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Biogeochemical cycles and biodiversity as key drivers of ecosystem services provided by soils

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Abstract

Soils play a pivotal role in major global biogeochemical cycles (carbon, nutrient and water), while hosting the largest diversity of organisms on land. Because of this, soils deliver fundamental ecosystem services, and management to change a soil process in support of one ecosystem service can either provide co-benefits to other services or can result in trade-offs. In this critical review, we report the state-of-the-art understanding concerning the biogeochemical cycles and biodiversity in soil, and relate these to the provisioning, regulating, supporting and cultural ecosystem services which they underpin. We then outline key knowledge gaps and research challenges, before providing recommendations for management activities to support the continued delivery of ecosystem services from soils.

We conclude that although there are knowledge gaps that require further research, enough is known to start improving soils globally. The main challenge is in finding ways to share knowledge with soil managers and policy-makers, so that best-practice management can be implemented. A key element of this knowledge sharing must be in raising awareness of the multiple ecosystem services underpinned by soils, and the natural capital they provide. The International Year of Soils in 2015 presents the perfect opportunity to begin a step-change in how we harness scientific knowledge to bring about more sustainable use of soils for a secure global society.

1 Introduction

Soils play a critical role in delivering a variety of ecosystem services (Scholes and Scholes, 2013). Management aimed at improving a particular ecosystem service can either provide co-benefits to other services or can result in trade-offs (Robinson et al., 2013). Examples of some of the synergies and trade-offs (Smith et al., 2013) and the role of soils in supporting ecosystem services, and their role in underpinning natural capital (Dominati et al., 2010; Robinson et al., 2009, 2014) have recently been re-

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viewed. The ability of soils to provide services is principally conferred by two attributes: the range of biogeochemical processes that occur in the soil; and the richness and functionaliy of soil biodiversity. In the following sub-sections we present the state-of-the-art understanding and knowledge gaps on carbon, nutrient and water cycling in soil, and their role as a habitat for organisms and as a genetic pool. We clarify how the biogeochemical processes provide regulating, provisioning and supporting services, and the role of biodiversity (genetic diversity, functional diversity, and abundance and activity of organisms) in supporting these services. These functions collectively confer soil health, which is critical for the underpinning of cultural services, among other things.

The Millennium Ecosystem Assessment (MA) (Millennium Ecosystem Assessment, 2005), classified ecosystem services into supporting, regulating, provisioning and cultural services, and this categorisation is widely used, and though the scheme was not designed to fit all assessments (Fisher et al., 2009), it has been modified for use in national ecosystem assessments (e.g. UKNEA, 2011). More recently, the Common International Classification of Ecosystem Services (CICES; Haines-Young and Potschin, 2012) was developed to support environmental accounting in the European Union and in the United Nations Statistical Division (European Commission et al., 2013; European Commission et al., 2014). A major difference between the MA and the CICES classification systems is that CICES does not include supporting services (see below), which are treated as intermediate steps in the delivery of final goods and services (Haines-Young and Potschin, 2012). In this review, we include supporting services, since they are often refered to in the literature, while accepting the CICES observation that supporting services are not of direct benefit of people, although they are of great indirect benefit. The MA (Millennium Ecosystem Assessment, 2005; supplemented by UKNEA, 2011 for supporting services) provides definitions and examples of provisioning, regulating, supporting, and cultural services as follows:

Provisioning services are "physical products obtained from ecosystems" and include: food (including wild-harvested seafood and game, cultivated crops, wild foods and spices); raw materials (including timber, pulp, skins, animal and vegetable fibres, organic matter, fodder, and fertilizer); genetic resources (including genes for crop improvement and health care); freshwater; minerals; medicinal resources (including pharmaceuticals, chemical models, and test and assay organisms); energy (hydropower, biomass feedstocks including biofuels, wood and charcoal,); and ornamental resources (including fashion, handicraft, jewellery, pets, worship, decoration and souvenirs like furs, feathers, ivory, orchids, butterflies, aquarium fish, shells, etc.).

Regulating services are "Benefits obtained from the regulation of ecosystem processes" and include: carbon sequestration and climate regulation; waste decomposition and detoxification; pollutant immobilization and detoxification; purification of water and air; regulation of water flow (including flood alleviation); and pest and disease control.

Supporting services are "Ecosystem services that are necessary for the production of all other ecosystem services" and include: soil formation; nutrient cycling; water cycling, primary production and habitat for biodiversity.

Cultural services are "Nonmaterial benefits people obtain from ecosystems through spiritual enrichment, cognitive development, reflection, recreation, and aesthetic experiences" and include: cultural (including use of nature as motif in books, film, painting, folklore, national symbols, architectural, advertising, etc.); spiritual and historical (including use of nature for religious or heritage value or sense of place); recreational experiences (including ecotourism, outdoor sports, and recreation); science and education (including use of natural systems for school excursions, and scientific discovery).

Figure 1 summarises the ecosystem services underpinned by soils. In the following sections, we examine the state-of-the-art understanding of carbon, nutrient and water cycles and biodiversity in soils, and show how these underpin the provisioning, regulating, supporting and cultural ecosystem services described above. We then discuss the knowledge gaps across all of these areas, recommend key foci for future research, and present recommendations for policies to support the continued delivery of these ecosystem services from soils.

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2 Soils and the Carbon cycle

2.1 Soil C stocks

Carbon (C) storage is an important ecosystem function of soils that has gained increasing attention in recent years. Changes in soil C impacts on, and feedbacks with, the Earth's climate system through emissions of CO_2 and CH_4, and storage of carbon removed from the atmosphere during photosynthesis (climate regulation). Soil organic matter itself also confers multiple benefits for human society e.g. enhancing water purification and water holding capacity, protecting against erosion risk, and enhancing food and fibre provision through improved soil fertility (Pan et al., 2013, 2014).

Soil is an important C reservoir that contains more C (at least 1500–2400 PgC) than the atmosphere (590 PgC) and terrestrial vegetation (350–550 PgC), combined (Schlesinger and Bernhardt, 2013; Ciais et al., 2013), and an increase in soil C storage can reduce atmospheric CO_2 concentrations (Whitmore et al., 2014). All three reservoirs of C are in constant exchange but with various turnover times, with soil as the largest active terrestrial reservoir in the global C cycle (Lal, 2008). Carbon storage in soils occur both in organic and inorganic form. Organic C stocks in the world's soils have been estimated to comprise 1500 Pg of C to 1 m depth and 2500 Pg to 2 m (Batjes, 1996). Recent studies showed that the soil C pool to 1 m depth may be even greater and could account for as much as 2000 Pg. These higher values are mainly based on increased estimates of the C stored in boreal soils under permafrost conditions (Tarnocai et al., 2009), in which decomposition is inhibited by low temperature, and lack of oxygen and low pH in waterlogged soils (e.g. peats; Smith et al., 2010). Although the highest C concentrations are found in the top 30 cm of soil, the major proportion of total C stock is present below 30 cm depth (Batjes, 1996). In the northern circumpolar permafrost region, at least 61 % of the total soil C is stored below 30 cm depth (Tarnocai et al., 2009). Peatlands are particularly important component of the global soil carbon store, covering only 3 % of the land area, but containing around

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500 PgC in organic rich deposits ranging from 0.5 m up to 8 m deep (Gorham, 1991; Yu, 2012).

In arid and semi-arid soils, significant inorganic C can be present as carbonate minerals (typically Ca / $MgCO_3$, called “calcrete” or “caliche” in various parts of the world), formed from the reaction of biocarbonate (derived from CO_2 in the soil) with free base cations, which can then be precipitated in subsoil layers (Nordt et al., 2000). Also soils derived from carbonate-containing parent material (e.g. limestone) can have significant amounts of inorganic C. The inorganic C pool globally is large, estimated to be ~750 Pg C to a depth of 1 m (Batjes, 1996). However, in most cases, changes in inorganic C stocks are slow and not amenable to traditional soil management practices, and do not play a significant role in terms of most ecosystem services (though a major exception is the geoengineering proposal to add finely-ground silicate minerals to soils, which will then weather to carbonates, taking up CO_2 in the process; Köhler et al., 2010). Thus, further discussion of soil C in this review will focus on soil organic C.

The net balance of soil C depends on the C inputs to soils relative to C losses. Losses can occur via mineralization (i.e. decomposition), leaching of dissolved C and carbonate weathering (Smith, 2012; Schlesinger and Bernhardt, 2013). Thus the soil organic C stock may either increase or decrease in response to changes in climate and land use practices. Furthermore, rates of SOC stock change in different parts of the profile can vary for different soils and types of perturbation, because some portion of the C stored in soil mainly in top soil turns over rapidly, while other soil C fractions in particular mineral associated C and subsoil C can have a long residence time, on the order of centuries or millennia (von Lützow et al., 2008; Rumpel and Kögel-Knabner, 2011). The accumulation of stabilised C with long residence times in deep soil horizons may be due to continuous transport, temporary immobilisation and microbial processing of dissolved organic matter within the soil profile (Kalbitz and Kaiser, 2012) and/or efficient stabilisation of root-derived organic matter within the soil matrix (Rasse et al., 2005). The process of soil formation – i.e. the development of depth, horizons and specific properties - is itself a supporting service.

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High SOC content also improves other chemical and physical soil properties, such as nutrient storage (supporting service), water holding capacity (supporting and regulating service), aggregation and sorption of organic or inorganic pollutants (regulating service). Carbon sequestration in soils may therefore be a cost-effective and environmentally friendly way, not only to store C for climate regulation, but also to enhance other ecosystem services derived from soil, such as agricultural production, clean water supply, and biodiversity (Pan et al., 2013) by improving soil organic matter (SOM) content and thereby soil quality (Lal, 2004). Moreover, processes which improve SOM may themselves provide services, e.g. use of cover crops which can provide provisioning or water regulation services while improving soil C. SOM or soil carbon are widely-used proxy variables for soil health (e.g. Kibblewhite et al., 2008).

2.2 C cycling

Carbon enters the soil as aboveground or belowground plant litter and exudates. C input is not homogenous within the soil profile. Whereas topsoil receives higher amounts of aboveground litter, subsoil C originates from root C as well as dissolved C, transported down the soil profile. Root C has a greater likelihood of being preserved in soil compared to shoot C, and was therefore hypothesised to account for most of the SOC (Rasse et al., 2005). The majority of plant litter compounds are processed by the soil biota. Thus SOM is composed of plant litter compounds as well as microbial and, to a smaller extent, faunal decomposition products (Paul, 2014). It is a complex biogeochemical mixture comprising molecules derived from organic material in all stages of decomposition. Some organic matter compounds, including microbial decomposition products, may be stabilised for centuries to millennia by binding to soil minerals or by physical occlusion into micro-aggregates (von Lützow et al., 2008), for example with iron oxyhydrates (Zhou et al., 2009), or through protection by occlusion within soil aggregates (Dungait et al., 2012). The inherent chemical recalcitrance of some plant litter compounds (e.g. lignin) has a minor influence on their longevity in soil Thevenot et al., 2010) whereas the location of SOM within the soil matrix has a much stronger

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control on its turnover (Chabbi et al., 2009; Dungait et al., 2012). Mineral-associated SOM is predominantly composed of microbial products (Miltner et al., 2012). Therefore, microbial use efficiency (Liu et al., 2011) of plant inputs largely determines SOM stabilisation through interaction with the mineral phase (Cotrufo et al., 2013). In peatlands, organic matter is stabilised by high water tables that slow down biological activity and decomposition. Eventually, all SOM is mineralized to carbon dioxide (CO_2) in aerobic environments, or reduced to methane (CH_4) in anaerobic environments. Soil CO_2 efflux, resulting from SOM mineralization, and from rhizosphere respiration and inorganic C weathering, is the largest terrestrial flux of CO_2 to the atmosphere (~ 60 PgC, an element of the climate regulation service; Smith 2004). This flux is an order of magnitude larger than anthropogenic CO_2 emissions due to fossil fuel burning and land use change (1.1 PgC yr^{-1}, Ciais et al 2013). Under anaerobic conditions, CH_4 is formed by methanogenic microorganisms. A proportion of this CH_4 is oxidised to CO_2 by methanotrophic microorganisms, but a proportion can be emitted from the soil surface. Since CH_4 is many times more potent as a greenhouse gas than CO_2 on a per-molecule of per-mass basis (Ciais et al., 2013), soil CH_4 emissions and their mitigation play an important role in climate regulation.

Fire may affect many ecosystem services including C sequestration. A decrease soil C storage has been observed initially, but through positive effects on plant growth as well as input of very stable pyrogenic C, C storage may increase at longer timescales (Knicker, 2007). An additional long-term C pool in many soils is pyrogenic-carbon (PyC), formed from partially combusted (i.e. pyrolysed) biomass during wildfires or other combustion processes (Schmidt and Noack, 2000). Globally, soils are estimated to contain between 54 and 109 Pg PyC (Bird et al., 2015). Some of this PyC has a highly condensed aromatic structure that retards microbial decay, and can thus persist in soils for relatively long periods (Sing et al., 2012). Soil amended with industrially produced PyC (biochar) as a climate mitigation technique often shows no increase in soil respiration despite the additional carbon, the reduced ecosystem carbon turnover results in increased soil carbon storage (Stewart et al., 2013). PyC additions to soil affect

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regulating ecosystem services, such as C sequestration, nutrient cycling and adsorption of contaminants. However, PyC properties, and as result their effect on ecosystem services, may be strongly dependent on fire conditions.

2.3 Factors influencing soil C storage

Fundamentally, the amount of C stored in a given soil is determined by the balance of C entering the soil, mainly via plant production but also through manures or amendments such as organic sludge or biochar, and C leaving the soil through mineralization (as CO_2), driven by microbial processes, and to a lesser extent leaching out of the soil of dissolved carbon and carbonate weathering. Locally, C can be lost or gained through soil erosion or deposition, leading to a redistribution of soil C, at landscape and regional scales (van Oost et al., 2007).

Consequently, the main controls on soil C storage are the amount and type of organic matter inputs, the efficiency by which this is used by microbes, and the capacity of the soil to retain it by physical or chemical stabilization (Cotrufo et al., 2013). In most natural and agricultural ecosystems, plant productivity and subsequent death and senescence of biomass provides the input of organic C to the soil system. Thus, higher levels of plant residue inputs will tend to support higher soil organic carbon stocks, and vice versa (Paustian et al., 1997). Plants also affect soil C cycling by their specific mycorrhizal associations (Brzostek et al., 2015). Shifts in specific mycorrhizal associations affect SOM storage by contributing to both SOM formation and decomposition. Ectomychorrizhal turnover is a dominant process of SOM formation (Godbold et al., 2006), possibly due to the more recalcitrant nature of the chitin in fungal tissues, compared to the cellulose and lignin in plant residues. In arbuscular mycorrhizal fungi, glomalin, a highly resistant glycoprotein, was found to have an active role in aggregate formation and SOM stocks (Rillig, 2004). Symbiotic mycorrhizal fungi can also directly impact the turnover of organic matter by the production of exo-enzymes (Averill et al., 2014; Finzi et al., 2015).

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In many regions of the world SOM accumulates because of inhibition of microbial SOM decomposition, due to cold, dry or anoxic conditions (Trumbore, 2009). In general, when water is not limiting, higher soil temperatures increase the rate of microbial decomposition of organic matter. Thus soil temperature is a major control of SOM storage in soil C cycle models (Peltoniemi et al., 2007). The temperature sensitivity of SOM decomposition is not, however, as straight-forward as represented in most models, but varies between the many different forms of chemical and physical protection of organic matter in soil (Conant et al., 2011; Zheng et al., 2012). Water influences soil C storage through several processes. Moist, but well-aerated, soils are optimal for microbial activity and decomposition rates decrease as soils become drier. However, flooded (saturated) soils have lower rates of organic matter decay due to restricted aeration and thus often have very high amounts of soil C (e.g. peat soils). High precipitation may also lead to C transport down the soil profile as dissolved and/or particulate organic matter, as well as lateral transport through soil erosion and deposition. During dry periods, SOM decomposition is decreased, but after rewetting there may be an accelerated pulse of CO_2 emission in aerobic soils (Borken and Matzner, 2008) , whereas drought and lowering water tables may increase decomposition in naturally anaerobic peats (Freeman et al, 2001; Clark et al., 2012). However, the effect of drought is not only direct via soil microbial activity. There are feedback loops concerning drought and C storage via plant activities, such as litter input and rhizodeposition. Drought was found to affect plant litter composition (Sanaullah et al., 2014), plant C flow and root exudation (Sanaullah et al., 2012), as well as the resulting enzyme activities in the rhizosphere (Sanaullah et al., 2011).

C cycling in soils is strongly linked to the cycling of N and P. Since the C : N : P stoichiometry in SOM is generally lower than in plant material – i.e. there is more N and P per unit C – C generally accumulates in aerobic soil where nutrients are not limiting (Alberti et al., 2014). Nevertheless, increase in organic C is often accompanied by increased N resource use efficiency in croplands (Pan et al., 2009), especially when SOC is increased with biochar (Huang, et al., 2013). In nutrient limited peatlands, in-

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puts of nitrogen and/or phosphorus within the tolerance levels of sensitive plant species have increased rates of carbon accumulation (Aerts et al., 1992; Turunen et al., 2004; Olid et al., 2014). The relationship between nutrients and C cycling is not straight forward, since nutrients are also needed by soil microbes to degrade SOM. Thus nutrient addition can either decrease or increase C storage, depending on the initial SOM stoichiometry, the ability of the soil minerals to stabilize microbial products of decomposition and the simultaneous effects on plant productivity and organic matter inputs to soils.

The amount and type of clay particles (and to a lesser extent silt particles) are the major factor controlling the quantity and composition of soil C (Sollins et al., 1996; von Lützow et al., 2006). Clays are mainly sheet-like crystals of silicon and aluminium, known as phyllosilicates, often located as skins coating soil aggregates. In clay-rich soils, higher organic matter content and a greater concentration of O-alkyl C derived from polysaccharides may be expected compared to sandy soil, which are characterised by lower C contents and high concentrations of alkyl C (Rumpel and Kögel-Knabner, 2011). Aliphatic material may be responsible for the hydrophobicity of soils, which can lead to reduced microbial accessibility and therefore increased C storage (Lorenz et al., 2007). Many of the OM-matrix interactions are driven by expandable and non- expandable phyllosilicates, which interact with organic compounds through their large surface areas, micro pores and micro aggregation, particularly in acid soils. By contrast, in neutral and calcareous soils, polyvalent cations (especially Ca^{2+}) predominate in the interaction mechanism, forming bridges between the largely negatively charged SOM and negatively charged phyllosilicates (Cotrufo et al., 2013). Short order silicates, like allophane, provide some of the strongest organo-mineral interactions and stabilize both proteins and carbohydrate monomers, though their occurrence is much more geographically restricted (Buurman et al., 2007; Dümig et al., 2012: Mikutta and Kaiser 2011). In rice paddies, iron oxyhydrates usually act as coating of soil mineral particles and stabilize carbon, contributing to a higher C storage and stability than in iron-poor soils (Song et al., 2012).

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Bioturbation (the mixing of soil by organisms) may further influence the amount as well as the chemical nature of soil C. It greatly influences the heterogeneity of soils by creating hotspots of carbon and biological activity. On biologically active sites, incorporation and transformation of organic compounds into soil is usually enhanced, leading to more organo-mineral interactions and increased C storage (Wilkinson et al., 2009).

Microbial decomposition of SOM may be stimulated by the input of labile (easily decomposed) organic matter, through the priming effect (Jenkinson et al., 1971). Positive priming refers to greater mineralisation of otherwise stable C through shifts in microbial community composition and activity (Fontaine et al., 2003). However, in some cases the addition of organic matter to soil may also impede mineralisation of native SOM (negative priming effect), thereby protecting SOM from its decomposition. Plant communities are the main controlling factors of these processes because they influence organic matter input and microbial activity by their effects on soil water, labile C input, pH and nutrient cycling (Kuzyakov et al., 2000).

By storing and cycling C, nutrients and water, soils provide supporting services like soil formation, nutrient retention and water retention, which underpin both primary production and landscape hydrology (the processes which deliver provisioning services such as food, fibre and water), in addition to the regulating services such as climate regulation already discussed (Fig. 1). To assure that soils continues to provide these key services soil will require to be managed both for C preservation – thus mitigating climate change – while simultaneously permitting continued SOM recycling. Janzen (2006) pointed to this dilemma, that there is a trade-off between improved soil fertility to support the provisioning services of food/timber production and the regulating service of soil carbon sequestration aiding climate regulation. Despite knowledge on which practices are likely to lead to improved SOC status, better understanding of the controls on SOM distribution, stabilisation and turnover will help to better target these practices. This will be an important contribution to the mitigation of greenhouse gases, while assuring decomposition and with it the cycling of nutrients necessary to

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support food production. Table 1 summarises management actions affecting the soil carbon cycle and their impacts on ecosystem services.

3 Soils and nutrient cycles

Soils support primary production among other services, which in turn delivers the provisioning services of food and fibre production. As such, soils are vital to humanity since they provide essential nutrients, such as nitrogen (N), phosphorous (P) and potassium (K) and many trace elements that support biomass production, which is essential for supply of human and animal food, for energy and fibre production and as a (future) feedstock for the chemical industry. Since the 1950s, higher biomass production and yield increases have been supported through fertilizers derived from mined minerals or industrially synthesised (Fig. 2). Intensification of agricultural practices and land use has in many regions resulted in a decline in the content of organic matter in agricultural, arable soils (Matson et al., 1997). In some areas, extensive use of mineral fertilizers has led to atmospheric pollution, greenhouse gas emissions (e.g. N_2O, very important for climate regulation), water eutrophication, and human health risks (Galloway et al., 2008), thereby negatively affecting the regulating services of soil, air and water quality (Smith et al., 2013). During the 21st century, it is likely that the human population and demand for food, feed and energy will rise. In order to sustain biomass production in the future, and to avoid negative environmental impacts, fertile soils need to be preserved and soil fertility needs to be restored where lost. This can be done through both the recycling and accumulation of sufficient amounts of organic matter in soils (Janzen, 2006), through a combination of plant production and targeted additions of organic and mineral amendments to soils (see Sect. 2).

The soil function "fertility" refers to the ability of soil to support and sustain plant growth; which relates to making available N, P, other nutrients, water and oxygen for root uptake in the amounts and proportions needed, when they are needed. This is facilitated by (i) their storage and buffering in soil organic matter, (ii) nutrient recycling

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from organic to plant available mineral forms, and (iii) physical – chemical processes that control their sorption, availability, displacement and eventual losses to the atmosphere and water. Managed soils are a highly dynamic system and it is this very dynamism that makes the soil work and supply ecosystem services to humans. Overall, the fertility and functioning of soils strongly depend on interactions between the soil mineral matrix, plants and microbes; these are responsible for both building and decomposing SOM, and therefore for the preservation and availability of nutrients in soils (Cotrufo et al., 2013). To sustain this service, the cycling of nutrients in soils must be preserved.

After carbon, N is the most abundant nutrient in all forms of life, since it is contained in proteins, nucleic acids and other compounds (Galloway et al., 2008). Humans and animals ultimately acquire their N from plants, which on land is mostly taken up in mineral form (i.e., NH_4^+ and NO_3^-) from the soil. The parent material of soils does not contain significant amounts of N (in contrast to P and most other nutrients, which largely originate from the parent material). New N mostly enters the soil through the fixation of atmospheric N_2 by a specialized group of microorganisms. However, the largest flux of N within the soils is generated through the continuous recycling of N internal to the plant-soil system: soil mineral N is taken up by the plant, it is fixed into biomass, and eventually N returns in the form of plant debris to the soil. Here microorganisms decompose it, mineralizing part of the N and making it newly available for plant growth, while transforming the other part into SOM, which ultimately is the largest stock of stable N in soil. Genrrally, N cycles tightly in the system with minimal losses. Nitrogen is lost from the soil to the water system by leaching and to the atmosphere by gas efflux (NH_4, N_2O and N_2). In most terrestrial natural ecosystems, N availability limits productivity. Through the cultivation of N_2 fixing crops, the production and application of synthetic N fertilizer, increasing application of animal manure from livestock and biowastes, and the unintentional deposition of atmospheric reactive N (ultimately derived from industrial-era human activities), humans have applied twice as much reactive N to soils as the N introduced by natural processes, significantly increasing biomass produc-

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tion on land (Vitousek and Mason, 1993; Erisman et al., 2008). However, since mineral fertilizer use efficiency is generally low, and fertilizers are generally applied substantially in excess of plant demand, a high percentage of N fertilizer is lost from the soil. This phenomenon is spread over most of the globe. However, in some regions of the world, in particular in Sub-Saharan Africa where economic constraints limit the use of fertilizers, productivity is still strongly limited by soil available N and other nutrients, notably P and K (N and P; Fig. 3).

Phosphorus derived from parent material, though weathering, cycles internally in the plant -soil system between biochemical molecules (e.g. nucleic acid, phospholipids, etc.) and mineral forms after decomposition (e.g. H_3PO_4). In soils, P is among the most limiting of nutrients, since it occurs in small amounts and is only available to plants in its dissolved ionic forms, which promptly react with calcium, iron and aluminium cations to form highly insoluble compounds. Largely in these forms, P is lost to the aquatic system through erosion and surface runoff. Losses may also occur in dissolved form, for instance via sub-surface flow and groundwater (McDowell et al., 2015). An important form of loss is in the export of organic P in agricultural products. Due to widespread agricultural P deficiencies, humans started to mine “primary” P from guano or rock phosphate deposits and added it to soils in the form of mineral fertilizer (Fig. 2). This external input has led to positive agronomic P balances (McDonald et al., 2011) and excesses or P and N in many regions (West et al., 2014; Fig. 3).There are large variations across the world, with high surpluses in the USA, Europe and Asia, and deficits in Russia, Africa and South-America (Fig. 3). Since plant P uptake is a relatively inefficient process with roughly 60 % of the total P input to soils not taken up in the short term, a threefold increase in the export of P to water bodies has been estimated, with significant impacts on water quality (Bennett et al., 2001).

Clearly, management practices need to be implemented that sustain, restore or increase soil fertility and biomass production by promoting the accrual of SOM and nutrient recycling, applying balanced C amendments and fertilization of N, P and other nutrients to meet plant and soil requirements, while limiting the addition of excess fer-

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tilizer and retaining nutrients in the soil-plant system. Carbon, N and P cycling in soils is coupled by tight stoichiometric relationships (e.g. relatively fixed C : N : P in plants and microorganisms; Güsewell, 2004), thus their management needs to be studied in concert. Nutrient management has been extensively studied, with the aim of identifying and proposing management practices (e.g. precision agriculture) that improve nutrient use efficiency and productivity, and reduce potentially harmful losses to the environment (van Groenigen et al., 2010; Venterea et al., 2011). Yet, our ability to predict the ecosystem response to balanced fertilization is still limited, and effectiveness and reliability would benefit from continued monitoring of efforts. Further benefits are anticipated from improved plant varieties with root morphologies that have better capacity to extract P from soils or use it more efficiently, perhaps in concert with mycorrhizal symbionts. Fertilization with nutrients other than N and P has been less well explored within the realm of understanding soil organic matter responses to agricultural C inputs and the potential to restore and increase soil organic matter (e.g. Lugato et al., 2006). Hence, we stress the importance of an integrated approach to nutrient management, which supports plant productivity while preserving or enhancing SOM stocks, and reducing nutrient losses to the atmosphere or water resources. Several issues exist where prediction and optimization of performance would benefit from relevant and continued data acquisition for the range of climate and environmental and agro-ecological conditions. Table 2 summarises some management actions affecting soil nutrient cycles and their impacts on ecosystem services.

4 Soils and the water cycle

Soils provide important ecosystem services through their control on the water cycle. These services include provisioning services of food and water security, regulating services associated with moderation and purification of water flows, and cultural services such as landscapes and water bodies that meet recreation and aesthetic values (Dymond, 2014). At the pedon to hillslope scale, water stored in soil is used for evapo-

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transpiration and plant growth that supplies food, stabilizes the land surface to prevent erosion and regulates nutrient and contaminant flow. At a catchment and basin scale, the capacity of the soil to infiltrate water attenuates stream and river flows and can prevent flooding, while water that percolates through soil can replenish groundwater that can maintain water supplies and sustain surface water ecosystems while promoting a continued flow during periods of reduced precipitation.

The soil functions of accepting, storing, transmitting and cleaning of water shown in Table 3 are inter-related. Soil water storage depends on the rate of infiltration into the soil relative to the rate of precipitation. Soil hydraulic conductivity redistributes water within and through the soil profile. The infiltration rate and hydraulic conductivity both depend on the water stored in the soil. The initially high rate of infiltration into dry soil declines as the soil water content increases and water replaces air in the pore space. Conversely, hydraulic conductivity increases with soil moisture content as a greater proportion of the pores are transmitting water. Water content and transmission times are also important to the filtering function of soil because contact with soil surfaces and residence time in soil are important controls on contaminant supply and removal.

The quantity of water which a soil can store depends on the thickness of the soil layer, its porosity and soil matrix-water physical interactions. The latter are expressed as a water retention curve, the relationship between the soil water content and the forces holding it in place. The porosity and water retention curve are in turn influenced primarily by the particle size distribution and the soil bulk density, but also the amount of SOM and the macropores created by biotic activity.

Optimum growth of most plants occurs when roots can access both oxygen and water in the soil. The soil must therefore infiltrate water, drain quickly from saturation to allow air to reach plant roots, and retain and redistribute water for plant use. An ideal soil for plant production depends on the climatic conditions. Soil structural stability and porosity are also important for the infiltration of water into soil. In addition to soil texture, organic matter improves soil aggregate stability. While plant growth and surface mulches can help protect the soil surface, a stable, well-aggregated soil struc-

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ture that resists surface sealing and continues to infiltrate water during intense rainfall events will decrease the potential for downstream flooding resulting from rapid overland flow. Porosity (especially macropores of a diameter ≥ 75 µm) determines the capacity of the soil to retain water and controls transmission of water through the soil. In addition to total porosity, the continuity and structure of the pore network are as important to these functions as they are in filtering out contaminants in flow. Furthermore, the soil must support biota that will degrade the compounds of interest or have sorption sites available to retain the chemical species. Soil organic matter is important for these roles and together with mineral soil (especially the clay fraction) provides sorption sites. Flow through macropores, which bypass the soil matrix where biota and sorption sites are generally located, can quickly transmit water and contaminants through the soil to groundwater or artificial drains, but for filtering purposes a more tortuous route through the soil matrix is more effective.

Management of soil alters the ecosystem services provided by water (Table 4). Soil conservation and sustainable management practices to combat desertification help to retain soil organic matter, structural stability, infiltration and profile water holding capacity. The promotion of soil as a C sink to offset greenhouse gas emissions generally helps to maintain or improve soil hydrological functions as well. Deforestation, overgrazing and excessive tillage of fragile lands, however, will lead to soil structural deterioration and a loss of infiltration, water retention and surface water quality (Steinfield et al., 2006). Anthropogenic modifications to the water cycle can aid soil function. In dry regimes, inadequate soil moisture can be mitigated through supplementary irrigation, and where excessive precipitation causes problems, waterlogging can be relieved by land drainage. However, irrigation and drainage can have consequences for water regulation services. Irrigation that enables a shift to intensive land use can increase the contaminant load of runoff and drainage (McDowell et al., 2011). Furthermore, drainage of wetland soils has been shown to reduce water and contaminant storage capacity in the landscape and can increase the potential for downstream flooding, and increase the potential for GHG emissions due to the rapid decomposition of SOC in soil and dissolved organic C in drainage water (IPCC, 2013). The removal of surface or groundwater for irrigation disrupts the natural water cycle and may stress downstream ecosystems and communities. Irrigation of agricultural lands accounts for about 70 % of ground and surface water withdrawals, and in some regions competition for water resources is forcing irrigators to tap unsustainable sources. Irrigation with wastewater may conserve fresh water resources, but the fate of water-borne contaminants in soil and crops is a potential concern (Sato et al., 2013).

5 Soils as a habitat for organisms and as a genetic resource

Soils represent a physically and chemically complex and heterogeneous habitat supporting high diversity of microbial and faunal taxa. For example, 10 g of soil contains about 10^{10} bacterial cells, representing more than 10^6 species (Gans et al., 2005). Up to 360 000 species of animals live predominantly in the soil – a large fraction of all animal species (Decaëns et al., 2006). These complex communities of organisms play critical roles in sustaining soil and wider ecosystem functioning, thus conferring a multitude of benefits to global cycles and human sustainability. Specifically, soil biodiversity contributes to food and fibre production, and is an important regulator of other soil services including greenhouse gas emissions, water purification (Bodelier, 2011) and supporting services such as nutrient cycling. Stocks of soil biodiversity represent an important biological and genetic resource for biotechnological exploitation. Previous methodological challenges in characterizing soil biodiversity are now being overcome through the use of molecular technologies, and currently significant progress is being made in opening the "black box" of soil biodiversity (Allison and Martiny, 2008) with respect to providing fundamental information on normal operating ranges of the biodiversity under different soil, climatic and land use scenarios. Addressing these knowledge gaps is of fundamental importance, firstly as a prelude to understanding wider soil processes, but also to better inform the likely consequences of land use or climatic change on both biodiversity and soil ecosystem services.

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The development of molecular technologies has led to a surge in studies characterizing soil biodiversity at different scales – from large landscape scale surveys to specific, locally-focused studies using manipulation, or contrasting of specific land uses. The large-scale surveys yield the broader picture, and conclusions are emerging identifying the importance of soil parameters in shaping the biodiversity of soil communities (Fierer and Jackson, 2006). In essence, the same geological, climatic and biotic parameters which ultimately dictate the supporting service of soil formation, are also implicated in shaping the communities of soil biota, thus regulating the spatial structure of soil communities observed over large areas (Griffiths et al., 2011). Locally focused experimentation typically reveals more specific changes with respect to local land use or climate. Most studies have focused on assessing one component of soil diversity. Next-generation high throughput sequencing now allows the analyses of "whole soil foodwebs", permitting a thorough interrogation of trophic and co-occurrence interaction networks. The challenge is to consolidate both approaches at various scales, to understand the differing susceptibility of global soil biomes to change.

It is essential to link these new biodiversity measures to specific soil functions in order to understand the pivotal roles of soil organisms in mediating soil services. The development of in situ stable isotope tracer methods (e.g. Radajewski et al., 2000) to link substrate use to the identified active members serves to clarify the physiological activity of these organisms. Additionally, whole genome shotgun metagenomic sequencing is now becoming an increasingly cost effective approach to assessing the biodiversity of functional genes in soils (Fierer et al., 2013), potentially allowing for a trait-based rather than taxon-based approach to understanding soil biodiversity, akin to recent approaches applied to larger and more readily functionally understood organisms aboveground. It is becoming increasingly apparent that functionality and biodiversity co-vary with other environmental parameters. Thus manipulative experimentation is required to determine the fundamental roles of soil biodiversity versus other co-varying factors in driving soil functionality. Table 5 summarises management actions affecting the soil biota and their impacts on ecosystem services.

6 Knowledge gaps and research needs concerning soil carbon, nutrient and water cycles, and the role of soil biodiversity

6.1 Soil carbon cycle

Substantial progress has been made in recent years towards more fundamental understanding of the processes controlling soil C storage and in improving and deploying predictive models of soil C dynamics that can guide decision makers and inform policy. However, it is equally true that many new (and some old) gaps in our knowledge have been identified and research needs articulation. New research on soil C dynamics has been driven in part by increasing awareness of (1) the importance of small scale variability for microbial C turnover (Vogel et al., 2014), (2) interactions between the C cycle with other biogeochemical cycles (Gärdenäs et al., 2011) and (3) the importance of soil C, not only at the field scale, but at regional to global scales (Todd-Brown et al., 2013). The most cited gaps in basic knowledge include plant effects on SOM storage and turnover, controls on microbial efficiency of organic matter processing, including biodiversity, association/separation of organic matter and decomposing microbial communities in the mineral soil matrix (Bardgett et al., 2008), role of soil fauna in controlling carbon storage and cycling, dynamics of dissolved organic carbon and its role in determining C storage and decomposition (Moore et al., 2013; Butman et al., 2014), black C stabilization and interactions of black C including biochar with native soil C and mineral nutrients, and the role of soil erosion in the global C cycle (Quinton et al., 2010). For predictive modelling and assessment, most frequently cited knowledge gaps are: closer correspondence of measured and modelled SOM fractions (Zimmermann et al., 2007), improved modelling of C in subsurface soil layers, distributed soil C observational and monitoring networks for model validation, more realistic and spatially-resolved representation of soil C in global-scale models, and the response to climatic extremes (Reichstein et al., 2013).

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6.2 Soil nutrient cycles

In the second half of the 20th century, higher biomass yields were supported by higher use of fertilizer (N, P) inputs. Today, at the beginning of the 21st century, this is not considered sustainable. Alternatives are needed that will use inherent soil fertility and improved resource use efficiencies, and to prevent losses of N and P. Examples in agriculture include ecological intensification and new crop varieties with improved ability to extract P and use from soils. At the food system level, more effective nutrient management would benefit from a focus on a "5R strategy": (1) re-align P and N inputs, (2) re-duce P and N losses to minimize eutrophication impacts, (3) re-cycle the P and N in bio – resources, (4) re-cover P (and N) from wastes into fertilizer, and (5) re-define use and use-efficiency of N and P in the food chain including diets and regional and spatial variability (e.g. Snyder et al., 2014).

6.3 Soil water

The soil management practices that maintain the ecosystem services of food and water provision, flow regulation, water purification, and aesthetic value within the soil and water cycle are well known. However, their application is not universal and poor management leads to a loss of function. Under scenarios of increased climatic variability with more extremes of precipitation and increased severity of droughts, soil functions will be stressed and the level of good soil management will be required to improve (Walthall et al., 2012). Research into these interactions, and future-proofing of current good practice is required.

6.4 Soil biota

Despite recent advances in knowledge regarding stocks and changes in soil biodiversity, global scale syntheses is still largely absent. Indeed many of these highly pertinent issues were raised more than 20 years ago (Furusaka, 1993), and to date none

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of these factors have been unravelled fully. Key barriers to syntheses are the lack of concerted soil surveys addressing multiple functions with standardized methodologies. New technologies for soil biodiversity assessment generate large datasets of gene sequences which are typically archived in publicly accessible databases. The adoption of such approaches for soil function measurements alongside deployment of agreed standard operating procedures (e.g. as developed in recent EU funded EcoFINDERS project), could serve to address these gaps. Ultimately, new methods are revealing the high sensitivity of change of soil biological and genetic resources from threats such as management, and we now need to recognize the distinct types of organisms found in different soils globally, and understand their functional roles in order to predict vulnerability of these resources to future change.

7 Recommendations for management activities to support the continued delivery of ecosystem services from soils

Best management practices that support one facet of soil functioning tend to also support others. Building SOM, for example, enhances soil C, soil nutrient status, improves water holding capacity and supports soil biota (Lal, 2004; Smith, 2012). Similarly, preservation of natural ecosystems, and prevention of degradation or conversion to intensive agriculture, almost always benefits soil C, nutrients, water and biota. These synergies, and the fundamental role of soil makes the goal of supporting soil function more straightforward than the goal of maximising multiple ecosystem services, which often involve trade-offs (Robinson et al., 2013; Smith et al., 2013). For example, in terms of the provisioning service of food, the highest per-area yields are often obtained under intensive cropping, with large external inputs of synthetic fertilizer, other agrochemicals (such a pesticides and herbicides) and sometimes water through irrigation (West et al., 2014), with the most intensive forms of agriculture occurring in greenhouses, where external inputs of fertilizers, water and energy can be extremely high (Liu et al., 2008). Though intensive cropping produces high per-area yields, it is not

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the best management system for a range of other ecosystem services, potentially adversely affecting supporting services (e.g. soil formation through erosion), regulating services (e.g. climate regulation through greenhouse gas emissions; air, water and soil quality through leaching of agrochemicals; pollination through adverse impacts on pollinators) and cultural services (e.g. reduced aesthetic value of the landscape through large scale monoculture; Smith et al., 2013). Balancing the trade-offs between different ecosystems services is, therefore, more difficult than designing management strategies that support soil C, nutrients, water and biota. Tables 1, 2, 4 and 5 present some examples of management activities that affect a range of soil functions, and a number of beneficial management actions occur in most/all of the tables. The most important of these beneficial management activities are described below.

7.1 Land cover and use change

A number of meta-analyses (Wei et al., 2014; Guo and Gifford, 2002; Don et al., 2011) show that natural systems lose carbon when converted to agriculture, with the exception of forest to pasture conversion where some studies indicate carbon gain (Guo and Gifford, 2002) while others indicate carbon loss (Don et al., 2011). Given the link between organic matter and soil carbon, nutrients, water and biota, conversion of natural systems to agriculture is likely to adversely impact all of these factors. Protection of natural ecosystems, therefore, benefits soil carbon, nutrients, water and biota. Rewilding of surplus agricultural land would be expected to enhance soil carbon, nutrients, water and biota, as seen in set-aside or reforestation of former cropland (Don et al., 2011). In the absence of land cover/land use change, improved management of agricultural soils can improve soil carbon, nutrient, water and biota, as described below.

7.2 Improved agricultural management

Reducing soil disturbance (e.g. through reduced or zero-tillage) can increase SOC stocks (West and Post, 2002; Ogle et al., 2005), though the C benefits of no-till may

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be limited to the top 30 cm of soil and some authors argue that the C benefits have been over-stated (Powlson et al., 2014). Baker et al., (2006) found similar soil C in conventional and no-till systems, suggesting that C accumulation is occurring at different depths in the soil profile under different management schemes. Given the tight coupling of soil C and N, increased organic matter also tends to increase nutrient supply, and also enhances water holding capacity (Lal, 2004), and can increase soil biota. Zero tillage also gives rise to greater earthworm and arthropod populations (House and Parmelee, 1985). Perennial crops also reduce the need for annual tillage, and can provide similar benefits. Cultivation of perennial plants with improved rooting systems are likely to increase soil C stocks in C depleted subsoil horizons (Kell, 2012). Land-use change, such as removal of perennial plants and subsequent cultivation, were found to affect both short-lived and long-lived C pools (Beniston et al., 2014).

Maintaining ground cover through improved residue management and use of cover crops during traditional bare fallow periods helps to improve C returns to the soil, prevent erosion and surface sealing, maintain soil nutrients and supports an active level of soil biota (Lal, 1997). Similar benefits can be achieved through well designed rotations and use of perennial crops or agroforestry (e.g. Mbow et al., 2014).

Use of organic amendments increases SOM content (Lal, 2004; Smith, 2012; Gattinger et al., 2012), which, as described above benefits soil C, nutrients, water and biota. Organic amendments traditionally include crop residues, animal manures, slurries and composts. These organic matter additions were found to improve C storage and other regulating ecosystem services if repeated regularly. Recent developments, such as the use of biochar or hydrochar from the pyrolysis or hydrothermal carbonization of crop residues or other biomass, can increase SOC stocks and can also reduce soil N_2O emissions and enhance soil fertility (Zhang et al., 2010), which could be effective over multiple years (Liu et al., 2014). However, the properties of these materials and their net effect on ecosystem services, is strongly dependent on production conditions (Wiedner et al., 2013; Naisse et al., 2015). Soil amendment with compost and biochar or their mixture may be particularly useful for increasing the regulating and

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supporting services of degraded soils (Ngo et al., 2014). Biochar, in conjunction with bioenergy production, is at this stage one of the most promising technologies for acieving the large-scale negative carbon emissions required by mid-century to prevent global mean temperatures from increasing above 2 °C, though this is controversial (Fuss et al., 2014).

Optimised timing and rate of fertilizer application: intensification has increased annual global flows of N and P to more than double natural levels (Matson et al., 1997; Smil, 2000; Tilman et al., 2002). In China, N inputs to agriculture in the 2000's were twice that in 1980's (State Bureau of Statistics-China, 2005). Optimising the timing and rate of fertilizer applications ensures that the nutrients are available in the soil at a time when the plant is able to take them up, which limits nutrient loss, hence reducing the risk of water pollution and downstream eutrophication (Carpenter et al., 1998). Fertiliser decision support tools can help to implement optimised nutrient management, as can soil testing (to establish soil nutrient status before fertilization), and precision farming, to ensure that nutrient additions are targeted where needed. Subsurface application of slurries to reduce ammonia volatilization can increase nitrous oxide emissions, so there can be trade-offs associated with this practice (Sutton et al., 2007).

Optimised use of agrochemicals: reduction in use of broad spectrum bioactive agrochemicals will benefit soil biota. The under-application of pesticides and herbicides could also plausibly have net negative environmental impact, if it means that more land needs to be brought into production (Carlton et al., 2010, 2012). Optimisation of agrochemical applications will also reduce water pollution through leaching.

Water management: irrigation of dryland agriculture can increase productivity and C returns to the soil, with the benefits to soil carbon, nutrients, water and biota discussed above, but it can decrease filtration potential and increase the risk of soil salinization (Ghassemi et al., 1995; Setia et al., 2011). In waterlogged marginal lands, drainage can increase productivity and thereby increase carbon returns to the soil, while decreasing methane and nitrous oxide emissions. If wetland soils are drained, oxidation of organic soils will lead to large losses of soil C and and the nutrients associated with it; and

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decrease the ability of these soils to carry out services like water purification (e.g. through denitrificiation). Drainage of peatlands has been associated with increased runoff and flood risk (Ballard et al., 2012). In terms of biodiversity, productivity of drained marginal lands can increase at the expense of plant genetic diversity.

Improved grazing management (e.g. optimised stocking density) can reduce soil degradation, and thereby maintain and enhance organic matter content (McSherry and Ritchie, 2013) benefiting soil C, nutrients, water and biota as described above. Higher productivity and deep rooted grasses can do similarly (Kell, 2012), while also modifying water use efficiency, but potentially at the expense of plant genetic diversity. Reduction in grazing density can reduce soil compaction, and therefore increase infiltration and water storage and reduce risk of runoff and flooding downstream (Marshall et al., 2009). Fire management can also increase soil C and nutrient status of soils (e.g. Certini, 2005).

8 Conclusions

Many practices are known to enhance all or most of the functions of soils considered in this review is encouraging for our efforts to protect soils into the future. There are still knowledge gaps (outlined in Sect. 6), and fundamental research is still needed to better understand the relationships between different facets of soils and the array of ecosystem services they underpin, but we know enough to start to make a difference now. Despite a growing population and increasing demands for resources, enough is known to discriminate the extremes of beneficial and detrimental agricultural practices, and their interactions with different types of soils. However, more knowledge is required on where specific agricultural systems are best placed to utilise and deliver ecosystem services most efficiency, to protect and enhance our soils in the long-term. A significant challenge is to find effective ways to share this knowledge with soil managers and policy makers, so that best management can be implemented. A key element of this knowledge exchange must be in raising awareness of the ecosystems services

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underpinned by soils, and thus the natural capital they provide (Robinson et al., 2013). The International Year of Soils in 2015 presents the perfect opportunity to begin this process.

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Table 1. Management actions affecting the soil carbon cycle and their impact on ecosystem services.

Management action or other driver of change	Provisioning service impact	Regulating service impact	Supporting service impact	Cultural service impact
Land-use change (conversion of forest/grassland/wetland to cropland)	Increased production of food, fibre, and energy crops; reduced availability of natural raw materials; potential change in hydrology/water availability	Decreased soil C sequestration and storage – increased GHG flux; increased erosion and sediment yield – reduced regulations of water flow and quality;,	Primary production may be changed; nutrient recylcing reduced if no inputs, increased if there are inputs;	Lower recreation value; may have impact on cultural value in recreating diverse landscapes
Land-use-change (establishment of forest or grassland on agricultural land)	Raw material provision may be increased; agricultural production likely decreased (but not alwasy e.g agroforestry)	Increased C sequestration; increased regulation of water flow and quality	Primary production may be changed, increased water recycling	Increased recreation value; may have impact on cultural value in recreating diverse landscapes
Intensified nutrient management through fertilisation and liming	Increased production of food and other raw materials	Effect on net soil C sequestration uncertain; increased GHG flux from fertiliser production and use; water and air pollution	Increased primary production; increased nutrient recycling	
Soil amelioration using organic amendments such as compost and biochar	Increased food production; more raw materials; more water available for plant growth	Increased C sequestration; increased water purification value	Increased primary production; increased nutrient cycling; improved water infiltration and retention	
Diversification of crop production systems (i.e., more perennials, reduced bare fallow)	Potential impact on agricultural production (±); more diverse products	Increased C sequestration; increased purification value	Changed primary production; increased nutrient retention; improved water infiltration and retention	Improved cultural value from more diverse landscapes
Replacement of hay forage production with pasture use on grasslands	No impact	Effect on C sequestration uncertain		Increased recreation value; may have impact on cultural value in recreating diverse landscapes
Improved grazing management	Increased food production; reduced runoff and improved water use	Increased C sequestration; increased purification value; water flow regulation	Increased primary production; improved water infiltration and retention	

Table 2. Management actions affecting soil nutrient cycles and their impact on ecosystem services.

Management action or other driver of change	Provisioning service impact	Regulating service impact	Supporting service impact	Cultural service impact
Intensive addition of mineral fertilizers	Increased food, fibre and feedstock production;	Reduced water quality through eutrofication, reduced air quality through emission and volatilization of reactive N gases	Increased primary production. Alteration of the nutrient and C cycling. Possible reduction of biodiversity	
Use of organic soil amendments (e.g. biochar)	Increased food, fibre and feedstock production; may increase water retention	Increase C sequestration	Increase nutrient retention	
Implementation of No- tillage			increase nutrient retention	
Precision agriculture	Increase efficient production of food	Reduced GHG emissions per unit production	Reduce consumption of water and nutrient, by improving use efficiency	
Prescribed use of fire for pasture management	Increase feedstock production	Increase C sequestration, by conversion to BC	Reduce N recycling, by storing black nitrogen	
Use of biological soil supplements	Stimulate productivity; act as fertilizers	May improve pest and disease control	Improved nutrient cycling	

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Table 3. Soil functions related to the water cycle and ecosystem services.

Soil Function	Mechanism	Consequence	Ecosystem service
Stores (Storage)	Water held in soil pores supports plant and microbial communities	Biomass production Surface protection	Food Aesthetics Erosion control
Accepts (Sorptivity)	Incident water infiltrates into soil with excess lost as runoff	Storm runoff reduction	Erosion control Flood protection
Transmits (Hydraulic conductivity)	Water entering the soil is redistributed and excess is lost as deep percolation	Percolation to groundwater	Groundwater recharge Stream flow maintenance
Cleans (Filtering)	Water passing through the soil matrix interacts with soil particles and biota	Contaminants removed by biological degradation/ retention on sorption sites	Water quality

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Table 4. Management actions affecting the soil water cycle and their impact on ecosystem services.

Management action or other driver of change	Provisioning service impact	Regulating service impact	Supporting service impact	Cultural service impact
Land use change (increase change of agricultural to urban)	Decreased biomass, decreased availability of water for agricultural use.	Increased impervious surface, decreased infiltration, storage, soil mediated water regulation	Decreased genetic diversity; reduction of rainfall recycling e.g. in the tropics	Decreased natural environment
Land use change (increase change of arable to intensive grassland)	Increased yield of animal over vegetable protein.	Increased C sequestration, greater requirement of water, stress on ecosystem health of downstream waterways	Increased genetic diversity associate with mixed pastures	Change in aesthetic value away from traditional norm
Irrigation (increase)	Increased biomass over dryland agriculture, decreased availability of water for urban use	Increased C sequestration, but decreased filtration potential	Improved habitat for plant species	Infrastructure alters landscape decreasing spiritual connection with catchment
Drainage (increasing in marginal land)	Decreased soil saturation, increased biomass, removal of wetlands	Decreased C sequestration, denitrification and flood attenuation	Better habitat for productive grassland plants, but loss of genetic diversity	Decreased recreational potential (e.g. ecotourism)

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Table 5. Management actions affecting the soil biota and their impacts on ecosystem services.

Management action or other driver of change	Provisioning service impact	Regulating service impact	Supporting service impact	Cultural service impact
Land use change of natural vegetation to agricultural intensification	Changed genetic resources, changed production of (precursors to) industrial and pharmaceutical products	Decreased C sequestration, changed pest and disease control	Changed elemental transformation	Changed diversity of soil organisms (elimination of some soil animals, etc.)
Use of organic amendments	Increased genetic resources, decreased production of (precursors to) industrial and pharmaceutical products	Increased C sequestration	Increased soil formation, increased primary production by phototrophs, changed elemental transformation	Increase of soil organisms
Use of broad spectrum bioactive agrochemicals	Decreased genetic resources, decreased production of (precursors to) industrial and pharmaceutical products	Possible decreased waste decomposition and detoxification	Decreased primary production by phototrophs, changed elemental transformation	Decreased diversity of soil organisms (elimination of some soil animals, etc.)
Pollution by heavy metals or xenobiotics	Decreased genetic resources, decreased production of (precursors to) industrial and pharmaceutical products	Possible decreased waste decomposition and detoxification	Decreased primary production by phototrophs, changed elemental transformation	Decreased diversity of soil organisms (elimination of some soil animals, etc.)
Climate change (global warming)		Possible decreased C sequestration	Changed elemental transformation	

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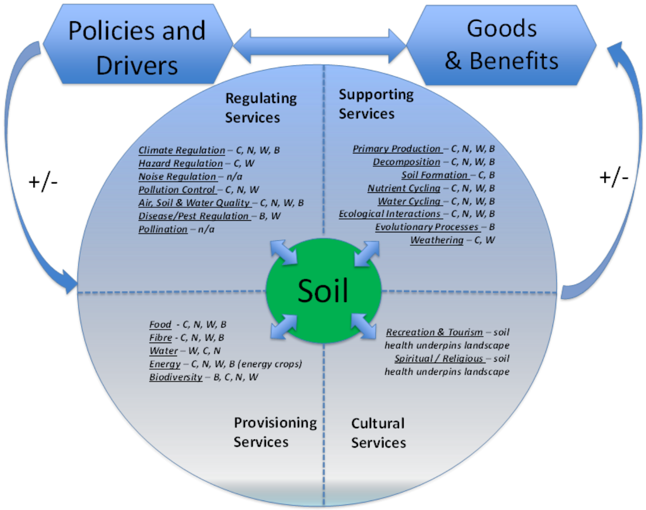

Figure 1. Schematic representation of where soil carbon, nutrient and water cycles, and soil biota underpin ecosystem services (adapted from Smith et al., 2014). Role in underpinning each ecosystem service shown by C = soil carbon, N = soil nutrients, W = soil water, B = soil biota.

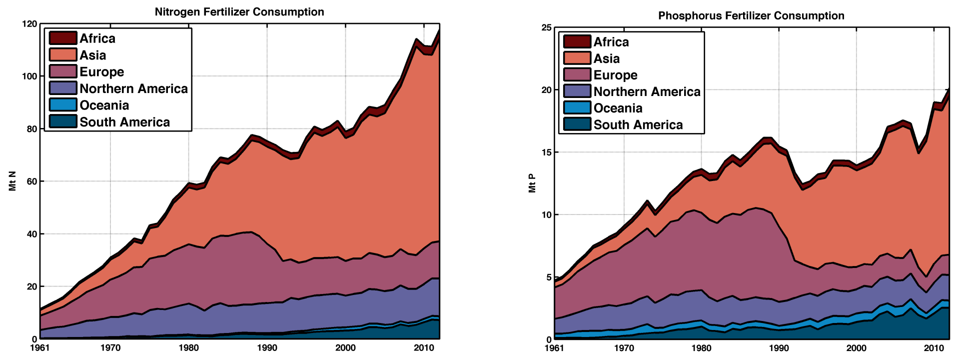

Figure 2. Global **(a)** nitrogen (N) and **(b)** phosphorus (P) fertilizer use between 1961 and 2012 split for the different continents in Mt P per year; plotted from FAOSTAT data (FAOSTAT, 2015).

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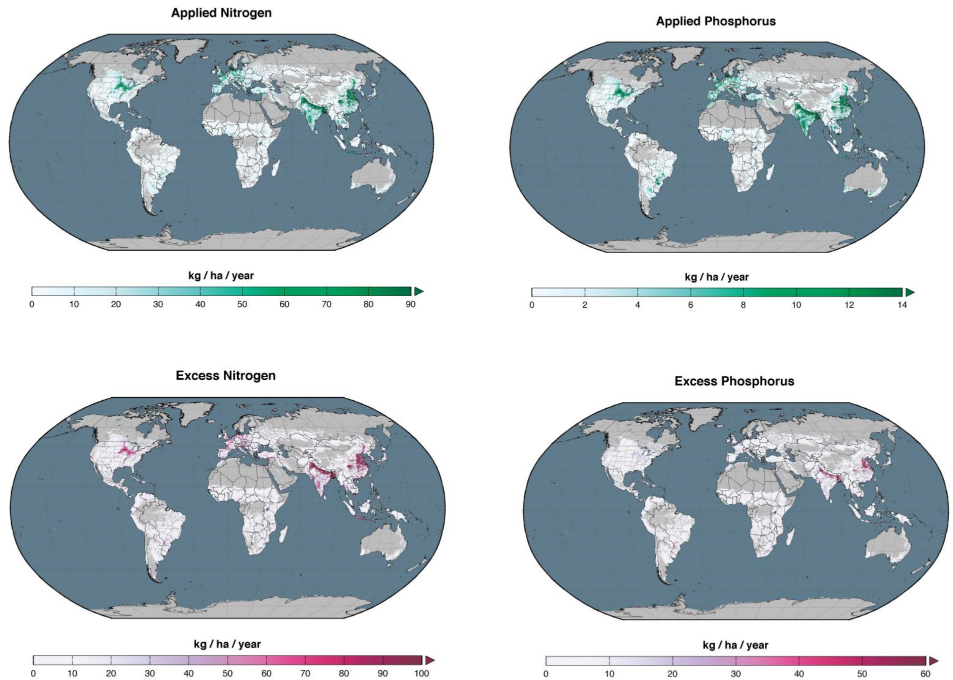

Figure 3. Applied and excess nitrogen and phosphorus in croplands. Nitrogen and phosphorus inputs and excess were calculated using a simple mass balance model (West et al., 2014), extend to include 175 crops. To account for both the rate and spatial extent of croplands, the data are presented as kg per ha of the landscape. **(a)** Applied Nitrogen, including N deposition; **(b)** applied Phosphorus; **(c)** excess Nitrogen; **(d)** excess Phosphorus.

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