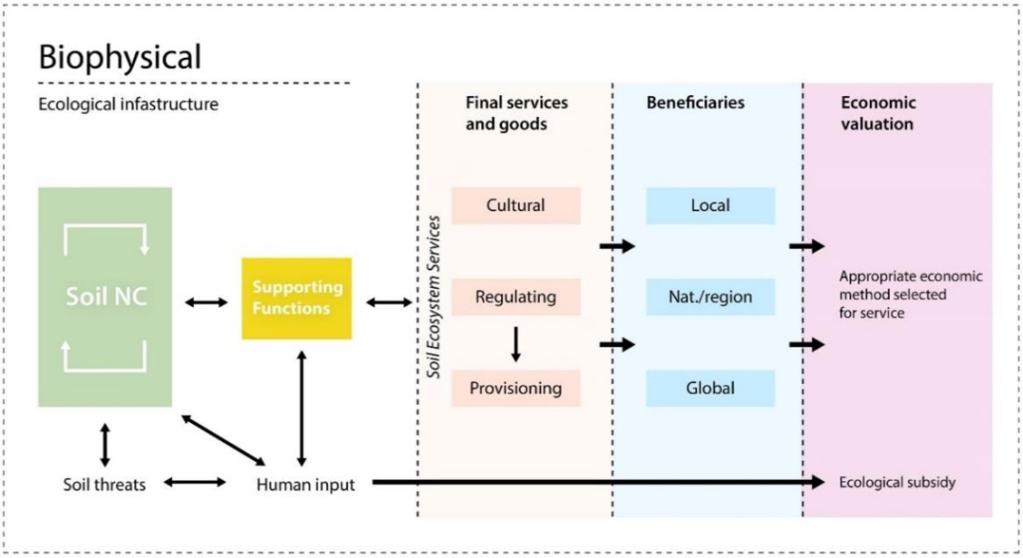

Figure 2 Soil ecosystem services framework (Jónsson et al., 2019)

A soil ES-based framework is needed for sustainable soil management but it has its limits and is not a panacea. Information on soil conditions, soil use, and other important soil factors are also needed to assess holistically whether soils are managed sustainably.

1.3 Indicators for sustainable soil management

Another way to account for the importance of soil is to monitor and assess what is happening with soils by using indicators. They are important for sustainability assessments as they show how a resource is developing (Ness et al., 2007). Sustainability indicators have their foundation in the concept of sustainable development, which became popularised in 1987 with the Brundtland Commissions' report, *Our Common Future*, and has since then been central to decision making worldwide (MEA, 2005; World Commission on Environment and Development, 1992). The Brundtland report defined sustainable development as a development that "*meets the needs of the present without compromising the ability of future generations to meet their own needs*" (Development et al., 1987, p. 43). It centres on the notion of equity, both intra — and intergenerational, and the importance of keeping humanity and its ecological impact within planetary boundaries (Rockstrom et al., 2009; Steffen et al., 2015; UNDESA, 2002). The need for the development of sustainability indicators is set out in Agenda 21 from the Rio UN Summit in 1992 and was taken up by the United Nations Commission for Sustainable Development (CSD) (Pinfield, 1996). Indicators are central to sustainability assessments as they can visualise phenomena and highlight trends and can thus simplify, quantify, analyse and communicate otherwise multifaceted and complicated information (Jesinghaus, 1999; Warhurt, 2002) which is very useful for policy and decision makers.

Indicators for soil management have been around for a long time but they have tended to focus solely on the biological, geological or chemical properties of soils (Jonsson et al., 2016a). Other important soil aspects that relate to the social and economic dimensions of soil have largely been left out and indicators sets have not covered holistically all three dimensions (environment, social and economic) of sustainable development. To further complicate the matter, soils vary a lot depending on their geographical location and conditions, such as the parent bedrock material, climate, vegetation, slope and fauna. One set of soil indicators might not be appropriate for every soil type or soil user. Users of soils range from single farmers to large companies, from policy makers to academics and scientists. These stakeholder groups have different views on how they approach soil management and how to monitor its development. Specific stakeholders might only focus on certain dimensions of sustainable development, for example policy makers might emphasise societal and economic issues, while farmers would be more interested on immediate properties, like soil's organic matter, linked to the environmental dimension. The need for location-based soil sustainability indicators is stated in the World Soil Charter (FAO, 2015c, p. 4) principle four where it is says that: "*[t]he implementation of soil management decisions is typically made locally and occurs within widely differing socio-economic contexts. The development of specific measures proper for the adoption by local decision-makers often requires multi-level, interdisciplinary initiatives by many stakeholders. A strong commitment to including local and indigenous knowledge is critical".* This is the approach that was adopted for the development of soil indicators for sustainable development, which is presented in paper three. The development of the indicators built on a transdisciplinary approach with multiple stakeholders who were local experts working in different sectors. The groundwork for the indicator development was based on an extensive literature review, various workshops with expert stakeholder groups, and then finally a Delphi survey for the final selection of soil indicators. This multi-step approach can serve as an example of bridging and bringing together the knowledge of multiple domains within the broad field of soil science, reaching a consensus among different stakeholder groups on what should be the key aspect to focus on regarding sustainable soil management.

1.4 Combination of approaches for sustainable soil management

Sustainable soil management is a central topic for the expected intensification of agriculture in the 21st century needed to fulfil food demand. This intensification calls for sustainable approaches to increase food production from existing farmlands in ways that "place far less pressure on the environment and do not undermine our capacity to continue producing food in the future" (Garnett et al., 2013, p. 33). The success of conventional agriculture in the 20th century by increasing yields and feeding more people has had tremendous effects on natural capital; on land and soil, water and climate, involving major negative externalities. The environmental cost of agriculture is considerable and further "*[e]xpanding agricultural land results in losses of vital ecosystem and biodiversity services, as well as damaging livelihoods for communities relying on these lands*" (Baulcombe et al., 2009, p. 7). According to the TruCost analysis (Trucost PLC, 2013), a study undertaken on the behalf of TEEB, where the revenue of a business activity is compared to the cost of natural capital, four agricultural sectors are placed in the top 10 list of the 100 global externalities, the largest being 1) Cattle ranching and farming in South America, which has a natural capital cost of 312.1 $BN compared to a revenue of 16.6 $BN; 2) Wheat farming in Southern Asia, which has 214.4 $BN in cost and 31.8 in revenue; and the other two follow close behind (Trucost PLC, 2013 p. 28). These very high cost to revenue ratios illustrate the immense cost of conventional agriculture and further underlines the need to radically rethinking approaches to lessen the impacts on the environment (Garnett et al., 2013), as a business-as-usual scenario will not work throughout the 21st century. Sustainable agriculture practices are needed, and intensification of production must work within the confines of agricultural sustainability. According to Baulcombe et al. (2009 p. 7), agricultural sustainability incorporates four key principles:

1. Persistence: the capacity to continue to deliver desired outputs over long periods of time (human generations), thus conferring predictability;
2. Resilience: the capacity to absorb, utilise or even benefit from perturbations (shocks and stresses), and so persist without qualitative changes in structure;
3. Autarchy: the capacity to deliver desired outputs from inputs and resources (factors of productions) acquired from within key system boundaries, [and]
4. Benevolence: the capacity to deliver desired outputs (food, fibre, fuel, oil) while sustaining the functions of ecosystem services and not causing depletion of natural capital (e.g. mineral, biodiversity, soil, clean water).

Besides these four principles, Baulcombe et al. (2009, p. 7) add: *"[a]ny system is by these principles and measures unsustainable if it depends on non-renewable inputs, cannot consistently and predictably deliver desired outputs, can only do this by requiring the cultivation of more land, and/or causes adverse or irreversible environmental impacts which threaten critical ecological functions"*. Sustainable soil management takes the central stage of any quest for agricultural sustainability involving the intensification of agriculture. The practical implication of sustainably intensifying agriculture on existing land without disrupting the flow of essential soil ES (Olson et al., 2017) and harming soil natural capital must involve new approaches to both manage and measure soil trends and conditions.

Paper four introduces a novel combination of approaches, so-called Tools for Sustainable Soil Management, where Ecosystem Service Analysis, Energy Return on Investment (EROI) and Cost-Benefit Analysis (CBA) are used together to assess, in a more holistic manner, the effect of different fertilisation treatments on an agricultural operation. The conventional approach in agriculture has been to look narrowly at the profitability of an operation, ignoring the multiple negative externalities associated with production. By using this combination of multiple approaches and soil indicators it is possible to investigate the ecological, energetic and economic aspects of an agricultural operation simultaneously and better establish the differences associated with various fertilisation treatments, given the aim of sustainable soil management and agricultural sustainability.

1.5 Study location Koiliaris watershed Crete, Greece

The Koiliaris watershed, on the Greek island of Crete, was selected as a venue for the pilot studies where the soil ES framework was put to the test on two separate occasions; first in a Watershed scale study (paper two) and second in a plot scale study (paper four). Koiliaris watershed was one of the Critical Zone Observatories (CZO) in the SoilTrEC project. The CZO was in the north-western part of the island, 25 km east of the city of Chania. The watershed is 132 km^2 and has a steep rise in elevation ranging from 0 m on the coast up to 2120 m AMSL in the White Mountains. Around 16% of land on the island is attributed to agricultural activities (Panagos et al., 2014) and it is the main consumer of water on the island (Chartzoulakis et al., 2001). Agriculture contributes up to 13% of the island's Gross Domestic Product (GDP) and employs 6.7% of the workforce growing fruit, vegetables and raising livestock. Fruit crops are dominated by olive production (approx. 90%), followed by citrus, almonds and avocados (Chartzoulakis et al., 2001). Economically, olive oil is Crete's most important agricultural product (Tsakiris et al., 2007). The main vegetable crops are tomatoes, cucumbers, potatoes, eggplants, onions, watermelons, melons, cabbages and peppers. Livestock farming (sheep and goats) is also important in Crete and more than a million sheep and goats are farmed there under low intensity systems (Stefanakis et al., 2007). Land use patterns have remained similar for the last 50 years but they have intensified, especially the number of livestock on the island, which has increased fivefold (Nikolaidis et al., 2013). This long history of land use has led to soil degradation (Nikolaidis, 2011; Nikolaidis et al., 2013), resulting in poorly developed soils (Panagos et al., 2014). Crete is considered a high-risk area for desertification due to inappropriate land use, like overgrazing (Panagos et al., 2014) and changing climate conditions. The Koiliaris CZO was selected to carry out the two soil ES valuations because it had the most developed data and models at the time that the research took place. The data for the research was provided by SoilTrEC partners at Technical University Crete and collected on site during two separate visits in 2013 and 2014.

1.6 Thesis focus and structure

The following research questions guided the overall thesis:

- Why should we value soil economically?
- How can we value soil ecosystem services economically?
- What indicators are necessary to promote sustainable soil management and, as a corollary, sustainable agriculture?
- What is needed to factor in soil natural capital into land-use decision making?

The aims of this PhD thesis were twofold: 1) to develop a methodological framework and carry out an evaluation of the economic value of soil ecosystem services. The purpose was to enable more thorough economic valuation of the consequences of different land use management decisions in agricultural systems, and 2) to develop indicators which could guide decision-makers on sustainable soil management.

The following milestones were related to the aims, of which each resulted in an academic publication: 1) Review research methods and valuation of soil ecosystem services (paper I); 2) Develop a framework for the economic valuation of soil ecosystem services (papers I and II); 3) Carry out a pilot study using the soil ecosystem services framework on the Greek Island of Crete, both on a watershed scale and on a plot scale (papers II and IV); and 4) Develop a set of indicators for sustainable soil management in partnership with expert stakeholders (paper III).

Paper one is a review study that provides an overview of the categories, methods and values for soil ecosystem services (ES). This is the first review study of soil ES economic value in the ecosystem services literature.

The second paper introduces a soil ES framework that links biophysical soil processing models, developed by a SoilTrEC partner, and economic methods to value soil ES. It reports on the outcome of a pilot study on soil ES valuation concerning three services in the Koiliaris watershed on Crete.

Paper three introduces a transdisciplinary approach to develop soil indicators for sustainable development and reports on the outcome of a Delphi survey among expert stakeholder groups, where scientists, policy makers and soil practitioners selected a set of indicators deemed most important for sustainable soil management.

The fourth paper reports on the outcome of a tomato plot study in Koiliaris watershed where three approaches, soil ES, Energy Return on Investment (EROI) and Cost-Benefit Analysis (CBA) were used to look at the economic, ecosystem services and energetic aspects of an agricultural operation focusing on the effects of different fertiliser application. This paper ties together the preceding three papers as it builds on the soil ES framework and uses two other soil indicators necessary for sustainable soil management. This study has wider implications for plant and livestock systems as it shows that it is possible to intensify production in an energy efficient manner using organic amendments, that are often unused by-products in agricultural systems, whilst improving soil ecosystem services.

Besides the four research papers the thesis includes an article by Robinson et al. where I am a co-author on the value of soil resources and natural capital and a book chapter for elementary schools on the importance of soil.

The research articles presented in this thesis contributed to the Soil Transformation in European Catchment (SoilTrEC) research project, which is a part of the European Commission's 7th Framework Programme. The aim of SoilTrEC was to understand the lifecycle of soils within the Critical Zone of the Earth's surface and to develop methodologies to maintain and enhance soil functions and manage them sustainably for future generations.

Professor Brynhildur Davíðsdóttir and I took part in Work Package 5 - *Quantifying Soil Function Impacts*, where the aim was to integrate the soil processes' model results and knowledge – a focus of the other work packages of the SoilTrEC project – into broader assessment frameworks and quantify impacts on soil functions, making these available as decision-support tools.

1.7 Summary of methods and results

The following are short summaries of the methods and results for the five papers and the book chapter presented in this thesis.

1.7.1 Paper I

Jónsson, J.Ö.G., Davíðsdóttir, B., 2016. Classification and valuation of soil ES. *Agricultural Systems 145*, 24-38.[2]

The first paper is a literature review of soil ES, which analyses the economic value of soil ES and methods to value different services.

The main research questions in this paper are:

- How are soils being evaluated within the current framework of ES?
- What economic methods are used to value them?
- What are the potential values of different soil ES?

The paper reviews the available literature of soil ES and soil functions, the economic methods used to value them, and values derived from studies under different land use[3]. The paper connects the recent advances in Earth Critical Zone research that are contributing to the development of soil ES valuation techniques within the ES framework. These developments have further underpinned the necessity to include the multiple aspects of soils in ES frameworks for their fundamental roles in ecosystem functionality and vitality. The current literature on ES seems to pay less attention to soils and its ecosystem services and functions than other ecosystem types. Examples are given on how soil ES can be classified and valued using standard economic methods and established analysis frameworks. We show how significant economic value is derived from soil ES and thus highlight the economic losses associated with soil degradation. Furthermore, we show the need to develop a comprehensive framework for the economic assessment of soil ES to better inform decision making at various levels of governance on land use and management.

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[2] Received: 24 March 2014 / Accepted 28 February 2016 Available online: 12 March 2016.
 The role of the doctoral student (Jón Örvar G. Jónsson) in this paper was to carry out all the research activity. Professor Brynhildur Davíðsdóttir guided the doctoral student during the research activity and writing process.

[3] The following keywords were used during the literature search: "soil ecosystem services", soil "ecosystem services", "soil services", "soil service" "soil functions", "soil function" "soil ecosystem functions", "ecosystem services of soil".

1.7.2 Paper II

Jónsson, J.Ö.G., Davíðsdóttir, B., Nikolaidis, N.P., 2016. Valuation of soil ecosystem services. *Advances in Agronomy 142*. 353-384.[4]

This paper presents a framework for the evaluation of soil ES, connecting it to biophysical models provided by TUC-HER. The paper also presents a case study where the framework is put to test in the Koiliaris Watershed, Crete, Greece where three soil ES were identified and modelled.

The main research questions addressed by this paper are:

- What information does the economic value of soil ES convey?
- What is the economic value of the selected three soil ES in the Koiliaris watershed?

Paper two introduces a framework to assess the relevance of sustaining soil functions that links together the concepts of soil natural capital, soil biophysical support functions, and soil ES with beneficiaries and economic valuation. We define and categorise different components of the framework, illustrating their functions within the framework and how these various parts interlink and create benefits for humans, which can be valued economically. We illustrate the use of the framework in a pilot study in the Koiliaris watershed on the Greek island of Crete, where we value three soil ES using economic methods. The estimated economic values of the respective soil ES were as follows: crop and livestock biomass 740–7560 id$ ha− 1 year− 1; filtering of nutrients and contaminants 0–278 id$ ha− 1 year− 1; and climate regulation − 2200 to − 5610 id$ ha− 1 year− 1. The paper illustrates the importance of using economic valuation of soil ES along with other metrics necessary for sustainable land management, as failing to do so might lead to land use recommendations based solely on the highest value yielding services.

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[4] Received: 15 September 2016 / Accepted 15 November 2016 Available online: 30 December 2016. The role of the doctoral student (Jón Örvar G. Jónsson) in this paper was to carry out the research activity and writing the paper. Professors Brynhildur Davíðsdóttir and Nikolaos Nikolaidis guided the doctoral student during the research activity and writing process.

1.7.3 Paper III

Jónsson, J.Ö.G., Davíðsdóttir, B., Jónsdóttir, E.M[5]., Kristinsdóttir, S.M., Ragnarsdóttir, K.V., 2016. Soil indicators for sustainable development: A transdisciplinary approach for indicator development using expert stakeholders. *Agriculture, Ecosystems & Environment 232*, 179-189.[6]

The main research questions addressed by this paper are:

- What are the core soil indicators for sustainable development?
- Is a transdisciplinary approach to indicator development with extensive stakeholder participation a good tool for indicator development?
- Are soil indicators needed for sustainable development?

Paper three presents the outcome of a transdisciplinary approach towards sustainability indicator development, where a set of soil indicators for sustainable development were created. The development process involved active stakeholder participation from scientists, policymakers and soil managers mainly based in Iceland. Using a Delphi survey technique, stakeholder groups evaluated 49 indicators and selected 30. Of these 30, 14 were common to all stakeholder groups and presented a final set of core soil indicators for sustainable development. The Delphi survey illustrated the usefulness and the need for relevant stakeholder involvement in an indicator's development process and the supportive role of survey-based instruments in selecting common indicators. We illustrated the need for soil indicators for sustainable development as soil is a central issue in at least half of the United Nations Sustainable Development Goals. We argued that an indicator development process as presented in the paper can serve as starting point for discussion on complex matters, such as the sustainable use of soils, as it brings forth the commonalities and the differences between stakeholder groups.

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[5] The role of Eydís María Jónsdóttir in this paper was to provide data and research documents on soil indicators.

[6] Received: 15 September 2016 / Accepted 15 November 2016 Available online: 30 December 2016. The role of the doctoral student (Jón Örvar G. Jónsson) in this paper was to carry out the research activity. Professors Brynhildur Davíðsdóttir and Kristín Vala Ragnarsdottir guided Eydís María and the doctoral student during the research activity and writing process. Sigrun María Kristinsdóttir assisted with a stakeholder survey.

1.7.4 Paper IV

Jónsson, J.Ö.G., Davíðsdóttir, B., Nikolaidis, N.P., Giannakis, G.V., 2019. Tools for Sustainable Soil Management: Soil Ecosystem Services, EROI and Economic Analysis. *Ecological Economics* 157, 109-119.[7]

The main research questions addressed by this paper are:

- What are the differences in economic, ES and energy parameters between the methods of fertilising?
- What is the most favourable treatment according to these three approaches?
- What is the least favourable treatment according to the three approaches?

Paper four compares the outcome of a multi-year experimental study of four different fertilisation treatments on soil ecosystems, that took place in a tomato plantation in the Koiliaris watershed on the Greek Island of Crete. Three approaches were used - Ecosystem Services Analysis (ES), Energy Return on Investment (EROI) and Cost-Benefit Analysis (CBA) to compare the outcomes of the four treatments. The treatments included the application of inorganic fertiliser (IF), manure (M), municipal solid waste compost (MSWC), and a 70:30 mix of MSWC + M. The results showed that MSWC + M was the best overall regarding economic, ES and EROI, while the IF one was the least favourable in all three approaches. The result from this study has a large implication for agriculture and livestock production as it shows that it is possible to have an economically profitable operation that provides an energy surplus and improves soil services and functions. The comparative analysis using these three approaches provides valuable information to facilitate sustainable soil use, which is needed to enable the foreseeable intensification of agriculture in the coming decades.

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[7] Received: 2 May 2018 / Accepted 12 November 2018 Available online 23 November 2018 The role of the doctoral student (Jón Örvar G. Jónsson) in this paper was to carry out the research activity and writing the paper. Professors Brynhildur Davíðsdóttir and Nikolaos Nikolaidis guided the doctoral student during the research activity and writing process. Georgos Giannakis assisted with data collection and compilation.

1.7.5 Paper V

Robinson, D.A., Fraser, I., Dominati, E.J., Davidsdottir, B., Jonsson, J.O.G., Jones, L., Jones, S.B., Tuller, M., Lebron, I., Bristow, K.L., Souza, D.M., Banwart, S., Clothier, B.E., 2014. On the Value of Soil Resources in the Context of Natural Capital and Ecosystem Service Delivery. *Soil Science Society of America Journal 78*, 685-700.[8]

The main research questions addressed by this paper are:

- Why should the economic value of the contribution of soils to the provision of ecosystem services be included?
- How can the valuation of soil be incorporated into land use decision support tools?

This article examines the valuation concepts and asks why we might attempt to economically value the contribution of soils to ecosystem services. We examine economic valuation methods and review data on the economic valuation of soils. By surveying prices of soils on the internet we can make a first, limited global assessment of the direct market value of topsoil prices. We then consider other research efforts to value soil. Finally, we consider how the valuation of soil can be used in the introduction of improved resource management mechanisms such as decision support tools on which valuation can be based, including the UN's System of Environmental and Economic Accounts (SEEA) and policy mechanisms like Payments for Ecosystem Services (PES). The paper highlights that soils make critical and essential contributions to the economy, and that soil loss represents a major environmental and economic loss. A survey of soil commodity prices on the internet indicates that the median direct market value of topsoil in terms of price per ton is ~$22 in the United States and Canada, and ~$47 in the UK. It is important to capture changes to the soil ecosystem and its functionality, and methods are needed to capture soil values under various uses, both

[8] Received: Jan 14, 2014 / Published: June 10, 2014. The role of the doctoral student (Jón Örvar G. Jónsson) in this paper was to carry out part of the research activity regarding the value of soil at different geographical locations. Dr. David Robinson was the main author, guided the research activity and the writing process and other authors contributed to the various part of the paper.

quantity and functionality. To work well, economists and soil scientists must work together to develop indicators that can be used to assess the state of soil functions. 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 the contribution of soils from an economic point of view.

1.7.6 Book chapter

Jonsson, J.O.G., Davidsdottir, B., Ragnarsdottir, K.V., 2015. 5. The Value of soil ecosystem services. Soil: The Life Supporting Skin of Earth, 40.[9]

This book chapter featured in an eBook on soil for secondary school students. The eBook – *Soil: The Life Supporting Skin of Earth* was an output from the SoilTrEC research project, with the target audience being school children from the ages 11 to 18. In the book chapter – *The value of soil ecosystem services*, we explain what natural capital and ecosystem services are and why they are fundamental for human wellbeing. We illustrate the importance of soil as a type of natural capital and what services and benefits soils bring us. We show how different valuation techniques, including economic methods, can be used to value soils and how they can be a useful tool in taking more informed decisions about land and soil management.

The structure of the succeeding chapters is as following: the summary, reflections and concluding remarks followed by five papers: I - Classification and valuation of soil ecosystem services; II - Valuation of Soil Ecosystem Services; III - Soil indicators for sustainable development: A transdisciplinary approach for indicator development using expert stakeholders; IV - Tools for sustainable soil use: Soil ecosystem services, EROI and economic analysis and V – On the Value of Soil Resources in the Context of Natural Capital and Ecosystem Services Delivery and lastly a book chapter – The value of soil ecosystem services.

[9] Published: January 2015. The role of the doctoral student (Jón Örvar G. Jónsson) in this paper was to carry out the research activity and writing the chapter. Professors Brynhildur Davíðsdóttir and Kristín V. Ragnarsdóttir guided the doctoral student and contributed to the writing of the chapter.

2 Summary, reflection and concluding discussion

2.1 Summary

This thesis focuses on providing methods and indicators for sustainable soil management in land-use decision making, specifically focusing on economic valuation of soil ecosystem services and the development and use of indicators for sustainable soil management. It provides an extensive literature review on the economic valuation of soil ES, bringing together the various soil ES, assessment of them and value ranges. The thesis provides two studies on how soil ES valuation can be used, how to link biophysical models with a framework for the economic valuation for soils and how to use soil ES valuation techniques, along with other approaches, in land-use decision making. It provides a methodology for developing indicators for sustainable soil management which incorporates extensive stakeholder participation, bringing forth a set of practical indicators that decision-makers can use. The thesis also shows how to use multiple approaches when estimating the sustainability of an agricultural operation by combining soil ES analysis along with Cost-Benefit Analysis (CBA) and an Energy Return on Investment (EROI) analysis. This combination provides more holistic information (ecological, economic and energetic) on the outcome of agricultural operations. The thesis outcome is a set of methods and examples that provides decision-makers with additional tools to make better-informed decision regarding land-use in agricultural systems, decisions that are more likely to consider the ecological, economic and energetic aspects that influence land-use.

Paper one (Jonsson and Davidsdottir, 2016) is one of only a few literature reviews that provides an overview of the categories, methods, and values for soil ES. The paper demonstrated what economic valuation methods are available to assess soil ecosystem services. Other ecosystems, like forests and wetlands, seem to be covered extensively in the ES literature (Kumar, 2010), but for soils, there is a gap and no consensus has been reached on how to classify and evaluate them using economic data. The article contributes to the literature by connecting soil natural capital and soil ES into a coherent soil ES framework that allows for economic valuation. It builds on the work by Dominati et al. (2010a); Dominati et al. (2010b); Robinson and Lebron (2010); Robinson et al. (2009). The paper helps to communicate the value of soil natural capital to soil stakeholders (soil practitioners, scientists and policy makers) who help to secure long-term soil protection and security (Breure et al., 2012). The conclusion of the paper is that there is a need to develop a holistic classification for the economic valuation of soil ES.

Paper two (Jonsson et al., 2016b) complements paper one as it introduces a comprehensive soil ES framework that links to biophysical models and methods for the quantification of a set of soil ES. Thus, it creates a seamless link from a measured and modelled soil service to an economic value. The framework builds on the quantification of soil functions and services made available through the measurement and modelling work produced within the SoilTrEC project and reported in the special issue of Advances in Agronomy (Giannakis et al., 2016; Kotronakis et al., 2017b). The paper shows how soil ES can be valued economically by valuing three soil ES in the Koiliaris watershed on the Greek island of Crete. The conclusion of paper two is that soils are an important type of natural capital and their value must be recognised as they provide multiple benefits for humans.

Paper three (Jonsson et al., 2016a) focuses on the development of indicators for sustainable soil management and it illustrates the usefulness of a transdisciplinary approach to consensus building concerning the sustainable management of soils. It uses a Delphi survey technique along with other methods to identify different stakeholders' opinions and to select several established indicators for soils, deemed appropriate estimators of sustainable soil use. The fact that the highest rated indicator from the selection process was the *Public awareness of the value of soil* shows the need of communicating further, both to the public and decision makers on all levels of society, the immense importance of soil to human wellbeing (Jonsson et al., 2016a). Soil security is a central issue of the Sustainable Development Goals and cannot be obtained without the use of indicators for sustainable soil management. The conclusion of article three was that a transdisciplinary approach with extensive stakeholder participation is a workable way to reach a consensus regarding the sustainable management of soils.

Paper four (Jónsson et al., 2019) builds on papers one, two and three by combining the soil ES evaluation framework along with two other approaches, Cost-Benefit Analysis (CBA) and Energy Return on

Investment (EROI). The paper thus uses three broad indicators as tools for sustainable soil management, and it showed how important it is to evaluate an agricultural operation from many aspects simultaneously, not solely looking at whether an operation is economically feasible as is commonly the case. Through a more holistic analysis (from an ecological, energetic and economical perspective) of an agricultural operation, it is possble to gain better insights into its viability. This broader perspective is necessary if sustainable soil management is to be achieved. The results showed that organic additions trumped inorganic fertilisers in all categories measured and then these results, along with the other primary studies (Giannakis et al., 2017; Kotronakis et al., 2017a), make a convincing argument for the wider adoption of organic additions.

The structure of the remainder of this chapter is as follows. Section 2.2 discusses in more detail the main outcome and implications from the research in the context of land-use decision-making regarding 1) soil natural capital and soil ES, 2) sustainability indicators, and 3) sustainable soil management. The contribution to academic and practical knowledge is covered in section 2.3. Section 2.4 provides recommendations based on the outcomes of the research. Section 2.5 evaluates the limitations of the research regarding research methods and data. Section 2.6 considers options for future research. Section 2.7 provides a concluding statement based on the thesis outcomes.

2.2 Discussion

In this chapter the results from the four papers will be discussed in the context of land-use in agriculture regarding:

- Soil natural capital and ecosystem services
- Sustainability indicators
- Sustainable soil management

2.2.1 Soil natural capital and soil ES

Papers one, two and three show how to classify and value soil ecosystem services using the soil ES framework proposed in paper two and biophysical data from TUC-HER. The work builds on the studies of Dominati and Robinson (Dominati et al., 2010a; Dominati et al., 2010b; Robinson et al., 2009), and I argue, as they do, that the soil is a form of natural capital that yields a flow of services, with this type of natural capital providing tremendous benefits to humans. This is illustrated by pointing out the multiple roles that soils play in human welfare and by bringing together valuations for almost all service categories, emphasising that soils play a fundamental role in the facilitation of human wellbeing. With this in mind and the results and discussion presented in this thesis, I argue in favour of valuing soil ES in monetary terms as a way of protecting soil natural capital. Some authors have argued the contrary, that they should not be valued in monetary terms as too many unknown variables exist and it is too difficult (Baveye et al., 2016). I disagree with this assertion and believe valuation is justifiable in certain situations concerning land use, as the alternative might be that soil ES would be ignored, under-valued and over-exploited. A valuation based on educated estimation is better than no valuation. As Costanza et al. (2017) put it, the question is not whether or not to value, but what type of valuation approach is appropriate. As soil ES are mostly non-market goods, it is easy to leave them out of economic decision making. As Daily (1997, p. 23) put it: *"To make rational choices among alternative uses of a given natural environment, it is important to know both what ecosystem services are provided by that environment and what those services are worth"*. I fully acknowledge the existence of other value systems that this research does not cover but a full assessment of the different value systems applicable to soil ES was beyond the scope for this thesis. As a result, they were omitted, although some of them were briefly mentioned. Monetary valuation of services, used in this research, is justifiable when trying to figure out what the best land use options are regarding the soil ES that are present. This valuation approach brings to the surface overlooked values and associated costs in land use decision-making. It does not, however, deliver perfect information. Nevertheless, the values reported are signalling the benefits that soils provide, and that soil use comes with a cost and its misuse may lead to potentially high costs. Many authors have tried to account for soils in economic decision making, and there is a diverse literature (Jonsson and Davidsdottir, 2016) on economic valuation of soil ES and functions. There are, though, relatively few studies that link soil properties to ES and then value these economically (Adhikari and Hartemink, 2016; Baveye et al., 2016). A few studies worth mentioning are by Dominati et al. (2014a); Dominati et al. (2014b); and Hewitt et al. (2015). It is important to note that the soil ES framework, and the associated monetary values report in the articles, are only annual flow values and thus snapshots reflecting just the current state. The true value of soil natural capital cannot be calculated by summing up the values for all the services. Doing this leads to the inherent risk of underestimating the true value of soils, as there are many unknown aspects of soils and an incomplete understanding of how they function (Parker, 2010). Their true value is, in fact, incalculable in monetary terms. Trying to come up with a single number for “the” total value of soils is an illogical pursuit. Like Norgaard et al. (1998, p. 37) so eloquently stated in their article “Next, the value of God, and other reactions:” *[N]ow that we know the exchange value of the earth, we wondered with whom we might exchange it and what we might be able to do with the money, sans Earth*.“ What this large scale evaluation of natural capital and ecosystem services does, though, is change the dialogue regarding the value of the benefits that human receive from nature, like what happened when the landmark article *The value of the world’s ecosystem services and natural capital* by Costanza et al. (1997) came out. This article reported for the first time a comparison between global GDP and the aggregate economic value of the world’s ES (Costanza et al., 2017). Although the true or final value of soil natural capital cannot be calculated, the various studies cited in paper one illustrated the economic importance of individual soil ecosystem services, giving a value range from id$2 to id$22,219 per hectare. The total value of soil ES is likely to be significant and the value of the multiple soil ES needs to be factored into land use decision making. This value is largely invisible to decision-makers, and as a result, there is a need to develop a comprehensive framework for the assessment of soil ES to incorporate into policymaking. This incorporation is necessary to maintain soil natural capital and provide the ES at a level desirable to society.

During the literature search on soil ES analysis, it became increasingly apparent that the value of soil has largely been left out of the ES approach and land use decision making. This can be seen by the omission of soils as an ecosystem from landmark large-scale studies (Kumar, 2010; Rockström et al., 2009). Even though the international community is more focused on the importance of soils now than before (see discussion in the introduction), the issue is still relevant as studies including any valuation of soil ES are few and far between. In 2017, according to Costanza et al. (2017), the relatively new journal of Ecosystem Services had published 405 research review and commentary papers on Ecosystem Services. Only one of them concerned soils as an ecosystem. Although soil ES are often included in other ecosystem analysis types, e.g. forests, this shows that the focus is, for the most part, not on soils. A small illustration - by looking at two journals that have a strong focus on ecosystem services analysis, i.e. *Ecosystem Services* and *Ecological Economics* and using the keywords given in Table 1, there is an indication that analysis of soil ES as an ecosystem is not high on the agenda within the ecosystem services community.

Table 1 Keyword results from ScienceDirect for the journals Ecosystem Services and Ecological Economics

	Journal of Ecosystem Services	Journal of Ecological Economics
Keywords		
"soil ecosystem services"	4	6
"soil services"	5	7
"wetland ecosystem services"	26	14
"wetland services"	14	35
"forest ecosystem services"	74	44
"forest services"	64	169

Sustainable soil management relies on having holistic information regarding the economic value of soil ES. Thus, research efforts with regard to soil ecosystem services need to increase.

2.2.2 Sustainability indicators

To establish whether soil management is heading in the right direction, soil indicators are required - indicators that measure the state of the system, in what direction it is heading and show the various essential aspects of soils (Jonsson et al., 2016a). This research is one of the first attempts to develop soil indicators for sustainable development that covered all three dimensions of sustainable development and used a transdisciplinary approach with extensive stakeholder engagement. Up to now, the procedures used for soil indicator development have either been a solely top-down selection of soil indicators by experts, thus leaving out many important soil stakeholders, or the selection criteria has focused mainly on the natural aspects of soil (chemical, physical, ecological), leaving out important economic and social dimensions (Jonsson et al., 2016a). Incorporating extensive stakeholder input, we developed an indicator set that covers the three dimensions of sustainability while highlighting the properties and functions of soils requiring most attention.

The process of selecting soil sustainability indicators is in line with the call of authors like Hak et al. (2016), who have demanded indicators in order to achieve the 2030 Sustainable Development Goals, as soil sustainability is necessary if the United Nations Sustainable Development Goals for 2030 are to be reached (IASS, 2015; Keesstra et al., 2016). Until now, the literature on soil indicators has mainly focused on one or two dimensions (Baulcombe et al., 2009). This approach is a good example of the usefulness of a transdisciplinary approach using multiple instruments at various steps to reach a conclusion among varied stakeholders. With soil, it is of particular importance to involve stakeholders across multiple levels of governance, as the soil is a resource out of sight and so its various fundamental properties in Earth's Critical Zone go unnoticed. By involving many stakeholder groups, it is more likely that they take the multiple facets of soil into consideration. The highest-ranking indicator overall among the groups was *Public awareness of the value of soil*, emphasising that the experts felt there was a need to further educate the public on the importance of soil. There is a need to inform a wider audience about soils' importance as drawing attention to soil and its biodiversity has failed to attract the attention of society (Breure et al., 2012). The research showed the usefulness of the Delphi approach as it can both accommodate what is common among groups and different, as each stakeholder group had their satellite indicator set that overlapped with the other groups in the core set. With this method, it is possible to reach a consensus on a complex matter, like sustainable soil management, among different stakeholder groups that might operate among different levels of society. The research did display some weaknesses. The main were high attrition rates and relatively low response rates among the participants. These weaknesses are commonly reported in Delphi Surveys (Powell, 2003). Nonetheless, the indicator development process brought together different stakeholder groups that might not otherwise communicate or might not know of each other (Breure et al., 2012), thus creating an opportunity for knowledge growth and the exchange of ideas on how to best manage soils sustainably. A core set of indicators for sustainable soil management agreed upon by farmers, scientists, NGO's and policymakers alike is an asset.

2.2.3 Sustainable soil management

Humans allocate more than 50 per cent of the world's vegetated land for agriculture (Foley et al., 2005). Agriculture affects the atmosphere, terrestrial ecosystems and water bodies, large and small. While being successful in providing food and feed for both humans and animals, agriculture also undermines its very own foundation. As Pretty (2008) points out, agriculture is unique as its action within the sector effects many of the assets it is built on, thus unsustainable agricultural practices negatively affect the soil natural capital that agriculture builds on. Unsustainable management practices have destroyed vast areas of fertile land and soil (Brantley et al., 2007; David et al., 2009; FAO, 2015d), and to feed the future 10 billion people on the planet, agriculture needs to intensify but the intensification must be done sustainably. History is full of examples of collapsed civilisations due to soil degradation and the loss of soil natural capital (Diamond, 2006; Montgomery, 2012). Agriculture, in its pursuit to create food, feed and fibre for humans, affects the soil ecosystem, and its essential functions, and with it the atmosphere and waterways. In this thesis (papers two and four), the focus was on valuing three of the essential ecosystem services or soil functions as defined by the European Commission (European Commission, 2002, 2006): climate regulation (carbon), food production (biomass), and storing and filtering function (nitrate retention). The reasons for selecting these three services is that they are fundamental to soil functionality and have a wide effect beyond the soil ecosystem. First is soil carbon sequestration and its corollary, soil organic matter (SOM). SOM is elementary for the structure and function of the soil. For a well-functioning soil, the foundation of sustainable agriculture, a certain percentage of soil organic matter must be in place but how much exactly is up for debate as the percentage of SOM is influenced by multiple variables (Oldfield et al., 2015). Currently soil is the reservoir of twice as much carbon as the biosphere and atmosphere combined (Bellamy et al., 2005). Soil organic carbon is the dominant matter of SOM and human agriculture has affected the carbon balance in the soil. It is estimated that the loss of soil carbon due to land use and land cover changes in the last 12,000 years is somewhere in the magnitude of 133 Pg. of carbon for the top 2 m of soil and the rate of loss has increased in the last two hundred years (Sanderman et al., 2017). With industrial agriculture, for the last 70 years, this rate of soil carbon loss has increased even still. As carbon leaks out of the soil into the atmosphere, it degrades soil fertility and fuels climate change. Keeping soil carbon in the soil and returning the organic matter back to the soil is a necessary step to maintain and increase soil carbon and thus help to keep the natural carbon cycles in their dynamic equilibrium (Lal, 2009, 2011). This aspect cannot be overstated as agriculture accounts for around 20% of global greenhouse gas emission (Stocker, 2014). As was shown in paper four (Jónsson et al., 2019), the organic additions (manure and compost) improved the soil carbon and structure (Kotronakis et al., 2017a), and should thus be preferable fertiliser(s) where they are readily available. Many authors have also specifically argued for sequestering carbon in soil through agriculture as a potential win-win, as increased carbon in soils helps food production (more fertility), tackles climate change simultaneously (Lal, 2004, 2011), increases aggregate size and stability (Lehtinen et al., 2015) and subsoil and above soil biodiversity (van Leeuwen et al., 2015), and that organic additions are an essential part of that equation (Milne et al., 2015). Climate change will affect the stability of the soil carbon pool (Schils et al., 2008) and land management practices need to make sure that they are not increasing carbon outflow from soils but rather that they are carbon neutral or, better still, carbon negative. Organic additions can play a role in helping to intensify agricultural production in a sustainable manner, as it is likely that current agricultural land needs to yield more to feed a projected 10 billion in this century (Baulcombe et al., 2009). The results from (Kotronakis et al., 2017a) showed that organic additions improved the carbon content in the soil, and are a way to protect and improve soil essential functions.

Soils can regulate, to some extent, water quality by absorbing, retaining and chemically transforming solutes, thereby avoiding their release into waterbodies. Nitrogen-based fertilisers are extensively used in agriculture and have contributed to the massive increase in yields since the middle of the last century. They have also caused widespread negative environmental effects. When not handled correctly, they leak from agricultural land and into waterways and waterbodies, creating pollution, algae bloom downstream, large "dead zones" in coastal areas, affect groundwater quality, and cause human and animal disease. Nitrogen pollution is one of the 9 planetary boundaries that humanity has crossed and retaining nitrates in agricultural operations is crucial (Powlson et al., 2011; Rockstrom et al., 2009). Nitrate retention by soils is therefore important for sustainable soil management. The results from paper four showed that the treatment that received the manure application had the highest nitrate retention, followed by compost/manure mix and the compost. The treatment that received the inorganic fertiliser treatment had a net leaching. This result has a wider relevance as, for example, in Europe in 2014 inorganic fertiliser accounted for 45% of the nitrogen input in agriculture while manure accounted for 38% (EUROSTAT, 2018).

The main role of agriculture is the creation of biomass for food, feed, fibre, and fuel. Since the introduction of inorganic fertilisers and industrial agriculture, there has been a long-standing debate surrounding the agricultural approach that is optimal. Industrial agriculture has had tremendous success in increasing biomass yields that result in feeding more people. It also causes large environmental problems and its critics call for more sustainable ways to address food production. The proponents of industrial agriculture have argued that alternative methods, like organic agriculture, cannot feed the world because the yields are much lower. The yields from agricultural operation translate into economic profitability, *ceteris paribus*, this is likely the most important issue for the farmer. The results from the tomato plot experiment show that it might not necessarily be the case that operations with inorganic fertiliser yield more and are therefore more profitable. Looking at the cost of inputs and the yield, the treatment that received the mixture of organic additions fared the best, and the one receiving the inorganic fertiliser performed least well.

One of the challenges with the future intensification of agriculture is how to optimise energy inputs. Agriculture accounts for about 3 percent of global energy consumption and while being an insignificant share on the fossil fuel energy market, it is highly dependent on fossil fuels regarding machinery in land management and in the production of fertilisers and pesticides. Then there is also embedded energy to consider in machinery, buildings and other infrastructure which is needed to support agricultural operations (Woods et al., 2010). There is a large literature on energy and economic efficiency in agriculture, but I maintain that this article is one of the first studies that combines the three approaches put forward in article four (Jónsson et al., 2019): soil ES, CBA and EROI. The article looked at the energy balance of different fertilisations and how they influenced soil natural capital and associated ES, and translated these human wellbeing benefits into economic values. By linking the multiple approaches, it was possible to gain a better insight into what could constitute sustainable agricultural practice; a practice that creates a positive energy balance, maintains or improves the soil natural capital and delivers soil ES that increase human well-being and private profits. The results showed that there need not to be a mismatch between private and social benefits regarding agricultural production, as including organic additions seems to benefit both (FAO, 2015d). Organic additions improved the soil ES, brought a positive energy balance and a greater economic benefit than applying inorganic fertiliser. Win–win solutions are therefore possible in agriculture, and by using multiple analytical approaches they can be revealed.

2.3 Contribution to academic and practical knowledge

2.3.1 Academic

The overarching theme of the thesis was to show how important soil is as a type of natural capital and contribute to the growing literature on soil ES, soil sustainability indicators, and sustainable soil management. I argue that soil ecosystem services need to be factored into land-use decision making processes, and that sustainable soil and land management need to be approached with multiple sustainability indicators to ensure that ecological, energetic and economic factors are all taken into consideration. Sustainable soil management and the future intensification of agriculture will not be possible without taking into consideration the multiple aspects of soil natural capital and ensuring its viability.

This thesis has made several contributions to academic knowledge. Paper one is the first study that the authors know of that synthesises many of the valuation studies available on soil ecosystem services. The paper shows how soil ES can be classified and valued with standard economic methods. It shows that significant economic value is derived from soil ES as it provides multiple services that benefit humans and how Earth Critical Zone research is contributing to the development of soil ES. The paper is a contribution towards the quest for more informed decision-making in land-use as it provides value and valuation methods for soil ES that are usually omitted in land-use decision making. It demonstrates the need to develop a comprehensive framework for economic assessment of soils ES to better inform decision-making on various levels of governances regarding land use and soil management.

Paper two builds on paper one and shows how to link biophysical models with soil ES by providing a soil ES framework and a pilot study where the framework is applied on a watershed scale. This is one of few studies to conduct economic valuation of biophysically measured soil ES. The study introduces a framework to assess the relevance of sustaining soil functions that link together the concepts of soil natural capital, soil biophysical support functions, and soil ES with human beneficiaries and economic valuation. It shows that soil ES are important and valuable. The paper proposes a method to estimate the “carbon cost of farming” by economically valuing the carbon mineralization rate of the soil. The carbon cost of farming is a way to estimate whether agricultural methods are enhancing the capacity of the soil system to store carbon or if they are causing the system to release more carbon into the atmosphere.

Paper three contributes to the growing field of sustainability indicators by developing a process for selecting sustainability indicators for soil by combining several methods along with active stakeholder group participation. The pre-development process involved a literature review and a World Café that was concluded with a Delphi survey. The paper presented a set of core and satellite indicators for soil sustainable management which addressed the three dimensions of sustainability, covering for the first-time aspects of sustainable soil management which until now have not been included in soil indicator sets. Historically, soil sustainability indicators have mostly been developed within the environmental dimension of sustainable development, focusing on the physical, chemical or biological aspects of soils (Jonsson et al., 2016a). The other two dimensions of sustainability, the social and economic, have largely been missing. This was the first broad stakeholder participation exercise in Iceland on the issue of soil sustainability indicators. The core and satellite set of indicators of soil sustainability are a novel contribution to the literature on soil indicators as they represent a way of communicating the complex and multi-dimensional issue of soil sustainability to and between different stakeholders' groups and possibly to a wider public audience. The development process and the indicators can also contribute towards the development of indicators for the 2030 sustainable development goals, as soils play a crucial role in many SDG goals and targets.

Paper four is the first study we know of that combines three approaches, i.e. soil ES analysis, Cost-Benefit analysis (CBA) and Energy Return on Investment (EROI) while evaluating an agricultural operation that is receiving four different types of fertiliser treatment. Combining these three approaches offers a new perspective on the sustainability of an agricultural operation, as it allows for the consideration and comparison of the ecological, economic and energetic aspects of the operations. This approach can be useful in the quest for sustainable intensification solutions for agriculture which need to be more energy efficient, yield more and do so with fewer environmental costs. The results from the combined approach in the tomato plot showed that the organic additions (manure and compost) outperformed inorganic fertiliser regarding their measured ecological, economic and energetic parameters. The inorganic fertiliser fared worst of all the treatments. This result is a valuable contribution to the academic debate on the viability of organic additions versus inorganic in sustainable agriculture and agro-ecosystems, which has been dominated by discussion about whether organic methods can yield as much as industrial agricultural methods. The positive outcomes of the organic additions should at least warrant a further investigation into its potential wider application.

2.3.2 Practical

This thesis provides practical information for use by decision makers in several ways. It provides, in papers one, two and four, examples of the economic value of soil ES and undertakes pilot studies to show how these types of valuation studies are performed. By bringing these values to light and showing how to perform such valuations, decision makers have at their disposal a value range for different soil ES and a precedent for their use. Although economic values for soil ES and ecosystem services, in general, are context and location specific, decision makers can use the methods, framework and valuations presented in this thesis as a guide and examples for estimating their own soil ES valuations in their area. The systematic classification of services and the methods of economic valuation, along with the soil framework, which provides a conceptual guide to the valuation process, should empower decision makers to include the value of soil ecosystem services in land-use decision making.

The research on soil indicators for sustainable development provided a methodology for the selection of soil indicators. This process can be duplicated and applied anywhere and could provide important information about how different stakeholder groups view the essential elements of soil sustainability needing our attention. These perspectives are different, as was illustrated by the results, but due to the research approach and the methods used, the stakeholder groups reached a consensus on the core indicators to focus on regarding soil sustainability. This facilitates communication between stakeholder groups and helps to inform policy making. The indicators, both the core and satellite lists, are directly applicable to the stakeholders that took part in the process because they are experts within their fields or management level and are likely to have selected indicators that most apply to their work. Each stakeholder group can, therefore, focus on their set and they can use the core set to communicate with other stakeholder groups.

Paper four illustrated the practical use of multiple approaches or indicators when evaluating an agricultural operation. Agriculture has mainly been valued by its economic profits from biomass production, disregarding many of the negative externalities associated with it. By combining approaches, it is possible to look at an agricultural operation in a more holistic manner, from ecological, economic and energetic perspectives. These combined approaches can be applied to agro-ecosystems large and small and should provide decision makers with better information regarding the outcomes of approaches to land use management. The multi-method approach adopted in paper four can be applied anywhere where data is available.

2.4 Recommendations

2.4.1 Soil ecosystem services

Using frameworks for soil ecosystem services can be an asset for land use decision makers. As an abstract guide on the complex reality of soils, their functions and services, the framework shows how biophysical outputs are translated into economic values. It can help to highlight soil values that have historically tended to be ignored or left out of land use decision making. Using a soil ES framework (figure 1), like the one proposed in paper two, in land-use decision making would acknowledge the importance of soil natural capital for human well-being and would be a step towards a more sustainable use of the resource. Frameworks for soil ES must be localised if they are to be applied in management and decision making. The biophysical outputs and economic values need, preferably, to be obtained from primary studies based on local conditions, be these regional or national, that are representative of the soils there and local markets. Benefit transfer methods should be a secondary approach, adopted if no primary sources are available as the benefit transfer method has its limitations (Richardson et al., 2015). When the services are categorised, great care should be taken to have a clear distinction between supporting functions and final services to avoid potential double counting. Recommendations by Fu et al. (2011, p. 1) should be taken to heart: *(1) identifying the spatiotemporal scales of ecosystem services; (2) valuing the final benefits obtained from ecosystem services; (3) establishing consistent classification systems for ecosystem services; and (4) selecting valuation methods appropriate for the study context.* The selection of biophysical proxies that represent soil services is also important to consider. I would advise consultation of the established literature on the proxies most commonly used for the selected service, using databases published from the *Ecosystem Services Partnership*[10] and the like. Using established proxies allows for comparison with other studies, and it is likely that the valuation method for the proxy is reported in a database or published academic literature. When using an analysis framework, such as the soil ecosystem, it is important to know that it is easier both to collect data on and economically value some services more than others. For example, it is easier to gather data on biomass production, a provisioning service, than on intangible cultural services, such as the aesthetics of soils (Jonsson and Davidsdottir, 2016). Assessing the economic value of biomass is rather straightforward as it is generally estimated using market

[10] https://www.es-partnership.org/

pricing data. With certain cultural services linked to soils, for example valuing the beauty of a magnificent landscape is a whole different matter and requires other types of methods, e.g. non-monetary approach. We report on various economic methods for valuing soil in Jonsson and Davidsdottir (2016).

The inherent danger is that the focus of economic valuation studies will only be on the low-hanging fruit, obtaining data on ES that is easy to come by, which very often are the provisioning services that have market values. With soils, many of the services are not traded in markets so they need non-market valuation techniques, or even non-monetary valuation. Thus, when establishing a management system based on an ES framework, decision makers should aim to value the broadest possible array of services, since focusing on only a few or even one service might give an incomplete picture and thus the wrong incentives for decision makers (Costanza et al., 2017). It is also important to not only to rely on the economic valuation of ES for ecosystem management. Economic valuation of soil ES reports only on the flow value of services taken into consideration and immature support functions are likely to be left out from this analysis. Many functions of the ecological infrastructure of soil are not suitable for economic valuation and need different types of valuation and reporting to ensure their inclusion in decision making protocols. I therefore advise using the soil ES framework along with other soil sustainability indicators to ensure that the ecosystem management system has a holistic view of the soil resource.

2.4.2 Soil Sustainability indicators

The process introduced in paper three and preceding work is useful for consensus building among various stakeholders, and it is particularly suited for soils, as so many stakeholders on different levels of society influence its use and it is a key type of natural capital that needs protection. A few recommendations are proposed regarding applying the methodology and how it could be extended:

- Extensive stakeholder participation was a prerequisite for the soil indicator development process to be successful. It is important to limit the dropout rate of stakeholders as high rates might skew the results. I speculate that in our case the large number of dropouts were caused, at least in part, by the high number of questions.

- To ensure the future use of the indicators there needs to be a venue or a platform for various stakeholders to meet and discuss how indicators are developing over time. Such collaboration could guide national and local policies on sustainable soil management and might form one of the foundational approaches towards securing and maintaining soil natural capital.

2.4.3 Combination of approaches

The combinations of approaches offer a more holistic insight into agricultural operations. Combining EROI, economic efficiency and relevant soil ES can be a useful approach to estimate the sustainability of an agricultural operation, and the potential for sustainable intensification and sustainable soil management. A few caveats regarding the use of EROI are necessary:

- It is important to establish proper system boundaries for the agricultural operation. Including too small or too large a boundary might skew the results.

- It is necessary to select the proper energy equivalent component with caution as they heavily influence the outcome of the study. There might even be a disagreement in the literature on what constitute the exact values for certain energy components.

- Those who use EROI or similar methods to estimate the energy balance of agricultural systems need to ensure that the energy equivalents are internally consistent and that they select values established by the scientific literature and based on actual measurements.

2.5 Limitations and future research

In this section, the limitations of this thesis are discussed along with potential future research:

- The research methods used
- Data use and data availability
- Future research

2.5.1 The research methods used

The following research methods were used in the thesis: 1) Literature review in paper one; 2) Ecosystem Services Analysis in papers two and four; 3) Energy Return on Investment in paper four; and 4) Delphi Survey Technique in paper three.

1) Literature review

Literature review was used in paper one to summarise the knowledge on soil ecosystem services valuation in the academic literature. The way that the literature review was conducted influences the outcomes of the review. Within traditional literature reviews there are known limitations, such as being restrictive to literature that is already known to the authors or literature that is found by conducting little more than cursory searches (Mallett et al., 2012). This means that the same studies are often cited, and this can introduce a persistent bias in a literature review. A systematic review helps to reduce this, and this approach is close to the one adopted in this thesis. Systematic review adopts a set of search strategies, a predefined search string and an inclusion or exclusion criteria. Systematic reviews often have a set of rules or review principles aimed to minimise the bias of the researcher. In such reviews, all decisions used to compile the information are meant to be explicit for the reader to assess the quality of the review. In the paper some of the selection criteria were not reported. This was an unfortunate oversight and including these would have been of benefit in terms of rendering the selection process transparent. However, it should be noted that systematic reviews retain shortcomings, as they can be biased if the selection or the emphasis of the researcher of certain studies is influenced by preconceived notions. Systematic reviews also require unrestricted access to databases and peer-reviewed journals relevant to the topic. They also tend to be resource-heavy and time-consuming, especially if the literature base is vast and covers multiple fields, as is the case with ecosystem services. When I started the work on the literature review there were very few overview articles that summarised the potential value of soil ecosystem services. I therefore conducted a rather broad search and used search terms to include as many potential sources as possible. I did not restrict myself to the known literature in the field, especially since the valuation of soil ecosystem services has only commenced in the last 10 years or so and is still a subset of ecosystem services analysis that is in its early days. I do not think that the results are biased in that regard.

2) Analysis of Ecosystem Services

Ecosystems are complex networks of living and non-living components and ES analysis generalises and simplifies these networks. In doing so, there is an inherent risk of leaving something important out. ES that flow from natural capital stocks do not have clear boundaries and are directly or indirectly linked to other ES and functions. This results in trade-offs and disservices between services and functions which increase the complexity of valuation, be it monetary, non-monetary or a pluralistic approach. The ES valuation approach used in this thesis uses monetary valuation based on an anthropocentric utilitarian interpretation of value, and ecosystem species or components without an economic value might be ignored or left out of the assessment. This also applies to ecological processes and functions that do not directly benefit people, and critical ecological functions might be undermined in the pursuit to optimise one service.

In the framework and classifications of soil ES, I attempted to be as clear as possible regarding the definition of soil natural capital, soil support functions and soil ES. I also tried to be as transparent and upfront on how the proxies used to represent the services were selected and where the values for the monetary valuation were derived from. I acknowledged that there are trade-offs between services, but did not include the value, or rather the cost, of disservices in the analysis. Ecosystem disservices are functions of ecosystems that humans perceive as negative and incur costs rather than benefits, e.g. pests, floods, loss of biodiversity etc. (von Döhren and Haase, 2015). If included in the analysis, disservices could potentially have affected the outcome of the two ES papers.

3) Energy Return on Investment

According to Hall (2017), the limitations regarding the use of EROI can be attributed to three main issues: a) clarity in values, b) differences in the methods of analysis, and c) what to include within the boundaries of the study. These three issues are often interlinked.

a) Clarity in values - sometimes the value used for the energy outputs are based on nameplate values instead of actual measured outputs that reflect real conditions. This can lead to overestimation of the EROI. Hall (2017) cites examples of installed photovoltaic panels where a large difference in the nameplate values, in energy output as estimated by the producer of the photovoltaic, are used at face value instead of the actual measured outputs.

b) Difference in the methods of analysis - Hall (2017) cites two examples where respected investigators did not completely agree on how to calculate the EROI for the same fuel. Both methods used by the researcher were proper methods of analysis and his conclusion was that there might not be just one way to do the analysis.

c) The boundaries issue relates to what and what not to include in the EROI assessment. According to Hall (2017), the boundaries should include all energy costs involved in any energy generating activity. This is not always the case with EROI assessments and can lead to overly high estimations. He concludes that what should be straightforward, that is, to decide the boundaries, is in fact quite difficult, because "the inclusion of many of the costs can be more philosophical than scientific" (Hall, 2017 p. 138).

Besides the three main issues mentioned above, EROI assessments also tend to be resource intensive both in terms of money and time.

In paper four I attempted to acquire as accurate data as possible for all the energy equivalents. All energy equivalent values are based on values obtained from literature. It is of course possible that I might have chosen the wrong value for some energy equivalents because there is not complete agreement in the literature on EROI concerning what some energy equivalent values should be. I tried, though, to select the ones that were the most commonly used. I was upfront and transparent on how I calculated the EROI values and what the system boundaries were. It is possible that something was left out that should have been included within the boundaries as I was not involved in the tomato field experiment throughout its duration and therefore had to rely on accurate reports from colleagues.

4) Delphi Survey Technique

Some of the common limitations that have been reported with Delphi survey techniques are: high time commitment, heavy workload, long questionnaires, low response rates in consecutive rounds, hasty decisions and potential bias of the facilitators (Shortall et al., 2015). Some of the limitations were observed in paper three. Delphi surveys require a considerable time commitment from the participants, the surveys are often addressing complex issues, and participants need time to go through the questionnaires and supporting material. This can lead to a heavy workload if the questionnaires are long. Some of the participants in the survey (Jonsson et al., 2016a) were spending up to an hour going through the survey in the first round, indicating that it might have been too long. In the consecutive rounds (2 and 3) the response rate dropped, especially with the policy makers, and one could see that some of the participants were making hasty decisions, e.g. going through the round in a very short time. Then there is the potential bias of the facilitators, which might have influenced the outcome of some indicators, that is, indicators that were perhaps on the borderline whether they would be accepted or not. The facilitators must then make the decisions about whether to include an indicator or not, based on the ratings and the comments from the participants.

2.5.2 Data use, data availability and uncertainty

A few caveats regarding the use and collection of data during the research are worth mentioning. In paper one, the search criteria for the soil ES, that is the keywords used, were missing from the paper in the published version. For future research it would have been preferable to have included these, so when new valuations appear in the literature they can be added to the meta-analysis. The article would also have benefitted from making a clearer distinction between land and soil. Land and soils are sometimes used interchangeably in the literature and it was not always clear regarding soil ecosystem services where soil ends, and the other building blocks of land begin. Using a clear definition like Koch et al. (2013 p. 437), where soil is defined as a *"distinct living entity that [is] one of the core building blocks of land"*, would have been helpful. Land then consists of soils, rocks, rivers and vegetation.

The paper looked at 33 studies and found a large value range both between the different categories of soil ES and within them. For example, biomass production in the provisioning services showed the greatest value range of any services. The value range of biomass reflects the specificity of each valuation study because the value of the biomass produced relates to the crop type, farming methods, and the local market price. It is a reminder that benefit-transfer methods, using values between locations, should be done with great care and some degree of reservation (Richardson et al., 2015).

The shortcomings of the second paper were that there was limited data availability at the time of the field study. The SoilTrEC project, which this thesis contributes to, relied on multiple partners to provide data, and at the date of the research presented in paper 2, there was only data available for three of the services proposed in the soil ES framework. It would have been preferable to include more ES. The services for which actual measured data existed represented three of the essential services or soil functions as defined by the European Commission (European Commission, 2002, 2006): climate regulation (carbon), food production (biomass), and storing and filtering function (nitrate retention).

Another issue was the selection of proxies to represent services. The proxy must be measurable and simultaneously a good representative of the services in question. There were difficulties with the proxies available for the climate regulation service. Only the carbon mineralisation rate of different soil profiles was available. In the ES literature, the standard approach has been to use either carbon sequestration or carbon addition as proxies and base the economic valuation on those biophysical parameters (Jonsson and Davidsdottir, 2016). This data was unavailable on a watershed scale for Koiliaris CZO at the time of the study, and thus carbon mineralisation was used instead. Carbon mineralisation is the outflow of carbon from the soil, not the inflow that is the customary flow to measure. The use of carbon mineralisation as a proxy makes the study hard to compare to other valuations in similar areas. It is unknown whether the different soil profiles in Koiliaris release more carbon than they sequester, although there is an indication that the system has been stable for some time (Nikolaidis, 2016). Although this is not the customary proxy for climate regulation services (Jonsson and Davidsdottir, 2016), it represents a insight into what has been called "the carbon cost of farming", i.e. the associated carbon release due to particular land use. With more data and better models, as introduced in Advances of Agronomy special issue on SoilTrEC (Sparks, 2017), it will be possible to get a better understanding of whether the soils are acting as sinks or sources in the Koiliaris watershed, and what benefits or costs this brings.

In paper three the main issue was the dropout rate that occurred during the Delphi survey, particularly among the policy makers. In hindsight, I speculate that the questionnaire had too many questions (45-50 in each round) and that this might have discouraged the participants, as it made the process of answering the questions too long, up to an hour in some of the rounds. The actual reason for low response rates is, though, unclear as there was no post-evaluation asking the dropouts why they opted out. This dropout rate does somewhat decrease the generalisability of the indicators. Despite this, the approach proved its usefulness as it brought together, working with the same domain, different stakeholders with different sub-domains and expertise, and helped them to reach a consensus on a complex issue which resulted in the creation of a set of soil indicators for sustainable development.

There are two caveats I feel are worth mentioning in relation to paper four. First, the scale of the agricultural plot that was studied. The tomato plot was a small experimental plot (192 m^2) and it was much smaller than a standard farm operation, and it only grew one crop. It would be preferable to verify the results on a larger scale, for example, 1 ha^{-1} and with more crops. The second caveat concerns the energy equivalent used for the EROI analysis. There is disagreement in the literature on the energy equivalent value for sheep manure. Authors like Michos et al. (2012) opt for using a value based on the heating value of sheep manure. This leads to a higher value, around 20.0 – 23.5 MJ/kg in comparison to (Alipour et al., 2012; Mohammadshirazi et al., 2012; Samavatean et al., 2011), who opt for using embodied energy value, a lower number of around 0.3 MJ/kg. I use the value based on embodied energy as it is consistent with the other values used to represent embodied energy. If the heating value were to be used, this would change the outcome of the study as sheep manure (energy value 0.3 MJ/kg) only accounts for 5% of the overall energy used for the entire tomato plot treatment, with gasoline dominating at 73%. If the sheep manure value would be changed to 23.5 MJ/kg then manure would account for 81% of the overall energy expenditure for the treatments. This would change the order regarding the most preferable EROI ratio to compost (MSWC) being the most favourable one, followed by inorganic fertiliser (IF), then compost/manure mix (MSWC + M), and the least favourable would be manure. The selection of a proper energy equivalent is thus very important.

It was impossible to find an energy equivalent value for a mixture of compost and manure in the literature, so I assumed it had the same embodied energy as the manure and the compost.

This collection of research articles opens several possible future research areas regarding soil ES, soil sustainability indicators, sustainable soil management and the combination of the approaches for evaluating the sustainability of land use practices and soil management.

2.5.3 Soil ecosystem services

Paper 1 reported 33 studies on soil ES valuation for most of the service categories except one, cultural services. In the watershed and plot studies, only three services were selected for each study. To gain further insight into the value of the annual flow of soil ES it would be preferable if future studies could include at least all the essential soil services as put forward in the Soil Thematic Strategy. Future research should thus focus on broadening the scope of soil ES and try to capture all the service categories, especially cultural services, with more valuation studies on individual services. A soil ES framework needs to be integrated with biophysical modelling to link soil natural capital, soil ES and its economic consequences in terms of human wellbeing. Thus, a further integration with dynamic modelling looking at different land use and climate change scenarios is a worthy pursuit. The soil ES framework, along with the SoilTrEC biophysical models, could be integrated with the special software packages available for ES mapping and valuation, like InVEST (Daily et al., 2009) and ARIES (Villa et al., 2009), which would allow it to be used more readily by land-use decision makers.

The SoilTrEC project created a large body of research data, as represented in a special edition in Advances in Agronomy featuring the SoilTrEC research project exclusively (Sparks, 2017). The core research took place in four Critical Zone Observatories (CZO) around Europe, with participation from CZO's in the United States and China. These CZO's modelled the whole "lifecycle" of soil from creation to erosion (Banwart et al., 2012). Due to time constraints and the fact that usable data from all the CZO's for analysis was not available when this research took place, it was only possible to analyse a part of the data that is now available after the project has been fully completed. Only a few services from one of the CZO's within the SoilTrEC project were used as they were available at the time of this research. As a result, ample opportunity exists to both study and compare soil ES between them. The SoilTrEC project was set up as a transect belt of European Critical Zones from north to south, and it gave an overview of the complete soil lifecycle from formation to erosion. It would be interesting in comparing the soil ES between the CZO's on different latitudes and in various ecosystems.

Furthermore, it would be interesting to evaluate the value of soil ES and how they change along different timespans, e.g. over multiple years and during the whole life-cycle of soil. Many of the soil ES and its functions are likely to receive a low value when they are underdeveloped or are intermediate in providing supporting services. An economic analysis might make some of them seem worthless as they are providing indirect economic benefits to humans. This depends, though, on the service selected and the economic valuation method, but this might be the outcome if an assessment is not conducted along the soil lifecycle. This research could help to point out the flaws in approaching the natural system with only one type of valuation tool, economic valuation in this case.

2.5.4 Soil sustainability indicators

There is much potential for future research indicators for sustainable soil management. Paper three presented data from stakeholders mainly in Iceland along with scientists from other countries. The outcome of the study is relevant for Iceland but cannot be generalised to other countries as soil conditions differ between places. This type of study could be carried out on a European-level on a country-by-country basis with relevant stakeholder groups. The results might be that different opinions exist within Europe on what soil indicators are important when assessing soil sustainability. The approach indicator development in this thesis to might bring to light the commonalities and differences that exist between stakeholder groups, both within and between countries. Establishing core and satellite national indicators for soil resources for every country could then potentially feed into a European-wide or global core and satellite indicators. This type of indicator set would be an asset for a common EU soil policy as it would help with communication and forming recommendations specific and non-specific to countries.

2.5.5 Future research options for sustainable soil management

There are many potential research options for sustainable soil management using the multiple approaches presented in paper four. With ample time and money, I would look at replicating these same results on a larger scale with a broader set of soil ES, with more crops and preferably longer time series. I would like to compare a fully organic system (no-chemicals) and a conventional and add livestock into the mix. The study would be at the whole farm level. The study would have to be at least 1 ha^{-1} so the results could be scaled more easily. I would like to add more soil ES to the analysis, especially the cultural services which tend to be left out. If possible, I would like to incorporate the biophysical models from TUC-HER into a software package like InVEST, so that soil services would always be taken into consideration when doing an ES Analysis for an area.

2.6 Conclusion

This thesis shows that soils are an important type of natural capital that provide multiple services and benefits to humans and has included economic valuation of these. Regrettably, the value of soil has historically not been included in land-use decision making, which has perhaps contributed to cases of unsustainable management. Soil ES can be valued economically as I show in this thesis, and their value must be recognised for sustainable soil management and sustainable agriculture, and these should be included in decision-making frameworks regarding land use. Soil security and soil sustainability are the basis for obtaining many of the UN 2030 Sustainable Development Goals and soil is central to food, feed and fibre production, and the stability of climate. Sustainable soil use is also central for future sustainable agricultural intensification, given the anticipated need to feed many more people this century. Soil sustainability indicators, as presented in this thesis, can play an important role in the pursuit of sustainable soil management. The way that sustainability indicators are developed needs to include all relevant stakeholders to facilitate the cooperation needed to safeguard soils and communicate their immense importance to a wider audience. Agriculture, as the major global land-use, has a tremendous influence on the soil, depending on its operations. It can degrade the soil and soil-related ES or enhance the vitality of the soil if based on more sustainable methods. In this thesis, I show that combining assessment approaches and indicators offers deeper, multi-dimensional and holistic insights into agricultural operations, which in turn are likely to be more sustainable. The fate of human civilisations throughout history has been intrinsically tied to the state of the soil resource. This thesis provides methods and tools that assist in land-use decision making and will hopefully contribute to a more sustainable use of soil ES and the protection of soil natural capital.

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Review

Classification and valuation of soil ecosystem services

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ABSTRACT

Soil ecosystem services (ES) provide multiple benefits to humans but to date no consensus has formed on a comprehensive framework for their classification and economic valuation, and therefore a systematic approach has not been developed to evaluate their importance. We present a literature review of soil ES and functions, the economic methods that have been used to value them, and values that have been derived from various studies under different type of land use. We illustrate how recent developments in the field of Earth Critical Zone research are contributing to the development of soil ES valuation techniques within the ES framework. These developments have further underpinned the necessity to include the multiple aspects of soils in ES frameworks because of their fundamental roles in ecosystem functionality and vitality. We provide examples on how soil ES can be classified and valued using standard economic methods and established analysis frameworks. We show how significant economic value is derived from soil ES and thus highlight the economic losses associated with soil degradation. Furthermore, we also demonstrate the need to develop a comprehensive framework for the economic assessment of soil ecosystem services in order to better inform decision-making at various levels of governance regarding land use and management.

Contents

1. Introduction

Soil is one of the more species-rich habitats of terrestrial ecosystems and its functions include biomass production, maintaining nutrient balance, chemical recycling and water storage to name a few (Blum, 2005).

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Soil is a very slow forming resource, and similarly to other habitats and ecosystems, it is coming under increasing pressures due to anthropocentric activities. The near exclusion of the importance and value of ecosystems and resources, such as soils, in economic decision-making is exacerbating degradation pressures. In order to resolve this, the concepts of natural capital and ecosystem services (ES) have been widely adopted by academics, NGOs and governments and have been gaining in momentum and acceptance (despite disagreement and confusion of terminology — see Table 1 for our definitions of key terms) for the last two decades (Gomez-Baggethun et al., 2010).

Even though many studies (Kumar, 2010) have been conducted valuing ES, services derived from soils have been partially or entirely omitted (Dominati et al., 2010a). We aim to communicate the importance of soil ES by reviewing the existing literature to identify the various ES that soils provide, and proceed to analyses of how concepts related to soil ES and soil natural capital have developed and how services from soils can be economically evaluated. Relying on the widely-adopted Millennium Ecosystem Assessment (MA) framework for ES, we report on various economic valuation methods for soil ES and the value of soil ES in different types of land uses.

2. Ecosystem services

2.1. Ecosystem service classification schemes

The environment or natural capital can be considered a stock, which similar to man-made capital yields, through its multiple functions, a flow of goods and services into the future (Costanza et al., 1997). Collectively the various services from natural capital have been referred to as 'ecosystem services' and are defined as the benefits people obtain from the ecosphere and its ecosystems (MEA, 2005). When assessing the economic value of ES, the services provided by an ecosystem must be identified, classified, and then valued economically (Kumar, 2010; MEA, 2005). Various classification schemes for ES have been devised such as those by De Groot et al. (2002), the Millennium Ecosystem Assessment (MEA, 2005) and The Economics of Ecosystem and Biodiversity (TEEB) (Kumar, 2010).

Three of the most common classification schemes divide ES into four categories: production/provisioning services, regulating services, habitat/supporting services and information/cultural services. These categories are broadly similar although there are clear differences, particularly concerning what are regarded as supporting or habitat services. The different categories can be explained in general terms through the MA definitions for each service category (see Table 2).

TEEB, the latest addition to the classification frameworks, builds on previous frameworks (e.g. Costanza et al., 1997; de Groot et al., 2002; MEA, 2005), and thus there are parallels between the three classification schemes as can be observed at the different stages of the ES concept development (Gomez-Baggethun et al., 2010) (see Table 3). The main difference between the TEEB and MA classification schemes is the omission of the supporting service category in TEEB and adoption of what is called the habitat category. In the TEEB framework, supporting services are seen as a subset of ecological processes rather than a specific category. TEEB nevertheless identifies habitat as a special category to highlight the importance of ecosystems as habitat providers for migratory species and biodiversity (Kumar, 2010).

Table 2
MA categories (MEA, 2005).

Provisioning services	Products people obtain from ecosystems, such as food, fuel, fibre, fresh water.
Regulating services	The benefits people obtain from the regulation of ecosystem processes, including air quality maintenance, climate regulation, erosion control, regulation of human diseases, and water purification.
Cultural services	The non-material benefits people obtain from ecosystems through spiritual enrichment, cognitive development, reflection, recreation, and aesthetic experiences.
Supporting functions	Services that are necessary for the production of all other ESs, such as primary production, production of oxygen, and soil formation.

In light of increasing pressures on soil natural capital stocks, which has impact on the flow of ES, it is vital to have a comprehensive economic valuation and accounting system to properly incorporate ES into decision-making (Costanza et al., 1997; de Groot et al., 2002). It is necessary to account for as many ecosystems and biodiversity aspects as possible, including the crucial roles of soils in delivering ES. This has not been the case with many of the frameworks, as most of them focus predominantly on what is happening in the above-ground ecosystems (Dominati et al., 2010a), leaving the important and complex soil ecosystems underground often partially or completely omitted from economic assessments. For example, in the classification of ecosystem types in TEEB, soil is excluded (de Groot, 2010). There is a growing need to incorporate the part played by soils into ES frameworks and acknowledge the important role that the pedosphere has in the Earth's ecosphere (Robinson et al., 2012). Critical Zone science might be one approach that could be helpful in highlighting the importance of soil functions in the Earth's ecosphere and thus open the possibility of integrating soil ES into established ES frameworks.

2.2. Earth's Critical Zone

In 2001 The National Resources Council (NRC, 2001) emphasized the importance of developing a better understanding of the Critical Zone to assess the impact of human activities on the Earth and to adapt to their consequences: "Earth's Critical Zone includes the land surface and its canopy of vegetation, rivers, lakes, shallow seas, and it extends through the Pedosphere, unsaturated vadose zone, and saturated groundwater zone. Interactions at this interface between the solid Earth and its fluid envelopes determine the availability of nearly every life-sustaining resource" (NRC, 2001, p. 31). Soil and its functions are an important component of

Table 1
Definitions of key terms.

Key terms	Definitions
Ecological infrastructure of soil	Soil natural capital, its properties; and soil support functions that underlie other ecosystem services and are in a dynamic relationship with soil processes and soil natural capital.
Ecosystem services	The benefits that people obtain from the ecosphere. Ecosystem goods and services are synonymous with ecosystem services.
Final goods and services	Benefits that flow from the Ecological Infrastructure (cultural, regulating and provisioning services)
Natural capital	Stocks of natural resources found on earth yielding a flow of valuable ecosystem goods or service into the future.
Soil ecosystem services	The flow from the Ecological infrastructure of soil. Soil ecosystem services refers to both ecosystem goods and services from soil.
Soil natural capital	Soil stocks on Earth which yield a flow of goods and services. Soil natural capital is characterized by soil properties.
Soil processes	Any change or reaction which occurs within soils, either physical, chemical or biological. The complex interactions among the biotic and abiotic elements of the soil.
Soil properties	The physical, chemical and biological characteristics of a soil. They can be inherent or manageable.
Soil support functions	A subset of interactions between the natural capital and soil processes that are required for the production of final soil ecosystem services and goods that satisfy human needs. The support functions are intermediate steps in the stock and flow chain and are therefore neither consumed directly nor valued economically.

Table 3
Comparison of different ES frameworks.

Service/functions	Services	De Groot (2002)	MA (2005)	TEEB (2010)
Habitat/support service	Refugium functions	x		
	Nursery	x		
	Nutrient cycling		x	
	Soil formation		x	
	Primary production		x	
	Maintenance of life of migratory species			x
	Maintenance of genetic diversity			x
Regulation function/service	Gas regulation	x		
	Climate regulation	x	x	x
	Disturbance prevention	x	x	x
	Water regulation	x		x
	Water supply	x		
	Soil retention	x		
	Soil formation	x		
	Nutrient regulation	x		
	Waste treatment	x		x
	Pollination	x		x
	Biological control	x	x	x
	Water purification		x	
	Air quality regulation			x
	Erosion prevention			x
	Maintenance of soil facility			x
Production function/service	Food	x	x	x
	Raw material	x		x
	Genetic resources	x		x
	Medicinal resources	x		
	Ornamental resources	x		x
	Fresh water		x	x
	Wood and fibre		x	
	Fuel		x	
Information/ cultural service	Aesthetic information	x	x	x
	Recreation	x	x	x
	Cultural and artistic information	x		x
	Spiritual and historic information	x	x	x
	Science and education	x	x	x

the Critical Zone. The pedosphere is the thin semi-permeable membrane at the Earth's surface that serves as an interface between the solid and fluid envelopes. These envelopes are the atmosphere, the hydrosphere, the biosphere, and the lithosphere. It is at this juncture between the spheres that soil forms. Soil, in combination with other envelopes, plays a key role in maintaining the many ES within the Critical Zone. The concept of the Critical Zone has though a broader spatial and temporal scale and thus a larger focus than the concept of ES. In Field et al. (2014) the authors compare (Table 4) the concept of ES to what they call the Critical Zone service perspective and emphasize the difference between these perspectives in both temporal and spatial terms. In particular, the Critical Zone service perspective includes "the full extent of the vertical weathering profile […], allowing improved integration of processes the determine constraints that limit provision of ecosystem services" (Field et al., 2014, p. 5). Soil ES are therefore only a part, albeit a crucial one, of the Critical Zone perspective.

The Earth Critical Zone is coming under increasing pressure due to human activity (Rockstrom et al., 2009) and several degradation

Table 4
Ecosystem services and Critical Zone services (Field et al., 2014).

Ecosystem services	Bio-focus, especially including biodiversity, time scale commonly day to years, more focus on renewable natural resources.
Critical Zone services	Geo-focus, especially including soils, regolith, time scales commonly minutes to millennia, more on non-renewable natural resources

pressures influence the state of the pedosphere within the Critical Zone. The EU which has recognized the vitality of soils has defined a few key environmental, economic and social functions that are vital for life: food and other biomass production; storing, filtering and transformation of minerals, organic matter, water and energy, and diverse chemical substances; habitat and gene pool which performs essential ecological functions; provision of a beneficial physical and cultural environment for mankind; and a source of raw materials (European Commission, 2002, 2006). According to the EU, these essential soil functions are under several degradation pressures due to human activities. These degradation pressures include: erosion; decline in organic matter; local and diffuse contamination; sealing; compaction; decline in biodiversity; salinization; floods and landslides (European Commission, 2006).

2.3. SoilTrEC

In order to address the degradation and soil losses in the Earth's Critical Zone, a scientific connection must be established between these degradation pressures and the resulting state of soils. A few Critical Zone projects seek to establish this link. One of them is Soil Transformation in European Catchments (SoilTrEC), which we are members of. SoilTrEC is a European Union Seventh Framework Programme (FP7) funded research project focused on addressing certain knowledge gaps regarding important soil ES and functions as listed by the EU soil thematic strategy (European Commission, 2006). The research in SoilTrEC is driven by the hypothesis that soil processes can be described along a life cycle of soil development – from pedogenenis to erosion – and that the processes and functions of soil can be quantified during this life cycle. The results from this research will provide a quantification of key flows of material and energy at catchment scale that contribute to the economic goods and services humans derive from soils, and these flows can then be classified and valued economically (Banwart et al., 2012).

2.4. Soil and Earth's Critical Zone

Although not often recognized as important, soils are complex, dynamic ecosystems that sustain physical processes and chemical transformations that are vital to terrestrial life, thus making the health of the soil and its biodiversity vitally important to humans. Soils serve as the main bases for biodiversity on Earth, as soils contain more species, both in number and quantity, than all other above ground biota put together (Blum, 2005). A handful of soil may contain more than 10 billion bacteria containing thousands of different species (Torsvik and Ovreas, 2002), and the activities of Micro-, Meso- and Marco-fauna are essential foundations for biodiversity in general (Artz et al., 2010; Wolters et al., 2000). In addition, the genetic diversity of soil is a source of many current and potential future pharmaceuticals and medical treatments (D'Costa et al., 2006; Minton, 2003; Turbé et al., 2010). Soils play an important role in the Earth's water cycle. They absorb, filter and store water, attenuating water flows. Soils provide nutrients, water and a physical environment conducive to terrestrial biomass production. Biomass production in turn is the foundation for economic activities in various agro-ecosystems, for example agriculture and forestry. Humans also consume soil animals directly (Decaens et al., 2006) and even soil itself (Abrahams, 2012). Indeed, the importance of soils in human food production systems is shown by that over 99% of all food (calories) consumed by humans comes from land-based ecosystems (Pimentel, 2000). Soils degrade and decompose organic matter and when functioning properly, they also have the capacity to degrade or reduce toxic or hazardous compounds (Andrews et al., 2004; Dominati et al., 2010a). Soils play an important role in climate regulation, helping to regulate global temperatures and precipitation levels, particularly through the sequestration of atmospheric carbon dioxide (CO_2) and its storage within major carbon sinks (Haygarth and Ritz, 2009; Lavelle et al., 2006; Turbé et al., 2010; Wall, 2004). About 25% of atmospheric carbon dioxide comes from biological oxidation reactions in the Pedosphere, which contains twice as much carbon as the atmosphere and up to three times the carbon found in all vegetation (Blum, 2005; NRC, 2001; Wilding and Lin, 2006). When used unsustainably, soils can be a major source of CO_2 emissions and the decline of carbon stored in soil has had impacts on the concentration of CO_2 in the atmosphere (UNEP, 2012). Soils play an important part in the regulation of atmospheric chemical composition, for example regulating (CO_2)/oxygen (O_2) balance, ultraviolet B (UVB) protection and sulphur oxide (SO_x) levels (Costanza et al., 1997; Haygarth and Ritz, 2009). Soil colours and its varied formations have, throughout the ages, provided inspiration to artists. Soil has played a major part in worldview and religious beliefs held by different societies thus playing a role in their cultural heritage. In the Mayan culture where soil was understood as a gift from the ancestors that must be reciprocated with human blood (Wells and Mihok, 2009) and in the Judeo-Christian religion, God created man from soil. The spiritual aspects of nature and its ecosystems are a well-known phenomenon (see Gould et al., 2014). Soil is an important foundation in different educational subjects, for instance the agricultural, geological, geographical and biological sciences. Soils provide a platform for entertainment and leisure activities (Decaens et al., 2006) and are an archaeological archive, containing one of the main sources of information on the history of man before the invention of writing (European Soil Bureau Network, 2005). In addition, soil and sediments act as geological archives giving insight into the climate and environmental conditions of the past. Given the multiple and extremely diverse services of soils, they are of critical importance for human-wellbeing and necessity to understand their role and function in the Earth's Critical Zone.

3. Frameworks for soil ES

Soil ES are defined here as the benefits that people derive from soils (Dominati et al., 2010a). Despite that various frameworks for classifying ES have been put forward and the evident importance of soil health to human welfare, soil's ES are overlooked and to date there is no agreed upon framework for identifying, classifying or valuing them (Robinson and Lebron, 2010). There have been a few attempts to identify and classify soil ES (Table 2). The first study that we know of appeared in Daily et al. (1997b) where the authors classified six services:

- Buffering and moderation of the hydrological cycle;
- Physical support of plants;
- Retention and delivery of nutrients to plants;
- Disposal of wastes and dead organic matter;
- Renewal of soil fertility; and
- Regulation of major element cycles.

This was an important first attempt but missing were the categorizations that developed later (in the MA or TEEB), where services are bracketed based on the type of services (supporting, provisional, regulating, and cultural) they provide. Further identification and classification of soil ES have been developed both in soil specific (Andrews et al., 2004; Barrios, 2007; Dominati et al., 2010a; Faber and van Wensem, 2012; Haygarth and Ritz, 2009; Lavelle et al., 2006; Palm et al., 2007; Smit et al., 2012; Wall, 2004; Warr and Ayres, 2004; Weber, 2007) and agroecosystem contexts (Porter et al., 2009; Sandhu et al., 2010a, 2010b; Swinton et al., 2007; Zhang et al., 2007). Table 5 sets out a further comparison based on the authors' categorization.

It is apparent that most of these frameworks are missing at least one of the following aspects: 1) the connection between soil natural capital, soil functions and soil ES; 2) categorization of the different services; 3) the potential beneficiaries of the soil ES; and 4) how to value economically the benefits.

Some of the frameworks fulfil up to three of the criteria (Daily et al., 1997b; Dominati et al., 2010a; Haygarth and Ritz, 2009), but all of them are missing one or more elements from their framework, and thus fall

Table 5
Frameworks for soil ESs.

Soil ESs	Daily et al. (1997a, 1997b)	Andrews et al. (2004)	Wall (2004)	Warr and Ayres (2004)	Lavelle et al. (2006)	Swinton et al. (2007)	Barrios (2007)	Weber (2007)	Palm et al. (2007)	Zhang et al. (2007)	Haygarth and Ritz (2009)	Porter et al. (2009)	Dominati et al. (2010a)	Sandhu et al. (2010a)	Faber and van Wensem (2012)	Smit et al. (2012)
Biodiversity pool		x		x				x		x	x	x	x		x	x
Nutrient cycling	x	x	x		x	x	x		x	x	x	x	x	x	x	x
Soil formation			x		x				x	x	x			x	x	x
Water cycling		x	x	x			x	x			x	x	x			x
Biolog. control of pests and diseases			x		x	x	x			x		x	x	x	x	x
Climate regulation	x		x		x	x	x	x	x	x	x	x	x	x		x
Hydrological control	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x
Recycling. of wastes and detoxification	x		x						x			x	x			x
Filtering of nutrients and contaminants	x	x	x	x					x				x			
Biomass production			x	x		x		x	x		x	x	x	x		
Clean water provision			x		x	x			x	x	x			x	x	
Raw materials				x				x			x					
Physical environment	x	x		x				x	x		x		x		x	
Heritage			x			x		x			x					
Recreation			x	x		x		x			x	x		x		
Cognitive			x	x		x					x	x	x	x	x	

short to value soil ES in a holistic manner. The frameworks listed above were created with various goals in mind, for example looking at management scenarios (Andrews et al., 2004), the importance of soil fauna for soil ES (Barrios, 2007; Lavelle et al., 2006), and the roles of ES in the context of agricultural production (Sandhu et al., 2010a, 2010b; Swinton et al., 2007; Zhang et al., 2007). Sometimes the categorization was based on the specific goal at hand and not necessarily put forward as a comprehensive soil framework, but rather explaining the specific soil ES relevant to their individual context. This context specificity is necessary when applying a soil framework to an area but before that can be done the general methodological approach has to be established. In our opinion, the most comprehensive framework to date is the approach proposed by Dominati et al. (2010a). There the authors tie together the concept of the natural capital resource of soil and the ES that flow from its stock. The authors state that the existing literature on ES tends to: "focus exclusively on the ecosystem services rather than holistically linking these services to the natural capital base from which they arise"(Dominati et al., 2010a, p. 2). Their framework builds on the MA (2005) and its categories. They define soil natural capital as a "stock of natural assets yielding a flow of valuable ecosystem goods or services into the future", building on Costanza and Daly's (1992, p. 38) definition of natural capital, and natural capital of soil is then characterized by its properties. Soil properties can be either manageable or inherent (Dominati et al., 2010a) and, like other capital, soil is formed, maintained and degraded. Works by Robinson et al. (2009, 2012, 2013); Robinson and Lebron (2010) along with the work of Dominati et al. (2010a, 2010c), have further linked the concept of soil ES and natural capital into one framework, thus moving towards a more holistic approach to soil management, i.e. how soil natural capital influences the provision of soil ES and how human pressure on soil and management decisions influence the dynamic equilibrium of soil stocks. Notably, the original, the framework omits how to value the services using economic techniques but Dominati et al. (2014a, 2014b) have since published research based on this framework and using economic techniques to value soil ES. We rely on the framework proposed by Dominati et al. (2010a) to organize the review of soil ES, and assessments of their economic value. One of the reasons for relying on the framework put forward by Dominati et al. (2010a) concerns their use of the MA framework as a basis, as "the M[E]A seeks to distinguish supporting ecosystem services, which are important for maintaining ecosystems, from those that provide direct benefits to people" (Alcamo et al., 2003). By adhering to the MA framework, the danger of double counting is avoided which some later frameworks tend to be more prone to. For example, in TEEB some of the services which are classified as supporting services in the MA (soil formation and nutrient cycling) are instead classified as regulating services, thus mixing into one category what are considered to be both intermediate and final services.

4. Soil ES and economic valuation studies

4.1. Value in economics

In this paper, examples of how soil ES can be classified and valued economically are highlighted with the purpose of integrating the economic value of soil ES into land-use decision-making processes. A *Preferences-based approach* to valuation is therefore opted for as opposed to *Biophysical approach* (for a detailed discussion on valuation of ES see chapter 5 of Kumar, 2010).

In *Preferences-based approaches*, which are common in economics, valuation is always based on anthropocentric values, i.e. the value to humans. Such values are split into two categories: use and non-use values (Fig. 1). Use values are divided into three categories: direct use values; indirect use values; and option values. Direct use values can be consumptive or non-consumptive. Examples of consumptive use values include the value of agricultural products from provisioning services and non-consumptive values include the value linked to the recreational experience that are a cultural service. Indirect use values include values linked to regulation services like flood mitigation or biological control of pests and diseases.

Non-use values, also referred to as "passive use" values, are values that are detach from the use of the resource, but can include the option to reserve the ability to consume its services in the future (option value), or the option to reserve the ability of future generations to use the service in the future (bequest value). Beyond these examples of future use is existence value, which denotes the non-use value that people place on a resource simply knowing that it exists, even if they will never see or use it.

Economists choose between various valuation tools they have at their disposal to place an estimated monetary value on the identified

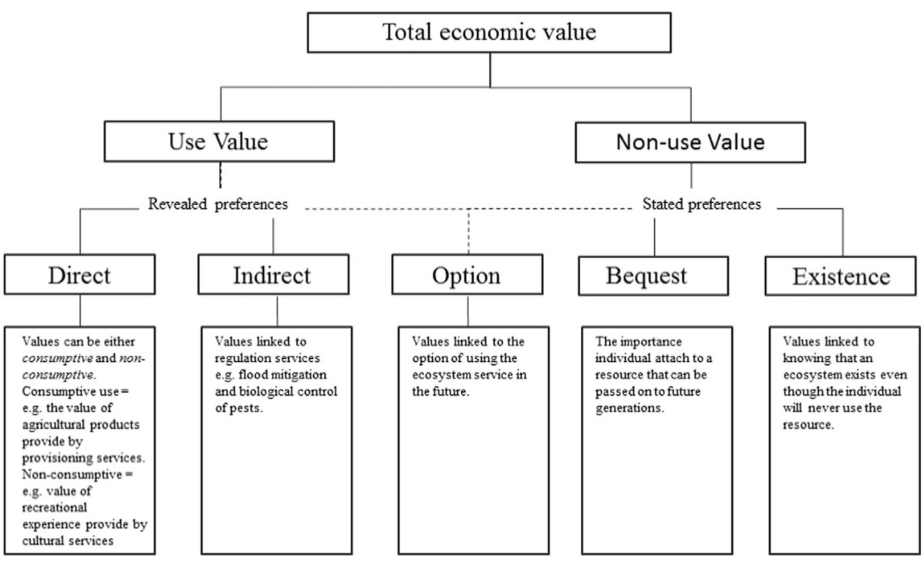

Fig. 1. Total economic value.

services, and this has been done for various soil ES (see Table 3). Total economic value is the sum of all the relevant use and non-use values for all the various services a particular ecosystem provides (Freeman, 2003; Hanley et al., 2006).

4.2. Valuation tools

Numerous economic valuation techniques have been developed and are categorized according to whether they rely on actual consumer behaviour (revealed preferences) or stated consumer behaviour (stated preferences). Revealed preference techniques look at actual decisions people make in reaction to specific ES or to changes in environmental quality, and are therefore used when assessing use values. Stated preference techniques elicit values through survey methods and capture both non-use values as well as use value. A short review of the main methods within each category follows below.

The most common methods for use values are: market prices, net factor method, cost based methods, travel cost method and hedonic pricing. The market pricing method estimates the economic value of ecosystem goods or services that are sold in markets. Most often this is used to obtain the value of provisioning services, since commodities are usually sold in markets, for example agricultural produce. The approach can value either changes in quantity or in the quality of a good or service. In the absence of distortions such as taxes or subsidies, market prices can be a good indicator of the value of ES (King and Mazzotta, 2012; Kumar et al., 2010). The net factor is based on the value of an ecosystem service sourced from its contribution to the output derived from its use in production, such as in recreational services or tourism. For example, how much of the added value generated by tourism is attributable to the existence of a particular ecosystem, as opposed to other inputs such as produced capital, material inputs, and labour. One example where this type of method could be applied is the valuation of water quality. Water quality affects fish production in river systems, or the costs of purifying municipal drinking water. Thus, the economic benefits of improved water quality can be measured by the increased revenues from growth in the spawning stock of fish or the decreased costs of providing clean drinking water (Freeman, 2003).

Cost-based methods rely on actual costs associated with the avoided cost, damage cost or replacement cost. These methods involve estimating the value of ES based on either the costs of avoiding damages due to lost services, the cost incurred because of damages to the ES, or the cost of replacing ES by providing substitute services (Kumar et al., 2010). These methods do not provide strict measures of economic values, which are based on individual and societal willingness to pay[1] for a product or service. Instead, they assume that the costs of avoiding damages or replacing ecosystems or their services provide useful estimates of the willingness to pay for these ecosystems or services. This is based on the assumption that, if people incur costs to avoid damages caused by lost ES, or to replace the services of ecosystems, then those services must be worth at least what people paid to avoid the damage or replace the services. In practice cost-based methods are most appropriately applied where supporting and regulating services are valued.

The travel cost method is based on using travel expenses as a proxy for the price of visiting outdoor recreational sites (Fletcher et al., 1990). The underlying rationale is that travel is a complementary good to recreation. A statistical relationship between the observed visits and the cost of visiting is used to approximate the demand curve[2] for recreation. Once a demand curve has been derived the value to the consumer can be assessed. This method has been widely used to estimate the value of the benefits of various recreational activities (Bowes et al., 1989).

[1] WTP = the maximum amount that a person is willing to pay for a good she does not have.

[2] A graph showing the relationship between the price of a good and the amount of demand for it at different prices.

The hedonic pricing method seeks to explain the value of a commodity as a bundle of valuable characteristics (Lancaster, 1966), with one of them being various environmental amenities. A classic example of such a commodity is real estate as the price of real estate depends on size, location, as well as various environmental amenities such as view, noise level, air quality, and proximity to green areas. The assessment method then illustrates to what extent the environmental qualities affect the price.

Non-use values are normally assessed by using survey based techniques such as contingent valuation (CV). CV is the most common valuation method of obtaining stated preferences for non-use values. A CV study is conducted by asking a sample of the affected population questions on well-specified hypothetical scenarios to identify the preferences[3] of each respondent to a particular environmental amenity or an ecosystem. Two key parts of any CV study are the description of the scenario in order to convey the hypothetically planned change in environmental quality, and the question eliciting the individual respondent's willingness to pay (WTP) or willingness to accept compensation[4] (WTA) for a change in quality (see e.g. Bateman and Willis, 1999; Hanley and Spash, 1993; Mitchell and Carson, 1989).

4.3. Valuing soil ES: reviewing values from the literature

When assessing ES in a particular setting, the following approach is applied. First the area is described (e.g. what is the dominant land use, climate conditions, land management practice, land cover, soil type, soil health etc.) to give context for the valuation and the soil services from the area. Second, a stakeholder assessment is performed to identify the beneficiaries of the services rendered from the area. The beneficiaries of ES can be at different scales (e.g. local, regional and global. For more information about beneficiaries and different scales see chapter 5 in Kumar (2010)). Third, the area is analysed with respect to the ES that are present, using the best information and tools available, and the ES that are selected for economic valuation are then quantified in biophysical terms using proxies. These proxies determine the economic value of each service for that specific land use, soil condition and management. The values for all the services constitute the total economic value of services in the area. The total economic value is reported in separated ES categories. This all of course depends on data availability, regarding both the biophysical and economic data. Supporting functions are excluded from the total economic value as they underpin other service categories build on and to value them would involve double counting. In spite of the many frameworks mentioned there is unfortunately to date, no agreed upon framework for identifying, classifying or economically valuing soil ES (Robinson and Lebron, 2010). There is though, a diverse literature on the economic valuation of various soil ES and functions. In the review of available literature on soil ES, 33 per reviewed studies, books and reports were looked at, comprising a total 86 soil ES (many of the studies valuated more than one soil ES) and categorized these into *supporting functions*, *regulating services*, *provisioning services* and *cultural services* using a modified MA framework (see Table 3). Among some of the parameters identified for the services were: category, service/function, method spatial scale, land use classification according LUCAS (EUROSTAT, 2016)[5] and MAES ecosystem type level 2 (EEA, 2016)[6] (see Tables 1–3 in Appendix A). Of those 86 services, 36 were classified as regulating services, 32 as provisioning services, 17 as supporting functions, and 1 as cultural services. The spatial scale registered for the services was global, country, province/region, regional, municipality and local (plot scale). Services registered for a country scale were most abundant or 30, followed by 23 province/region scale, 18 on

[3] What people want.

[4] WTA = The minimum amount that a person is willing to receive to give up a good in her possession.

[5] A classification system used by EUROSTAT on the state and dynamics of changes in land use and cover in European Union.

[6] An ecosystem type classification system used by MAES (Mapping and Assessment of Ecosystems and their services) analytical framework.

local (plot), 13 on regional and 5 on municipality level. The dominant LUCAS land use category was agriculture taking place in either in a cropland or grassland ecosystem according to the MAES categories. As can be seen from this short comparison, studies on the cultural value are almost non-existent and soil ES studies on a city scale are few.

The values for soil ES were standardized into a common unit of international dollars. The international dollar (id$) is a hypothetical unit of currency used to standardize monetary value across countries by correcting to the same purchasing power that the U.S. dollar had at any given time. First, all values were converted into 2012 values in local currency, and then the local currency was transformed into the US dollar using the currency exchange for 2012, with this value then corrected to the same purchasing power that the U.S. dollar had using the World Bank's PPP conversion factors[7]. By using international dollars it is possible to directly compare the value of soil ES between countries (de Groot et al., 2012). The official exchange rates and PPP conversion factors were obtained from the 2013 World Bank Development Indicators (World Bank, 2013). Following are examples of different soil ES quantification, economic methods used and the value derived from the studies.

4.3.1. Support functions

Support functions underpin other services and as such are not to be economically evaluated as this would involve double-counting. Nevertheless, there are economic valuations studies in the literature of what could be considered as soil support functions and we have therefore opted to include them as they are interesting in the context of illustrating the immense importance and value of soil natural capital and soil ES and functions.

The economic valuation literature concentrates on the role of soil in four different functions: *biodiversity pool*; *nutrient cycling*; *soil formation*; and *water cycling*.

4.3.1.1. Biodiversity pool. Soils are probably the most species rich habitats of terrestrial ecosystems (Science, 2004; Weber, 2007) providing a source of habitat to millions of species, enabling them to function and develop. This reservoir of biodiversity is important and many of these species serve as an essential part of the functional diversity and resilience of the soil. Van der Putten et al. (2004) reviewed the value of soils, based on Pimentel et al. (1997), as a biodiversity pool and illustrated an estimated annual value of id$2.1 trillion (Table 6).

4.3.1.2. Nutrient cycling. Nutrient cycling maintains soil fertility and is a process whereby chemical elements are moved through the biotic and abiotic parts of the soil. Microorganisms are key moderators of this service. This cycling is the foundation for many other processes of the soil (Brussaard et al., 2007; Dominati et al., 2010a; Zhang et al., 2007). When assessing the value of nutrient cycling, most authors have used replacement cost, relying on the market price of restoring lost nutrients (see e.g. Drechsel et al., 2004; Pimentel et al., 1995; Pimentel et al., 1997; San and Rapera, 2010). The values range from id$24 to id$180 kg/ha^{-1}/yr^{-1} (Table 6).

4.3.1.3. Soil formation. The chemical, physical and biological activities that lead to the formation of soil over time by weathering of rocks and minerals. This process is affected by relief (terrain), parent material, climate and geography (Brantley et al., 2007). The value of soil formation has been assessed based on the price of topsoil, using market prices (Pimentel et al., 1997; Sandhu et al., 2008), with prices ranging from id$18 to id$28 ton/ha^{-1}/yr^{-1} (Table 6).

4.3.1.4. Water cycling. Water cycling involves the physical process of water moving through the soil. This movement influences the geo-, bio- and chemical processes in the soil and affects the development of its biodiversity and functions (Dominati et al., 2010a). The value of the water cycling function has been assessed based on the replacement cost of the service with irrigation, illustrating a value between id$62 and id$126 ha^{-1}/yr^{-1} (Sandhu et al., 2008) (Table 6).

4.3.2. Regulating services

The regulating services of soil natural capital that have been valued include: *biological control of pests and diseases*; *climate and gas regulation*; *hydrological control*; *filtering of nutrients and contaminants* in addition to *recycling of wastes and detoxification*.

4.3.2.1. Biological control of pest and diseases. A healthy soil community keeps pests and harmful disease vectors at bay through competition, predation and parasitism (Barrios, 2007; Dominati et al., 2010a). Sandhu et al. (2008) assessed the value of biological control based on the avoided cost of artificial pest control and found a value of id$59–id$268 ha^{-1}/yr^{-1} (Table 6).

4.3.2.2. Climate and gas regulation. This service includes the production and sequestration of greenhouse gases (Haygarth and Ritz, 2009; Lavelle et al., 2006; Turbé et al., 2010; Wall, 2004), as well as the regulation of atmospheric chemical composition (Costanza et al., 1997; Haygarth and Ritz, 2009). The methods assessing the economic value of climate regulation services differ between cases. They are based on: the cost of carbon sequestration in various contexts based on the market price of carbon quotas; the market cost of sequestration methods; or choice experiments illustrating the willingness to pay for enhanced soil carbon sequestration. The value ranges from id$20 to id$268 /ha^{-1}/yr^{-1} (Table 6).

4.3.2.3. Hydrological control. Regulation of water runoff through water storage and retention. This lessens the impact of flood, drought and erosion events (Dominati et al., 2010b; Lavelle et al., 2006). The value of hydrological control has been evaluated based on the replacement cost of topsoil; defence expenditures based on the mitigation cost linked to soil degradation, the prevention of floods or the cost of dredging waterways; contingent valuation revealing the willingness to pay for preventing soil erosion; hedonic pricing; and avoided cost (Bond et al., 2011; Colombo et al., 2006; Eastwood et al., 2000; see e.g. Miranowski and Hammes, 1984; Pimentel et al., 1995; San and Rapera, 2010). These studies reveal a wide range of service values, ranging from id$30 to id$1175 ha^{-1}/yr^{-1} (Dominati et al., 2014a; San and Rapera, 2010) (Table 6).

4.3.2.4. Filtering of nutrients and contaminants. Soils have the ability to control water quality by, to some extent, absorbing and retaining solutes and 'contaminants', therefore avoiding their release in water bodies such as ground water, lakes and rivers (Andrews et al., 2004; Dominati et al., 2010b). Dominati et al. (2014a, 2014b) have evaluated this service using provision cost and defensive expenditure. The values range from id$544 to id$6402 ha^{-1}/yr^{-1} based on the nutrient or pollutant filtered (Table 6).

4.3.2.5. Recycling of wastes and detoxification. Soils degrade and decompose organic matter. The texture of soil and its drainage qualities are important regarding the retention of pollutants, pathogens and heavy metals (Andrews et al., 2004; Dominati et al., 2010a). Soil biota also plays an important role in breaking down toxic or hazardous compounds (Massaccesi et al., 2002), and is a low cost alternative to the standard environmental contamination clean-up following excavation and transportation (Das and Chandran, 2011; Singh, 2008). The value of recycling of wastes and detoxification has been estimated for grazing land for sheep and cattle production by Dominati et al. (2014a) and Dominati et al. (2014b). They report values depending on the type of soil and land characteristics ranging from id$77 and id$330 ha^{-1}/yr^{-1} (Table 6).

[7] For more information about the Purchasing Power Parity and how it is formed and used see discussion on page 152 - 154 in Ecological Economics an Introduction by Michael Common and Sigrid Stagl (Common and Stagl).

Table 6
Soil ES and economic valuation summary — based on Table 1 in Appendix A.

Soil service category	Services/functions	Valuation method	International dollar (id$) 2012	id$ units
Support functions	Biodiversity pool	Various methods	2.1 trillion	id$/yr^{-1}
	Nutrient cycling	Replacement cost, market price, hedonic price	24–180	id$/ha^{-1}/yr^{-1}
	Soil formation	Market price	18–28	id$/ha^{-1}/yr^{-1}
	Water cycling	Market price	62–126	id$/ha^{-1}/yr^{-1}
Regulating services	Biological control of pests and diseases	Avoided cost, provision cost	59–268	id$/ha^{-1}/yr^{-1}
	Climate regulation	Choice experiments, market price, replacement cost	2–268	id$/ha^{-1}/yr^{-1}
	Hydrological control	Damage cost, hedonic cost, replacement cost, benefit transfer, defensive expenditure, provision cost, contingent valuation, choice modelling	30–1175	id$/ha^{-1}/yr^{-1}
	Recycling of wastes and detoxification	Provision cost	77–330	id$/ha^{-1}/yr^{-1}
	Filtering of nutrients and contaminants	Provision cost, defensive expenditure	544–6402	id$/ha^{-1}/yr^{-1}
Provisioning services	Biomass production	Market price, producers price	231–22,219	id$/ha^{-1}/yr^{-1}
	Clean water provision	Damage cost, net factor, hedonic cost	34–101	id$/ML
	Raw materials	Producers price	9–147	id$/t
	Physical environment	Defensive cost, replacement cost, provision cost	32–110	id$/ha^{-1}/yr^{-1}
Cultural services	Heritage	Net factor	ND	No data
	Recreation	Damage cost	571,720	id$/yr^{-1}
	Cognitive	No data	ND	No data

4.3.3. Provisioning services

Provisioning services include *biomass production*, *clean water*, *raw materials* and *physical environment.*

4.3.3.1. Biomass production. Soils provide nutrients, water and physical environment for terrestrial biomass production. Humans use biomass in the form of food, wood, fuel, and fibre. The value of biomass production is frequently based on market values or producer prices of the biomass produced or the raw materials in question (Decaens et al., 2006; Haley, 2006; Porter et al., 2009; see e.g. Sandhu et al., 2008). Values range widely based on the product, its location, and the quantity sold. The value range from id$1.6 per kg of biomass of soil animals sold to id$22,219 ha^{-1}/yr^{-1} for food biomass produced on organic farms (Table 6).

4.3.3.2. Clean water provision. The soil's services of buffering and filtering are crucial for establishing the quality and quantity of our subterranean and surface water reserves (Clothier et al., 2008). The value of the provisioning of clean water is most often based on the cost of cleaning the water and making it suitable for human consumption. As in the other categories, values vary widely. In the United States, for example, values range from id$34 to id$101 per million litres (Tegtmeier and Duffy, 2005) (Table 6).

4.3.3.3. Raw materials. Topsoil, clay and peat are examples of raw materials from soil. Soils are also consumed directly and serve an important role as source of minerals and medicine in some areas of the world (Abrahams, 2012). Values are normally based on the market price of raw materials per tonne and vary widely based on the raw material in question (see e.g. Dolley and Bolen, 2000; Jasinski, 2000; Virta, 2004). The values from this service range from id$9 to id$147 per tonne of material (Table 6).

4.3.3.4. Physical environment. Soils provide a physical environment for human infrastructure, plants and animal species. They also provide a suitable living and reproduction space for different types of flora and fauna (Andrews et al., 2004; Daily et al., 1997a; Topp et al., 1997; Weber, 2007). Values for this service range from id$32 to id$110 ha^{-1}/yr^{-1} (Dominati et al., 2014a; Dominati et al., 2014b) (Table 6).

4.3.4. Cultural services

Cultural services include services such as *cognitive services*, *heritage services* and *recreational services.* In the ES literature there many studies that look at the cultural services in different ecosystems (Kumar, 2010) but when it comes to cultural services from soils there are very few studies that the authors are aware of.

4.3.4.1. Heritage services. Soils maintain our geological, ecological and archaeological archive. Studies on the economic value of heritage services are largely missing from the literature (Table 6).

4.3.4.2. Cognitive services. These include various non-commercial activities such as aesthetics, spirituality and education. Soil supports various types of vegetation in different landscapes that have been a source of aesthetic influence for artists throughout the ages (Wells and Mihok, 2009). Studies on the economic value of cognitive services of soils are also missing from the literature (Table 6).

4.3.4.3. Recreation services. Soils provide an environment for recreational activities, for example ecotourism (Decaens et al., 2006) and different sports. Recreational value is commonly assessed using the travel cost method. However, studies of soil recreational services are largely missing with the exception of Eastwood et al. (2000), who assessed the reduction in recreational value due to soil erosion. The authors found it to be 1% of the operations cost for national conservation estates in New Zealand, amounting to id$571,720 a year (Table 6).

4.4. Summary

This overview illustrates that soils provide various valuable ES across all service categories. There is no consensus on a holistic classification system for economically valuing soil ES, and no comprehensive assessment has been made of their worth. Various studies, however, have illustrated the economic importance of individual services giving a value range from id$2 to id$22,219 per hectare, revealing that the total value of soil ES is likely to be significant.

5. Concluding discussion

Provisioning of ES is necessary for maintaining economic systems, and is directly and indirectly linked to human well-being. As many ES, including soil ES, are largely non-market goods, they are excluded from formal economic decision-making and therefore undervalued and overexploited. In the case of soils and soil ES, this is revealed through significant threats to soil natural capital. It is clear that soil degradation is driven or exacerbated by human activities such as unsustainable agricultural and forestry practices, industrial activities, tourism, urban and industrial sprawl, road building, soil sealing and construction work (European Commission, 2006). These threats then translate to a loss of soil natural capital that carry significant societal and economic costs and impact human welfare (Initiative, 2015).

We reviewed the main categories of soil ES, the frameworks proposed for classification and provided a comprehensive overview of existing economic valuation studies of soil ES. We also illustrated that economic valuation tools exist for the assessment of soil ES, and various studies have illustrated the value of individual soil ES (see Table 6 and Table 1 in Appendix A). The value of the services in individual categories varies significantly as this represents valuations from various land-uses in different locations and ecosystem types. A comparison of id\$/ha^{-1}/yr^{-1} in different service categories illustrates the following:

- Supporting functions value range from 24 to 180 id\$/ha^{-1}/yr^{-1} with both the lowest and highest value applying to nutrient cycling.
- Regulating services values range from 2 for climate regulation to 6402 id\$/ha^{-1}/yr^{-1} for filtering of nutrients and contaminants.
- Provisioning services value range from 32 for physical environment to 22,219 id\$/ha^{-1}/yr^{-1} for biomass production.
- Cultural services value range is non-existent since studies on the cultural value of soils are missing from the literature.

The most common function in supporting functions was nutrient cycling, totalling 12 of the 17 studies represented. The biodiversity pool function was the most underrepresented and the only economic value available is an aggregated value of the contribution of soil biodiversity to a number of ES. This is to be expected as soil biodiversity is a support functions and is inherently difficult to value and should not be valued. Nevertheless there exists a valuation based on expert opinion (van der Putten et al., 2004) and it estimates the contribution of soil biodiversity to be 2.1 trillion id\$/yr. In 2008 there were approximately 4.9 billion hectares of agricultural land (arable land, permanent crops and pasture) worldwide (FAO, 2010) and given the assumptions that the 2.1 trillion id\$ refers to ES in agricultural land (Pimentel et al., 1997; van der Putten et al., 2004), the 2.1 trillion id\$/yr^{-1} converted to ha^{-1}/yr^{-1}, gives a value of approximately 430 id\$/ha^{-1}/yr^{-1} that soil biodiversity is contributing to ES worldwide. Although we have some reservation regarding this value, it underlines the potential immense importance of soil biodiversity for agro-ecosystems and for the Earth Critical Zone. In the regulating services, hydrological control was the most represented with 17 of 36 of the values followed by climate regulation with eight values. Biological control of pests and diseases and recycling of wastes and detoxification had only three services each. For the provisioning services category, the most values are for raw materials with 11 of 31 values and followed by biomass production with nine values. The fewest are for the physical environment service, with five. With regard to cultural services, there is only one value and that is the annual country value for recreational services. The most obvious gap in the literature is within the cultural service category as they are almost non-existent. This is also reflected in some of the soil frameworks (Table 5) where cultural services are left out. It is interesting to note the large differences in values, especially with regard to biomass production, 231 to 22,219 id\$/ha^{-1}/yr^{-1} (see Table 1 in Appendix A). This has to do with the different land uses and the economic value of the biomass produced. As these values are location specific (also farm system specific, market system specific etc.) care must be taken when using methods like benefit transfer which use available information from studies already completed in another location. Although the range of values for each soil ES varies greatly, their identification is sufficient to reveal that their total economic value is likely to be considerable. Valuation like this can prove to be important for land use management as it illustrates the benefits that soil natural capital is providing.

Caution has to be exercised when using this kind of valuation, in particular for management purposes. Soil like many other ecosystems has a life-cycle and during this life-cycle, i.e. from pedogenesis to erosion (Banwart et al., 2012), some soil processes and functions that underlie and provide ES are absent during certain parts of its life-cycle. Younger developing soil has not the same output of soil ES as more mature soil because there is difference in the state of the natural capital. If this kind of economic valuation would be used, without taking into consideration the life-cycle of soil and difference in the natural capital stocks, it could be detrimental to areas where soil development is in its infancy, as the methodology has a bias towards more output-orientated agro- ecosystems (e.g. primary production from cropland), due to valuing the soil ES in monetary terms. If the multiple roles of soils and values are invisible to the land-manager, he might make decisions based on incomplete information. Say, for example, the land manager decides to sell soil from a particular area as a raw material. When making that decision he might miss the trade-off involved between soil as a raw material sold off one time and the hydrological control services that soil provides, which might be more valuable to him in the long run. Here, a holistic approach, based on a soil ES economic valuation might help him to make more informed decisions.

Double-counting is also an issue that must be addressed in ES valuation. Double counting happens when the classification system confuses the ends with the means, such as when both intermediate and final ES are classified within the same set. The difference needs to be distinguished between the benefits that people receive from an ecosystem and the ecosystem processes that provide those benefits. According to Wallace (2008), any classification system that confuses the ecosystem processes with the outcome of the processes will create redundancy. This is a problem because there is the danger of exaggerating the value of the output from the ecosystem and thus its total value. The issue of double counting is addressed by using the MA as the base framework, an approach involving a clear distinction between intermediate and final services, and also by not aggregating any values between categories.

As soil ecosystems provide multiple services a holistic assessment framework, that can help to illustrate the economic value of soil ES, is needed. Yet, to date there is no consensus on a comprehensive framework for the classification and economic valuation of soil ES (Robinson et al., 2012), and simultaneously no systematic way has been developed to evaluate their importance (Anderson et al., 2004). The value of soil ES is invisible and as Kumar (2010) states "[m]any people benefit from ES without realizing it and they fail appreciate their value". As a result, there is a need to develop a comprehensive framework for the assessment of soil ES that can be incorporated into decision support tools for decision-makers at various levels. "Being able to place a value on ecosystem services is fundamental to designing policies [and incentive mechanisms] to induce agricultural land managers to provide (or maintain) ES at levels that are desirable to society" (Swinton et al., 2007).

We have provided a literature review of soil ES, shown that soil ES can be valued by using standard economic methods, and has brought forward examples of valuations for a diverse range of soil ES across a range of land uses. We also showed how the concept of soil ES ties together with the emerging concept of the Critical Zone. The importance of the soil ES in the Critical Zone shows the necessity of properly accounting for the value of soil ES in land-use decision making. A holistic framework and a methodology is required in order to tie together soil natural capital, soil ES and economic valuation. In a forthcoming paper we will introduce a contribution towards such a framework.

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Appendix A

Table 1
Soil ES studies.

Category	Services/functions	CTR	Scale	LOC	CONT	LUCAS	MAES	Year	MTD	Value	Orig. value	CCY	Original Units	ID 2012	ID units	Ref
SF	Biodiversity pool	W	GL	Global	GL	Most land types	Most land types	1997	VM	AV	1.5 trillion	USD	USD/yr^{-1}	2.1 trillion	id$/yr^{-1}	31
	Nutrient cycling	US	C	US	NA	Agriculture	Cropland	2012	RC	AV	0.59	USD	USD/t/yr^{-1}	0.6	id$/t/yr^{-1}	10
		US	C	US	NA	Agriculture	Cropland	2011	RC	AV	1.13	USD	USD/t/yr^{-1}	1.2	id$/t/yr^{-1}	10
		SSA	P	Sub-Saharan Africa	AF	Agriculture	Cropland	2004	RC	AV	20	USD	USD/ha^{-1}/yr^{-1}	24	id$/ha^{-1}/yr^{-1}	9
		NZ	P	Canterbury	OC	Agriculture	Cropland	2005	MP	AV	40	USD	USD/ha^{-1}/yr^{-1}	47	id$/ha^{-1}/yr^{-1}	28
		NZ	P	Canterbury	OC	Agriculture	Cropland	2006	MP	AV	43	USD	USD/ha^{-1}/yr^{-1}	49	id$/ha^{-1}/yr^{-1}	28
		GH	M	Kumasi	AF	Agriculture	Cropland	2004	RC	AV	45	USD	USD/ha^{-1}/yr^{-1}	55	id$/ha^{-1}/yr^{-1}	9
		GH	M	Kumasi	AF	Agriculture	Cropland	2004	RC/HC	AV	50	USD	USD/ha^{-1}/yr^{-1}	61	id$/ha^{-1}/yr^{-1}	9
		MM	L	Nyaung Shwe	AS	Agriculture	Cropland	2006	RC	AV	50	USD	USD/ha^{-1}/yr^{-1}	87	id$/ha^{-1}/yr^{-1}	27
		US	C	US	NA	Agriculture	Cropland	1992	RC	AV	100	USD	USD/ha^{-1}/yr^{-1}	164	id$/ha^{-1}/yr^{-1}	22
		MM	L	Nyaung Shwe	AS	Agriculture	Cropland	2006	RC	AV	103	USD	USD/ha^{-1}/yr^{-1}	178	id$/ha^{-1}/yr^{-1}	27
		MX	M	Guanajuato	NA	Agriculture	Cropland	2000	RC	AV	135	USD	USD/ha^{-1}/yr^{-1}	180	id$/ha^{-1}/yr^{-1}	29
		US	C	US	NA	Most land types	Most land types	1995	MP	AV	8 billion	USD	USD/yr^{-1}	12 billion	id$/yr^{-1}	23
	Soil formation	US	C	US	NA	Several types	Several types	1995	MP	AV	12	USD	USD/ha^{-1}/yr^{-1}	18	id$/t/ha^{-1}/yr^{-1}	23
		NZ	P	Canterbury	OC	Agriculture	Cropland	2005	MP	AV	24	USD	USD/ha^{-1}/yr^{-1}	28	id$/t/ha^{-1}/yr^{-1}	28
	Water Cycling	NZ	P	Canterbury	OC	Agriculture	Cropland	2006	MP	AV	54	USD	USD/ha^{-1}/yr^{-1}	62	id$/ha^{-1}/yr^{-1}	28
		NZ	P	Canterbury	OC	Agriculture	Cropland	2005	MP	AV	107	USD	USD/ha^{-1}/yr^{-1}	126	id$/ha^{-1}/yr^{-1}	28
RS	Biological control of pests and diseases	NZ	P	Canterbury	OC	Agriculture	Cropland	2005	AC	AV	50	USD	USD/ha^{-1}/yr^{-1}	59	id$/ha^{-1}/yr^{-1}	28
		NZ	L	Waikato	OC	Agriculture	Grassland	2011	PC	AV	210	NZD	NZD/ha^{-1}/yr^{-1}	206	id$/ha^{-1}/yr^{-1}	8
		NZ	R	Hawke's Bay	OC	Agriculture	Grassland	2011	PC	AV	328	NZD	NZD/ha^{-1}/yr^{-1}	334	id$/ha^{-1}/yr^{-1}	7
	Climate regulation	NZ	R	Hawke's Bay	OC	Agriculture	Grassland	2011	MP	AV	2.20	NZD	NZD/ha^{-1}/yr^{-1}	2.2	id$/ha^{-1}/yr^{-1}	7
		NZ	R	Hawke's Bay	OC	Agriculture	Grassland	2011	MP	AV	6.00	NZD	NZD/ha^{-1}/yr^{-1}	5.9	id$/ha^{-1}/yr^{-1}	7
		NZ	P	Canterbury	OC	Agriculture	Cropland	2005	MP	AV	20	USD	USD/ha^{-1}/yr^{-1}	24	id$/ha^{-1}/yr^{-1}	28
		NZ	P	Canterbury	OC	Agriculture	Cropland	2005	MP	AV	22	USD	USD/ha^{-1}/yr^{-1}	26	id$/ha^{-1}/yr^{-1}	28
		ES	P	Andalusia	EU	Agriculture	Cropland	2012	CE	AV	17	EUR	EUR/ton- CO_2/yr^{-1}	20	id$/ ton- CO_2/ yr^{-1}	26
		UK	C	Scotland	EU	Agriculture	Cropland and Grassland	2009	CE	NPV	38	GBP	GBP/ton-$CO_{2\text{-eq}}$/yr^{-1}	72	id$/ ton-$CO_{2\text{-eq}}$ /yr^{-1}	13
		UK	C	UK	EU	Agriculture	Cropland and Grassland	1996	MP	AV	63	GBP	GBP/ton-CO_2/yr^{-1}	149	id$/ton-$CO_2$ - / ha^{-1}/yr^{-1}	25
		CN	L	Shanghai	AS	Agriculture	Cropland	2005	MP/RC	AVR	531–2000	CNY	RBM/ha^{-1}/yr^{-1}	71-268	id$/ ha^{-1}/yr^{-1}	33
	Filtering of nutrients and contaminants	NZ	L	Waikato	OC	Agriculture	Grassland	2011	DE	AV	554	NZD	NZD/ha^{-1}/yr^{-1}	544	id$/ha^{-1}/yr^{-1}	8
		NZ	R	Hawke's Bay	OC	Agriculture	Grassland	2011	PC	AV	1800	NZD	NZD/ha^{-1}/yr^{-1}	1,796	id$/ha^{-1}/yr^{-1}	7
		NZ	R	Hawke's Bay	OC	Agriculture	Grassland	2011	PC	AV	2227	NZD	NZD/ha^{-1}/yr^{-1}	2,189	id$/ha^{-1}/yr^{-1}	7
		NZ	L	Waikato	OC	Agriculture	Grassland	2011	DE	AV	2924	NZD	NZD/ha^{-1}/yr^{-1}	2,871	id$/ha^{-1}/yr^{-1}	8
		NZ	L	Waikato	OC	Agriculture	Grassland	2011	PC	AV	6513	NZD	NZD/ha^{-1}/yr^{-1}	6,402	id$/ha^{-1}/yr^{-1}	8
	Hydrological control	US	P	Willamette Valley	NA	Agriculture, Forestry, Recreation	Cropland, Woodland, Urban	1984	DC	AV	0.85	USD	US/t	1.9	id$/t	19
		US	C	US	NA	Agriculture	Cropland and Grassland	2002	DC	AV	2.5	USD	USD/m^3/yr^{-1}	3.2	id$/m^3/yr^{-1}	30
		US	L	Ohio state park lakes	NA	Agriculture	Cropland	1999	HC	AVR	0.15–5.18	USD	USD/ac-ft./yr^{-1}	18	id$/ac-ft./yr^{-1}	1
		NZ	C	NZ	OC	Agriculture	Cropland	1998	DC	AV	15	NZD	NZD/m^3/yr^{-1}	20	id$/m^3/yr^{-1}	18
		MM	L	Nyaung Shwe	AS	Agriculture	Cropland	2010	RC	AV	17	USD	USD/ha^{-1}/yr^{-1}	30	id$/ha^{-1}/yr^{-1}	27
		BZ	L	Rio Bravo	NA	Agriculture	Cropland	1989	BT	AV	23	USD	USD/ha^{-1}/yr^{-1}	43	id$/ha^{-1}/yr^{-1}	27
		US	P	Iowa	NA	Agriculture, forestry	Woodland and forest, sparsely vegetated land	1978	HC	AV	6	USD	USD/ac-t/yr^{-1}	49	id$/tons -h/yr^{-1}	11
		US	C	US	NA	Agriculture	Cropland	1992	RC	AV	30	USD	USD/ha^{-1}/yr^{-1}	49	id$/ha^{-1}/yr^{-1}	22
		MM	L	Nyaung Shwe	AS	Agriculture	Cropland	2010	RC	AV	36	MMK	USD/ha^{-1}/yr^{-1}	63	id$/ha^{-1}/yr^{-1}	27
		US	P	Kern County	NA	Agriculture	Cropland, Heathland Scrub	2007	BT	AV	134	USD	USD/ac/yr^{-1}	367	id$/ha^{-1}/yr^{-1}	21
		NZ	R	Hawke's Bay	OC	Agriculture	Grassland	2011	PC	AV	911	NZD	NZD/ha^{-1}/yr^{-1}	895	id$/ha^{-1}/yr^{-1}	7
		NZ	R	Hawke's Bay	OC	Agriculture	Grassland	2011	PC	AV	1,155	NZD	NZD/ha^{-1}/yr^{-1}	1134	id$/ha^{-1}/yr^{-1}	7
		NZ	L	Waikato	OC	Agriculture	Grassland	2011	PC	AV	1,196	NZD	NZD/ha^{-1}/yr^{-1}	1175	id$/ha^{-1}/yr^{-1}	8
		US	C	US	NA	Energy production	Urban	1997	RC	AV	0-1.38	USD	USD/t	0–2	id$/t	15
		US	C	US	NA	Agriculture	Cropland	2007	RC	NPV	0-1.38	USD	USD/t/yr^{-1}	0–1.5	id$/t/yr^{-1}	2
		ES	P	Andalusia	EU	Agriculture	Cropland	2005	CV/CM	AVR	95 - 160	EUR	EUR/ha^{-1}/yr^{-1}	124–209	id$/ha^{-1}/yr^{-1}	3
		US	P	Colorado	NA	Agriculture	Cropland	2011	CV	AVR	6.21 - 7.12	USD	USD/ac/yr^{-1}	16–18	id$/ha^{-1}/yr^{-1}	2
	Recycling of wastes and	NZ	L	Waikato	OC	Agriculture	Grassland	2011	PC	AV	78	NZD	NZD/ha^{-1}/yr^{-1}	77	id$/ha^{-1}/yr^{-1}	8

(continued on next page)

Table 1 (*continued*)

Category	Services/functions	CTR	Scale	LOC	CONT	LUCAS	MAES	Year	MTD	Value	Orig. value	CCY	Original Units	ID 2012	ID units	Ref
	detoxification	NZ	R	Hawke's Bay	OC	Agriculture	Grassland	2011	PC	AV	127	NZD	NZD/ha^{-1}/yr^{-1}	125	id$/ha^{-1}/yr^{-1}	7
		NZ	R	Hawke's Bay	OC	Agriculture	Grassland	2011	PC	AV	336	NZD	NZD/ha^{-1}/yr^{-1}	331	id$/ha^{-1}/yr^{-1}	7
PS	Biomass production	MX	C	Mexico	NA	Agriculture	NA	2006	MP	AV	1.50	EUR	Euro/kg	1.6	id$/kg	5
		DK	L	Taastrup	EU	Agriculture	Cropland and Grassland	2009	PP	AV	216	USD	USD/ha^{-1}/yr^{-1}	231	id$/ha^{-1}/yr^{-1}	24
		NZ	R	Hawke's Bay	OC	Agriculture	Grassland	2011	MP	AV	484	NZD	NZD/ha^{-1}/yr^{-1}	475	id$/ha^{-1}/yr^{-1}	7
		DK	L	Taastrup	EU	Agriculture	Cropland and Grassland	2009	PP	AV	515	USD	USD/ha^{-1}/yr^{-1}	551	id$/ha^{-1}/yr^{-1}	24
		NZ	R	Hawke's Bay	OC	Agriculture	Grassland	2011	MP	AV	745	NZD	NZD/ha^{-1}/yr^{-1}	732	id$/ha^{-1}/yr^{-1}	7
		MX	C	Mexico	NA	Agriculture	Cropland	2004	MP	AV	875	USD	USD/ha^{-1}/yr^{-1}	1,063	id$/ha^{-1}/yr^{-1}	14
		NZ	P	Canterbury	OC	Agriculture	Cropland	2005	PP	AVR	3,220	USD	USD/ha^{-1}/yr^{-1}	3785(987-16,458)	id$/ha^{-1}/yr^{-1}	28
		NZ	L	Waikato	OC	Agriculture	Grassland	2011	MP	AV	4,757	NZD	NZD/ha^{-1}/yr^{-1}	4671	id$/ha^{-1}/yr^{-1}	8
		NZ	P	Canterbury	OC	Agriculture	Cropland	2005	PP	AVR	3,990	USD	USD/ha^{-1}/yr^{-1}	4690 (1.352-22.219)	id$/ha^{-1}/yr^{-1}	28
		CO	M	Santafé de Bogotá	NA	Agriculture	NA	2006	MP	R	1.00-1.50	EUR	EUR/100 gr	11-16	id$/kg	5
	Clean water provision	US	C	United States	NA	Water supply and treatment	Urban	1984	DC/HC	AV	17	US Dollar	USD/KT	38	id$/KT	16
		US	P	Willamette Valley	NA	Agriculture, Forestry, Recreation	Cropland, Woodland, Urban	1984	DC	AV	20	USD	USD/MGD	44	id$/MGD	19
		UK	C	UK	EU	Agriculture	Cropland and Grass	1996	DC	AV	52.3 million	GBP	GBP/yr^{-1}	124 million	id$/ yr^{-1}	25
		KE	M	Nairobi	AF	Agriculture, Forestry	Cropland, Woodland	2008	DC	AV	9,910,000	KES	KES/yr^{-1}	80792	id$/yr^{-1}	20
		NZ	C	NZ	OC	Recreation, Leisure, Sport and Water supply and treatment	Urban	1998	NF	AV	2,8 million	NZD	NZD/yr^{-1}	3.81 million	id$/yr^{-1}	12
		US	C	US	NA	Agriculture	Cropland and Grassland	2002	DC	AVR	26 - 79	USD	USD/ML	34-101	id$/ML	30
	Physical environment	NZ	L	Waikato	OC	Agriculture	Grassland	2011	RC	AV	17	NZD	NZD/ha^{-1}/yr^{-1}	17	id$/ha^{-1}/yr^{-1}	8
		NZ	R	Hawke's Bay	OC	Agriculture	Grassland	2011	DE	AV	33	NZD	NZD /ha^{-1}/yr^{-1}	32	id$/ha^{-1}/yr^{-1}	7
		NZ	R	Hawke's Bay	OC	Agriculture	Grassland	2011	DE	AV	53	NZD	NZD/ha^{-1}/yr^{-1}	52	id$/ha^{-1}/yr^{-1}	7
		NZ	L	Waikato	OC	Agriculture	Grassland	2011	PC	AV	112	NZD	NZD/ha^{-1}/yr^{-1}	110	id$/ha^{-1}/yr^{-1}	8
		W	GL	Global	GL	Agriculture	Cropland	1990	PC	NPV	55,000	USD	USD/ha	96,615	id$/ha^{-1}	4
	Raw materials	US	C	US	NA	Mining and quarrying	Urban	2004	PP	AV	7	USD	USD/t	9	id$/t	32
		US	C	US	NA	Mining and quarrying	Urban	2000	PP	AV	15	USD	USD/t	20	id$/t	6
		US	C	US	NA	Mining and quarrying	Urban	2000	PP	AV	18	USD	USD/t	24	id$/t	17
		US	C	US	NA	Mining and quarrying	Urban	2000	PP	AV	20	USD	USD/t	26	id$/t	6
		US	C	US	NA	Mining and quarrying	Urban	2000	PP	AV	24	USD	USD/t	32	id$/t	17
		US	C	US	NA	Mining and quarrying	Urban	2004	PP	AV	28	USD	USD/t	34	id$/t	32
		US	C	US	NA	Mining and quarrying	Urban	2000	PP	AV	27	USD	USD/t	36	id$/t	17
		US	C	US	NA	Mining and quarrying	Urban	2004	PP	AV	44	USD	USD/t	54	id$/t	32
		US	C	US	NA	Mining and quarrying	Urban	2004	PP	AV	44	USD	USD/t	54	id$/t	32
		US	C	US	NA	Mining and quarrying	Urban	2004	PP	AV	101	USD	USD/t	123	id$/t	32
		US	C	US	NA	Mining and quarrying	Urban	2004	PP	AV	121	USD	USD/t	147	id$/t	32
CS	Recreation	NZ	C	NZ	OC	Recreation, Leisure, Sport and Water supply and treatment	Urban	1998	DC	AV	420,000	NZD	NZD/yr^{-1}	571,720	id$/yr^{-1}	12

Table 2
Explanations of categories abbreviations in studies table.

ES Category	Country (CTR)	Scale of research	Continent	MAES (level 2)	LUCAS Categories	Method (MTD)	Currency (CCY)	Value type
Cultural services = CS Provisioning services = PS Regulating services = RS Supporting functions = SF	BZ = Belize CN = China CO = Colombia DK = Denmark ES = Spain GH = Ghana KE = Kenya MM = Myanmar MX = Mexico NZ = New Zealand SSA = Sub-Saharan Africa UK = United Kingdom US = United States W = World	C = Country G = Global L = Local M = Municipality/City P = Province/region R = Regional	AF = Africa AS = Asia EU = Europe GL = Global NA = North America OC = Oceania SA = South and Central America	Urban Cropland Grassland Woodland and forest Heathland and shrub Sparsely vegetated areas Wetlands Marine inlets and transitional waters Rivers and lakes Marine	Agriculture Forestry Aquaculture and Fishing Mining and Quarrying Manufacturing and Energy Energy production Industry and Manufacturing Transport, Communication Networks, Storage, and Protective works Water and Waste Treatment Construction Commerce, Finance and Business, Community services Recreation, Leisure and Sport Residential Unused and Abandoned areas	AC = Avoided cost BT = Benefit transfer CE = Choice experiments CM = Choice modelling CV = Contingent valuation DC = Damage cost DE = Defensive expenditure HC = Hedonic cost MP = Market price NF = Net factor PC = Provision cost PP = Producers price RC = Replacement cost VM = Various methods	CNY = China Yuan Renminbi EUR = Euro Member Countries GBP = United Kingdom Pound KES = Kenya Shilling MMK = Myanmar (Burma) Kyat NZD = New Zealand Dollar USD = United States Dollar	AV = Annual value AVR = Annual value range NPV = Net Present Value R = Range

Table 3
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Paper II

CHAPTER TWELVE

Valuation of Soil Ecosystem Services

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Contents

Abstract

Soil natural capital and soil ecosystem services (ES) are under increasing pressure because of human activities. Soils provide multiple benefits to humans, and the role of soil in Earth's Critical Zone is fundamental to its functions that provide these benefits. Despite their importance, soils are rarely appreciated for the values they provide. One reason is the absence of their economic value in land-use decision making. We present a framework for categorizing and economically valuing soil ES and illustrating the use of the framework in a case study for three soil ES in the Koiliaris watershed on the Greek island of Crete. The value of the soil ES estimated was crop and livestock biomass 740–7560 id$ ha^{-1} year^{-1}; filtering of nutrients and contaminants 0–278 id$ ha^{-1} year^{-1}; and climate regulation −2200 to −5610 id$ ha^{-1} year^{-1}.

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Highlights

- Soils provide multiple economic benefits that are rarely accounted for.
- A framework for the classification and economic valuation of soil ecosystem services (ES) is presented.
- The soil framework is applied at Koiliaris, Crete, Greece for three soil ES.
- This chapter illustrates soil ES values from −5610 to 7560 id\$ ha^{-1} $year^{-1}$ depending on the specific soil ES.

1. INTRODUCTION

The delivery of ecosystem services (ES) is necessary for maintaining economic systems and is directly and indirectly linked to human well-being. As ES are largely nonmarket goods, they are excluded from formal economic decision making and therefore undervalued and overused. Assessment and economic valuation of ES is an attempt to reverse this trend (Kumar, 2010; MEA, 2005), but comprehensive valuation of soil ES has been somewhat ignored in the literature until now (see Jónsson and Davíðsdóttir, 2016 for an overview). Soils are an important kind of natural capital that has specific functions providing multiple, essential ES (Haygarth and Ritz, 2009). More than 99% of human food (calories) comes from the land and the soil (Pimentel, 2006). Soils filter and clean our drinking water; they deliver plant nutrients needed for vegetation growth (Daily et al., 1997); and they host soil biota that decompose plants when they die. Soils provide habitats for millions of species (Science, 2004) and store twice as much carbon as the biosphere and the atmosphere combined (Bellamy et al., 2005; Scharlemann et al., 2014). Soils also regulate water flows and thereby prevent floods. They provide us with building materials as well as provide the structural foundation for human activities (Frossard et al., 2006). Soils are a source of many current medicines, probiotics and antibiotics (D'Costa et al., 2006; Minton, 2003; Turbé et al., 2010). They store our history, in buried archaeological artifacts and sediments (European Soil Bureau Network, 2005). They are a source of recreation (Decaens et al., 2006) and are a fundamental part of world religions, for example, in Christianity God created man from soil. Derived from this discussion it is clear:

(i) how soils contribute to all ES categories as defined by the Millennium Ecosystem Assessment (MEA, 2005) and

(ii) that soil resources, as other natural resources, are important economic assets, whether or not they are valued via marketplaces.

Given the importance of the multiple services derived from soils, it is evident that they need to be maintained and the only way to do that is to protect soil structure and processes that are the base for their functionality. Unfortunately, soil, as per other types of natural capital, is coming under increased pressure because of human activities. One of the culprits for these degradation processes is the omission of soil ES valuation in land-use decision making. In order to address this deficiency, we propose a valuation framework for soil ES and then illustrate how soil and its various functions that provide ES can be valued economically through a case study in the Koiliaris watershed on the island of Crete, Greece.

1.1 Framework Development

The conceptual framework of a life cycle of soil development describes that soil processes go through a cycle from pedogenenis to degradation of soil functions and eventual physical erosion and loss (Banwart et al., 2012). This life cycle serves as the starting point for the soil ES framework (Fig. 1). This has some parallels with the concepts of Dominati et al. (2010) on formation, maintenance, and degradation of soil natural capital. To expand these concepts, the framework depends heavily on work undertaken by Robinson et al. (2009) and Dominati et al. (2010). There the authors tie together the concept of natural capital of soil and soil ES and how soil threats and soil management influence the soil's natural capital stocks and thus the provision of soil ES (see Jónsson and Davíðsdóttir, 2016). This paper builds on, and adds to, their work by showing how soil ES are linked to beneficiaries and can be valued using standard economic methods. The framework proposed in Fig. 1 consists of three interconnected sections: a biophysical

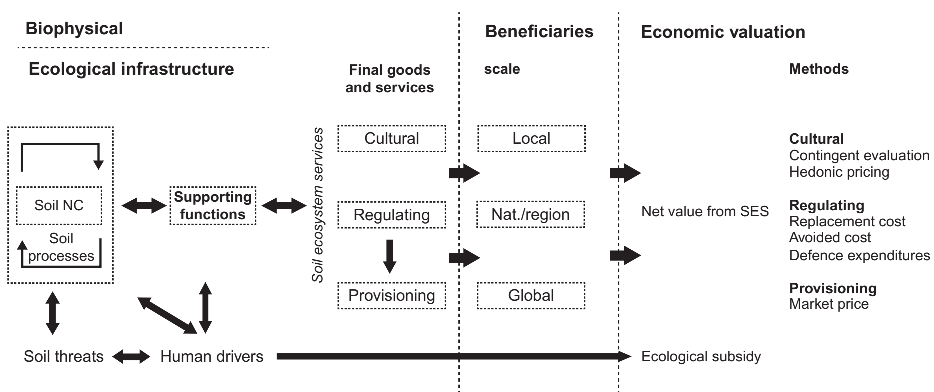

Fig. 1 Soil ecosystem services framework.

model, the identification of beneficiaries, and an economic valuation. The framework is then further subdivided into components within each section. The biophysical section has two main components:

Ecological infrastructure (Ei) of soil (soil natural capital and supporting functions) and *final goods and services* (cultural, regulating, and provisioning) that are the benefits that flow from the Ei (see Table 1).

Table 1 Key Terms

Key Terms	**Definitions**
Ecological infrastructure of soil	Soil natural capital, its properties, and soil support functions that underlie other ecosystem services and are in a dynamic relationship with soil processes and soil natural capital. Adapted and modified from Bristow et al. (2010)
Ecosystem services	The benefits that people obtain from ecosystems. Ecosystem goods and services are synonymous with ecosystem services (MEA, 2005)
Final goods and services from soil	Benefits that flow from the ecological infrastructure (cultural, regulating, and provisioning services). Adapted and modified from MEA (2005)
Natural capital	Stocks of natural resources found on Earth yielding a flow of valuable ecosystem goods or service into the future (Dominati et al., 2014c)
Soil ecosystem services	The flow from the ecological infrastructure of soil. Soil ecosystem services refer to both ecosystem goods and services from soil. Adapted and modified from MEA (2005) and Bristow et al. (2010)
Soil natural capital	Soil stocks on earth which yield a flow of soil goods and services. Soil natural capital is characterized by soil properties (Dominati et al., 2010)
Soil processes	Any change or reaction which occurs within soils, either physical, chemical, or biological. The complex interactions among the biotic and abiotic elements of the soil. Adapted and modified from de Groot et al. (2002)
Soil properties	The physical, chemical, and biological characteristics of a soil. They can be inherent or manageable (Dominati et al., 2010)
Soil support functions	A subset of interactions between the soil natural capital and soil processes that are required for the production of final soil ecosystem services and goods that satisfy human needs. The support functions are intermediate steps in the stock and flow chain and are therefore neither consumed directly nor valued economically. Adapted and modified from MEA (2005)

Final goods and services are called soil ES in the framework and refer to both ecosystem goods and services from soil. The beneficiaries part shows at what scale (local, regional/national, or global) the recipients of the ES are based. The third section is the economic valuation, where the value for soil ES is established using standard economic methods (e.g., market pricing, avoided cost, and associated methods; see Jónsson and Davíðsdóttir, 2016 for more detail on the different methods for the economic valuation of soil ES).

1.2 Biophysical Section

Soils, along with rivers, aquifers, wetlands, and other landscape elements, are "key components of an 'ecological infrastructure' that supports the continuing delivery of ecosystem services required by natural systems for their survival, and mankind for human well-being" (Bristow et al., 2010, p. 13). This chapter draws on this definition and defines Ei of soil as: soil natural capital, its properties, and soil processes; and soil support functions that underlie other ES and are in a dynamic relationship with soil natural capital. Soil natural capital refers to soil stocks on Earth, which yield a flow of benefits in the form goods and services. Soil natural capital builds on the definition of natural capital (Costanza and Daly, 1992), and it has physical, chemical, and biological properties (Table 2) that can be measured in qualitative or quantitative terms.

The soil support functions (Fig. 1) included in the framework are *soil formation*, *nutrient cycling*, *biodiversity pool*, and *water cycling* (Fig. 1). Soil's Ei creates soil ES (final goods and services).

Soil formation: The chemical, physical, and biological activities lead to the formation of soil over time by weathering of rocks and minerals. This process is affected by relief (terrain), parent material, climate, and geography (Brantley et al., 2007).

Nutrient cycling: The maintenance of soil fertility whereby chemical elements that are essential to the production and functioning of living organisms is moved and chemically reacted through the biotic and abiotic parts of the soil. Microorganisms are key moderators of this service. This cycling is the foundation for many other processes of the soil (Brussaard et al., 2007; Zhang et al., 2007).

Biodiversity pool: Soils are probably the most species-rich habitats of terrestrial ecosystems (Science, 2004; Weber, 2007). This reservoir of biodiversity is important to many soil processes, and many of the species

Table 2 Examples of Soil Properties

Indicators	**Metrics**
Pedodiversity	Number of soil classes within an area
Aggregate diversity	Mean weight diameter of various aggregates, and aggregate diversity measured with the Shannon–Wiener index
Bulk density	g/cm^3
Changes in topsoil depth	cm
Change in cation exchange capacity (CEC)	Milliequivalents/100 g
Soil contamination	Concentrations in topsoil
Change in topsoil pH	pH
Changes in microbial biomass	C ($mg\ kg^{-1}$)
Change in and absolute level of net N mineralization	$mg\ kg^{-1}$ soil%
Change in total soil organic matter (TSOM)	%

of organisms serve as an essential part of the functional diversity and resilience of the soil.

Water cycling: Water cycling involves the physical process of water moving through the soil. This movement influences the geo-, bio-, and chemical processes in the soil and affects the development of its biodiversity and functions (Dominati et al., 2010).

Soil ES are the flow of benefits from the Ei of soils. The soil ES are split into three categories similar to the MEA (2005) (Table 3): provisioning, regulating, and cultural. For more information about individual soil ES and service category, see Jónsson and Davíðsdóttir (2016).

When assessing soil ES, it is necessary to capture the dynamics of the soil Ei. Thus, the valuation of ES and functions requires a framework that relates land-use management practices to soil functions such as biomass production, carbon and nutrient sequestration, biodiversity, and water transformation as well as to how changes in land management affect these functions. The complexity of soil interactions necessitates the quantification of these functions to be determined through quantification of soil processes, e.g., through empirical measurement and mathematical modeling.

Table 3 MEA Categories (MEA, 2005)

Services	Definitions
Provisioning services	Products people obtain from ecosystems, such as food, fuel, fiber, freshwater
Regulating services	The benefits people obtain from the regulation of ecosystem processes, including air quality maintenance, climate regulation, erosion control, regulation of human diseases, and water purification
Cultural services	The nonmaterial benefits people obtain from ecosystems through spiritual enrichment, cognitive development, reflection, recreation, and esthetic experiences

1.3 Beneficiaries

ES are an anthropocentric concept and are therefore only relevant in the context of human beneficiaries. Beneficiaries are individuals that receive positive change in well-being through final goods and services. Before a value can be placed on ES, the beneficiaries need to be identified and established in their spatial scale. That can be done by looking at the value chain of soil ES and establishing the direct beneficiaries for the final services. This approach follows the distinction between intermediate and final services (Boyd and Banzhaf, 2007) and the establishment of direct beneficiaries (Johnston and Russell, 2011). The beneficiaries can be individuals, commercial entities, or the public sector; they can be distributed at local, national/regional, or global scale, and the benefits can be short or long term (eftec, 2005). The benefits also have ecological, sociocultural, and economic dimensions. An individual beneficiary on a local scale might be a farmer that gains economic benefits from cultivating crops (provisioning services), ecological benefits from the biological control of pests provided in part by a healthy soil biota (regulating services), and sociocultural benefits from living in a pastoral landscape which is considered beautiful (cultural services).

1.4 Economic

The economic valuation of soil ES takes place at the right side of the framework (Fig. 1) where the final goods and services that flow from the soil Ei are valued based on the benefits derived from those services. The economic valuation illustrates the net value of the soil ES. The economic value is derived by, for example, (a) looking at the biophysical data of the particular service (e.g., a regulating service), (b) seeing what would change in the absence of the soil ES (that is, complete loss of the service), and (c) comparing that to the current state of the soil ES. The difference in the biophysical quantity of the service is the foundation for the economic valuation. Support functions within the soil Ei are not valued as this would lead to double counting and this is consistent with standard economic principles (Boyd and Banzhaf, 2007). For a list of potential economic evaluation methods of soil ES, see Jónsson and Davíðsdóttir (2016). Within the economic valuation section, the ecological subsidy is also subtracted for certain services that have a direct human input. This could, as stated before, include, e.g., the use of fertilizer in agricultural fields to increase biomass yields. Here the fertilizer is only partly responsible for the overall output and that needs to be taken into account. If this information is not available from the biophysical proxies, then the price of these inputs is simply subtracted. This is particularly important for valuing trade-off in soil ES as soil ES receiving an ecological subsidy that increases its output and overall value potentially sends the wrong signals to land managers and policy makers about its effectiveness and value.

1.5 Soil Threats and Human Drivers

The status of the soil Ei is affected by external forces, including soil threats and human drivers (Fig. 1). These external forces influence the soil natural capital and the support functions and thus in turn influence the provisioning of final goods and services from soils. As a result, soil threats and human drivers must be included in a framework intended to capture the value of soil ES, and the potential costs associated with their degradation. The soil framework proposed here is able to pick up the influence of these external forces through (i) the ecological subsidy and (ii) the net economic value from soil (see Section 1.4).

1.5.1 Soil Threats

In 2002, the European Commission, in its communication "Towards a Thematic Strategy on Soil Protection," identified eight main threats to soils in the EU (Table 4). Soil degradation is driven or exacerbated by human activities such as inadequate agricultural and forestry practices, industrial activities, tourism, urban and industrial sprawl, road building, soil sealing, and construction work (European Commission, 2006).

These activities negatively affect the status of soil's Ei and its provision of soil ES. The soil threats are made visible in the framework by looking at how they affect the provision of the soil ES and functions and how that results in

Table 4 Soil Threats and Implications for Soil Functions

Soil Threats	Implications for Soil Functions
Erosion	Implications for all functions and services
Decline in organic matter	Climate regulation, biomass production
Soil contamination	Filtering of nutrients and contaminations
Soil sealing	Implications for all functions and services
Soil compaction	Biodiversity pool
Decline in soil biodiversity	Biodiversity pool
Salinization	Filtering of nutrients and contaminations
Floods and landslides	Hydrological control

changes in economic value. Table 4 shows how the soil threats affect the soil functions and services. The soil threats that the framework is able to capture at this moment are organic matter decline and erosion.

1.5.2 Ecological Subsidy

Ecological subsidies are direct human inputs that are intended to influence the supply of certain soil ES (Dale and Polasky, 2007; Tilman et al., 2002; Vitousek et al., 1997). The inputs are *fertilizers*, *pesticides*, and *irrigation*, and they can cause disservices and trade-offs (Power, 2010; Swinton et al., 2007). These inputs need to be accounted for when looking at the outputs of soil systems. Ecological subsidies differ from the human activities that drive or exacerbate soil degradation as those activities overall adversely affect soil Ei and soil ES, while the ecological subsidies enhance the supply of certain soil ES. The ecological subsidy is accounted for in the economic valuation section by looking at the changes in the biophysical output of the soil ES that are attributed to the ecological subsidy. If the information about changes in the biophysical outputs is unavailable, the costs of the inputs can be used as a proxy instead.

1.5.3 Fertilizers

The application of fertilizers, both natural and artificial, influences the provision of soil ES. Their application in an agricultural context makes certain nutrients available for biomass production that might not be so readily available, or even be considerably unavailable, in the natural landscape. This input increases biomass yields, thus potentially increasing the amount of harvestable provisioning services. At the same time, this application can negatively influence many processes of the soil, for example, the supporting functions provided by soil biodiversity (Cao et al., 2011; DEFRA, 2009; Mozumdera and Berrens, 2007; Sarathchandra et al., 2001). The effects of fertilizers are made visible in the framework by looking at the changes in the biophysical outputs of the ES due to the fertilizer inputs. If the information about changes in the biophysical outputs is unavailable, the costs of the inputs can be used as a proxy instead.

1.5.4 Pesticides

Pesticides are any substances or mixture of substances used to prevent, destroy, or control any pest, including vectors of human or animal disease, and unwanted species of plants or animals causing harm or interfering with agricultural food and agricultural commodity production (FAO, 2002). Pesticides can change the natural landscape, flora, and fauna in such a way that it is more suitable to products that humans want, thus increasing yields. This application has its drawbacks as its application is not precise and thus affects biodiversity by killing beneficial organisms (Stockdale et al., 2006) and it can affect the provision of ES (Sandhu et al., 2008). It is difficult to estimate exactly how much pesticides affect the provision of soil ES. Some estimates, based on the energy input in agriculture, attribute 10% of the overall energy inputs to pesticides (Bardi et al., 2013). Because of the lack of quantifiable biophysical data on the effects of pesticides on the provision of soil ES, the costs of the pesticide inputs are used as a proxy for the ecological subsidy.

1.5.5 Irrigation

Irrigation is the artificial application of water to the soil. It is used in agricultural production where there is insufficient water at the right time for crop production and can also be needed where the particular crop is out of its natural range. There are potential risks associated with irrigation. If either the amount of water or the quality is incorrect, the farmer runs the risk of not only wasting water but also damaging the soil, particularly through salinization (Ashman and Puri, 2002). The costs of the irrigation inputs are used as a proxy for the ecological subsidy. When quantifiable biophysical data on the effects of irrigation on the provision of soil ES are available, they are used.

This section has illustrated how the soil framework connects soil natural capital to the provisioning of soil ES. The next section illustrates the use of the framework.

2. METHODOLOGY

In 2013, the soil ES framework was tested in a pilot study in the Koiliaris watershed, a Critical Zone Observatory (CZO) on the island of Crete, Greece. Koiliaris CZO is a partner in the European Union 7th Framework Programme funded research project SoilTrEC (http://www.soiltrec.eu). The rational for selecting Koiliaris CZO as a location for the pilot study was because of data availability and to facilitate synergies with other parts of the SoilTrEC project. SoilTrEC (Soil Transformation in European Catchments) is focused on addressing certain knowledge gaps regarding important soil ES and functions and the importance for mathematical modeling of these as listed by the EU soil thematic strategy (European Commission, 2006). The research in SoilTrEC is driven by the hypothesis that soil processes can be described along a life cycle of soil development (from pedogenenis to erosion) and that the processes and functions of soil can be quantified during this life cycle (Banwart et al., 2012). The biophysical data used for the soil ES economic valuation were obtained from the Koiliaris CZO.

2.1 Area Description

Koiliaris CZO is located in Koiliaris watershed in the northwestern part of Crete, approximately 25 km east of the city of Chania, and the CZO is operated by the Technical University of Crete. The watershed is 130 km^2 and has a very steep rise in elevation ranging from 0 m AMSL on the coast to 2120 m in the White Mountains. Agriculture is the main land use (Fig. 2) and includes cropland and pasture (35%), olive and orange groves (32.1%), shrub land and brush land (32.3%), and mixed forest (0.6%) (Giannakis et al., 2014).

The Koiliaris watershed is of particular interest in the SoilTrEC project because it is an example of long-term intensive land use through farming and intensive grazing. The soils at Koiliaris are under threat both because of these agricultural practices (Nikolaidis et al., 2013) and because of the effects of climate change (IPCC, 2014a). For detailed description of the hydro-, bio-, and geological aspects of Koiliaris CZO, see Moraetis et al. (2015).

2.2 The Integrated Critical Zone Model

The Integrated Critical Zone (ICZ) model developed by the SoilTrEC (Banwart et al., 2012) project is used to quantify soil functions, and it is an

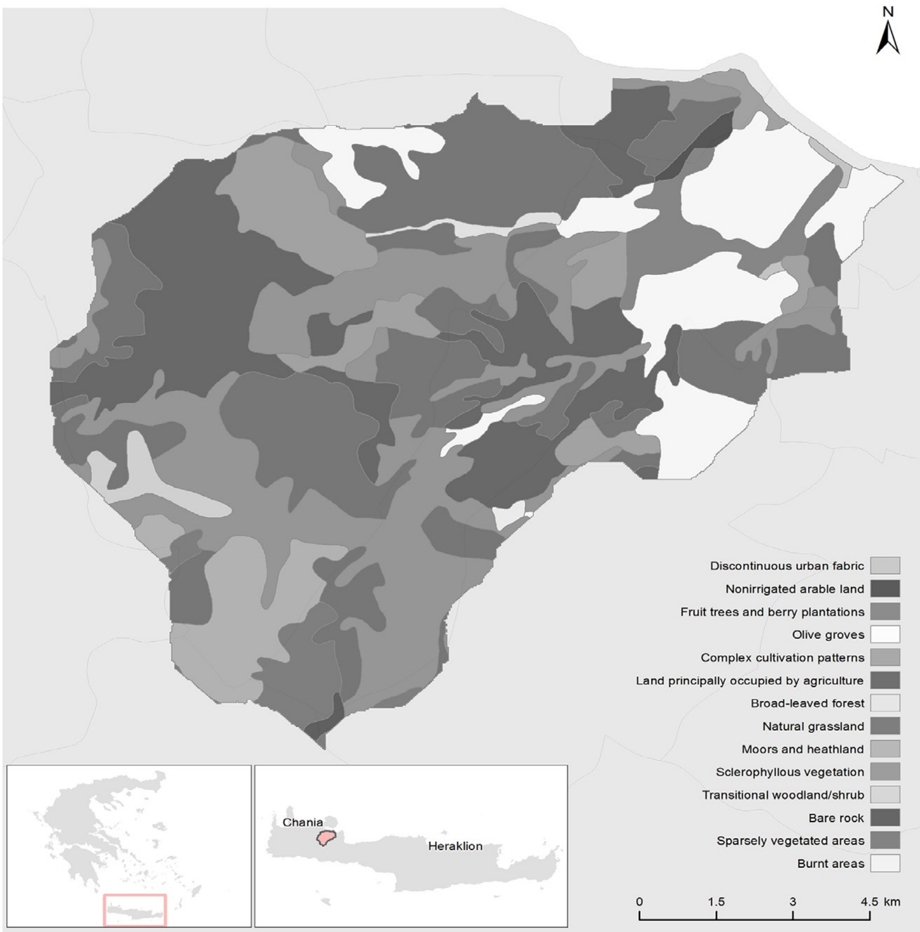

Fig. 2 CORINE land-use map of Koiliaris watershed.

integral part of the proposed framework. The ICZ is a mathematical model that links soil aggregate formation and soil structure to nutrient dynamics, biodiversity, water filtration and transformation, and biomass production. The ICZ model consist of a flow, transport, and bioturbation component; a chemical equilibrium and soil weathering component; a soil aggregation and carbon and nutrient sequestration dynamics component; and a plant growth and nutrient uptake component. The ICZ model was developed in two versions: the 1D-ICZ where it is coupled with HYDRUS-1D and it can simulate soil functions at the soil profile level (Giannakis et al., 2014); and the SWAT-ICZ where it is coupled with the watershed model SWAT to estimate in a semidistributed basis soil functions at the watershed level (Nikolaidis et al., 2013). A brief description of the ICZ follows (Fig. 3) and the details of the model can be found in Giannakis et al. (2017).

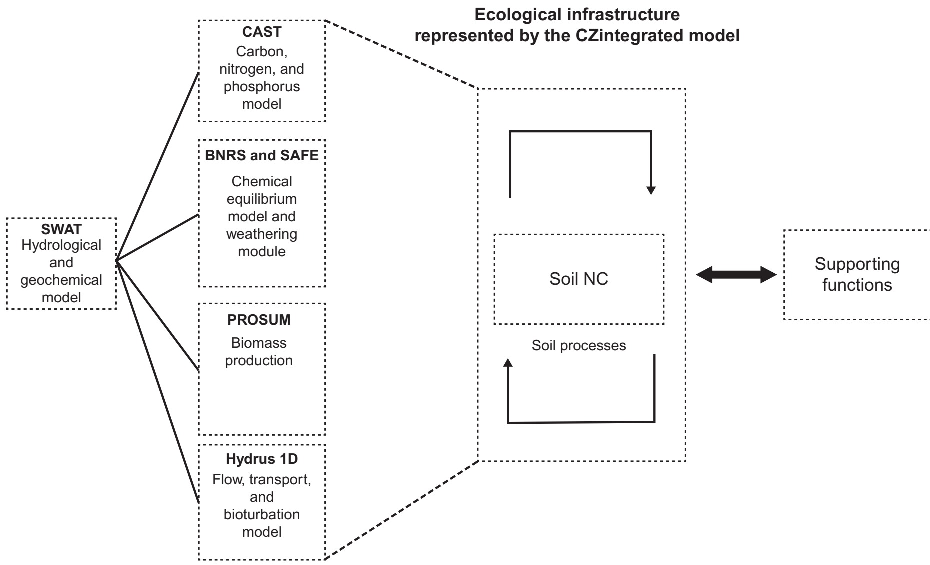

Fig. 3 Submodels of ICZ.

2.2.1 Carbon Sequestration and Nutrient Dynamics

The soil carbon, aggregation, and structure turnover (CAST) model and a simplified mechanistic N and P model (Stamati et al., 2013) were developed for the SoilTrEC project. The CAST model is based on the macroaggregates that are formed around particulate organic matter, followed by the release of microaggregates. In CAST, the transformations of organic matter have been linked with a dynamic model of soil aggregation/disaggregation, a simplified terrestrial ecology model that is comprised of saprophytic fungi, microorganisms (BIO pool), consumers and predators, and a plant/mycorrhizal root/fungi dynamics model. The carbon pools of ROTH-C were adapted to simulate the various organic matter pools which in turn account for the C, N, and P pools using stoichiometric C:N:P ratios. For more information on the CAST model, see Stamati et al. (2013) and Giannakis et al. (2014). This model estimates the dynamics of processes such as vegetation growth and carbon and nitrogen sequestration within the Ei.

2.2.2 Water Filtration and Transformation Dynamics

The chemical equilibrium model BRNS (Regnier et al., 2002) was adapted to account for the effect of water saturation as it varies with time, and exchange equilibria with the gas phase. The adapted code makes use of the SAFE chemical weathering module adapted from the ForSAFE modeling code (Belyazid, 2006). The mineral dissolution kinetics components used within the weathering model are source minerals and the ionic weathering products Ca^{2+}, Na^{+}, K^{+}, Mg^{2+}, Al^{3+}, H_4SiO_4, and PO_4^{3-}. This model relates to soil functions of water filtering and transformation within the Ei.

2.2.3 Plant Dynamics

A simple plant productivity module, PROSUM, was developed and adapted for the SoilTrEC project for inclusion in the 1D-ICZ model. Fixation of carbon (C) by plants is a key process in soil formation. The model is based on the theoretical production ecology principles and predicts the dynamics of key variables (e.g., above- and belowground production of litter C and N; nutrient and water uptake) in response to key physiological drivers and limitation (temperature; availability of light, water, atmospheric CO_2, and the nutrient elements N, P, Ca, Mg, and K; and grazing and management events). Mycorrhizal fungi effects on nutrient acquisition are incorporated into PROSUM on the basis of soil volume explored by the produced microbial biomass, and root exudate fluxes are calculated for use in the weathering component of the 1D transport model.

The ICZ model is coupled in the 1D version to the HYDRUS-1D (for more information on HYDRUS-1D, see Šimůnek et al., 2009) to simulate the flow of water as well as heat and solute transport in the unsaturated zone.

For the upscaled version, the ICZ model is coupled to the watershed model SWAT to estimate soil functions at the watershed level (Nikolaidis et al., 2013). In the SWAT-ICZ model, the SWAT model determines the hydrologic fluxes for every hydrologic response unit (HRU) for three layers (the upper soil, the unsaturated zone, and shallow aquifer) on a monthly basis which are then used by the ICZ model to simulate soil functions. The ICZ model can simulate dynamically soil stocks (such as carbon and nutrient sequestration, biomass production, and bacterial and fauna stocks) and soil fluxes (such as CO_2 emissions and nutrient fluxes to groundwater) and in this way quantify directly the soil functions.

2.3 Soil ES in Koiliaris Watershed

Three soil ES were chosen for evaluation based on the biophysical and modeling data available from our SoilTrEC partners and the soil functions and services emphasized in the SoilTrEC project (Banwart et al., 2012). The services selected were two regulating services (climate regulation and filtering of nutrients and contaminants) and one provisioning (biomass production from crop and livestock).

2.3.1 Filtering of Nutrients and Contaminants

Soils have the ability to control water quality by, to some extent, absorbing, retaining, and chemically transforming solutes (including contaminants), therefore avoiding their release in water bodies such as groundwater, lakes, and rivers. For the Koiliaris watershed, the retention of nitrate (NO_3^-) was assessed as a proxy for the service of filtering nutrients and contaminants. Nitrate retention is of particular interest because nitrates in supplies of drinking water are a well-known health hazard, and their concentration is regulated by the EU and municipalities need to abide by these regulatory limits (Environment, 2000). Another point of interest is that the amount of nitrates in the stream water can be an indication of intensive land use for grazing. Certain parts of the Koiliaris watershed are under intense grazing, especially areas in higher altitudes (Stamati et al., 2011). Well-functioning soils can retain and transform nitrates and prevent their release to waterways. This kind of soil ES therefore provides a benefit to the municipality which would, in the absence of the service, have to remove the nitrates from the drinking water supply. The data on nitrate (NO_3^-) retention were provided on a watershed scale in a tabular and spatial format (ArcGIS) by project partners at the Technical University of Crete, who operate the CZO. The hydrological and chemical aspects of the Koiliaris watershed have been studied in detail (see Kourgialas et al., 2010; Moraetis et al., 2010; Nikolaidis et al., 2013; Sibetheros et al., 2013; Stamati et al., 2011), and the data used here on nitrates (NO_3^-) are derived from that work. The metric representing the service is nitrates flux (NO_3^-) kg ha^{-1} year^{-1}, and it captures the function of soil to retain nitrates (NO_3^-) from the groundwater. The potential maximum amount of leaching in the absence of soil ES vs the current state of nitrates leaching was used. The difference between the potential maximum amount of leaching and current state is the ES that the soil provides. The beneficiaries in this case would include the municipality as there is substantial cost involved cleaning nitrates (NO_3^-) from water sources. Derived from Jónsson and Davíðsdóttir (2016), the method of economic valuation is the avoided cost method, which bases the value of the service on the costs that would be incurred in the absence of the service.

2.3.2 Climate Regulation

The second regulating service assessed is climate regulation, which is defined as the role of soils in regulating global temperatures and precipitation through sequestration of C and N compounds and emissions of the related greenhouse gases. Climate regulation is of concern because of multiple factors. The soils in Crete are under threat because of intensive land use (Nikolaidis, 2011) and climate change (IPCC, 2014b). Soil carbon plays an important role in the stability and fertility of soils (Milne et al., 2015), and soils are an important sink for carbon globally. This can also be a significant source of CO_2 emissions due to mineralization of soil organic matter, if mismanaged (Milne et al., 2015). Soil carbon management is an essential part of climate change mitigation and adaptation, and therefore, soil carbon mineralization was selected as a proxy for climate regulation services. The metric chosen for this service is kg CO_2 ha^{-1} $year^{-1}$, that is, the carbon mineralization rate of the soil. Carbon mineralization can be considered a proxy for CO_2 emissions as it illustrates the amount of CO_2 that is needed to be sequestered at a minimum, in order to keep a carbon neutral balance within the soil system. Data on carbon mineralization (C kg ha^{-1} $year^{-1}$) were provided by project partners at the Technical University of Crete on a watershed scale in a tabular and spatial format (ArcGIS) and were translated into an economic value using the avoided cost method. The avoided cost method applied is based on the cost of GHG mitigation, which refers to the expected cost of mitigating CO_2 emissions. The beneficiaries are regional and national authorities, as they are responsible for carrying out climate change mitigation projects. The use of carbon mineralization rates as a proxy for climate regulations means that costs applied to these rates reflect the maximum value that can be achieved, i.e., from mitigating the entire emissions of CO_2 from soil. Because the data are a proxy for actual CO_2 emissions due to soil carbon mineralization (rather than carbon storage), the value derived from this proxy represents *potential* value of benefits to be achieved by implementing interventions in soil management in order to reduce carbon mineralization rates. It does not reflect the value of climate regulation ES under current practices. Carbon sequestration or net annual C addition to the soil would have been a better proxy for this service, but unfortunately was unavailable for the site at the watershed scale within the current dataset of the SoilTrEC project.

2.3.3 Biomass Production

The third service assessed is biomass production from crops and livestock. Soils provide nutrients, water, and the physical environment for terrestrial biomass production. Humans use biomass in the form of food, wood, fuel, and fiber. Intensive land use in the form of agriculture and livestock production plays an important role in the Crete economy, and this land use impacts the functionality of the soils. Koiliaris watershed produces a plethora of different agricultural products in the form of arable crops, horticulture crops, and livestock. The metric chosen for this service was biomass in kg ha^{-1} $year^{-1}$, that is, the biomass from crop and livestock produced annually in the watershed. The data on crop production were obtained from the Greek national statistical database on agricultural products (2014), and the data on livestock (sheep and goats) were provided by project partners at the Technical University of Crete. The output for each crop and livestock category was linked to the corresponding CORINE (EEA, 2007) land-use category using expert judgment and values from the literature, resulting in a spatial biomass map for the watershed. Four years (2002–2006) of agricultural statistical data for Chania municipality, obtained from the Greek national statistics database (2014), were used to calculate the mean average production rate in units of kg ha^{-1} $year^{-1}$ for each crop category used in the survey for Chania municipality. The biomass map provided the basis for the economic valuation. The data on biomass production, measured in kg per biomass per production type, were provided on a watershed scale in a tabular and spatial format (ArcGIS) and were then translated into economic value. The economic method chosen for converting biomass production values to economic values utilized producers' prices (see Jónsson and Davíðsdóttir, 2016).

3. RESULTS

3.1 Filtering of Nutrients and Contaminants

The following assumptions are made: (1) nitrate retention is a good proxy for the filtering of nutrients and contaminants, (2) the cost numbers for nitrate pollution removal obtained from the European Nitrogen Assessment (Brink and van Grinsven, 2011) are a good representation of the pollution cost in the watershed, and (3) the biophysical data provide a good approximation of this soil ES service. Nitrate retention as a proxy for filtering of nutrients and contaminants service was previously used by Dominati et al. (2014a,b) (Fig. 4).

The European Nitrogen Assessment was the first continental-scale assessment of reactive nitrogen in the environment, and it included the first cost–benefit analysis for the different forms of reactive nitrogen (Sutton et al., 2011). The biophysical data were derived from a modified SWAT model adapted to the Koiliaris watershed. SWAT models are commonly used when modeling hydrological and geochemical properties of watersheds.

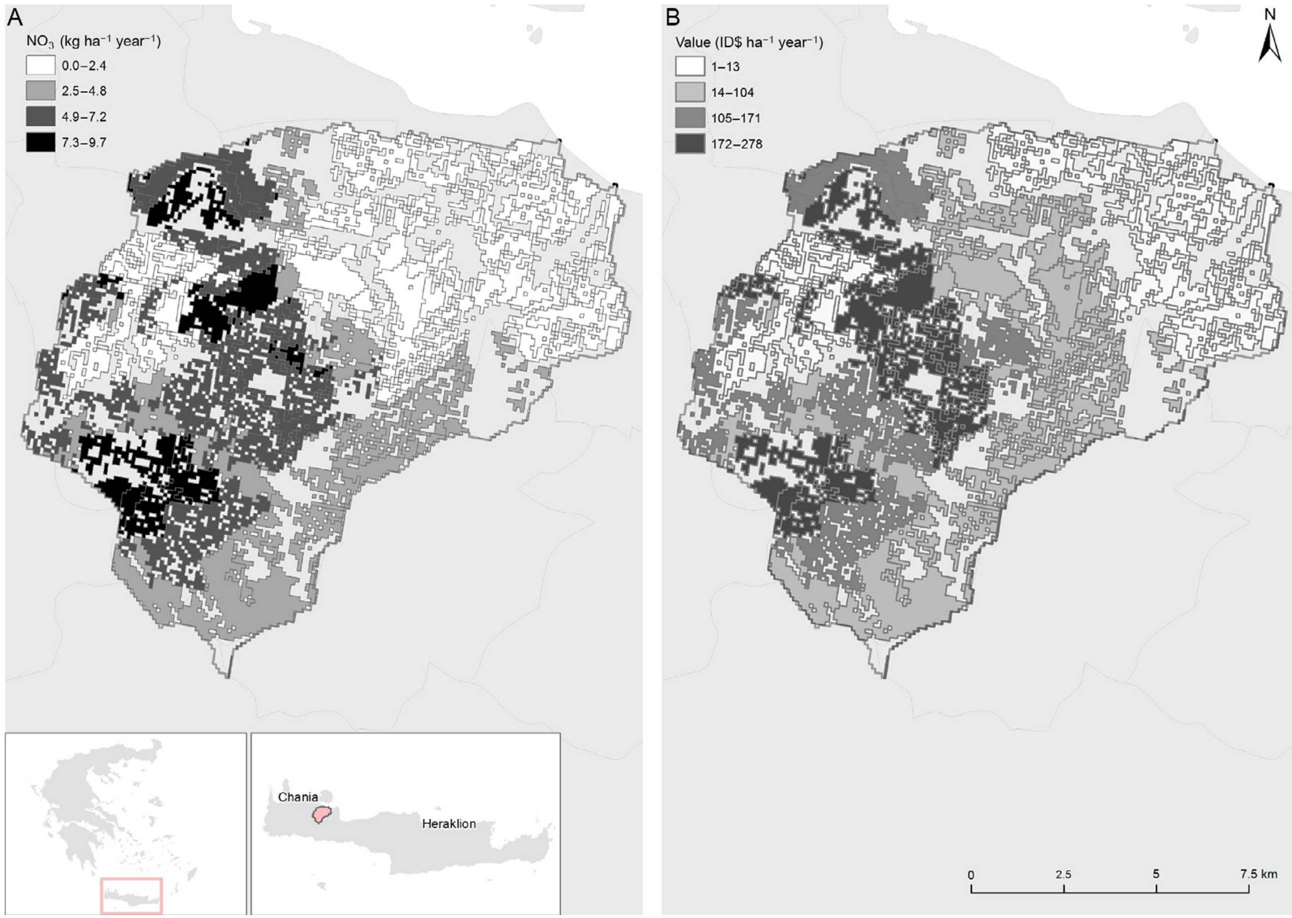

Fig. 4 Filtering of nutrients and contaminants—nitrate retention of soil. (A) Nitrate retention in kg ha^{-1} year^{-1}. (B) Value of nitrate retention in id$ ha^{-1} year^{-1}.

Table 5 Filtering Nutrients and Contaminants in Koiliaris Watershed

Input (kg ha^{-1} year^{-1})	Output (kg ha^{-1} year^{-1})
Nitrogen through fertilizer	Denitrification
NO_3^- through rain	NO_3^- in surface runoff
Organic N in manure from animal grazing	NO_3^- in lateral flow
	NO_3^- leached from the soil

The calculations and results for Koiliaris watershed were reported in Nikolaidis et al. (2013). In the SWAT model, Koiliaris watershed was broken down into 140 HRU. HRU are the smallest units of the SWAT model. The calculations of the HRU as a base unit of the SWAT model involved a number of threshold parameters, including how much each HRU had to contribute to the overall water flow in the watershed. If the HRU contribution was below a certain threshold, the area was ignored (left empty). The input and the output for the calculation of the nitrates flow in the watershed are shown in Table 5.

The soil ES were the difference between the input and the output, thus what the soil was able to retain of the NO_3^- in kg ha^{-1} year^{-1}. The potential leaching range maximum (absence of the soil ES) ranged from 0.54 kg to 26.12 (NO_3^-) ha^{-1} year^{-1}. The current state of leaching ranged from 0.5 to 16.73 (NO_3^-) ha^{-1} year^{-1}. Thus the soil ES retained from 0.03 to 9.65 (NO_3^-) ha^{-1} year^{-1} depending on the HRU. The economic cost numbers are based on estimations on nitrate unit damage cost of the major N pollutants, based on a number of studies reported in the European Nitrogen Assessment (Sutton et al., 2011). This unit damage cost ranged from a low of 6 id\$ to a high of 29 id\$ kg^{-1} N. Based on the soil retention and the unit damage cost, the estimated avoided cost provided by the soil ES filtering of nutrients and contaminants was in the range 0–280 id\$ ha^{-1} year^{-1}.

3.2 Climate Regulation

The following assumptions are made: (1) carbon mineralization can be considered a proxy for climate regulation service, (2) the CO_2 mitigation cost numbers obtained from the Greek national bank are an appropriate representation of the value of the service, and (3) the measured range of mineralization in each soil type is an accurate representation of the mineralization of this soil type in the watershed.

Carbon sequestration and carbon accumulation have been used as a proxy for climate regulation service in number of studies (Dominati et al., 2014a; Glenk and Colombo, 2011; Pretty et al., 2000; Rodriguez-Entrena et al., 2012; Sandhu et al., 2008; Xiao et al., 2005). In this study, data on annual soil carbon accumulation were unavailable, and therefore, data on carbon mineralization rate were used to represent CO_2. Agriculture has a significant impact on the state of the carbon in the soil, and this method might serve as way of estimating the carbon cost of different farming methods, seeing whether the agricultural methods are enhancing the capabilities of the soil system to store carbon or if they are causing the system to release more into the atmosphere, thus creating a disservice (for more on ecosystem disservices and agriculture—see, for example, Kragt and Robertson, 2014; Power, 2010; Zhang et al., 2007). The mitigation cost numbers were obtained from the Greek national bank and represent the range of cumulative mitigation cost of a ton of CO_2 over the period of 2010–2050: "The total average cost of reducing greenhouse gas emission under the Mitigation Scenarios is estimated at between €190 and €240 tons of CO_2 (2008 prices), cumulatively for the period 2010–2050" (Bank of Greece, 2011: p. 435). The mitigation cost numbers were converted into 2012 id$, giving a range of 248–314 id$ per tons of CO_2. The mineralization rate of each soil types was based on measurements of the top 10 cm layer of the soil at three sample locations, for three of the four soil types in the watershed. Table 6 shows the soil type and the mineralization rate in t ha^{-1} $year^{-1}$ and the conversion to CO_2 in t ha^{-1} $year^{-1}$ using the carbon to CO_2 mass multiplier of 3.667.

Fig. 5A shows the carbon mineralization rate in C t ha^{-1} $year^{-1}$ on a watershed scale for the different soils (for more information different soil profiles, see Moraetis et al., 2015) at the Koiliaris watershed representing the value of climate regulation services. Fig. 5B maps the estimated value of the carbon mineralization. By using carbon mineralization rates as a proxy for climate regulation, this approach yields negative values (costs) for current practices and reflects the potential for cost avoidance that could be achieved if agricultural practices were to be implemented to reduce to zero this proxy measurement for CO_2 emissions from soil.

The results for climate regulation are in the range of −2200 to −5610 id\$ ha^{-1} $year^{-1}$, depending on the mineralization rate of the soil types and the mitigation cost range, high vs low (Table 7). There is uncertainty regarding whether the annual carbon sequestration is enough to keep the soil carbon stock in balance, whether soil carbon is increasing, or if it is

Table 6 Soil Type and Carbon Mineralization Rate

Soil Type	**C Mineralization ($t\ ha^{-1}\ year^{-1}$)**	**CO_2 ($t\ ha^{-1}\ year^{-1}$)**
Calcareous rendzines soil and Mediterranean brown		
Low	2.42	8.89
Middle	2.57	9.42
High	4.71	17.27
Brown and red-brown alkaline Mediterranean		
Low	2.46	9.02
Middle	2.77	10.15
High	2.81	10.30
Podzols mixed with forest acid red		
Low	2.75	10.07
Middle	2.83	10.38
High	4.87	17.87

insufficient to match the carbon loss. The measurements that would substantiate this were unavailable at the time of the study.

3.3 Biomass Production From Crop and Livestock

The following assumptions are made: (1) biomass from crop and livestock is a good proxy for the biomass production service, (2) the biomass numbers from crops and livestock allocated to the different CORINE land-use categories are a sound representation of the biomass production from soils in the Koiliaris watershed, and (3) Greek national prices on agricultural commodities obtained from the FAO database (FAO, 2014) are representative of the producers' prices in the Koiliaris watershed. Agricultural products (from crops and livestock) are often used as proxies for biomass production from ecosystems (Dominati et al., 2014a,b; Porter et al., 2009; Sandhu et al., 2010). The biomass numbers from crops and livestock are formed by an expert judgment, by the authors, and by project partners at the Technical University of Crete, on the biomass output of Koiliaris watershed and are the best information available for the area. The expert judgment is based on national agricultural production statistics for the Chania region

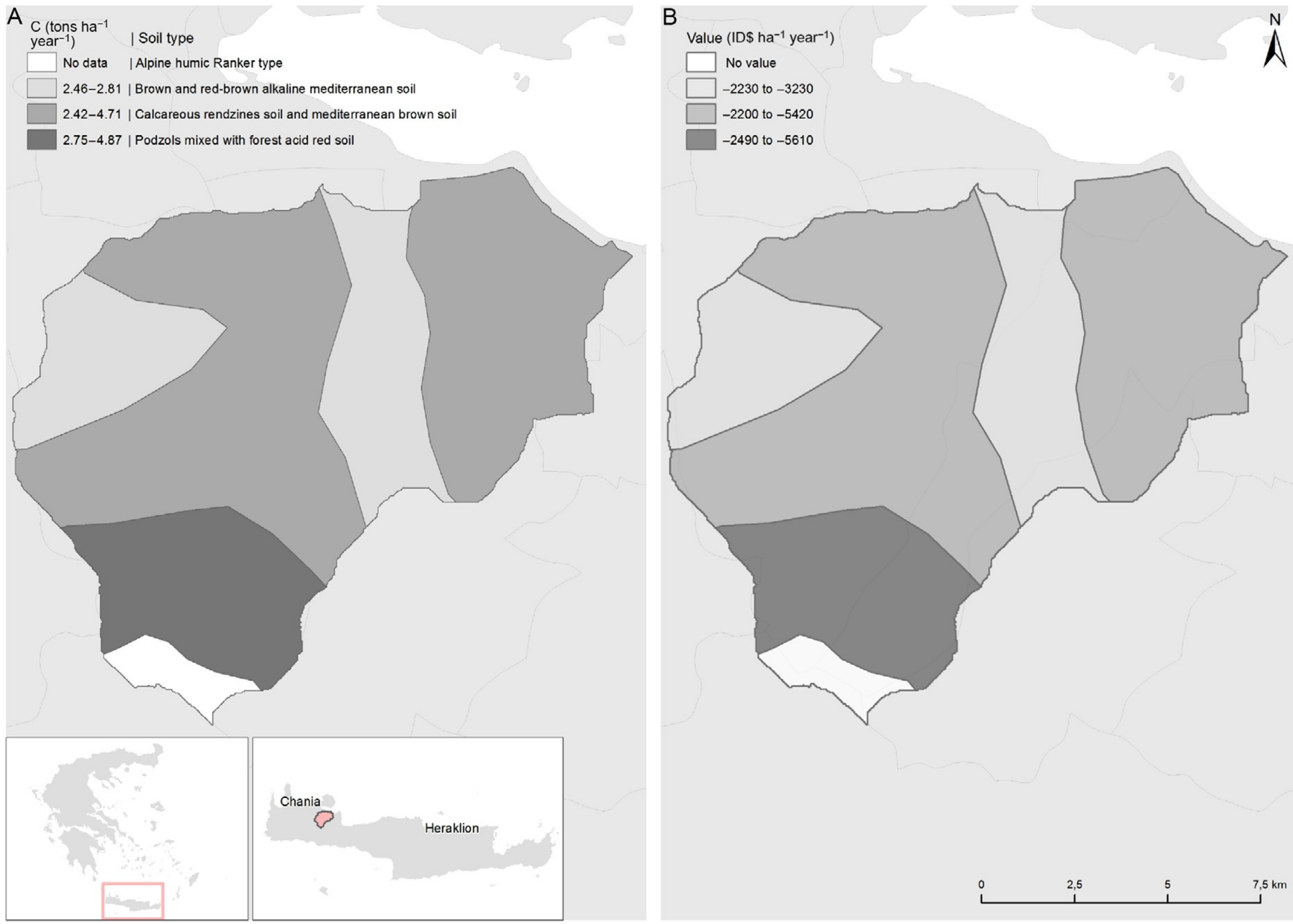

Fig. 5 Climate regulation. (A) Carbon outflow in tons ha^{-1} $year^{-1}$. (B) Value of climate regulation in id\$ ha^{-1} $year^{-1}$.

Table 7 CO_2 and Mitigation Costs

	CO_2 in Soil Types (t ha^{-1} year^{-1})	id\$248 Mitigation Cost Low	id\$314 Mitigation Cost High
Min	8.89	−2200	−2790
Max	17.87	−4430	−5610

(Hellenic Statistical Authority, 2014), taking into consideration current land use and agricultural practices (see Section 2.3.3 for more detail).

The Greek national prices on agricultural commodities represent the most complete dataset on agricultural commodities for the products produced in Koiliaris. Fig. 6B shows the biomass production and it ranges from 0 to 69,148 in kg ha^{-1} year^{-1}.

The estimated average economic values are reported in four main categories:

1. Grassland, shrub land, and pasture 740 id\$ ha^{-1} year^{-1} (270–1580).
2. Nonirrigated arable land 2230 id\$ ha^{-1} year^{-1} (440–4730).
3. Fruit and berry plantations 7500 id\$ ha^{-1} year^{-1} (6100–7580).
4. Olive groves 10,810 id\$ ha^{-1} year^{-1} (6540–10,910).

Price data were obtained from FAO database on Greek agricultural products for a 5-year period (FAO, 2014). Many of the areas are fertilized with inorganic fertilizer, and therefore, a proportion of the economic value can be attributed to the inorganic fertilizer (see Section 1.5.2). The following list is the soil ES contribution for the four main categories (provided by Technical University Crete):

1. Grassland, shrub land, and pasture 740 id\$ ha^{-1} year^{-1} or 100% of the biomass production is attributed to the soil ES.
2. Nonirrigated arable land 1115 id\$ ha^{-1} year^{-1} or 50% of the output is attributed to the soil ES.
3. Fruit and berry plantations 4500–5250 id\$ ha^{-1} year^{-1} or 60–70% can be attributed to soil ES.
4. Olive groves 6480–7560 id\$ ha^{-1} year^{-1} or 60–70% of the economic value can be attributed to soil ES.

Some of the areas on the biomass map (Fig. 6A) have no value data as no economic value could be found for the kind of crops allocated to these areas. It is then safe to assume that the values of the biomass production in the area are underrepresented. There is some uncertainty involved in allocating the different crops from the Greek national statistics on the agricultural output in Chania municipality to the correct CORINE land-use categories to obtain

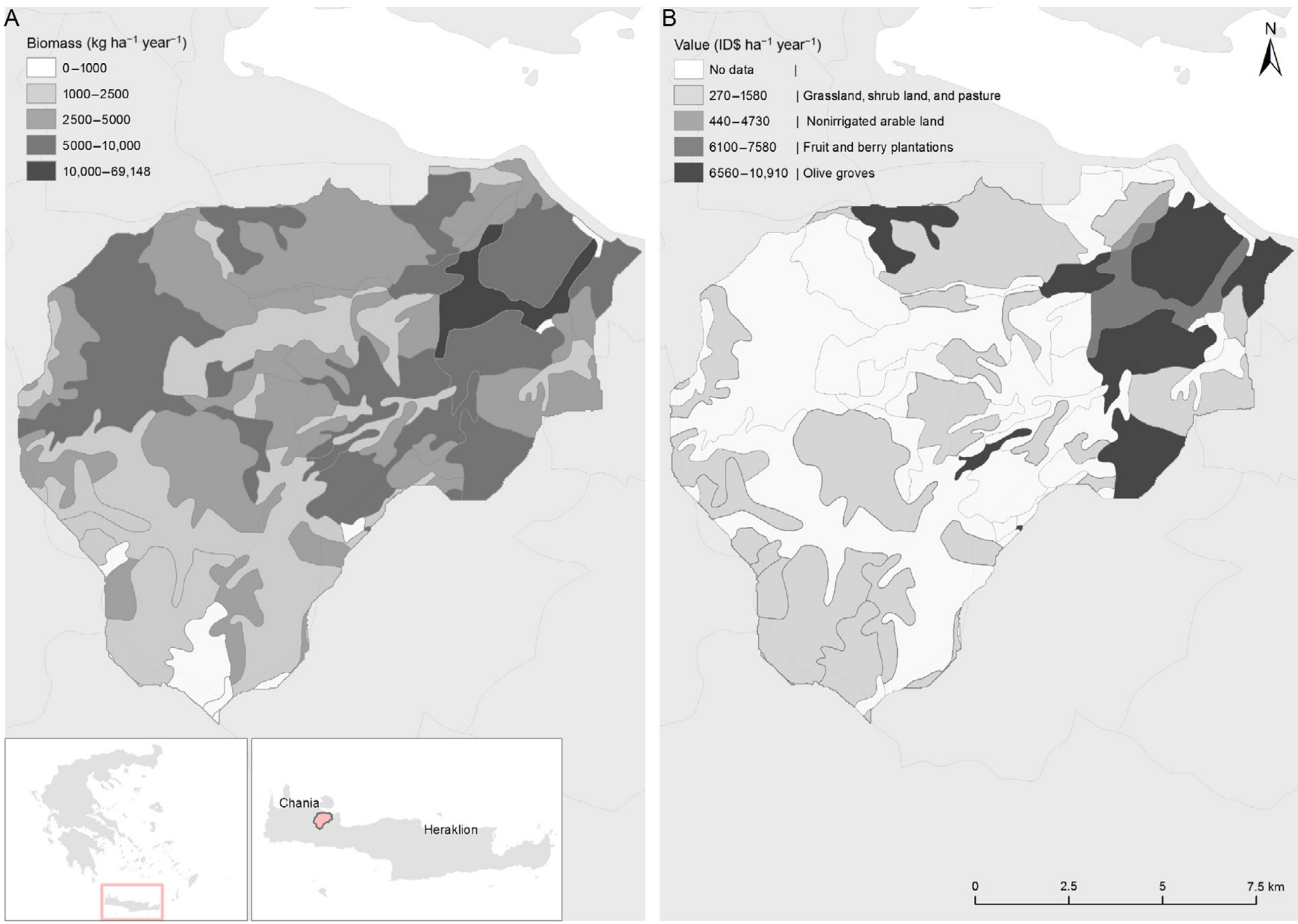

Fig. 6 Biomass provision from crop and livestock. (A) Biomass provision in kg ha^{-1} $year^{-1}$. (B) Value of biomass provision in id$ ha^{-1} $year^{-1}$.

the biomass values. The national statistics is not spatially allocated, thus expert judgment had to be used to allocate the crop categories, and this could potentially lead to misplacement. The allocation of the livestock biomass production faced similar problems as there was incomplete information available regarding the species of sheep and goats in the watershed. As different species produce different quantities of milk and meat, an average number for milk and meat production, for both goats and sheep, was assumed.

3.4 Summary

Owing to different scales of the data, it is difficult to evaluate trade-offs between services. The data on filtering of nutrient and contaminants are the most detailed, followed by the biomass provision and then climate regulation. Nevertheless, the following observation can be made: the value of the biomass production is highest in the lower part of the watershed, where the filtering service is low and the climate regulation service is high. The high values of the filtering service in the watershed are in the intensive grazing areas in the higher altitudes of the watershed, and also grazing in sloping areas. On flat land, as in the lower part of the watershed where most of the crop production takes places, there are low values for the filtering service. On the other hand, where there is intensive grazing on pasture and grassland in the higher altitudes and sloping areas of the watershed, there is a trade-off between the biomass production from livestock and the nitrate flow that results from the grazing activity. Grazing increases the livestock's biomass, but because of its intensity, it potentially decreases the capability of the soil system to retain nitrate as much as it could from the livestock manure. As we base the filtering service on the data that is available, it is only measuring the quantity of $(NO_3^{-})\,ha^{-1}\,year^{-1}$ captured by the soil (the difference between the current output and the maximum output in the absence of soil ES). Therefore, it is hard to generalize about the relationship between inputs, retention, and outputs as it depends on many different soil parameters. For example, an area that receives high input (rain and fertilizer) might have high retention relative to other areas, measured in $kg(NO_3^{-})\,ha^{-1}\,year^{-1}$, and still have high output of nitrate. More data are needed to establish the exact nature of this relationship.

In summary, the value for the three soil ES is:

- Filtering of nutrients and contaminants 0–278 id\$ ha^{-1} $year^{-1}$
- Climate regulation −2200 to −5610 id\$ ha^{-1} $year^{-1}$
- Biomass provision 740–7560 id\$ ha^{-1} $year^{-1}$

All of the services show a substantial range in value with the highest valued services being biomass provision. The regulation services, filtering of nutrients and contaminants, and climate regulation are assumed to be pure soil ES; that is, they are unaffected by the ecological subsidy that influences the biomass provision service.

3.5 Discussion

We have introduced a framework to assess the relevance of sustaining soil functions that links together the concepts of soil natural capital, soil biophysical support functions, and soil ES with beneficiaries and economic valuation. We have defined and categorized the different components of the framework, illustrating their functions within the framework and how these various parts interlink and create benefits for humans, which can be valued economically. As a proof of the concept, the soil framework was applied to a pilot study in Koiliaris watershed on the island of Crete, Greece. The results from the pilot study show that there are important and valuable soil ES in Koiliaris that are delivering benefits to humans. The relative economic values were demonstrated by biomass provisioning, climate regulation, and then filtering of nutrients and contaminants, in the order of highest to lowest value, respectively. When comparing the results to other studies (Jónsson and Davíðsdóttir, 2016), the economic values for biomass provision fall within the range that has previously been reported. However, the values for climate regulation and filtering of nutrients and contaminants are substantially different. Studies have reported values for biomass production ranging from 231 to 22,219 id$ ha^{-1} year^{-1} (Decaens et al., 2006; Dominati et al., 2014a,b; Haley, 2006; Porter et al., 2009; Sandhu et al., 2008) compared to 740 to 7560 id$ ha^{-1} year^{-1} for the biomass provision in Koiliaris. The reported values for filtering of nutrients and contaminants services range from 544 to 6402 id$ ha^{-1} year^{-1} (Dominati et al., 2014a,b), compared to 0 to 278 id$ ha^{-1} year^{-1} for Koiliaris. The climate regulation service from soil has been reported from 2 to 268 id$ ha^{-1} year^{-1} (Dominati et al., 2014b; Glenk and Colombo, 2011; Pretty et al., 2000; Rodriguez-Entrena et al., 2012; Sandhu et al., 2008; Xiao et al., 2005) compared to −2200 to −5610 id$ ha^{-1} year^{-1} at Koiliaris. The most noticeable difference can be seen in the climate regulation service from soils. This is because of the use of the annual mineralization rate of carbon, not the annual addition to its stock, along with the Greek climate mitigation cost estimates. By using annual mineralization rate of carbon as a proxy for CO_2 emission, we are looking at the potential value of the soil to prevent GHG emissions. As we do not know the annual carbon sequestration rate, we assume that it has to be at least equal to the emission rate to keep the system in balance. If GHG emissions from the soil profile would be totally unmitigated, it would result in cost that would have to be mitigated with other means. We thus report negative values for the carbon emitted from the different soil profiles. We use the mitigation cost numbers for climate regulation instead of market price of carbon, it better reflects the social cost of climate change as shown by the WTP for mitigation. The carbon stock exchanges have gone through turmoil during recent years, and there is substantial disparity between what the market is willing to pay for carbon and the social or mitigation cost. Thus, mitigation cost is opted for instead of market prices. If the market price of CO_2, based on the price of 6 id$ per ton of CO_2 in the EU carbon stock exchange in December 2012 (World Bank, 2013), had been used, then the value of the climate regulation services for Koiliaris based on the carbon mineralization rate would have been in the range -63 to -127 id$ ha^{-1} year^{-1}, depending on different soil types.

As in other valuation studies, there are study limitations as follows. Local economic data were difficult to find, and thus cost and price values had to be obtained from national databases and other sources. Although Koiliaris watershed has a plethora of biophysical measurements on the different aspects of the watershed, some of the biophysical data used in the pilot were unavailable as direct measurements or from modeling, and thus expert judgment was needed regarding the allocation and output of the services. Expert judgment was thus used regarding livestock concentration, livestock biomass yields, and crop biomass allocation within the watershed. With an improved land-use map of the watershed, the allocation of livestock and crop biomass can be improved.

With better data on land-use and carbon stocks and flow within the watershed, it would be interesting for future studies to look at the effects of different management decisions on soil ES and how climate change will affect the provisioning and economic value of services in the Koiliaris watershed. This could be done by further linking the soil framework proposed here to the modeling put forward in the SoilTrEC project (Banwart et al., 2012), thus tying further together the biophysical and economic aspect of soils, so both the physical and economic flows are visible.

Despite these limitations, the soil ES framework presented in this chapter can serve as a useful promoter of the sustainable management of soils, as it highlights in a holistic manner the economic benefit of different soil ES, in addition to estimating the economic cost of soil degradation and the associated loss of soil ES. This kind of information should be useful for land managers and policy makers as it gives them a better understanding of the possible trade-offs in land-use management. There are important caveats though. When relying solely on the economic value of soil ES in the context of land management, there is the inherent danger of focusing solely on provisioning services that are usually the highest yielding service in economic value, with possible negative consequences in the long run for other soil functions and services, such as soil biodiversity. This could lead to the mismanagement of the soil resource, and significant economic costs in the long run. Furthermore, a young soil that contains underdeveloped soil functions and services is likely to receive a low value and therefore be deemed unimportant. Therefore, it must be stressed that the economic value of soil ES certainly conveys important information, but should be presented in context with other metrics necessary for sustainable land management (see Jónsson et al., 2016, for examples, of soil sustainability indicators).

3.6 Conclusion

This chapter illustrates the importance of soils and their economic value and introduces a framework for the economic valuation of soil ES. A holistic framework for defining and assessing the economic value of soil ES is an important contribution in the quest for creating usable tools for decision makers that can illustrate the importance of fertile soils, and the economic costs of soil degradation. Soil ecosystems provide multiple benefits for humans but are often neglected in land-use decision-making processes. Soils play a central role in Earth's Critical Zone, which supplies this most important life-sustaining resource, and one way of protecting resources such as soil is to value their economic contribution and include it in land-use decision-making process.

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Soil indicators for sustainable development: A transdisciplinary approach for indicator development using expert stakeholders

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ABSTRACT

Sustainable management of soils is needed to accomplish many of the United Nations' Sustainable Development Goals, but it can be problematic in practice as soils are complex and to manage them sustainably requires the co-operation of multiple stakeholders on various level of society. We present the outcome of a transdisciplinary approach towards indicator development, where we created a set of soil indicators for sustainable development with stakeholder group participation from scientists, policy-makers and soil practitioners. The groups evaluated 49 indicators, through a Delphi survey technique, and selected a set of 30 indicators. Of these 14 were common to all stakeholder groups and represented a final set of core soil indicators for sustainable development. The Delphi survey did suffer from high attrition rate and low response rate, especially among the policy makers, which limits somewhat its findings. Nevertheless, the survey illustrated the usefulness of relevant stakeholder involvement in an indicator development process and the role of survey based instruments in aiding the selection of common indicators, whilst showing the different views of stakeholders groups. Given that the stakeholder groups have to consider a multitude of variables and impacts on soil and may have different focus and management goals in mind, a process such as this can serve as a starting point for discussion between stakeholder groups on various levels of governance about how to manage soil sustainably and help to fulfil the UN's Sustainable Development Goals.

1. Introduction

Soils supply us with food and clean water, they recycle nutrients, decompose contaminants, control water fluctuations, sequester and store significant amount of carbon and provide habitats for the largest number of species of any ecosystems on Earth (Science, 2004; Brevik et al., 2016). Owing to the multiple roles soils have in Earth's ecosystems, humans use them extensively and are thus exerting pressures that have resulted in their degradation (European Commission, 2002; Keesstra et al., 2016). In 2008 there were approximately 1.38 billion hectares of arable land worldwide (FAO, 2010) and up to 5 million hectares are lost every year because of degradation. Soil degradation impacts negatively on the multiple functions of soils (Table 1) and in turn affects more than 1.5 billion people in over 110 countries; 90% of which live in low-income countries (Nellemann, 2009).

In the European Commission's Towards a Thematic Strategy for Soils (European Commission, 2002, 2006) eight main threats to soils are listed (Table 2), illustrating that human activities such as agriculture and forestry practices, industrial activities, road building and soil sealing are major causes of degradation (Turbe' et al., 2010).

With a growing world population, the need for food, clean water and biofuels is on the rise. The demand for food and water is expected to increase by 50% and 30% respectively by the year 2030 (Godfray et al., 2010). Soil degradation presents a serious threat to fulfilling this likely increased demand (Bindraban et al., 2012), and as a result the protection and sustainable management of the soil resource becomes even more important.

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Table 1
Soil functions in Towards a Thematic Strategy for Soils (European Commission, 2002, 2006).

Soil Function Number	Soil functions (SF)
SF1	Food and other biomass production
SF2	Storing, filtering and transformation
SF3	Habitat and gene pool
SF4	Physical and cultural environment for mankind
SF5	Source of raw materials
SF6	Acting as a carbon pool
SF7	Archive of geological and archaeological heritage

Table 2
Soil threats according to the Towards a thematic strategy for soils (European Commission, 2002, 2006).

Soil Threat Number	Soil threats (ST)
ST1	Erosion
ST2	Decline in organic matter
ST3	Soil contamination
ST4	Soil sealing
ST5	Soil compaction
ST6	Decline in soil biodiversity
ST7	Salinisation
ST8	Floods and landslides

1.1. The sustainable development concept

The concept of sustainable development became known in 1987 with the Brundtland Commission's report, *Our Common Future*, and has since then been central to decision-making worldwide (Environment and Development, 1987; MEA, 2005). The 'Brundtland Report' defined sustainable development as development that "*meets the needs of the present without compromising the ability of future generations to meet their own needs*". It centres on the notion of equity, both intra- and intergenerational, and the importance of keeping humanity and its ecological impact within planetary boundaries (UNDESA, 2002; Rockstrom et al., 2009; Steffen et al., 2015).

1.2. Sustainability assessment and indicators

The need for the development of sustainability indicators is clearly set out in Agenda 21 from the Rio UN Summit in 1992 and was taken up by the UN Commission for Sustainable Development (CSD) (Pinfield, 1996). In addition, academics have called for the use of indicators as a means of measuring steps towards sustainability (Bell and Morse, 2008; Easdale, 2016). An indicator demonstrates in what direction something or someone is heading (Ness et al., 2007). By visualizing phenomena and highlighting trends, indicators simplify, quantify, analyse and communicate otherwise complex and complicated information (Warhurt, 2002), and as such they are meant to make complex realities more transparent (Jesinghaus, 1999). Indicators are important tools of sustainability assessment. Sustainability assessment is an iterative, continuing, collaborative process that is an important tool to aid in the shift towards sustainability, helping decision-makers consider the actions that should or should not be taken (UNDESA, 2007). Indicators and assessment tools are therefore essential to reach the various targets and goals relating to sustainable development.

1.3. Sustainable development goals

The United Nations' *Transforming our World: the 2030 Agenda for Sustainable Development* lists 17 Sustainable Development Goals and 169 targets that will stimulate action in critical areas for humanity and the planet until 2030 (United Nations, 2015). Sustainable management of soils has direct relevance for at least half of them and might also be relevant for other goals but in an indirect manner (see Table 1 in Supplementary material). Bouma (2014) and Keesstra et al. (2016) have emphasised the important role of soils in obtaining these goals. It is safe to assume that indicators are needed to report on how sustainably soils are managed in pursuit of the Sustainable Development Goals.

1.4. Soil indicators

Until now indicators for sustainable soil management have mostly been developed within the nature dimension of sustainable development, focusing on the physical, chemical or biological aspects of soils. What has been lacking are the other two dimensions: the social and well-being, and the economic. A plethora of soil indicators for different soil properties, qualities and functions exists in the nature dimension: Arshad and Martin (2002) proposed a minimum data set for soil quality, Marinari et al. (2006) and Fließbach et al. (2007) compared conventional and organic agriculture by using soil properties, and Roldán et al. (2007) used a biological properties of soil approach to compare till and no-till management systems. Rüdisser et al. (2015) proposed linking soil quality indicators with the occurrence of certain soil organism groups and Ritz et al. (2009) looked at national soil monitoring focusing on biological indicators. Muscolo et al. (2015) proposed using biochemical indicators looking at changes in soil organic matter as an early warning system in soil ecosystems. Huber et al. (2008) linked soil indicators directly to soil threats and Thomsen et al. (2012) used soil indicators as chemical stressors in soil systems. These are just a few examples of soil indicators from the literature but as stated before, there is predominance of nature based indicators in the soil sets or frameworks and there is a need to combine indicators from the nature dimension of soil, like soil quality with non-soil biotic, abiotic and socio-economic indicators (Herrick, 2000).

This is the first attempt that we know of that builds a set of soil indicators covering all of the three overarching dimensions of soil sustainable development, using a transdisciplinary approach with active stakeholder participation. In this paper we describe the second stage of developing soil indicators for sustainable development (SIFSD) using a survey based technique involving expert stakeholder involvement.

2. Methods

The complete SIFSD development process is illustrated in Fig. 1. The pre-development aspects, as completed by Jónsdóttir (2011), are indicated in steps 1–5 and that process is not covered in this paper.[1] The pre-development work resulted in 44 theme-based indicators that were used as potential indicators for a Delphi survey that took place in Iceland. Steps 6–8 relate to the Delphi survey outcomes and are the main focus of this paper. Steps 9–10 are only implemented when the indicators are applied to a specific study location and are therefore beyond the scope of this paper.

2.1. The Delphi survey technique

The Delphi survey technique is a vehicle for stakeholder engagement. The technique has been used to address sustainable development issues in many diverse sectors, including mining (Azapagic, 2004), forestry (Sharma and Henriques, 2005), transportation (Mihyeon Jeon and Amekudzi, 2005), environmental

[1] Information on the pre-development work can be found at: http://skemman.is/stream/get/1946/8865/24238/1/jonsdottir_msc_2011.pdf

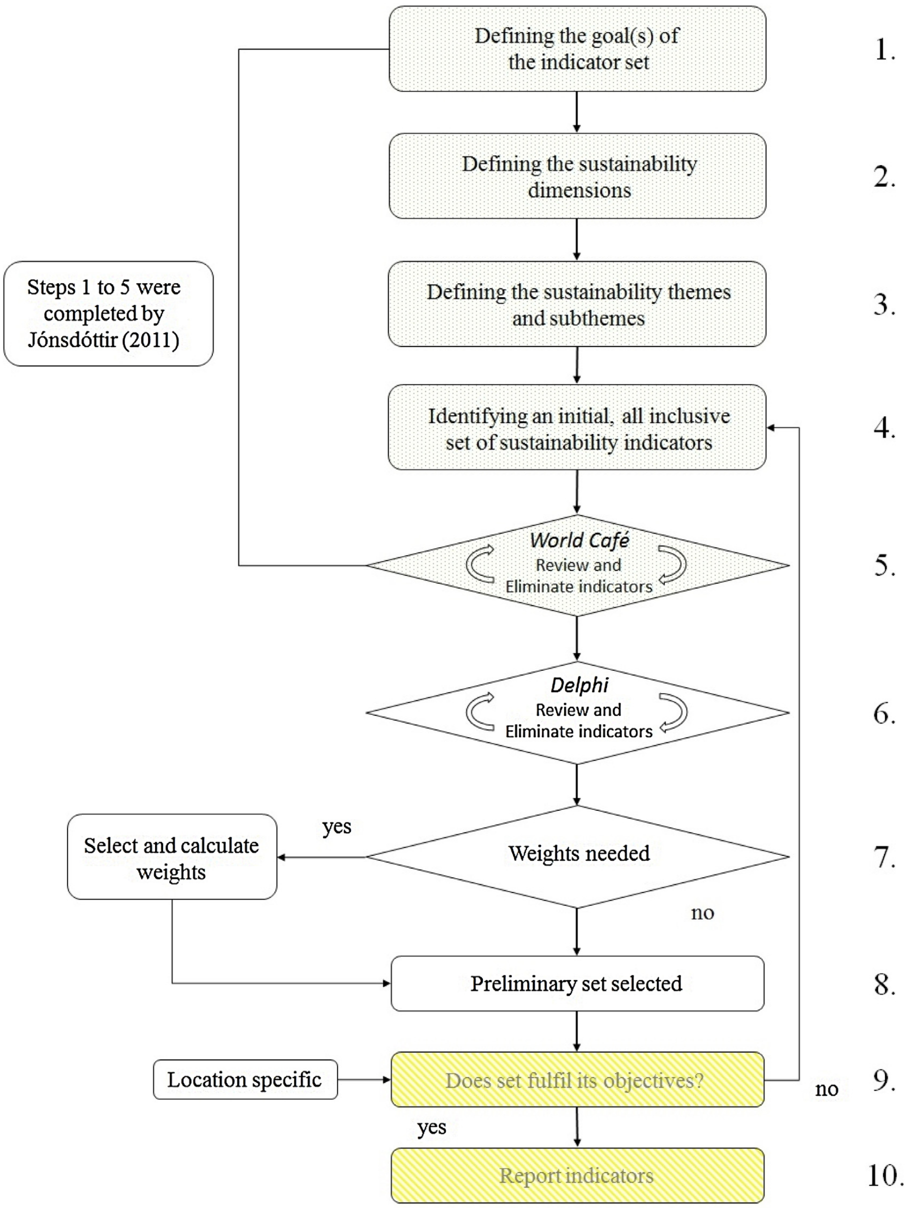

Fig. 1. The SIFSD development process.

management (Bailey et al., 2012) and energy development (Shortall et al., 2015). It is an established survey method for seeking unbiased opinions and consensus on a complex issue, and involves sequential questionnaires answered anonymously by a group of experts. This approach has been used as a consensus-building instrument in fields where opinion is needed from a selected audience with varied views, such as in program planning and policy determination (Gupta et al., 2013; Shortall et al., 2015). The Delphi survey technique has several advantages, including:

- Enables participation of a wide range of individuals with diverse backgrounds;
- Enables participation of individuals located in various regions;
- Provides a more cost effective approach than having on-location workshops;
- Ensures anonymity and thereby reduces the probability of personal conflicts affecting group dynamics; and
- Minimizes bandwagon effects as participants cannot see each other's voting.

Conversely, disadvantages may include a high time commitment; potential hasty decisions by participants as they must vote on each indicator; the risk of a lack of accountability for opinions through anonymity; or the potential for low response rates.

The Delphi technique consists of a structured written survey that is sent to participants, seeking both evaluations (scores) and comments related to specific indicators (Gupta et al., 2013; Shortall et al., 2015). During the first rounds new indicators also can be suggested by the survey participants. During each round of a Delphi survey, the participants give each indicator a score on a scale from 1 – 5, reflecting how relevant it is with 1.00 being Irrelevant, 2.00 Somewhat irrelevant, 3.00 Neither relevant nor irrelevant, 4.00 Somewhat relevant and 5.00 being Extremely relevant. The participants can also give an optional comment in response to each indicator. After each round, the indicator's scores and comments are incorporated into the next round of the survey by facilitators if their scores are high enough and if consensus is on their relevance as reflected by the standard deviation of the scores received. In general, indicators that receive a mean score below 3.00 and with a low consensus are discarded after each round. This process is repeated a few times, until a broad consensus has been reached among the participants for the suggested indicators (Shortall et al., 2015). After the final round, if the mean score minus the standard deviation is less than 3.00, the indicator is rejected. The indicators can then be further reviewed to identify those common to all stakeholder groups and the remainder, which are stakeholder group specific. One drawback of the method is that during the survey process, the mean score, the standard deviation and comments from the participants are all taken into consideration by the facilitators when deciding whether an indicator passes to the next round of the survey. This involves subjective value judgement by the facilitators in some cases.

In this study the Delphi survey was used to engage stakeholder groups and to help to identify i) a set of core soil indicators that all stakeholder groups agree on are important, and ii) a set of satellite indicators that are stakeholder group specific.

2.2. Method implementation

The Delphi survey took place in September/October 2014 in Iceland and ran for three rounds, each taking a week. The survey was distributed via the online survey management system Survey Monkey (surveymonkey.com). Three stakeholder groups participated: scientists (in Iceland and in the SoilTrEC project), soil practitioners (in Iceland), and policymakers (in Iceland). A stakeholder mapping exercise was carried out at the start of the research, with the intention of identifying individuals and organisations that might have an interest in the indicators, or considerable knowledge thereof. Stakeholders were selected based on different characteristics, as recommended in the Australian government stakeholder engagement practitioner handbook (Australian Government, 2008) namely: 1) Responsibility – Stakeholders to whom soil sustainability indicators have a responsibility, such as the local community, the general public, community representatives, environmental organisations and NGOs, local businesses and future generations 2) Influence – Stakeholders with influence or decision-making power when it comes to soil sustainability indicators, such as different levels of government 3) Proximity – Stakeholders that had participated in the first stages of the project, and that have most interaction with soil sustainability, such as researchers, different stages of the government and farmers of various kind 4) Dependency – Stakeholders who are directly or indirectly dependent on soil sustainability, such as farmers of various kind, researchers or food producers 5) Representation – Stakeholders who through regulation, custom or culture can legitimately represent a constituency when it comes to soil sustainability, such as NGOs representing the environment, local authorities, trade unions or local leaders and 6) Policy and strategic intent – Stakeholders that are directly or indirectly address by soil policy or practice, such as farmers, food producers, NGOs or financiers.

Table 3
Stakeholder groups.

	Invited	Round 1	Round two	Round three
Scientists	**27**	15	14	12
- Educational	5			
- SoilTrEC	20			
- Students	2			
Soil practitioners	**41**	17	8	6
- Farmers	13			
- Farmer's association	3			
- NGOs	11			
- Private company	14			
Policy-makers	**20**	11	6	3
- Government institutions	10			
- Policy making/Government	10			
Total	**88**	43	28	21

Initially, 220 people were contacted via email and telephone, prior to being sent a formal invitation to participate via email. All had been identified due to their expertise or work experience within the broad field of soil sciences. Of these, 88 people agreed to participate in the survey. The participants were then invited to an introductory meeting on Wednesday September 3rd, 2014, at the University of Iceland, or to join in that meeting online via Skype. About 20 people attended or joined via Skype.

In the first round an invitation was sent out to 88 people. 43 people finished the first round, 28 the second round and 21 the third (Table 3). Answers from only those who fully completed a round were included in the analysis.

3. Results

3.1. All stakeholder groups

During the three rounds the participants suggested five indicators in addition to the initial 44, and so the total number of indicators evaluated for three rounds was 49 (see Table 2 in Supplementary material). The Final SIFSD set resulted in 30 indicators that were selected by the stakeholder groups. In the nature dimension 17 of 20 indicators were accepted, in the society and well-being dimension 7 of 16 indicators were accepted, and in the economy dimension 6 of 16 were accepted.

3.2. Nature dimension

Of the seventeen indicators selected by the stakeholder groups from the nature dimension, fourteen indicators were included in the soil properties theme, two in the atmosphere theme, and one in the biodiversity theme. The highest scoring indicator in the nature dimension after round three was *Change in total soil organic matter*. The lowest scoring indicator was *Soil iron oxides content compared to reference value* (Table 4).

3.3. Society and well-being dimension

Of the seven indicators selected by the stakeholder groups from the society and well-being dimension, three indicators were in the Institution framework and capacity theme, two in Awareness and

Table 4
Nature indicators scores after each round (R1 – R3), statistics and results for all participants.

	[a]Theme	Sub – Theme	[b]Indicator	[c]R1	R2	R3	[d]Results
NATURE	Atmosphere	Atmosphere	Net carbon sequestration in soil	4.49 (0.55)	4.57 (0.57)	4.48 (0.68)	Accepted
			Extreme weather events	3.60 (1.22)	3.64 (1.03)	3.81 (0.75)	Accepted
			Temperature daytime temperature during the growing season	N/A	3.29 (1.24)	3.48 (0.93)	Rejected
	Biodiversity	Biodiversity	Pedodiversity	3.95 (0.90)	3.82 (0.77)	4.00 (0.89)	Accepted
	Soil Properties	Physical	Aggregate diversity	4.38 (0.79)	4.29 (0.76)	4.25 (0.64)	Accepted
			Bulk density	4.16 (0.75)	4.21 (0.83)	4.24 (0.77)	Accepted
			Change in topsoil depth	4.33 (0.97)	4.37 (0.69)	4.10 (0.89)	Accepted
			Soil sealing	4.17 (1.03)	4.44 (0.70)	4.38 (0.74)	Accepted
			Strata composition and buffer capacity	N/A	4.07 (0.94)	3.76 (1.14)	Rejected
			Soil erosion	N/A	3.96 (1.00)	4.19 (0.93)	Accepted
		Chemical	Change in cation exchange capacity (CEC)	4.14 (0.86)	3.93 (0.73)	3.85 (0.75)	Accepted
			Soil contamination	4.44 (0.96)	4.46 (0.96)	4.38 (1.12)	Accepted
			Change in topsoil pH	4.14 (1.01)	4.36 (0.78)	4.33 (0.73)	Accepted
			Soil iron oxides content compared to reference value	N/A	3.61 (0.69)	3.24 (1.14)	Rejected
		Biological	Change in microbial biomass	4.17 (1.08)	4.39 (0.57)	4.24 (0.94)	Accepted
			Change in and absolute level of net N mineralization	4.16 (1.04)	4.21 (0.79)	4.24 (0.62)	Accepted
			Soil protective cover	4.44 (0.93)	4.50 (0.75)	4.24 (0.77)	Accepted
			Change in total soil organic matter (TSOM)	4.70 (0.56)	4.64 (0.49)	4.48 (0.68)	Accepted
			Change in flora diversity above ground	N/A	4.14 (0.71)	4.30 (0.57)	Accepted
			Change in fauna diversity above ground	N/A	4.04 (0.79)	4.14 (0.73)	Accepted

[a] Themes within one of the overarching dimensions of sustainable development.
[b] Proposed soil indicator for sustainable development.
[c] Rounds one to three with mean score, standard deviation in parenthesis.
[d] Results after round three considering score, standard deviation and comments from participants.

public participation, one in Health, and one in Demographics. The highest scoring indicator in this dimension was *Public awareness of the value of soil,* which also was the highest scoring indicator in the survey after round three. The lowest scoring indicator was *Age diversity in rural areas* (Table 5). Two indicators, *Armed conflicts* and *contaminated Soils*, were moved to the nature dimension after round two after suggestions from many of the participants in the survey.

Table 5
Society and well-being indicators scores after each round (R1 – R3), statistics and results for all participants.

	[a]Theme	Sub - Theme	[b]Indicator	[c]R1	R2	R3	[d]Results
SOCIETY AND WELL-BEING	Institutional framework and capacity	Governance level	Access to information and justice	3.53 (1.32)	3.54 (1.29)	3.33 (0.86)	Rejected
			Government policies	3.85 (1.15)	4.21 (1.26)	4.10 (0.89)	Accepted
			Land tenure security	3.74 (0.82)	3.71 (0.98)	3.62 (1.12)	Rejected
		Science, technology and education	Expenditure on soil related research and development	3.77 (1.31)	4.04 (1.07)	4.10 (1.00)	Accepted
			Literacy	3.21 (1.49)	3.43 (1.20)	3.43 (1.40)	Rejected
			Education on sustainability	3.91 (1.34)	4.18 (1.02)	4.05 (0.92)	Accepted
	Awareness and public participation	Awareness and public participation	Public awareness of the value of soil	4.12 (1.18)	4.43 (0.92)	4.52 (0.68)	Accepted
			Public participation	3.72 (1.32)	3.86 (1.08)	3.95 (0.86)	Accepted
			Public access to nature areas	N/A	3.25 (1.17)	3.52 (0.98)	Rejected
	Health	Health	Human health (healthy life years)	3.37 (1.29)	3.39 (1.50)	3.33 (1.02)	Rejected
			Bioavailability of essential major and trace elements	3.51 (1.28)	4.32 (0.90)	4.00 (1.00)	Accepted
			Suicide rate of farmers	2.53 (1.32)	N/A	N/A	Rejected
	Demographic	Demographic	Age diversity in rural areas	3.19 (1.62)	3.32 (1.12)	3.19 (1.08)	Rejected
			Population growth	3.77 (1.32)	3.96 (1.04)	4.00 (0.77)	Accepted
	Security	Security	Armed conflicts	3.47 (1.42)	3.71 (1.21)	N/A	Moved*
			Contaminated soils	3.70 (1.30)	4.25 (0.89)	N/A	Moved*

[a] Themes within one of the overarching dimensions of sustainable development.
[b] Proposed soil indicator for sustainable development.
[c] Rounds one to three with mean score and standard deviation in parenthesis.
[d] Results after round three considering score, standard deviation and comments from participants. *The indicator was combined with Soil contamination in the nature dimension after recommendations from participants.

3.4. Economy dimension

Of the six indicators selected by the stakeholders in the economy dimension, four indicators were in the theme Industry specific indicators for agriculture and forestry, one in Consumption patterns, and one in economic value of soil ecosystem services. The highest scoring indicator after round three in the economy dimension was *Soil salinity due to irrigation* and the lowest scoring indicator was *Labour intensity*, which was also the lowest scoring indicator overall after round three. Based on suggestions from participants, two indicators were merged with others within the economy dimension: *Economic loss due to loss of soil ecosystem services* was merged with *Economic value of soil ecosystem services* and *Diversity in land management* was merged with *Change in land use diversity* (Table 6).

3.5. Core indicators and satellite sets

Table 7 shows the core indicators (highlighted) that are common to all stakeholder groups, the soil functions that the indicators represent and the threats they address. The core indicators selected by the stakeholder groups cover all the functions and threats, with most of them covering multiple functions and threats (see Table 2 in Supplementary material provided for more information on each indicator). Table 7 also shows the metrics for measuring all the indicators suggested, along with the scale of the measure and frequency. The metric, scale of measurement and the frequency of measurement are based on the outcome of stakeholder inputs both from Jónsdóttir (2011) and from participants in the Delphi survey.

Some of the stakeholder groups included particular indicators that the other groups did not contain, and are referred to here as satellite indicators. The stakeholder group *policy makers* had the most satellite indicators, eight in total. The group had three in the economy dimension, five in the society and well-being dimension, and none in the nature dimension. In the economy dimension two of the three indicators for the policy makers belonged to the consumption patterns theme but different sub-themes. In the society and well-being two of the five indicators belonged to the Institutional framework and capacity theme but different sub-themes. The satellite set for soil practitioners had two indicators: *Extreme weather events* and *Strata composition and buffer capacity,* both belonging to the nature dimension. The satellite set for scientists contained only one indicator: *Expenditure on soil related research and development.*

4. Discussion

Based on extensive stakeholder engagement, the final results consist of a core set of 14 soil indicators that all stakeholder groups deemed important in the context of sustainable development, in addition to satellite sets specific to each stakeholder group. The indicator set is markedly different from the soil indicator sets mentioned in section 1.4 as it covers all the dimension of sustainable development, not solely the nature dimension like the other sets do. The results show that the opinions of the

Table 6
Economy dimension indicators scores after each round (R1 – R3), statistics and results for all participants.

	[a]Theme	Sub - Theme	[b]Indicator	[c]R1	R2	R3	[d]Result
ECONOMY	Economic value of soil ecosystem services	Economic value of soil ecosystem services	Economic value of soil ecosystem services	3.77 (1.17)	3.46 (1.10)	4.00 (0.89)	Accepted
			Economic loss due to loss of soil ecosystem services	3.81 (1.17)	3.32 (1.16)	3.81 (0.87)	Merged*
	Consumption patterns	Land use	Change in land use diversity	3.67 (1.25)	3.61 (1.23)	4.00 (0.71)	Accepted
			Local food consumed	N/A	3.37 (1.31)	3.62 (1.07)	Rejected
		Waste	Waste generation intensity	3.35 (1.49)	3.32 (1.36)	3.33 (1.02)	Rejected
			Organic waste composted and returned to soil	3.74 (1.35)	3.75 (1.21)	4.10 (1.14)	Rejected
	Industry specific indicators for agriculture and forestry	Productivity	Yield, given no change in fertilization	4.02 (1.24)	3.85 (1.03)	4.05 (0.59)	Accepted
		Economic viability	Return on equity (ROE)	3.40 (1.31)	2.96 (1.20)	N/A	Rejected
			Debt to asset ratio	2.88 (1.24)	N/A	N/A	Rejected
		Input intensity	Energy returns on investment (EROI)	3.21 (1.35)	3.54 (1.17)	3.43 (0.87)	Rejected
			Fossil energy intensity	2.81 (1.31)	N/A	N/A	Rejected
			Chemical fertilizer use intensity	3.77 (1.34)	3.96 (1.00)	4.19 (1.03)	Accepted
			Pesticide use intensity	4.05 (1.27)	4.18 (1.16)	4.19 (0.81)	Accepted
			Soil salinity due to irrigation	3.95 (1.23)	3.89 (0.96)	4.24 (0.70)	Accepted
			Labour intensity	3.40 (1.29)	3.29 (1.15)	3.00 (1.03)	Rejected
		Industry practices	Diversity in land management	3.95 (1.19)	4.04 (1.07)	N/A	Merged**

[a] Themes within one of the overarching dimensions of sustainable development.
[b] Proposed soil indicator for sustainable development.
[c] Rounds one to three with mean score and standard deviation in parenthesis.
[d] Results after round three considering score, standard deviation and comments from participants.
* The indicator was merged with Economic value of soil ecosystem services as it is representing the same thing.
** The indicator was merged with Change in land use diversity after recommendations from participants.

Table 7
Soil indicators for sustainable development. Core set of soil indicators is highlighted.

Dim.[a]	Theme[b]	Sub-theme[c]	Indicator[d]	Metrics[e]	Link to soil functions (Table 1)[f]	Link to soil threats (Table 2)[g]	Scale[h]	Measurement frequency[i]	SC[j]	SP[k]	PM[l]
NATURE	Atmosphere	Atmosphere	Net carbon sequestration in soil	C equivalent gC/m^2/yr.	SF1–SF3, SF6	ST1 – ST6	Plot	Annual, seasonal, monthly	X	X	X
			Extreme weather events	Days/season, quantity/ intensity	SF1 – SF7	ST1 – ST3, ST6 – ST8	Plot	Seasonal		X	
	Biodiversity	Biodiversity	Pedodiversity	Number of soil classes within an area	SF1 – SF7	ST1-ST8	Plot	Annual or less frequently	X	X	X
	Soil Properties	Physical	Aggregate diversity	Mean weight Diameter of various aggregates, and aggregate diversity measured with the Shannon Wiener index	SF1 – SF3, SF6	ST1, ST2, ST5, ST6, ST8	Plot	Annual, seasonal	X	X	X
			Bulk density	g/cm3	SF1 – SF3, SF6	ST1, ST2, ST5	Plot	Annual or less frequently	X	X	X
			Change in topsoil depth	cm	SF1 – SF7	ST1, ST2, ST6, ST8	Plot	Annual or less frequently	X		X
			Soil sealing	% of total land area, excluding land under water and ice	SF1 – SF7	ST4, ST5, ST8	Plot	Annual	X	X	X
			Strata composition and buffer capacity	Absorption and permeation in the strata, and chemical composition. Field capacity. Water retention.	SF1, SF2	ST2, ST4, ST5	Plot	Annual		X	
			Soil erosion	µg/m3 of particulate	SF1 – SF7	ST1, ST2, ST6	Plot, national	Hourly	X	X	X
		Chemical	Change in cation exchange capacity (CEC)	C mole/kg.	SF1 – SF3, SF6	ST2, ST3, ST6, ST7	Plot	Once every 5 years		X	X
			Soil contamination	Concentrations in topsoil	SF1 – SF6	ST2, ST3, ST6	Plot	Annual		X	X
			Change in topsoil pH	pH	SF1 – SF3, SF5 – SF7	ST2, ST3, ST6, ST7	Plot	Several times a year	X	X	X
		Biological	Change in microbial biomass	C ($mg\,kg^{-1}$)	SF1 – SF4, SF6	ST2, ST3, ST6	Plot	Annual		X	X
			Change in and absolute level of net N mineralization	mg/kg soil	SF1 – SF3, SF6	ST2, ST6	Plot	Annual, seasonal	X	X	X
			Soil protective cover	% per season	SF1 – SF4, SF6	ST1, ST2, ST6	Plot	Annual, seasonal	X		X
			Change in total soil organic matter (TSOM)	%	SF1 – SF4, SF6	ST2, ST6	Plot	Annual	X	X	X
			Change in flora diversity above ground	Shannon's index and Simpson's index	SF1 – SF3, SF6	ST6	Plot	Annual	X	X	X
			Change in fauna diversity above ground	Shannon's index and Simpson's index	SF1 – SF3, SF6	ST3	Plot	Annual	X	X	X
Number of indicators in nature dimension									*13*	*16*	*16*
ECONOMY	Economic value of soil ecosystem services	Economic value of soil ecosystem services	Economic value of soil ecosystem services	€	SF1 – SF7	ST1 – ST8	Plot, national	Annual	X		X
		Land use		% of land cover	SF1 – SF7	ST1 – ST8	Regional		X		X

Table 7 (*Continued*)

Dim.[a]	Theme[b]	Sub-theme[c]	Indicator[d]	Metrics[e]	Link to soil functions (Table 1)[f]	Link to soil threats (Table 2)[g]	Scale[h]	Measurement frequency[i]	SC[j]	SP[k]	PM[l]
	Consumption patterns		Change in land use diversity					Annual, seasonal			
			Local food consumed	The percentage of local food (produced within certain radius from sales point) bought by consumers	SF1, SF4	ST2	Plot	Annual			X
		Waste	Waste generation intensity	ha	SF1 – SF5	ST3, ST4, ST6	Local	Annual			X
			Organic waste composted and returned to soil	%	SF2 – SF6	ST2, ST6	National, larger international regions	Annual		X	X
	Industry specific indicators for agriculture and forestry	Productivity	Yield, given no change in fertilization	Tonnes/ha	SF1, SF2, SF6	ST1 – ST8	Farm	Annual	X	X	X
		Input intensity	Energy returns on investment (EROI)	kcal out/kcal in	SF1, SF5	ST1 – ST8	Farm, local	Annual			X
			Chemical fertilizer use intensity	Kg/ha per yield (kg) by crop type/ha	SF1 – SF3, SF6	ST2, ST3, ST6	Farm, national, larger international regions	Annual	X		X
			Pesticide use intensity	Kg/ha per yield (kg) by type/ha	SF2 – SF4	ST2, ST3, ST6	Farm	Annual	X	X	X
			Soil salinity due to irrigation	g/kg (Na, K, Ca, Mg salts)	SF1 – SF3, SF6	ST1 – ST3, ST6 ST7	Farm	Annual	X		X
Number of indicators in economy dimension									*6*	*3*	*10*
SOCIETY & WELL-BEING	Institutional framework and capacity	Governance level	Access to information and justice	Has the Aarhus Convention on Access to Information, Public participation in Decision-making and Access to Justice in Environmental Matters been ratified or not	SF1 – SF7	ST1 – ST8	National	Annual or less frequent			X
			Government policies	Existence of soil related policies	SF1 – SF7	ST1 – ST8	National	Annual or less frequent	X		X
			Land tenure security	Long term (> 30 years) versus short term (< 30 years)	SF1 – SF7	ST1 – ST8	National	Annual or less frequent			X
		Science, technology and education	Expenditure on soil related research and development	% of overall research expenditure	SF1 – SF7	ST1 – ST8	National	Annual	X		
			Literacy	% population	SF1 – SF7	ST1 – ST8	National	Annual or less frequent			X
			Education on sustainability	%	SF1 – SF7	ST1 – ST8	National	Annual	X		X
	Awareness and public participation	Awareness and public participation	Public awareness of the value of soil	% of population, measured with survey	SF1 – SF7	ST1 – ST8	National	Annual or every five years	X	X	X

		Public participation	% of population, measured with survey	SF1 – SF7	ST1 – ST8	National	Annual			X
		Public access to nature areas	Proximity to cities, amount of public and national parks	SF1 – SF7	ST1 – ST8	National	Seasonal	X		
Health	Health	Human health	Healthy life years	SF1, SF2	ST1, ST2, ST6	National	Annual			X
		Bioavailability of essential major and trace elements	Mg/kg	SF1, SF2	ST1, ST2, ST6	Farm	Annual or less frequent		X	X
Demographic	Demographic	Population growth	%	SF1 – SF7	ST1 – ST8	National	Annual	X		X
Number of indicators in society and well-being dimension								*6*	*2*	*10*
Total number of indicators in each group								**25**	**21**	**36**

[a] Dimensions of sustainable development.
[b] Themes within the selected dimensions of sustainable development.
[c] Sub-theme within one of the themes.
[d] soil indicator for sustainable development.
[e] What is used to measure the indicator.
[f] Soil functions in the Soil Thematic.
[g] Soil threats according to the Soil Thematic.
[h] The measurement scale.
[i] How often measured.
[j] Scientists.
[k] Soil practitioners.
[l] Policy makers.

stakeholder groups somewhat vary (Table 7) is to be expected, as the indicators considered important by one group may not be deemed as important by another as their level of decision-making differs. The most obvious difference occurs between the soil practitioners and policy makers. On the one hand the selection of the soil practitioner's group shows a clear focus on the soil system itself, as 16 of the 21 indicators are placed in the nature dimension, leaving only 6 indicators that are divided between the other two dimensions. The soil practitioner's group also had the lowest number of indicators. The policy makers, on the other hand, had the largest group of indicators, 36 in total, and the most diverse set, including indicators in all three dimensions. The scientist group selected 25 indicators in total, with similar to the soil practitioners most of these located in the nature dimension. When looking at what issues all stakeholder groups deem important it becomes apparent that the indicators that all three groups have in common are mostly linked to the nature dimension, clearly capturing the links to the various functions that the soil performs and especially relating to soil physical and biological properties. This result is in line with other soil sustainability indicator sets that have been designed that largely focus on nature-based indicators (see Section 1.4). Furthermore, the proposed indicators for soil properties within the nature dimension are generally strongly affected by human activities, such as land use, land management, emissions, waste disposal etc. as well as representing key measures of soil quality. A surprise was the selection of *Public awareness of the value of soil* as the highest scoring indicator, which conflicted with our expectation that a nature based indicator would be deemed the most important. But as Brussaard et al. (2007) point out, "the values of soils are largely hidden and are usually less appreciated than those of above-ground assets". This leads to a lack of awareness and then to limited ability to connect the importance of soil protection to broader environmental, social and environmental outcomes (Bennett et al., 2010). The awareness of our stakeholder groups to this fact explains, in our view, the importance of this indicator and why it ranked number one.

In order for indicators to be influential, consensus must exist among actors that the chosen indicators are legitimate, credible and salient. This means that the indicators must not only answer questions that are relevant to each actor, but also provide a scientifically plausible and technically adequate assessment. To be legitimate, the indicators must be developed through a politically and socially acceptable procedure. The Delphi process used in this study lends legitimacy, credibility and saliency to the indicators that were produced. This can be seen by scrutinizing the change in standard deviation between rounds 1 and 3 for all the indicators (Tables 5–7), as this decreased for all indicators apart from five, indicating a development of a consensus in the group of respondents. Although it is difficult to evaluate whether true group learning or social learning occurred as a result of the Delphi, without doing a post-Delphi survey, it can be assumed that participants most likely benefitted from the Delphi process through the unfolding of greater understanding of the issues surrounding the sustainable development of soil. This is reflected through the Delphi process as it provides both quantitative and qualitative information from the stakeholders. The stakeholder input for the Delphi survey was also useful to the authors in designing better soil indicators generally, as the process illuminated problems with the theory behind and definition of certain indicators or reference values.

The main weaknesses of our approach were the high attrition and relatively low response rates in the Delphi survey. For example, participant's number went from 43 down to 21 between round one and three, with a particularly severe impact on the policy makers group that had a particularly high drop-out rate. As this certainly affects our ability to generalize from our survey, this seems not to have biased the final selection of core-indicators, as selections by the policy makers were not the limiting group when selecting the indicators as they chose the largest and most diverse set.

As the indicators have not yet been formally implemented, it is difficult to evaluate their practical suitability. However, many of the indicators are already used, just not in the specific context of soil and sustainable development (see for example: the various agri-environmental indicators in the OECD, Eurostat and FAOSTAT agri-environmental indicator sets where they report on fertilizer, pesticide, land use and soil erosion and others; the Human Development Index indicator on literacy; and World Bank's World Development Indicators where the World Bank reports on research and development expenditure and population growth among other things). To create a generalizable and universally applicable indicator set further studies are needed. It will be necessary to run the same indicator development process in different national and development contexts to evaluate indicator applicability for sustainable development given diverse economic, social and natural environments. This is an important process, as no one indicator set or framework can cover all soil systems and study locations (Van Cauwenbergh et al., 2007). Interesting results may emerge through a comparison of the outcomes from national studies focused on perceptions of the most important aspects of soil to monitor.

There are clear advantages of maintaining a core set of indicators and the stakeholder specific sets (SDSN, 2015) as this caters to soil managers that operate at different scales, decision-makers and the public. As reported in Dahl (2012), the general public and decision makers prefer a limited set of 10–15 indicators of the most relevant trends, but other stakeholder groups prefer a broader set. Selecting stakeholder specific indicators is a delicate matter as Van Cauwenbergh et al. (2007) asserts – selecting too few indicators promulgates the danger of omitting some important aspects, whilst selecting too many complicates data processing, data collection and interpretation. Our final results provide a core indicator set of 14 indicators and then broader satellite sets for specific stakeholder groups, ranging from 21 to 36 indicators (Table 7). We believe that by having both the core and satellite sets, we might be able to thread the narrow path between having too few and too many.

The UN 2030 Sustainable Development Goals cannot be reached with the continuing degradation of soils. Rapidly depleting soil resources will severely limit the options of future generations to fulfil their needs. How we humans safeguard soils for current and future generations is of utmost importance and needed for that are methods and tools to monitor soils systematically and holistically. In this paper we have presented a process for developing soil indicators for sustainable development. Indicators are important tools that can be used to monitor soil resources, and by using an expert-based stakeholder approach where soil managers across various levels of society participate it is more likely to reach a consensus on what constitute the elements of soils that should be monitored. We believe that by using an indicator development process with extensive stakeholder participation and consultation on different levels of soil management gives legitimacy and credibility to the final outcome: the core Soil Indicators for Sustainable Development (SIFSD). Many of the chosen indicators have established methods and are currently used, though some of them have perhaps not been used before in the context of soil. A few, such as *Economic valuation of soil ecosystem services*, have so far lacked established methods for evaluation, but there have been recent developments seeking to address this (see for example: Dominati et al., 2014; Graves et al., 2015; Jónsson and DavíÐsdóttir, 2016). The issue of soil sustainability is fundamental to reach the SDG as "the quality and health of soil determine agricultural sustainability, environmental quality, and as a consequence of both, plant, animal and human health" (Doran and Zeiss, 2000). Koch et al. (2013) call for a soil-centric policy framework "to generate policies that raise awareness of, and reverse, soil degradation and simultaneously recognize co-benefits for sustainable development". We believe that soil indicators for sustainable development might serve as contribution towards that end.

5. Conclusion

We presented a set of soil indicators for sustainable development – developed by using a transdisciplinary approach with extensive stakeholder participation. This type of indicator set can be a useful tool to assist decision-making regarding soil management. It can serve a communication medium or a middle ground, as the core soil indicator set represents something that all stakeholders agree on as being relevant for sustainable development in the context of soil. It is, therefore, a starting point. The use of an indicator set in decision-making, regardless of its suitability is, however, never guaranteed. We assert that the extensive stakeholder participation involved in the soil indicator development process lends credibility to the selected core set and, furthermore, will increase the likelihood of its future adoption.

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Appendix A. Supplementary data

Supplementary data associated with this article can be found, in the online version, at http://dx.doi.org/10.1016/j.agee.2016.08.009.

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Analysis

Tools for Sustainable Soil Management: Soil Ecosystem Services, EROI and Economic Analysis

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1. Introduction

Soils are an important form of natural capital that provide multiple benefits to humans (Banwart et al., 2012; Dominati et al., 2010; Jonsson and Davidsdottir, 2016) and are, along with water, the basis for human civilisation (Diamond, 2005; Montgomery, 2012). Soils are the foundation of agriculture and deliver multiple soil ecosystem services (ES) that together provide the necessary conditions for food, fibre and fuel production (Jonsson and Davidsdottir, 2016). Soils provide us with 95–99% of our food and their long-term sustainability is essential for sustained food production and the future of agriculture (Bindraban et al., 2012). It is expected that global food production will grow by 50% by the year 2030 and by 100% by the year 2050 in order to fulfil the demands of a growing world population and changing food consumption patterns (Godfray et al., 2010). At the same time, soil loss in some locations is up to 100 times faster than soil formation rates (Brantley et al., 2007; Montgomery, 2007), leading to questions as to whether soils can meet this demand (Banwart, 2011). Industrial agriculture, with its mechanisation and use of fossil-fuel based fertilisers, has for the last seventy years managed to increase food production in line with the growing population by enhancing yields, albeit at substantial cost to the soil resource and natural ecosystems (Brantley et al., 2007; David et al., 2009). Based on past trends and given the future projections of human population growth and consumption patterns, Tilman et al. (2001) predict the continued conversion of natural ecosystems to agriculture, 2 to 3-fold increase in fertiliser use and a comparable increase in pesticide use up to the year 2050. The long-term viability of current agricultural systems is uncertain as it is eroding the natural capital it relies on (Brantley et al., 2007). As a result, there is a pressing need to mitigate the negative effects of agriculture on natural capitals stocks. Various authors (Baulcombe et al., 2009; Godfray et al., 2010) have been calling for what has been termed 'sustainable intensification' of agriculture to address these trends. Sustainable intensification means increasing the productivity of current agro-ecosystems while sustaining natural capital stocks and minimizing effects on the environment. That is, "to increase production from existing farmland in ways with far less pressure on the environment and that do not undermine our capacity to continue to produce food in the future" (Garnett et al., 2013). Koch et al. (2013) and Lal (2009) emphasize that it is especially critical that intensification proceeds without damaging soil natural capital. Projections indicate that 80% of crop production growth in developing countries up to 2030 will be derived through intensification (FAO, 2009). There is evidence indicating that increasing agricultural intensification can erode ecosystem services (Power, 2010; Tilman et al., 2002; Tilman et al., 2001). For intensification to be realized without degrading soil natural capital and ecosystem services, current practices will need to change to more sustainable practices (Baulcombe et al., 2009). Conversion to sustainable intensification will require a shift from the use of non-renewable resources (fuel, fertiliser, pesticides) to renewable inputs (Sandhu et al., 2015), such as organic fertiliser (manure, compost), renewable energy and biologically based technologies for pest control.

The growth of agriculture since the middle of the last century has largely been due to substantial increases in the application of inputs such as fertilisers (Hazell and Wood, 2008). Two types of fertilisers are commonly applied, inorganic fertilisers (IF), derived from fossil fuels, which make up the bulk of fertilisers used, or organic amendments (OA), which are often derived from manure (M) or Municipal Solid Waste Compost (MSWC).

The aim of this study was to look at the effects of fertilisers on the soil ecosystems in a multi-year tomato plantation experiment by comparing the impact of the application of four different fertilisation treatments on soil ecosystem services, the energy balance of the operation and benefit-cost ratios. By assessing the impact from these three different approaches we can establish the effects of the different types of fertiliser on the trade-offs between the ecological, energy and economic perspectives of the tomato plantation. Comparing results from the three approaches; Quantification of the Ecosystem Services (the benefits received from ecosystems), the energy output from energy invested (EROI) and benefit-cost ratios, we aim to reveal if fossil fuel derived fertilisers or organically sourced fertilisers are preferable in each case.

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Kotronakis et al. (2017) measured all three ES, carbon and nitrogen content in the soil and biomass production throughout the experiment. These data were used to calibrate the model. This effect of organic matter on soil structure and functions is well established within the broader soil literature. Soil organic matter catalyses the maintenance of soil functions such as soil structure, nutrient turnover, biomass production and soil biodiversity. Soil carbon depletion due to agricultural practices has led to degradation of soil and water quality in arable land (Banwart et al., 2014). Adoption of agro-ecological practices, including carbon addition either by plant litter incorporation or inputs from external sources such as livestock (manure) and industry (MSWC) can be shown to be beneficial in reversing soil degradation and enhancing natural restoration. Bronick and Lal (2005) and Lehtinen et al. (2015) report that OA, in the form of compost and manure, can enhance soil structure, its functions and hence fertility. Lal (2011) points out that the application of OAs builds up the SOC pool which enhances the quality of the soil, water and combats climate change, and Christensen et al. (2009) report on the positive effects of OA on soil ES.

We aim to contribute to the ongoing discussion on the sustainability of agricultural practices and contribute to management choices in pursuit of sustainable intensification of agriculture. By relying on an experimental plot the study offers a novel approach to assess the sustainability of the soil system under use and complements other SoilTrEC[1] research (Sparks and Banwart, 2017), especially regarding the sustainable use of soil for agriculture (Giannakis et al., 2017; Kotronakis et al., 2017; Panakoulia et al., 2017).

2. Material and Methods

2.1. General Information

Data for this research was drawn from the Koiliaris Critical Zone observatory (Koiliaris CZO) situated near Chania, Crete, Greece (alt 15 m latitude 35.437139° long: 24.141889). A detailed description of Koiliaris CZO is in Kotronakis et al. (2017). Crete's long history of agricultural land-use has witnessed soil degradation (Nikolaidis, 2011; Nikolaidis et al., 2013), resulting in degraded soils (Panagos et al., 2014). For the last 50 years, land-use patterns have remained largely unchanged but with intensification, especially in terms of the number of grazing livestock on the island which has increased five-fold (Nikolaidis et al., 2013). Areas in Crete prone to high levels of erosion are attributable to overgrazing, especially in natural grasslands with a high density of livestock (Panagos et al., 2014). Crete is considered a high-risk area for desertification due to intense land-use and changing climate conditions.

2.2. The Study Site

The study site was established as a part of the SoilTrEC project with the aim to assess the effects of soil organic amendments on soil properties and functions. An experimental field run for four years by the Hydro-Geochemical Engineering and Remediation of Soils laboratory (HERS-lab, Technical University of Crete), was established in 2011 on leased land of about 850m^2. The site was a former orange tree orchard that had been set-aside for about 30 years, situated in the plain of Koiliaris river basin within the study area of Koiliaris CZO. The experimental plantation occupied an area of about 192 m^2 and included four fertilisation treatments to produce tomatoes, the common annual cultivations in this area. The fertilisation treatments included: inorganic fertilisers (IF); manure (M); Municipal Solid Waste Compost (MSWC); and MSWC (Municipal Solid Waste Compost) + M (manure) and thus the experimental site enabled comparison of organic amendments with inorganic fertilisation as the standard approach for crop cultivations. Table 1 provides information on the management practices for the tomato plantation. The data used in this study is based on simulation results derived from measured data from the four-year tomato plantation experiment, which included triplicates for each treatment. Only the results from the last three years of the experiment are used in this paper because the first year the planting started late (August), it was a short season and was used it as a warm up period for the experiment. The simulations were performed using the 1D-Integrated Critical Zone (ICZ) model that has been described in Giannakis et al. (2017) and the results of the simulations by Kotronakis et al. (2017). It is a physical based model that simulates dynamically the changes of soil structure, carbon and nutrient sequestration, plant and below ground biological dynamics and the geochemical changes in both the aqueous and solid phases. The model was calibrated using the data collected from the tomato field experiment where each treatment was conducted in triplicate. Measurements were taken during the growing season and after it. The field measurements included changes in soil structure and soil particle aggregation, sequestration of nutrients, biomass production above and below ground, soil chemistry as well as detailed characterization of all the water, carbon and nutrient inputs to the system. These data were used to calibrate the model and assess the effectiveness of use of organic fertilisers. Further descriptions of the experimental site, data collection and modelling can be found in Kotronakis et al. (2017).

The SoilTrEC project collected other soil micro and macro health indicators for the Koiliaris CZO, which included the tomato plot and the results can be found in Kotronakis et al. (2017). Additional data regarding energy and economic aspects related to the experiment were collected on site, at the end of the experiment (November 2014).

2.3. Soil Ecosystem Services

This study uses an ecosystem services framework (Fig. 1) approach for analysing the ecological effects of the four fertiliser treatments on soil natural capital. The soil ecosystem services framework (Jonsson and Davidsdottir, 2016; Jónsson et al., 2017) is based on the notion that soils are natural capital that deliver multiple benefits to humans through the flow of ecosystem services. The framework begins with a description of the soil's ecological infrastructure, where various soil processes, properties, and functions interact with each other and deliver soil ecosystem services, sometimes with the help of external inputs. The framework acknowledges the multiple services derived from soils as well as the various beneficiaries, illustrating the importance of keeping the soil ecological infrastructure intact, which is a prerequisite for sustainable intensification.

In this paper, we investigate how the four different fertiliser applications affect three soil ES and economically value them (for valuation methods and ranges for soil ES see Jonsson and Davidsdottir (2016)). The three services included are: climate regulation, biomass production, and filtering of nutrients and contaminants. They were based on selected soil parameters from Kotronakis et al. (2017) and were selected for three reasons: (1) importance; these three soil ES are defined as essential soil services by the European Commission (2002, 2006); soils are one of the key factors in climate regulation (Knorr et al., 2005; Lal, 2004, 2011; Milne et al., 2015); there is need for increased biomass production to feed more people in the 21st century (Amundson et al., 2015; Baulcombe et al., 2009; Godfray et al., 2010); and the capacity of soils to buffer some of the detrimental effects of nitrogen use in agriculture, like retaining nitrates is crucial (Powlson et al., 2011; Rockström et al., 2009); (2) data availability; time-series data was available for them from the Koiliaris CZO and (3) comparability and context; Jónsson et al. (2017) have examined these same services on a watershed scale for the same area.

[1] SoilTrEC (Soil Transformation in European Catchments) is a EU/FP7 funded research project focused on addressing certain knowledge gaps regarding important soil ES and functions and the importance of mathematical modelling (Banwart et al., 2012) of these as listed by the EU soil thematic strategy (European Commission, 2006).

Table 1
Management practices for the tomato treatments.

Agricultural practices	Description			
	IF	M	MSWC	MSWC + M
Plot size	48 m^2, three subplots 16 m^2 each with 20 plants of tomatoes, *Solanum lycopersicum* L., 'Bobcat'	Same practices	Same practices	Same practices
Land preparation	The plot was tilled with a tractor at the beginning of the experiment	Same practices	Same practices	Same practices
Average tilling number	The plot is tilled in the spring with a rototiller before planting so the organic matter from the year before is incorporated into the soil	Same practices	Same practices	Same practices
Planting period	May each year except the first year of the experiment when it was August	Same practices	Same practices	Same practices
Length of growing season	118–146 days	Same practices	Same practices	Same practices
Fertiliser application	30% N, 10% P, 10% K and 20% N, 20% P, 20% K with a ratio of 1:3 approx. 200 kg N/ha	Manure – approx. 870 kg TN/ha	MSWC – approx. 870 kg TN/ha	70% MSWC, 30% M – 870 kg TN/ha
Fertilisation period	May to September			
Average number of fertilisation	9 (during the tomato growing period)	Same practices	Same practices	Same practices
Hoeing period	May to September	Same practices	Same practices	Same practices
Average number of hoeing	4–6	Same practices	Same practices	Same practices
Irrigation period	May to October with a drip irrigation system	Same practices	Same practices	Same practices
Average number of irrigation	30–45 irrigation events	Same practices	Same practices	Same practices
Spraying pesticides period	May to September	Same practices	Same practices	Same practices
Average number of spraying	4–8 times	Same practices	Same practices	Same practices
Harvesting period	October–November	Same practices	Same practices	Same practices
Harvesting frequency	Regularly from mid-July to November	Same practices	Same practices	Same practices

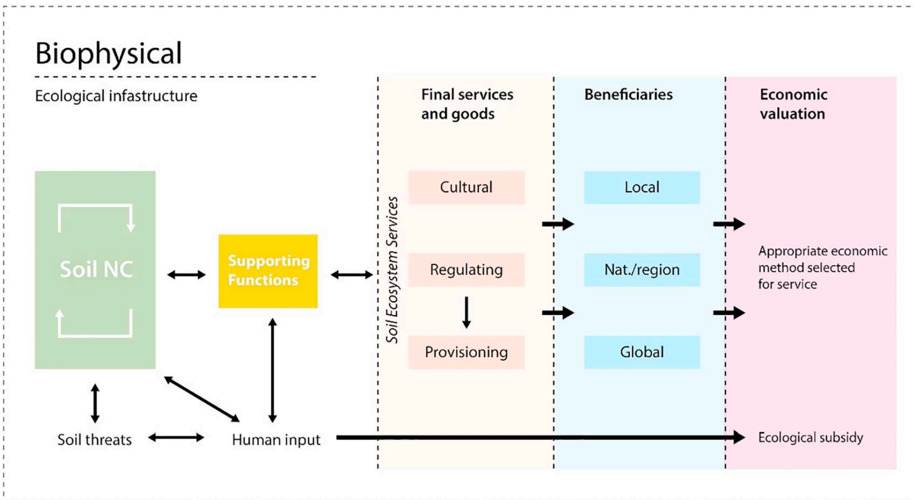

Fig. 1. Soil ecosystem service framework – adapted and modified from Jónsson et al. (2017).

Table 2
Soil services, metrics, valuation method and values.

Soil ES	Metric	Valuation method	Value US$ 2013
Carbon (CO_2)	t/ha^{-1}/yr^{-1}	Mitigation cost	111 averages
Biomass (tomato)	t/ha^{-1}/yr^{-1}	Producer's price	604 averages
Nitrate (NO_3^-)	t/ha^{-1}/yr^{-1}	Avoided cost	5150 (low)–24,730 (high)

2.3.1. Climate Regulation

Climate regulation is a soil ES defined as the regulation of global temperatures and precipitation through sequestration of carbon and methane (Jónsson et al., 2017). Soil organic carbon plays a key role in the physical structure, stability and fertility of soils (Milne et al., 2015), which in turn comprises a significant CO_2 sink. Soil organic carbon sequestration was used as a proxy for this service with the metric chosen as t C/ha^{-1}/yr^{-1} i.e. the net annual addition to the soil. The simulation data were provided by Kotronakis et al. (2017). The economic assessment of the service was based on avoided cost estimates, that is CO_2 mitigation (abatement) cost which was obtained from Vermont and De Cara (2010). This cost represents the average marginal GHG abatement cost estimate for agriculture for a tonne of CO_2. Although agriculture accounts for only 3% of global energy consumption, it is responsible for more than 20% of global greenhouse gas emissions (Moomaw et al., 2001). The abatement cost was converted to 2013 US dollars, giving a value of 111 US$ per tonne of CO_2. The total amount of CO_2 mitigated was derived from the C stock in the field, multiplied by the conversion to CO_2 in t/ha^{-1}/yr^{-1} using the Carbon to CO_2 multiplier of 3.667 (Table 2).

2.3.2. Biomass Production From Crop

The second ES assessed is the provisioning service of biomass production. Soils provide nutrients, water and the physical environment for terrestrial biomass production. The metric chosen for this service was biomass in kg/ha^{-1}/yr^{-1}, i.e. the biomass from the tomato crop harvest (Table 2). The simulation data for this metric were provided by Kotronakis et al. (2017). The method for converting biomass production quantities to economic values relies on farm gate prices (Jonsson and Davidsdottir, 2016). The price for tomatoes was obtained from the FAO database on national producer's prices. The average Greek national price of tomatoes was 604 US$ per tonne in 2013 (FAO, 2015).

2.3.3. Filtering of Nutrients and Contaminants

The third service is another regulating service, the filtering of nutrients and contaminants. Soils can control water quality by, to some extent, absorbing and retaining solutes and contaminants, therefore avoiding their release to water bodies. For the study, potential nitrate (NO_3^-) retention in kg/ha^{-1}/yr^{-1}was used as a proxy for this service. Potential nitrate retention captures the potential ability of soil to retain nitrates rather than these reaching the groundwater. The simulation data for this metric were provided by Kotronakis et al. (2017). The amount of organic nitrogen (in the form of humus and biomass assimilated) that is sequestered by the micro and macro aggregates and thus protected from mineralization simulated by the model was used to estimate the annual nitrate retention in the various treatments.

Nitrates in drinking water supply are a well-known health hazard (Compton et al., 2011), and their concentration is regulated by the EU (European Commission, 2000). Derived from (Jonsson and Davidsdottir, 2016), the economic valuation method for this ES is similar to climate regulation, based on avoided cost. The cost of nitrate pollution was obtained from the European Nitrogen Assessment (Brink and van Grinsven, 2011) and is in the range of 5150 to 24,730 2013 US $ per tonne (Table 2).

2.4. Energy Return on Investment

Energy is essential in agriculture and the growth of food production over the last 50 years can be directly and indirectly linked to increased energy use in agriculture, especially in through the use of fossil fuels in the production of fertilisers, pesticides and as fuel for machinery (Woods et al., 2010). Efficient use of energy is one of the conditions needed for sustainable intensification in agricultural production. This paper uses an approach called Energy Return On Investment (EROI) to estimate the energy balance associated with use of the four different fertilisers. The approach compares the amount of energy returned from one unit of energy invested in an energy producing system (Hall, 2011). EROI has been used as a parameter in energy system assessments capturing the sustainability of the resources in agriculture (Atlason et al., 2015b; Pimentel et al., 2005; Schramski et al., 2013) and food production (Perryman and Schramski, 2015; Veiga et al., 2015). When an EROI reaches one, then an equal amount of energy is retrieved as is invested. In general, the equation for EROI calculations can be described as (Murphy et al., 2011):

$$EROI = \frac{ED_{out} + \Sigma v_j o_j}{ED_{in} + \Sigma y_k I_k}$$

where ED_{out} is the direct energy output, v_j is a set of well-defined co-efficient outputs, o_j is the energy per unit of the given output co-efficient, ED_{in} is the direct energy input, y_k is a set of well-defined input co-efficients, and I_{rk} is the energy per unit of the given co-efficient (Atlason et al., 2015b). The EROI boundaries for the study were chosen based on Murphy et al. (2011). Inside the EROI boundaries, the following inputs are included: the diesel used for the rototiller; the gasoline for the water irrigation system (pump); the inorganic fertiliser used; the MSWC used; the M used; the human labour used, the pesticides used and the tomato crop biomass. Outside the boundaries were the production and transportation of materials to the site and the solar energy captured in the growing crops. The EROI input and output is measured in Megajoules per kilogramme (MJ/kg^{-1}) or per litre (MJ/l^{-1}) (Table 3).

Table 3
Energy equivalents of inputs and output in tomato production.

Particulars	Unit energy	Equivalent (MJ unit^{-1})	References
Inputs			
Human labour	h	1.96	Mohammadi and Omid (2010)
Diesel fuel	l	35.8	US Department of Energy (2015)
Gasoline	l	32.8	US Department of Energy (2015)
Chemical fertilisers	kg		
(a) Nitrogen		72.13	Helsel (1992)
(b) Phosphate		10.37	Helsel (1992)
(c) Potassium		8.2	Helsel (1992)
Pesticides			
Insecticides	kg		
(a) Miscible oil		357.90	Helsel (1992)
(b) Granules		283.00	Helsel (1992)
Fungicides	kg		
(a) Miscible oil		267.10	Helsel (1992)
(b) Wettable powder		113.50	Helsel (1992)
(c) Granules		190.90	Helsel (1992)
Manure (M)	kg	0.3	Samavatean et al. (2011)
MSWC	kg	0.3	EPA (2016)
MSWC + M	kg	0.3	Assumed by the authors of this paper
Output			
Tomatoes	kg	0.8	Ozkan et al. (2004b)

Most of the energy equivalents (Table 3) are established numbers derived from the literature or scientific studies, except for the MSWC + M used in this study. Given that both M and MSWC have the same energy equivalent, as reported in the literature (EPA, 2016; Samavatean et al., 2011), the assumption is made that the MSWC + M mix has the same energy equivalent as M and MSWC.

2.5. Economic Analysis of the Tomato Production

In this paper, we use conventional Cost Benefit Assessment (CBA) to estimate the economic feasibility of the different fertiliser applications, by comparing the capital and running cost to the benefits of the different fertiliser treatments. This is presented as a simple Benefit to Cost ratio. CBA is a tool to aid decision-making, measuring the efficiency of investments and helping to make a comparison between alternative investments in monetary terms (Boardman, 2011; Hanley and Spash, 1993).

The economic inputs for the study include both capital cost (Table 4) and the annual operation cost (Table 6). The capital cost was transformed into uniform annual payments (Table 5) based on the standard lifetime of a tomato operation (one farming generation) and a

Table 4
Capital cost in 2011 for the 192 m^2 in 2013 US$.

Inputs	Cost ($)
Mechanical land clearing	209
Plough and level	139
Fences	3475
Irrigation system	695
Installation for tomato plants	278
Hoe	28
Pesticide pump	139
Crates for carrying products	584
Bags for distributing M and MSWC	58
Shovels	28
Total sum	5634

Table 5
Parameters for capital cost recovery for the 192 m^2 in 2013 US$.

Parameters	Value
P (capital cost)	$5634
i (interest rate)	7.1% APR
N (years of operation)	35
Annual cost for the whole operation	$440
Per treatment plot	$110

loan interest rate available to farmers in Crete.[2] The economic output is the income from the tomato crop which was 604 US$ t/ha^{-1}. All prices for economic inputs and outputs were converted into 2013 US$.

The entire experimental plantation received the same treatment during the years of operation, with the exception of the fertiliser treatment. The capital cost expenditure was split into four parts, 25% for each experimental plot. The years 2012–2014 were identical in annual operational cost (Tables 4–6).

3. Results

All results presented below were scaled up from the 48 m^2 experimental plot size to a ha^{-1} to allow comparison with other studies. As the data used in the paper was simulated data from modelling, no statistical analysis was conducted. We report separately on each of the soil ES and then on net economic benefits relying on CBA. The economic assessment of soil ES and the assessment of conventional economic benefits and costs were kept separate to prevent double-counting, as well as to illustrate separately the importance of the individual ES which commonly remain excluded from conventional economic assessments.

3.1. Soil Ecosystem Services

3.1.1. Climate Regulation

Fig. 2 shows the annual net change of SOC for the four treatments for the three-year period, 2012 to 2014. All the treatments showed an increase in SOC content over the three-year period; the highest amount being from the MSWC + M with 33.98 t/ha^{-1}, followed by M with 32.39 t/ha^{-1}, MSWC with 24.84 t/ha^{-1} and IF with 9.54 t/ha^{-1}.

Table 7 shows the economic value of the annual addition (net changes) of SOC converted into sequestered CO_2 with average value of 111 US$. The annual economic value of CO_2 sequestered ranged from low −278 to high of 8036 US$ (Table 7). For the three-year period the highest total value was for the MSWC + M treatment of 13,841 US$, as it had the highest amount of carbon sequestered. The IF treatment had the lowest total value of 3897 US$ for the three-year period (Table 7).

3.1.2. Biomass Production From Crop

Fig. 3 shows the biomass yield of the tomato crop from the four treatments for the three-year period. The highest tomato biomass produced for the three years was in the M treatment with 627 t/ha^{-1}, followed by MSWC + M with 609 t/ha^{-1}, IF with 535 t/ha^{-1} and MSWC with 513 t/ha^{-1}.

Table 8 shows the economic value of biomass production. The highest values were for the M treatment, 378,425 US$, followed by the MSWC + M with 367,425 US$, IF with 323,055 US$ and MSWC with 310,025 US$, respectively.

3.1.3. Filtering of Nutrients and Contaminants

Fig. 4 shows the changes in potential NO_3^- retention from the tomato crop for the four treatments for the three-year period. The highest

Table 6
Annual operation cost + annual capital cost per treatment in 2013 US$.

Inputs	IF	MSWC	M	MSWC + M
Labour cost[a]	275	275	275	275
Seeds and plants	26	26	26	26
M	0	0	97	29
Fertilisers	20	0	0	0
MSWC	0	27	0	19
Pesticides	10	10	10	10
Diesel	2	2	2	2
Gasoline[b]	40	40	40	40
Lubricants	3	3	3	3
Metal wire to tie plants	17	17	17	17
Annual capital cost	110	110	110	110
Total	503	509	579	530

[a] Labour cost includes all cost associated with seeding, planting and harvesting and regular check-up and maintenance of the experimental plot.
[b] This is mainly the cost of running the irrigation system.

accumulated potential nitrate retention was the M treatment with 0.18 t/NO_3^- retained for the three-years, followed by MSWC + M with 0.16 t/NO_3^- and MSCW with 0.13 t/NO_3^-. The IF treatment had a net leaching of −0.06 t/NO_3^-.

Table 9 shows the economic value range of nitrate retention. The highest values were for the M treatment with 2671 US$ ha^{-1}, followed by M + MSWC with 2349 US$ ha^{-1} and MSW with 2102 US$ ha^{-1}, all in 2012. The IF treatment had no nitrate retention of all the three years and, in fact, had a net loss of nitrogen that was sequestered in the soil prior to the initiation of the treatments.

3.2. EROI

The energy content of the tomato crop at the farm gate (output) is aggregated and divided by the aggregated inputs. The treatments with the most favourable EROI ratio were the MSWC + M and M, 1:1.19, followed by the MSWC, 1:1.09, and IF, 1:1.07 (Table 10). The highest energy input for the individual treatment was the gasoline used in the pump for the irrigation system, ranging from 72% to 75%, followed by M (16%) in the M treatment, MSWC (12%) in the MSWC treatment and chemical fertiliser (11%) in the IF treatment. The M treatment had the highest energy output (tomato yield) of 939 MJ, followed by MSWC + M with 912 MJ (Table 10).

The IF had the lowest energy input of 754 MJ, followed by MSWC with 758 MJ. The fractions of inputs derived from fossil fuels (fuels, fertiliser and pesticides) ranged from 77% for the M treatment to 92% in the IF treatment.

3.3. Economic Analysis

The biomass from the tomatoes was upscaled to a value per hectare, and Table 11 compares the cost and benefits of all the treatments. The treatment that had the highest cost was the IF, with 497.33 US$ per t/ha^{-1} of tomatoes. The treatment that had the lowest cost was the MSWC + M, with 464.72 US$ per t/ha^{-1}. The treatment with the highest total revenue was the M treatment, with 147,726.32 US$. The treatment with the lowest total revenue was the IF, with 127,142.00 US$. The treatment with the most favourable Benefit to Cost ratio (total revenue divided by total cost), was the MSWC + M with 1.30. The treatment with the least favourable Benefit to Cost ratio was the IF with 1.21.

3.4. Ranking and Comparing the Different Methods

Table 12 shows the ranking of the four fertiliser treatments by the three approaches, EROI, CBA and Soil ES, with 1 being the best rank and 4 the worst. The three approaches (economic (CBA), ecological

[2] Farming loans are only available for Greek farmers at Piraeus Bank, Greece. See: http://www.piraeusbank.gr/en/agrotes/agrotika-daneia/anoixto-daneio.

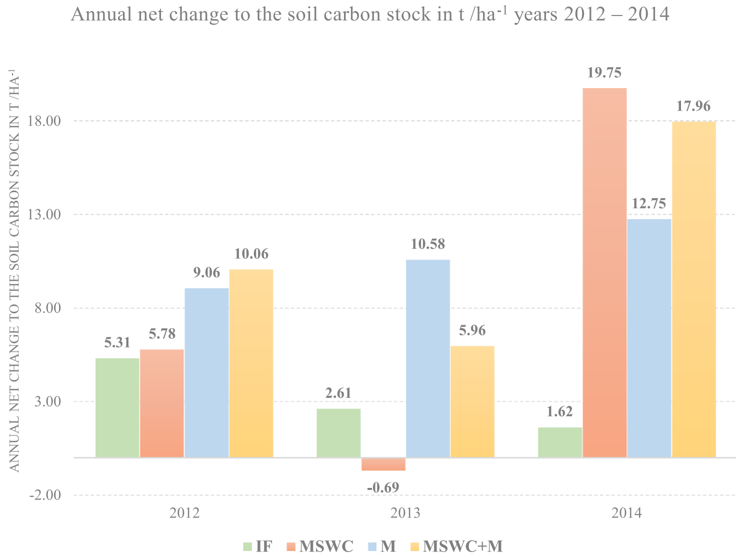

Fig. 2. Annual net change to the soil carbon stock 2012–2014. Negative values indicate decreases in carbon stock.

Table 7
Economic value of CO_2 sequestered 2012–2014 in 2013 US\$ t/ha^{-1}.

Treatment	2012		2013		2014	
	Value ($)	QTY (t/ha^{-1})	Value ($)	QTY (t/ha^{-1})	Value ($)	QTY (t/ha^{-1})
IF	2165	19.5	1066	9.6	666	6
MSWC	2353	21.2	−278	−2.5	8036	72.4
M	3685	33.2	4307	38.8	5184	46.7
MSWC + M	4095	36.9	2431	21.9	7315	65.9

Table 8
Economic value of tomato biomass for 2012–2014 in 2013 US\$ per t/ha^{-1}.

Treatment	2012		2013		2014	
	Value ($)	QTY (t/ha^{-1})	Value ($)	QTY (t/ha^{-1})	Value ($)	QTY (t/ha^{-1})
IF	113,232	187	126,849	ca	82,974	137
MSWC	107,948	179	130,112	215	71,965	119
M	131,097	217	147,705	245	99,623	165
MSWC + M	133,362	221	143,406	237	91,194	151

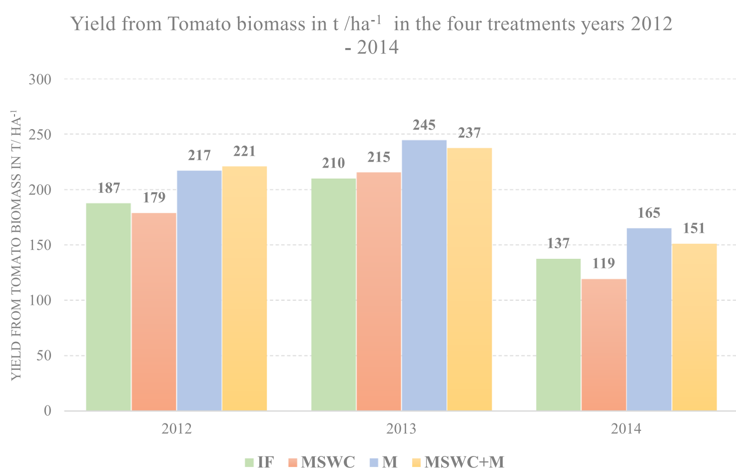

Fig. 3. Yield from tomato biomass 2012–2014.

(Soil ES), and energetic (EROI) aspects) were equally weighted in the study when determining rank to maintain transparency as is common in sustainability assessments, and in the spirit of Triple Bottom Line accounting. Therefore, the mean ranking of the three soil ES was used in the overall ranking calculation and as a result, each soil ES accounts for 0.11 in the overall ranking. The treatment that had the lowest (most favourable) overall ranking was the MSWC + M, followed by M (Table 12). The MSWC + M and M treatments have the most favourable EROI ratio (1.19), the MSWC + M has the highest Benefit to Cost ratio (1.30), and M had the lowest mean in ES (1.33). IF ranked worst in all

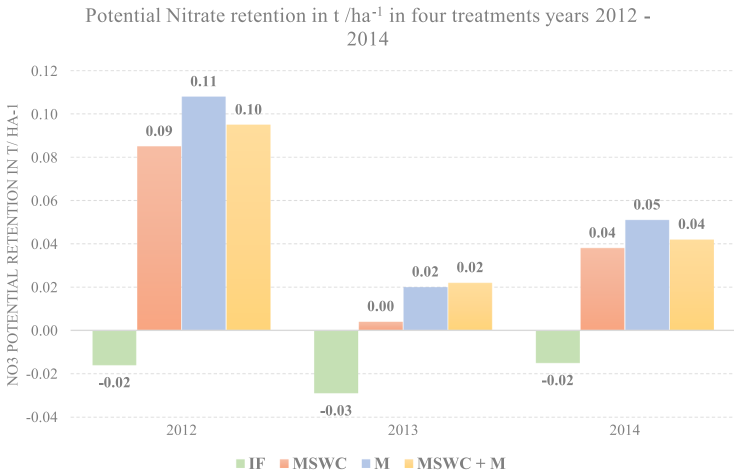

Fig. 4. Potential nitrate retention in t/ha^{-1} in four treatments for 2012–2014.

Table 9
The economic value of potential NO_3^- retention for 2012–2014 in US$ per t/ha^{-1}.

Treatment	2012			2013			2014		
	Value ($)		QTY (t/ha^{-1})	Value ($)		QTY (t/ha^{-1})	Value ($)		QTY (t/ha^{-1})
	Low	High		Low	High		Low	High	
IF	−82	−396	−0.016	−149	−717	−0.029	−77	−371	−0.015
MSWC	438	2102	0.085	21	99	0.004	196	940	0.038
M	556	2671	0.108	103	495	0.020	263	1261	0.051
MSWC + M	489	2349	0.095	113	544	0.022	216	1039	0.042

Table 10
Energy equivalents of inputs and outputs per treatment.

	IF			MSWC			M			MSWC + M		
	MJ	%	QTY	MJ	%	QTY	MJ	%	QTY	MJ	%	QTY
Input total												
Human labour (h)	58.0	8%	30.0	58.8	8%	30.0	58.8	7%	30.0	58.8	8%	30.0
Diesel (l)	35.8	5%	1.0	35.8	5%	1.0	35.8	5%	1.0	35.8	5%	1.0
Gasoline (l)	567.0	75%	17.5	567.0	75%	17.5	567.0	72%	17.5	567	74%	17.5
Chemical fertilisers (kg)												
-30/10/10	26.0	3%	1.09	0.0	0%	0.0	0.0	0%	0.0	0.0	0%	0.0
-20/20/20	59.0	8%	3.25	0.0	0%	0.0	0.0	0%	0.0	0.0	0%	0.0
Pesticides (kg)												
-Insecticides	4.33	1%	0.02	4.33	1%	0.02	4.33	1%	0.02	4.33	1%	0.02
-Fungicides	3.33	0%	0.01	3.33	0%	0.01	3.33	0%	0.01	3.33	0%	0.01
M (kg)	0.0	0%	0.00	0.0	0%	0.0	123.0	16%	411.0	37.0	5%	123.0
MSWC (kg)	0.0	0%	0.00	89.0	12%	296.0	0.0	0%		62.0	8%	207.0
Total	754.0	100%		758.0	100%		792.0	100%		768.0	100%	
Output												
Tomatoes (kg)	807.0		1008.0	827.0		1034.0	939.0		1174.0	912.0		1140.0
EROI 1:x	1.07			1.09			1.19			1.19		

approaches and for two out of three soil ES.

4. Discussion

The MSWC + M had the highest EROI (1:1.19) along with M, the highest Benefit to Cost ratio (1.30) and the second highest soil ES (1.67). The MSWC + M treatment ranked between 3.67 and 4.67 overall (Table 12). The least beneficial treatment overall was inorganic fertilisation (IF).

When comparing the results of soil ES for an individual year to other studies, the results of climate change regulation and biomass production fall outside the range reported in other studies, both in higher and lower values. In this paper we report values for the climate regulation service for individual years ranging from low −278 to high 8036 US $ ha^{-1}/yr^{-1} while other studies have reported a range of 3–426 US $ ha^{-1}/yr^{-1} (Dominati et al., 2014b; Sandhu et al., 2008; Xiao et al., 2005). The value for biomass production in this paper was from low 71,965 to high 147,705 ha^{-1}/yr^{-1}. Other studies have reported a range of 235–5926 US$ ha^{-1}/yr^{-1} (Dominati et al., 2014a; Dominati et al., 2014b; Haley, 2006; Pretty et al., 2000; Sandhu et al., 2008). For

Table 11
Cost and benefits for the four treatments in 2013US$ in ha^{-1}.

Cost and benefits per treatment[a]	IF	MSWC	M	MSWC + M
Yield in t/ha^{-1}	210.5	215.45	244.58	237.47
Revenue in t/ha^{-1}	604	604	604	604
Total revenue from production	127,142.00	130,131.80	147,726.32	143,431.88
Cost of production in t/ha − 1	497.33	492.04	492.93	464.72
Total cost of production	104,687.85	106,011.02	120,561.54	110,357.53
Profit in t/ha^{-1}	106.67	111.96	111.07	139.28
Benefit to cost ratio	1.21	1.23	1.23	1.30

[a] All the treatments were upscaled from 48 m^2 to ha^{-1}.

Table 12
Ranking and comparison of the four fertiliser's treatments with the three approaches for 2012–2014.

Methods	IF	Rank	MSWC	Rank	M	Rank	MSWC + M	Rank
EROI	1.07	4	1.09	3	1.19	1–2	1.19	1–2
B/C	1.21	4	1.23	2–3	1.23	2–3	1.30	1
ES $ t/ha^{-1}								
Climate regulation	*3987*	*4*	*10,111*	*3*	*13,176*	*2*	*13,841*	*1*
Biomass production	*323,055*	*3*	*310,025*	*4*	*378,425*	*1*	*367,962*	*2*
Filtration of nutrients	*(−) 717–(−) 77*	*4*	*21–2102*	*3*	*21–2671*	*1*	*113–2349*	*2*
Mean rank of soil ES		3.67		3.33		1.33		1.67
Total[a]		11.67		8.33–9.33		4.33–6.33		3.67–4.67

Italic indicates individual soil ES. Soil ES were treated as a group in ranking calculation and the mean rank for all three soil ES was used.

[a] Lower number indicates a preferable option in relation to these three methods.

filtering of nutrients and contaminants, represented by NO_3^- retention potential, we report values from low −717 to high value of 2671 US $ ha^{-1}/yr^{-1}. Other studies have reported values for this service in the range of 690–8114 US$ ha^{-1}/yr^{-1} (Dominati et al., 2014a; Dominati et al., 2014b).

A few reasons are behind these differences. The valuation method used for the climate regulation service is mitigation cost based on average abatement cost for agriculture and not the market price of carbon, which is more often used and is substantially lower than the mitigation cost used in this paper. We use the agriculture abatement cost numbers for climate regulation instead of market price of carbon, as it better reflects the cost of mitigating GHG (CH_4, N_2O and CO_2) emissions associated with agricultural activities. These agricultural emissions are not a part of the current EU ETS trading system[3] and thus using a quota market price for them is not representative of their value/cost. The value we use is in line with other studies reported in the literature (Bourne et al., 2012). Some authors (De Cara and Jayet, 2001; Domínguez, 2006) have even reported higher marginal abatement cost associated with Greek agriculture. However, acknowledging that there is a substantial disparity between what the market is willing to pay for carbon and the mitigation cost and thus using the market price[4] of CO_2, then the value of the climate regulation services for different treatments would have been in the range −14 to 685 US$ 2013 t/ha^{-1}. This value range is closer to what has been reported in the literature.

The value of the biomass production service is dependent on three factors: 1) the production volume of the biomass type, as this varies between primary industries and crop types; 2) the valuation method; and 3) the value selected. The value for the biomass services reported above are from other primary industry, and they illustrate the wide range that such valuations can give and are context and crop specific. A more insightful comparison for our study is to compare the tomato yield with other reports from Crete and elsewhere in Greece. According to a survey by Valogiannis (2012), farmers' tomato yields are 60–80 t/ha^{-1} from outdoor crops and around 120–160 t/ha^{-1} in greenhouses. Lychnaras and Schneider (2011) report a yield of 45–115 t/ha^{-1} for industrial tomatoes in central Greece and Kantor - Management Consultants (2015) reports an average yield of 92.2 t/ha^{-1} in the period 2003–2012 for greenhouse tomato cultivation. The yields from all treatments reported here were substantially higher than these studies and thus we obtained a substantial higher value than reported elsewhere. This is because the field used for the experiment was set aside for 30 years and thus had accumulated a substantial amount of nutrients in addition to the ones provided as part of the treatments.

What is evident when looking at the results from the soil ES is that the input of organic amendment C + N (OA) in the form of compost and manure improved the three soil ES. The IF treatment, on the other hand, neither improved the structure as reported by Kotronakis et al. (2017) nor increased the SOC stock, and is thus not a viable long-term strategy for soil sustainability nor sustainable intensification. Farmers have opted for using IF because of its low cost but, as Drechsel et al. (2004) point out, its use can mask declining fertility and soil degradation, as farmers find it more cost-effective to increase the amount of fertiliser used to maintain biomass outputs or open new plots rather than build up fertility. There are many environmental and health issues related to continuous use of IF in agricultural systems such as nitrate pollution of waterbodies (European Commission, 2000) and the potential accumulation of heavy metals to high levels (Czarnecki and Düring, 2015).

According to calculations done by Nikolaidis (2011), the potential of producing OAs (by processing MSWC, manures, agricultural residues etc.) in Crete could cover the fertiliser needs of the island. Despite this, farmers are reluctant to use OA's as part of more sustainable agricultural management practices (Giannakis, 2014, Personal Communication, November 10), which can improve soil C + N status. Returning organic matter back to the soil is beneficial for farmers in multiple ways as the results presented here show. OA's are the low hanging fruit for intensive vegetable production systems as they can be

[3] https://ec.europa.eu/clima/policies/ets_en.

[4] Based on the price of 5.5 US$ per tonne of CO_2 in the EU carbon stock exchange in December 2013.

made onsite and thus provide an alternative energy resource for farmers.

A plethora of studies (Canakci et al., 2005; Ghorbani et al., 2011; Mandal et al., 2002; Mobtaker et al., 2010; Mohammadshirazi et al., 2012; Ozkan et al., 2004a) exist that have investigated the relationship between energy inputs versus outputs balance and economic profitability. Although there are no other studies that the authors are aware of which address the same four-treatment setups as in this study, it is possible to use similar studies on organic versus conventional farming practices that look at both energy balance and economic output. Organic farming systems are generally more energy efficient than conventional systems (Atlason et al., 2015a; Kasperczyk and Knickel, 2006; Scialabba and Hattam, 2002; Tuomisto et al., 2012) but there is a large amount of evidence to show they yield less (Badgley et al., 2007; Connor, 2008). Our results contradict those results, as they indicate that OA can be more efficient and also deliver larger yields (Kotronakis et al., 2017). Two other studies support this conclusion. Kaltsas et al. (2007) found in converting to organic olive groves production it was possible to decrease energy inputs without suffering a reduction in yield. Gundogmus (2006) compared energy use in apricot production in organic and conventional farms in Turkey and found the EROI for the organic farm was 1:2.22 while 1:1.45 based on conventional production. The benefit to cost ratio was near identical (2.13 and 2.14) for the two systems.

Fertilisers increase the productivity of the soil system and usually increase biomass output (crop yield), which through sales brings larger direct economic benefits. Although environmental stewardship is a factor influencing farmers' decisions, the economic viability of an operation is often the overriding concern (Philip Robertson et al., 2014). Farmer's attitude towards OA's is also an issue that drives farmer's decisions. Cerda et al. (2017); Cerdà et al. (2018) have showed that Spanish farmers mainly see the cost associated with applying OA's (mulch) on their farms and are reluctant to do so without receiving subsidies for it, even though soil conditions improve by applying it.

Sustainable intensification of agriculture must be profitable for farmers. If not, there is less incentive to adopt methods that might involve greater energy efficiency and environmental stewardship. As Philip Robertson et al. (2014) comment: "On one hand are farmer's need for practices that ensure a sustained income in the face of market and consumer pressures to produce more for less; on the other are societal demands for clean and healthful environment. Most growers are caught in the middle".

Schemes like PES (payment for ecosystem services) might help to incentives farmers to ensure the provision of ES (DG Environment, 2012). Such schemes need to incorporate soil based ES frameworks (Jonsson and Davidsdottir, 2016; Jónsson et al., 2017) to provide farmers with incentives to maintain and protect soil natural capital, which is the basis for agriculture and its sustainable intensification. ES (Power, 2010; Ribaudo et al., 2010; Sandhu et al., 2010), EROI (Atlason et al., 2015a; Atlason et al., 2015b; Martinez-Alier, 2011; Pelletier et al., 2011) and CBA (Boardman, 2011; Hanley and Spash, 1993) have all been used in the context of agricultural production systems. However, according to our knowledge, this is the first time that a study looks at these three approaches simultaneously, specifically concerning the role of fertilisers. This combined approach provides a novel insight into the sustainability of an agricultural operation as it considers the effects of fertilisers through ecological, energy and economic perspectives.

This combined approach, however, is not a full-fledged sustainability assessment on its own, and this would require additional indicators to address the various aspects of sustainable intensification (Jonsson et al., 2016) and integrate the broader context associated with the Sustainable Development Goals (Bouma, 2014; Jonsson et al., 2016; Keesstra et al., 2016).

Some caveats are in order regarding the upscaling of the study plot, especially regarding the CBA and EROI assessments. Concerning the CBA analysis, we acknowledge there might be some scale economies involved in turning a small plot operation into a hectare based that we are not factoring in. There is a possibility that we are overestimating the cost associated with the tomato operation, which in turn has lowered our Benefit to Cost ratio. As a result, we argue that our calculations are likely to be on the conservative side and this possible overestimation does not change the conclusion that OA fertilisers have a more positive Benefit to Cost ratio than inorganic fertilisers. The same caveat applies regarding the upscaling of the EROI; we might be overestimating the energy use as we assume that upscaling would involve a more energy efficient[5] irrigation system as it has a high input cost. Other limitations to our analysis include the exclusion of possible ecosystem dis-services associated with agricultural operations in agroecosystems (Power, 2010; Zhang et al., 2007). While we acknowledge that there might be relevant dis-services associated with tomato production, their study was beyond the scope of this paper, which was to look at the effects of crop amendment on soil properties and tomato growth, illustrated by the selected ES, EROI and CBA. Future studies could look at replicating these same results on a larger scale with both a broader set of soil ecosystem services and dis-services, preferably covering all the essential soil functions and services put forward in the EU's Soil Thematic Strategy (European Commission, 2006) and a broader set of overall indicators similar to those proposed in Jonsson et al. (2016).

Sustainable agriculture should consider all four pillars of sustainability, which are the environment, economy, society and human well-being. Considering the global challenge to increase food production, sustainability in agriculture will have to deal with the trade-offs between the environmental impacts, the economic valuation as well as the societal acceptance of the proposed practices that will result in an overall improvement to human well-being. This societal challenge has to be addressed while many parts of the world will have to cope, adapt and mitigate the impacts of climate change which in the case of western Mediterranean will be quite significant (Nerantzaki et al., 2015; Nerantzaki et al., 2016; Nikolaidis et al., 2013) and the areas are prone to desertification because of climate change.

An agricultural operation cannot be considered sustainable unless it sustains or improves the ecological infrastructure of the soil, maintains a positive energy balance and provides economic benefits. We believe the results of the four treatments have wider implications for agricultural practices and sustainability, both for crops and livestock systems as it shows it's possible to intensify production in an energy efficient manner using OA's, that are often unused by-products in agricultural systems, while improving soil ecosystem services.

5. Conclusion

The combined use of the three methods presented in this paper provides additional insight into the sustainable use of soil natural capital from ecological, energy and economic perspectives. This study showed that OAs are fertilisers that can provide multiple benefits; they are energy efficient and, as shown by the EROI assessment, they can enhance sustainable soil use as they contribute to multiple soil ES and provide direct and indirect economic benefits for their users. The MSWC + M treatment was deemed the best overall because it had the highest EROI, the highest Benefit to Cost ratio and the second highest soil ES and was thus listed as the most beneficial overall in the multi-criteria ranking. The comparative analysis using these three methods provides valuable information to facilitate sustainable soil use, which is needed to enable the foreseeable intensification of agriculture in the coming decades.

[5] It is certainly a possibility the irrigation system might not be more efficient in a larger system leading to more gasoline use and a lower EROI. This would not though change the order of the results.

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Paper V

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Soil Science Issues

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

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The ecosystem services approach endeavors 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 macroeconomic 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).

Abbreviations: CAP, common agricultural policy; DST, decision support tool; GAEC, good agricultural and environmental conditions; LCA, life cycle assessment; NAC, natural asset check; PES, payments for ecosystem services; REDD, reduced emissions from deforestation and degradation; SEEA, system of environmental and economic accounts; SFM, sustainable forest management; SNA, system of national accounts; TEV, total economic value, WTP, willingness to pay.

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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., 2010; 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 treetop to bedrock, the biogeochemical engine at the heart of the earth's life support systems, with soil formation underpinning ecosystem services (Millennium Ecosystem Assessment [MEA], 2005). Yet, soil science appears slow in refocusing and mobilizing our creative talents (Bouma, 2005) 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[1] (in relation to science, social science, and policy making). The ecosystem services concept goes beyond ecosystem 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.

[1] 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).

As a result of the pressure on policymakers to consider soil multifunctionality 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 policymaker's 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 Economics of Ecosystems and Biodiversity approach, does not 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 recognize 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 nonmarket 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 as follows: "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 macroeconomic national accounting level as well as micro policy mechanisms such as 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 judgments 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 judgment 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.

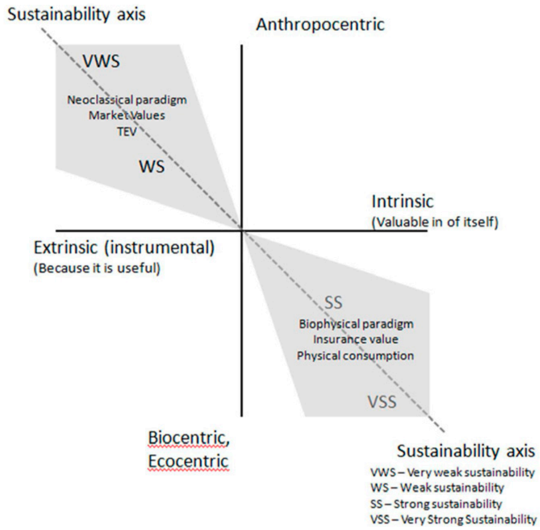

Fig. 1. Dimensions for value frameworks based on value type on the horizontal axis and moral standing of humans and biota on the vertical axis. The dashed line represents the sustainability axis indicating where within the value dimensions different sustainability world views tend to be located. Economic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

The Meaning of Economic Value

Economic value (neoclassical) is based on a framework for valuation that people are most familiar with as affecting our everyday lives. Total economic value (TEV) is the sum of all relevant use and non-use values generated now and in the future, that is, 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 Fig. 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 nonmarketable. These are the consumptive and nonconsumptive 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 Fig. 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 (i.e., 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 1 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 recognized. 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

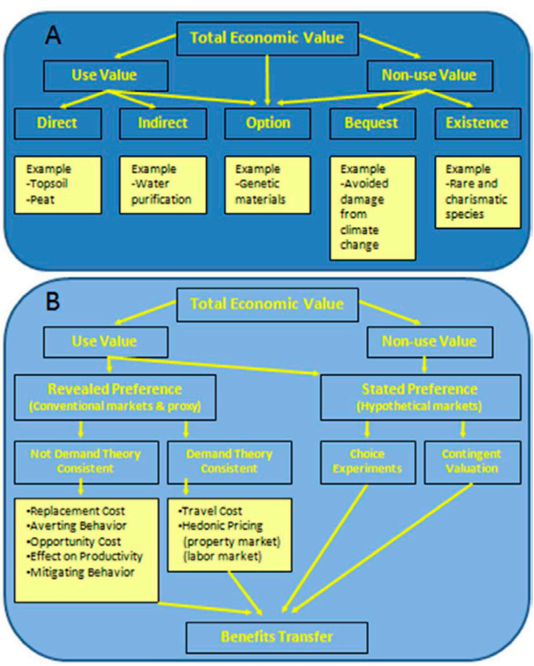

Fig. 2 (a) The total economic value framework (TEV) showing different types of economic value. Note price comes under direct use. (b) Economic methods used to estimate different types of value.

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 nonmarket 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 nonmarket goods, preferences are based on imperfect knowledge of ecological processes and functions. There are also long-standing disagreements within economics about the meaning of nonmarket value estimates generated using some of the most popular methods (e.g., 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 nonmarket valuation whereas Hausman, who is a leading researcher within the wider field of economics, considers efforts at nonmarket valuation dubious if not plain worthless. Finally, there are whole swathes of moral, ethical, and philosophical criticisms that have been made against nonmarket 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 nonmarket 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, p. 31): "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."

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 behavior 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 databases 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 WTP 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

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

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

al. (2000) argue that documenting ecosystem service values is useful because it does the following:

1. Highlights the importance of ecosystem functioning for mankind.

2. Highlights the specific importance of unseen, unattractive, or unspectacular ecosystems.

3. At a local level it can aid in identifying ecosystem services and acting as a help to decision making.

4. Can aid in understanding the impacts of change and feeding back to models to improve our understanding of ecosystem function.

5. Is a way of communicating value by translating to a common reference, for example, dollars.[2]

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 517, "Roadmap to a Resource Efficient Europe," which highlights the need to value human intervention regarding natural capital, to promote a more sustainable use of resources (European Comission, 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.

[2] 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.

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 to be rational, or functional, and on how to resolve conflict between different goals (Edvardsson, 2004; Edvardsson and Hansson, 2005; Edvardsson, 2007).

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, p. 75). 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 do not 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; and 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 attain-

Table 2. European Union soil threats based on the Impact Assessment [SEC(2006)620].†

Soil threat	Estimated annual cost
1) Erosion	€0.7–14.0 billion USD 1.05–21.03 billion, 2013
2) Organic matter decline	€3.4–5.6 billion USD 5.11–8.41 billion, 2013
3) Compaction	no estimate possible
4) Salinization	€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

† Conversions to 2013 USD use an exchange rate for the given year (1.3, 2006) and inflation using a CPI index calculator (Areppim, 2014).

able" 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 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 Fig. 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 realizes 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 U.S. economy (Christians, 2011) is by far the most visibly valuable. Peat by comparison is only \$13 million in the United Sates (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^3 of peat for gardening each year (DEFRA, 2011), though it is hoped to phase this out by 2020. Given the U.S. average price for peat of \$24.4 per short ton (\$26.84 per ton) in 2010 and assuming a bulk density of 0.2 tons m^{-3}, 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 and Le (1994) who estimated how much earth (soil, sediment, and rock) humans moved in 1988 based on U.S. house construction (HC; 0.8 Gtons yr^{-1}); mining (3.8 Gtons yr^{-1}, of which 0.86 Gtons yr^{-1} was sand and gravel [SG]); and road building (RB; 3.0 Gtons yr^{-1}), giving a total of 7.6 Gtons yr^{-1}. If we

Fig. 3. Geospatial assessment of soil prices around the globe based on a web survey of sites selling bulk topsoil. Median price in the United States and Canada \$22.25 per ton, Median price in the UK is \$47.09 per ton. 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.

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^{-1} is activity related to moving unconsolidated material, or about one third of earth material moved. Hooke and Le (1994) also estimated that agriculture moves 1.5 Gtons yr^{-1} 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 6.4 US$ 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 ton^{-1}, 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 Gtons) of soil was required for reconstruction and defenses. According to Hooke and Le (2000) an exponential increase in earth moving has occurred during our industrialized 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 tons of topsoil with their 7.5 million tons 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 1980s a soil factory was established by the Scottish Development Agency and the Clyde Port Authority along the River Clyde, Scotland, which produced 2000 tons of topsoil per week; feasibility studies have also been conducted in the United Sates 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 nonmarketable 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 on-site loss of production from the soil; consideration of off-site 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 na-

Table 3. Estimated annual cost of soil degradation at different administrative scales.†

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

† For detailed references see (1, Adhikari and Nadella, 2011; 2, Kuhlman et al., 2010; 3, Cohen et al., 2006; 4, Environment Agency, 2002; 5, Jones et al., 2008). Conversions to 2013 USD use an exchange rate for the given year (in parentheses) and inflation using a CPI index calculator (Areppim, 2014).

tional 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 effect on the resource base (i.e., 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 micronutrients 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 Yangzte is becoming more socially unacceptable. Moreover, the importance of soils in terms of their multifunctional use, for example, carbon storage, waste recycling, water filtration, and 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) 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, which 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 and Crilly, 2000). In the United States, 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 (e.g., Colombo et al., 2005, 2006; Almansa et al., 2012; Rosario-Diaz et al., 2013). It is these methods that cause so much tension and debate in relation to nonmarket 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 skepticism 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 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, for example, 1 ton plus for landscaping, as price is highly variable for small quantities sold in shops. Values were calculated for 1 ton 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 ton in the United States and Canada, and $47 per ton 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

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.†

	Commodity price per ton	T/ha to 30 cm	Cost, 30 cm of topsoil/ha
Sand Wanlip sand and 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 Dairy Co market information	£ 350	2	£ 700 $ 1085
Water ($25m^3m^{-3}$) Utility retail price metered m^3	£ 1.57	750	£ 1178 $ 1826
Worms (USA) Red worm composting blog Lowest retail price ($15/lb) Range ($15–40)	£ 4300	2	£ 8600 $ 13,300
Reconstituted topsoil		Total	£ 70,830 $ 109,787
Bulk recycled screened topsoil, Wanlip sand and 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 $1m^3$/~1 ton, Rolawn loam topsoil, Tesco.com	£ 100	3900	£ 390,000 $ 604,500

† Soil bulk density assumed to be ~$1.36g/cm^3$ (Loam: 40% sand 60% clay and silt); prices exclude taxes; conversion to USD uses exchange rate of 1.55 for 2013.

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 is potentially a sizeable equivalent replacement cost. The price used for carbon (£150) reflects the approximate current abatement cost for a ton 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 tons 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 fertilizer inputs are the major input they buy, but although the cost per ton 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 capital (Ekins et al., 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 nonspatial 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. Life cycle assessment 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," and "acidification"), resource flows are associated with different impacts (midpoints) and damages (so-called "endpoints") on the environment.

European Commission (2011) 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, 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 monetization 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 modeling 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 LCAs.

An alternative suite of DSTs seeks to make a fuller assessment of ecosystem services through greater biophysical assessment and modeling, 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 utilize soils data and predict soil change to some extent, for example, 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 modeling approach to predict ecosystem service dynamics, while 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, 2013b). At the regional to national level the Land Utilization and Capability Indicator model (LUCI) is another emerging tool optimized to quickly use nationally available data sets to determine ecosystem services (Jackson et al., 2013). Land Utilization and Capability Indicator 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, while 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 and 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 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, for example, extensive farm surveys, soil quality consulting, and scientific survey data, which could be released and collated centrally (after a suitable period), even exploiting crowdsourcing 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, for example, 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 (>2 m) 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.

Macroeconomic 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 and net national product, 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 nonmarket 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 macroeconomic 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 (i.e., 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 (System of Environmental-Economic Accounting [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" (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, p. 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, p. 175); and, as we have 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 to 2 m, 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, p. 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" (SEEA, 2012, p. 176). 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 southeast 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 ecosystem 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 biophysical data to allow for the construction of more comprehensive biophysical ecosystem accounts.

Valuation for Payments for Ecosystem Services

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 U.S. federal government "rents" ~140,000 km^2 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 (e.g., 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, p. 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 of rural development policy), 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. 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, for example, 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 ton is ~$22 in the United States and Canada, and ~$47 in the UK. Most direct value assessment in the literature is based on replacement costs and relates to erosion, while 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 recognized 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, for example, carbon levels, or salinity levels, etc. In these situations, economic assessments would require more frequent soil functional monitoring on which to base valuation. 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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Book chapter

5. THE VALUE OF SOIL ECOSYSTEM SERVICES

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1. Introduction

"It suddenly struck me that that tiny pea, pretty and blue, was the Earth. I put up my thumb and shut one eye, and my thumb blotted out the planet Earth. I didn't feel like a giant. I felt very, very small" Neil Armstrong.

Ecosystems and their importance

From space it is obvious to see that the Earth is what is called a closed system; there are no significant inputs coming from the outside except the energy from the sun. The sun is the basis for the living ecosystems and humans use energy and raw materials from natural systems to build their societies and economies. As the laws of thermodynamics prescribe, energy and materials can neither be created nor destroyed, and therefore any waste that human economies produce goes back to the surrounding natural systems. Furthermore, **the physical inputs derived from natural systems are limited, as the Earth is a closed systems, and so is its capability to assimilate waste.** This means that how human economies operate and what rules they operate by has tremendous consequences for the biosphere. The condition of the biosphere also has consequences for human wellbeing and economic development. **The Millennium Ecosystem Assessment clearly illustrated the importance of maintaining functioning of the natural systems, to ensure continued human wellbeing.** In the book *Limits to Growth* the consequences of the interaction of population rise and limited resources were studied with systems dynamics models, showing that endless growth is impossible. The results from this study are still relevant today, but the results clearly illustrate the problems that arise with limited resources and increased environmental impact of human actions.

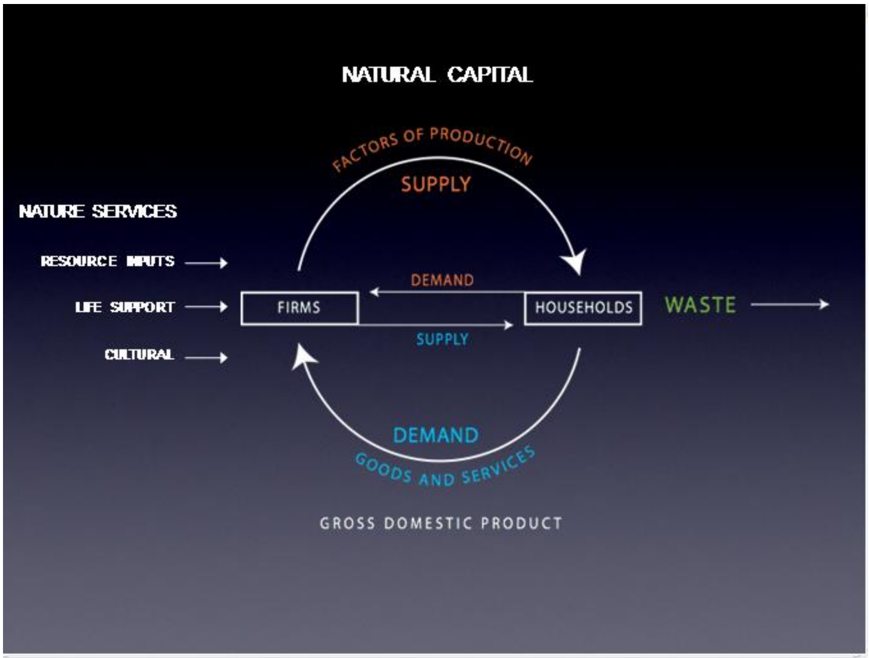

Figure 24. *The economy imbedded in natural systems.*

Natural capital and ecosystem services

What is natural capital? Ecological economists refer to natural systems, as natural capital (Figure 24). Natural capital, as other forms of capital (financial capital, human capital, built capital and social capital), yields a flow of goods and services of what has been collectively called ecosystem services***. Ecosystem services are simply put the benefits that humans derive from nature/natural capital.*** Humans use these services both directly and indirectly in their social and economic systems. **A direct service is something that is visible and often tangible, for example a food item such as fruit, fibres such as cotton, fresh water, energy and materials. Indirect services, however, are often invisible and intangible but no less important. Examples include; carbon sequestration in plants and in the soil, the formation of soil by natural processes, filtering and provisioning water which takes place out of sight by for example forests, wetlands and soils, and the sustenance of biodiversity.** An ecosystem can provide simultaneously many different ecosystem services that vary both spatially and temporally. If natural capital is degraded it loses its ability to provide us with the services needed for humans and other living beings to thrive, affecting wellbeing of all. This relationship was clearly illustrated in the Millennium Ecosystem Assessment as shown in Figure 25.

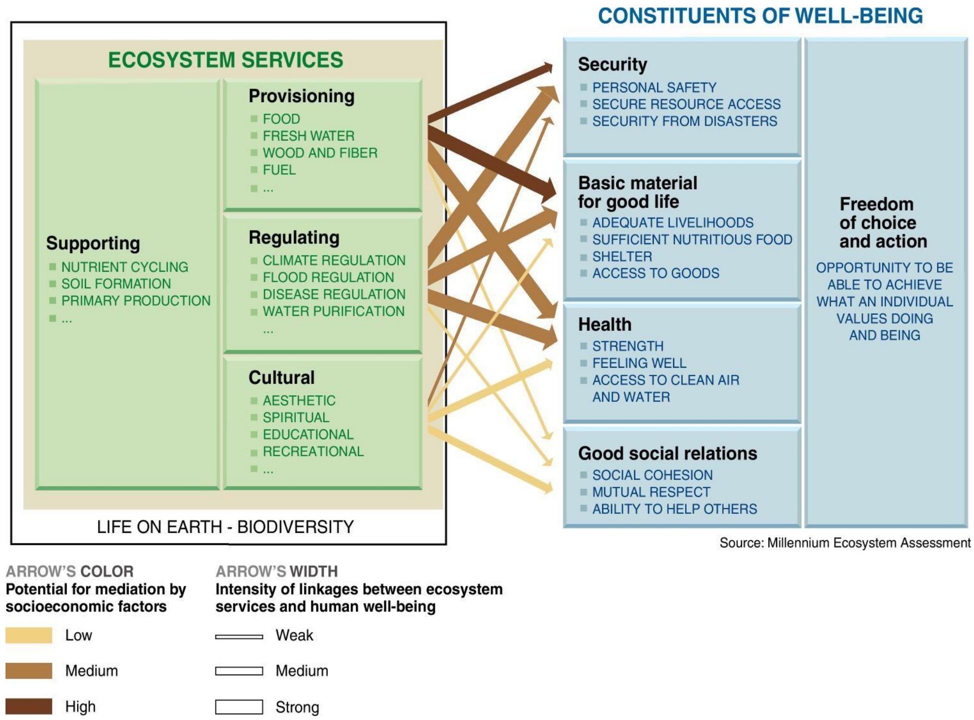

Figure 25. *The relationship between ecosystem services and human well being (Source: Millennium Ecosystem Assessment 2005).*

In the Millennium Ecosystem Assessment ecosystem services were categorized in four main groups depending on what services they provide. The groups are: supporting, regulating, provisioning and cultural services. ***Supporting services*** provide the necessary intermediate services for the other service groups, and include primary production, nutrient cycling and creating the living conditions for biodiversity. ***Regulating services*** are services that maintain and regulate essential ecological processes and life support systems through bio-geochemical cycles and other biospheric processes. Regulating services include climate regulation through for example carbon sequestration, flood prevention, prevention of outbreaks of pests and diseases and water purification. ***Provisioning services*** are services that provide direct inputs into social and economic system such as food and fibre, raw materials and energy. ***Cultural services*** are the nonmaterial benefits obtained from ecosystems such as recreational, educational, spiritual and aesthetic services. Maintaining and nourishing our natural systems and thereby growing our natural capital, will ensure that we continue enjoying the services provided by nature. As a result, **maintaining natural capital is necessary for continued human wellbeing.**

Soil ecosystem services

What are soil ecosystem services *(Figure 26)***?*** Soils are an important type of natural capital that has specific functions that provide multiple important ecosystem services (see also Chapter 2). ***Some even call soils the living skin of the Earth.*** **Around 99% of all our food comes from the land and the soil. Soils filter and clean our drinking water, they deliver the nutrients that plants need for growth and decompose them when they die. Soils provide**

habitats for millions of species where they can grow and flourish and if you would take teaspoon of soil from you backyard there probably would be billion/millions of microbes, thousands of funguses in that single teaspoon. Soil stores twice as much carbon as the biosphere and the atmosphere combined. Soils help to keep our climate stable by sequestering and releasing greenhouse gases like CO_2, they also can buffer heat waves and ameliorate local climate. Soils also regulate water flows and thereby prevent floods. Soil thus acts as a natural filter for water ensuring safe drinking water for us. Soil particles help with cloud formation released from the Earth´s surface through intensive agriculture and deforestation and provide nutrients for the smallest creatures in the ocean. Soils provide us with materials, which we use to build our cities and industries as well as provide the structural foundation needed. They provide us with medicine, probiotics and antibiotics, which makes us healthy. Immune systems of healthy adults "remember" germs to which they have never been exposed.

Figure 26. *Ecosystem services provided by soils (source: http://www.nature.com/scitable/knowledge/library/what-are-soils-67647639)*

Soils store our history, in buried ruins and sediments and they give us the opportunity to look into the past by studying layers of soil (pedology), so we can educate ourselves about our ancestors' discovery. Examples include the people preserved in peat bogs and clay-covered graves in Denmark and Germany for 3,000 years, with skin, hair and clothes conserved. The soil is our largest historical archive, and most of the artifacts stored have yet not been seen, read or discovered. This applies for all countries on Earth. If the soil is damaged or destroyed, then our largest historical archive are harmed.

Soils build and support magnificent landscapes and give us the chance to experience the marvels of nature. They have been a source of entertainment for children through the ages (play in the mud anyone?!?) and a source of recreation for old and the young - both easy going like gardening or intense like dirt bike racing. They are a fundamental part of our religion, the indoeuropean pantheon, later also the Judeo-Christian faith (God created man from soil) and our connection to the deity, for instance the ancient Mayan culture believed the soil was a gift from the ancestors. For the Incas of ancient Peru, the Earth Goddess (Pachamama), personified the Earth. The religions of the Middle East had their Earth Goddesses (Artemis, Asshera, Astarte, Demeter, Kybele, Ninhursagh) and Earth Gods (Enki, Ea). Derived from this discussion you can clearly see how soils contribute to all service categories as defined by the Millennium Ecosystem Assessment.

Given the importance of the multiple services derived from soils, it is clear that they need to be maintained and the only way to do that is to protect our soil natural capital. Soil, the skin of the Earth, is delicate; it is thin (on the average 15 cm) and forms slowly. It can take over 1000 years for 15 cm of soil to form in some areas but it can disappear in an instant, for example, during flash floods (see Chapter 3). Unfortunately, soil, as other types of natural capital, is coming under increased pressure because of human activities. We pave over them, pollute them with toxic substances, compress them with heavy agricultural machinery so they are as hard as concrete, leave them unprotected from the sun and let the wind blow them away and the rain wash them away.

International agencies tell us that desertification, land degradation and drought have an negative impact on more than 1.5 billion people in over 110 countries, 90% of them live in low-income countries, and that every year around 10 million hectares of agricultural land are lost because of soil erosion; this is equivalent to 1.5 times the size of Lake Victoria, Africa´s largest lake. Given how soils have been treated in the past **it is as if our economic decision-making frameworks do not recognize the multiple importances of our soil natural capital and its derived ecosystem services.**

Value and soil ecosystem services

Ecosystem services are fundamentally important for economic prosperity and human well-being. In the market economy, a dominant form of an economic system in the western world, decision-making is largely based on signals provided by the market through prices. Prices of goods and services are set by the interaction of supply (sellers) and demand (buyers), determining optimal quantities of output, as well as the optimal use of various inputs to the production process. Value is derived from the willingness to pay for a particular good or a service, illustrating relative economic importance and its relative scarcity.

Unfortunately, not all goods and services are captured by markets, and this is specially the case with many goods and services derived from natural capital. Such services are called non-market goods; **soils as natural capital and many soil ecosystem services are considered non-market goods and services. Their nature does not easily lend itself to be traded in markets and thus they have no market price, but are regardless immensely important for our economy**.

The value of non-market ecosystem services has been evaluated since the 1990s. It was found that for the entire biosphere, the value (most of which is outside the market) is estimated to be in the range of US$16–54 trillion ($10^{12}$) per year, with an average of US$33 trillion per year. Because of the nature of the uncertainties, this must be considered a minimum estimate. Global gross world product total in 1994 was around US$18 trillion per year – indicating that nature gives us for free at least as much value as global production of goods and services. Since then many estimates have been conducted for ecosystem services, further supporting the importance of formally accounting for these services in economic decision-making through valuation.

Unfortunately as our economies are managed as market economies, non-market goods are invisible in the market and thus are largely excluded from economic decision-making. This fact has often resulted in misguided economic decisions as they are based on incomplete information, resulting in the degradation of natural capital such as soils.

This absence of value can be addressed with assessment methods that relate to how economics treat the concept of value. The theory of value in economics relates to the idea of human well-being and that well-being is based on economic benefits which economic decision making aims to maximize. **Economic benefits, and thus value is assessed through our willingness to pay for a particular good or a service.** This notion of willingness to pay is used to assess the value of non-market goods and services derived from natural capital.

Values derived from natural capital such as soils are broken into several types. The two main types of values are what are called **use value and non-use value.** Use values are broken into direct and indirect use values. Direct use values include consumptive uses such as food (collection of berries, mushrooms, herbs and plants) and fibre, whereas non-consumptive uses include for example recreation, photography and view from a dwelling. Indirect use values include use values that are not consumed such as carbon sequestration, hydrological buffering, filtering of nutrients and contaminants and biological control of pests and diseases.

Non-use values include *option*, *bequest* and *existence* values. The concept of non-use value refers to the value that people assign to economic goods and services (including public goods, public assists or public resources) even if they never have and never will use them. Option value is individual willingness to pay for maintaining natural capital such as soils even if there is little or no likelihood of the individual actually ever using its derived services, but there is value in maintaining the possibility that it may someday be used. Bequest value is the willingness to pay for maintaining or preserving natural capital that has no use now, so its services are available for future generations. Existence value reflects the benefit people receive from knowing that a particular natural capital and its associated services exist. The total economic value of soil ecosystem services is the sum of all use and non-use values.

Economic valuation methods

The notion of value used to **obtain use value and non-use values relies on people's willingness to pay for ecosystem services, reflecting their importance** (Figure 27). Several valuation methods exist, varying what they measure and the data required. The methods are categorized according to whether preferences and thus willingness to pay are expressed in surveys or revealed through actual behaviour.

Revealed preference techniques base the value of ecosystem services on actual observed behaviour linked to the service or associated services or products or the revealed willingness to pay for a mechanism or a product that somewhat replaces the ecosystem service. The main methods are: *Market prices*; most commonly used to value provisioning services, *Cost based metrics;* including avoidance cost, replacement cost, and *damage expenditures*; most commonly used to value supporting and regulating services, *Travel cost*; used to value cultural services such as recreational value, and *Hedonic pricing*; used to value cultural services such as amenities.

Stated preference techniques elicit values directly through survey methods where subjects are asked about their willingness to pay to conserve a particular ecosystem service or to conserve an entire ecosystem or their willingness to accept a fee for losing a service or an ecosystem. Contingent valuation methods or choice experiments are the most commonly used stated preference methods, and are used to capture non-use values such as existence value as well as they can be used to assess all use values.

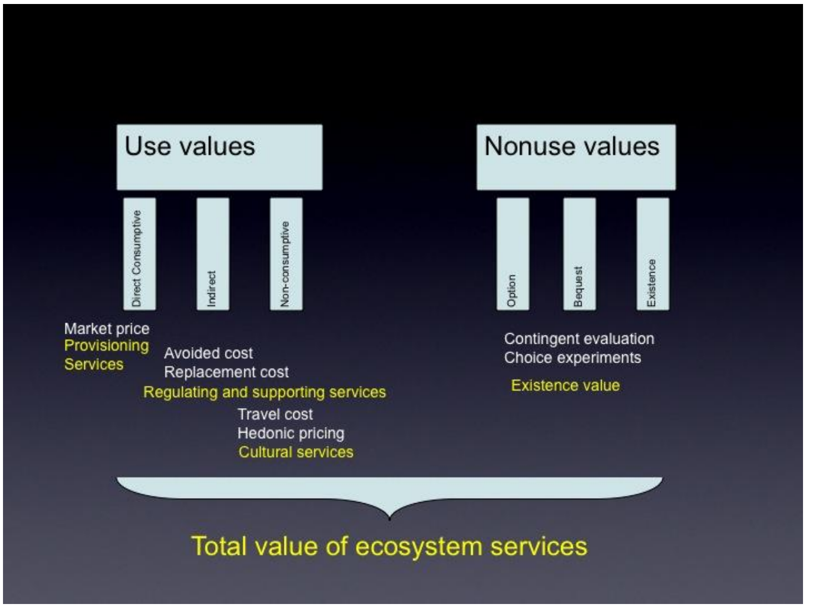

Figure 27. *Types of values derived from ecosystem services and valuation methods.*

Decision-making regarding sustainable land use and soil management

As with many other natural systems, the **services from soils suffer from the lack of proper economic valuation,** be it monetary or some other and many of the soils services are not even taken into consideration when ecosystems are analysed with the conventional ecosystem services approach.

In the next 50 years we need to grow more food than we have done for the last 10 thousand years and we have to do that with less land. This will put enormous pressure upon soils. **What decisions we make regarding land use and soil management are therefore of the outmost importance.** If we continue to overexploit the land and degrade the soil this will lead to reduction in the future provision of the services soils provide.

We need to change our design of decision making processes in such a way that the essential services that soils provide are factored into the process and that it results in decisions that sustain healthy and functioning soils. Valuing soils with some of the methods mentioned in this chapter is a step towards such a change, though of course we cannot fully price the total value of the natural world, nor do we want to. **By using the tools of economics that are at our disposal, along with other social and environmental tools such as soil sustainability indicators we can move towards more holistic approach regarding sustainable soil and land management.**

Beyond money

Existential values of soil beyond money also exists, where the value is determined in the expressions of Shakespeare's character Hamlet's fundamental question "to be or not to be." In face of existence or not existence, if society cannot persist with a certain type of consequence, then any form of money or discussion thereof is redundant, and we have to make a decision based on existential and ethically based choices.

Discussion

Natural capital such as soil is very important for our continued wellbeing. Soils provide us with essential soil ecosystem services that must be maintained, and the only way to secure their maintenance is to protect soil natural capital. Since many soil ecosystem services do not carry a market price, we do not think about them when making decisions every day. Therefore, soils tend to be overused, and soil natural capital degraded. To get us to think about the economic importance of soil natural capital, economists have recently developed methods to assess the economic value of soil ecosystem services. Hopefully such assessments will illustrate the immense economic importance of soils, and enable us to reverse the trend of soil degradation that is bound to harm our future well-being.

Exercises

1. Think about all the different things soils do for you and your wellbeing and try to place them in the classes defined by the Millennium Ecosystem Assessment. What is the service that is the most important to you and why?
2. Go to your local gardening shop or online and find how much we pay for soils in our daily lives. Considering that soils form at the rate of only millimetres per 100 years, do you think that the price of soil in the market reflects their value?

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6. CONCLUDING REMARKS

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As can be seen in this book, soils are one of our most important natural resources and yet we do not look after soil as we should. The reasons are many as highlighted in the five chapters above. It would appear that we did not learn from history as outlined in Chapter 1. There are many agro-ecological approaches that can be adopted that have been shown to increase both soil resilience and stability, but also crop yield. Chapter 2 outlines what soil does for us, soil function, soil impact on the water cycle and regulation of the global clime, soil provision of habitat, importance of soil for the carbon cycle, soil nutrient transformations and medium for plant growth and soil as a natural filter. In Chapter 3 the processes that cause soil degradation are outlined and solutions are suggested for soil protection. In Chapter 4 we consider the importance of understanding the life cycle of soils as well as steps to assess impacts on soil quality. Finally in Chapter 5 natural capital is introduced, the concept of soil ecosystem services is outlined, showing the many services they provide: They provide food, filter our drinking water, deliver nutrients for plants, and decompose organic matter in soil. Soils provide habitats for millions of species, stores twice as much carbon as the biosphere and atmosphere combined. Soils buffer climate and heat waves. Soils regulate water, soil particles aid in cloud formation, provide building material and structural foundations. They also provide us with medicine, and strengthen our immune system.

Given the importance of soils for survival of ecosystems and humans alike, what would you as a pupil at school suggest that we do to change direction?

Exercises

1. What are the agroecological approaches that you think are the most important for soils in your area? If you live in the city, focus on the soils in your garden, nearby park or allotments.
2. Have you ever gone into your garden and played in the soil? What did you see?
3. Does your family have a compost bin? If not could you sent one up?
4. Does your school have a garden? Have you ever tried to grow anything in soil? If not, why not try?
5. What are the most important soil erosion processes that you have seen in your area?
6. Have you ever thought about what life cycle assessment of a product or a service?
7. Do you think that it is important to economically value soil and their services?
8. How come that soils are not better protected for our own well being and future generations? What can you do to help protect soils?

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