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Soil resource efficiency in urbanised areas

Analytical framework and implications for governance

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Executive summary

Soil is a limited and non-renewable resource — implying that its supply is limited — of which a lot of functional demands are made. As well as the use of the soil resource for growing food, which has been done for millennia, the many and diverse services that can be derived from it are in particularly high demand in urbanised spaces, as these areas are where economic development and associated consumption related to the soil resource are especially evident. In the urban fringe — the transition zone between the core urban and rural zones — competing demands for the soil resource are likely. Depending on the demands in relation to the qualities of the soil resource in an urbanised area, soil may occasionally be considered a scarce resource.

In this report, we have explored the notion of soil as an integral part of ecosystems and natural capital, and thus focused on the stock of the soil resource and the flows of valuable goods and services that can be derived from this stock. The concept of natural capital recognises soil as an asset that is of use and benefit to society (also called a 'productive' asset). Putting soil within the framework of the land system allows a connection to be made with governance, including soil resource efficiency. Emphasis is put on place- and asset-based governance, implying that multiple governance levels are taken into account: a frame that fits the European context well. Soil resource efficiency in its simplest form expresses how efficiently society is using its soil resource without degrading it; the aim is to balance the supply of and demand on the soil resource.

Taking the place- and asset-based approach involves obtaining a valuation of the soil resource and the flows of goods and services that can be derived from it. In this report, we use 'valuation' in the broadest sense of 'recognising the importance of', namely recognising additional dimensions of valuation. The key question on which the report is centred is whether or not the importance of soil is integrated in current soil governance practice (decision-making) in urbanised areas. We use existing evidence (including governance practice) to assess whether or not the existing knowledge base is sufficient for valuation. To that effect, we use the following guiding questions:

- Where/how do soils and their use and management make a difference in delivering services?
- How are the (degradation) costs and benefits distributed?
- How can the demand on the soil resource be managed optimally (land resource efficiency)?

The knowledge base on urban soils that would allow ecological soil valuation is largely missing or, at best, fragmented. Nevertheless, several local, national and European activities are already contributing to a knowledge base on urban soils. To date, the focus has been mostly on pressures, such as soil contamination and soil sealing, but has more recently shifted towards responses, such as land recycling (reuse of brownfields) and the development of green infrastructure. To evaluate soil functions for supporting, provisioning, regulating and cultural services, the soil information base must be further improved. There is a need to adapt soil function and soil-based service descriptions to an urbanised context, and to develop geo-spatial maps of soil characteristics and soil functions that are relevant to urban demand.

Cost–benefit analysis can be used to assess the costs and benefits of serving human needs at specific sites. Impacts on the local/regional economy (e.g. on economic growth) are better captured by economic impact assessment methods, such as multiplier analysis. However, monetary valuation of the soil resource in urbanised settings has thus far been limited, presumably because monetary valuation methods hold quite a few methodological challenges. It is also difficult to separate soil from the natural (ecosystem) and anthropo-natural (land system) systems to which it belongs. Where available, economic valuations are often too site-specific to allow a broader understanding to be gained of the contribution of the soil resource to the local/regional economy. Additional analysis at the level of the regional economy may complete the picture. Nevertheless, neither site-specific nor regional economic assessments have been designed for benefit transfer to other contexts.

Examples from diverse urbanised contexts in Europe illustrate that public and private policy instruments for soil protection and land resource efficiency in urbanised areas exist. The public instruments include regulatory measures, price- and market-based economic incentives and broader awareness-raising instruments, including public financing for innovation or for outreach and education. By focusing on soil sealing and soil contamination — two degradation processes particularly relevant to urbanised contexts — it emerges that a mix of policy instruments is the best option. Overall, it seems that policy design can benefit from (further) integration of the natural capital concept. For example, monetary valuation of soil may actually be most meaningful in connection with a bigger (investment) project (e.g. green infrastructure). Going beyond the ecosystem services aspect of the soil resource, approaches from waste management could also be included, applying principles of reuse and recycling to soil from construction sites (emphasising the abiotic component of soil).

Overall, the current knowledge base does not, in most cases, facilitate asset-based governance; instead, the soil resource continues to be degraded, losing its potential to deliver valuable goods and services, the costs of which are mostly borne by the public/society. The natural capital and ecosystem services approach helps to appreciate the multi-functional role of soil and its ability to support a range of different benefits to different stakeholders simultaneously. For the most part, the degree of multi-functionality of soil and the range of ecosystem services provided largely depends on the way that land is used and managed. The challenge is to recognise the value of the multiple flows of services to which soil contributes and either to build these into policy instruments (including market-based ones) that reward the delivery of services, or to penalise their loss.

1 Introduction

1.1 Scope of this report

Soil is a component of natural capital, one of five capital types that support our economy. All capital types — financial, natural, manufactured, human and social — are stocks that have the capacity to produce flows of economically desirable goods and services (Goodwin, 2003).

Humans can benefit from the structure and functions of soil through a number of ecosystem goods and services, such as biomass (food, feed, fibre and fuel) production, water and temperature regulation, or waste assimilation. The last two are particularly relevant in urban contexts, as they contribute to the quality of living. Accepting that soil is an essential component of natural capital, and thus a natural resource that is instrumental in delivering valuable goods and services to society, implies that the importance of soil needs to be recognised.

The valuation of the soil resource and the flows of goods and services that can be derived from it is an approach that allows such recognition, highlighting that soil is an asset to the economy. Valuation refers, in principle, to the act of deciding how much money something might be sold for, or to the actual amount of money decided on; thus, valuation is about defining a monetary value for soil: an economic valuation. However, in this report, we adopt a broader view on value, defining valuation as 'recognising the importance of'. The key question on which this report is centred is whether or not the importance of soil is integrated in current soil governance practice (decision-making) — thus allowing for the inclusion of soil resource efficiency — in urbanised areas.

In order to identify the steps required in achieving asset-based governance of soils in urbanised contexts, this report will:

1. propose an analytical framework for land governance (Section 1.2);

2. describe the soil resource in the framework of the land system, taking account of the cause–effect links within that system, highlighting the particularities of an urbanised setting, and indicating options for ecological valuation of soils (Chapter 2);

3. in line with the monetary valuation approach, describe and evaluate the natural processes in soils that are decisive in deriving valuable goods and services; identify who benefits from them or bears the costs (if the soil resource is degraded); and comment on and critique the knowledge base and methods available to evaluate the soil resource from a monetary point of view (Chapter 3);

4. assess the consequences of the knowledge base (or lack thereof) on soils and their ecological and monetary valuation in urbanised contexts for asset-based governance, and present ways to include the soil resource in governance options and action (Chapter 4).

1.2 An analytical framework for soil resource efficiency

In this chapter, we propose an analytical framework for soil resource efficiency based on three main components, each with underlying frameworks and concepts:

- the land system, incorporating the concepts of natural capital and ecosystem services, and resource efficiency;

- place- and asset-based multi-level governance;

- decision-support tools for governance, including valuation.

1.2.1 The land system

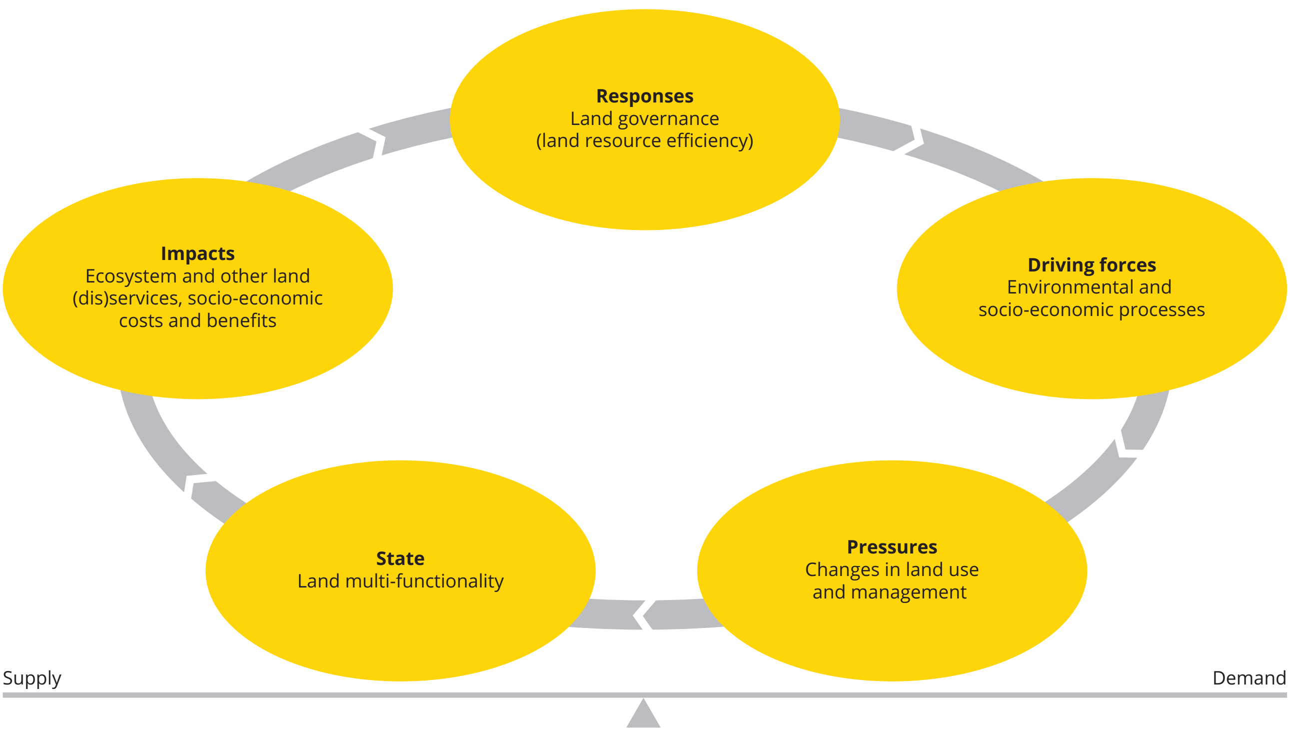
'The land medium integrates three spatial dimensions: the two horizontal ones of land cover/land use, and the third, the vertical one of soil and the underlying geology' (EEA, 2015a). The *land system* embodies the relationship between human activities on land, socio-economic conditions, the natural environment and the systems of governance that manage these interactions (Foresight Land Use Futures Project, 2010). 'Linking its components through cause and effect, it thus refers to the chain of driving forces, pressures, state, impacts and responses (or DPSIR) to which the land is subject' (Figure 1.1) (EEA, 2015a). Following this logic, land cover (biophysical cover), land use (the functional use of the land), land management (indicating a different use intensity) and soil characteristics, along with their contextual attributes such as climate, altitude, topography and hydrology, jointly define the *land's functions.*

The multi-functional character of land is central to the land system. The land resource and the many goods and services that can be derived from it (*supply*), are subject to many functional *demands* (e.g. from agriculture, industry, urban development). When land is used and sealed for the development of housing, industry, commerce or transport infrastructure, these services are lost or deteriorate following disrupted water and nutrient cycles. Likewise, land can be degraded as a consequence of inappropriate use and management (e.g. agricultural practices resulting in soil erosion or soil organic matter decline). Such events can be referred to as dysfunctions and disservices (EEA, 2015a).

Land governance is central to reaching a balance between the supply and demand of land resources and the valuable goods and services that can be derived from them. Land resource efficiency, in its simplest form, expresses how efficiently society is using its land resources; the aim is to balance the supply of, and demand on, the land resource. Originally an economic concept, resource efficiency seeks optimal resource allocation to maximise long-term social welfare, while acknowledging that the soil resource is non-renewable and finite and its degradation should be avoided. This requires a corresponding level of land use and management to preserve the land's potential to deliver natural goods and services, so that it is only depleted/used at the rate at which it can renew itself (see 'Soils and natural capital' in this section). This also implies that a sufficient amount and quality of soil must be available to contribute to the well-being of citizens, and requires an understanding that failing to do so may come at a cost.

The following sections focus on the particular role that soils play in the different components of the land system framework.

Figure 1.1 The land system



Source: EEA, 2015a.

Soils and ecosystem services

Soil is a living system in which a myriad of mainly microbial life forms use carbon derived from photosynthesis as an energy source to support a complex ecosystem (Kibblewhite et al., 2008). The physical structure, water regime, nutrient status, organic matter content and other biotic and abiotic factors define the habitat for this ecosystem.

The concept of an *ecosystem function* describes the combination of ecological structures and processes as well as the potential of ecosystems to deliver one or more services (Braat and de Groot, 2012). A *soil function* can then be defined as the output of a soil process or a set of soil processes, where the context is the soil system (Hewitt et al., 2015). The different definitions of *ecosystem services* all make the connection between the 'work done' by ecosystems and the benefits or well-being humans derive from it (Braat and de Groot, 2012). Soil functions thus define the potential performance of the soil to deliver services to other parts of the terrestrial ecosystem, to other ecosystems and to society at large.

Soils are thus at the heart of ecosystem service delivery. Soil-related ecosystem services depend on soil characteristics and their interactions (through soil processes), and are highly influenced by land use and land management (Adhikari and Hartemink, 2016). There are many ways in which the contribution from soils to ecosystem services delivery can be described and classified (Box 1.1). For the purpose of this report, we use the ecosystem service classification as proposed by the Millennium Ecosystem Assessment (2005) to group soil-based services.

Box 1.1 Linking soil functions to ecosystem services: approaches to classification

A variety of methods have been used to define and classify ecosystem services depending on the ultimate aim of the classification: two common objectives are ecosystem assessment and ecosystem accounting.

The most notable and also most cited classification is the one applied in the Millennium Ecosystem Assessment (2005). One particular point of interest here is that some ecosystem 'functions', that is the underlying processes of nature, support other functions that, in turn, provide flows of services that more directly benefit people. This led to a classification of four types of services (see Table 3.2), three of which are perceived to directly affect people, namely the provision of material goods and services, the regulation of natural processes and the provision of non-material goods and services, the so-called cultural services. A fourth type — supporting services — is essential to maintain the other three.

The Common International Classification of Ecosystem Services (CICES, 2013) was developed to support environmental accounting (EEA, 2015b):

CICES takes the Millennium Ecosystem Assessment classification of ecosystem services as a starting point but modifies the approach to reflect more recent research and does not include supporting services to reduce the risk of double-counting of benefits. The three main ecosystem service categories under CICES are provisioning services (e.g. biomass, water, fibre); regulating and maintenance services (e.g. soil formation and composition, pest and disease control, climate regulation); and cultural services (the physical, intellectual, spiritual and symbolic interactions of humans with ecosystems, lands and seascapes) (EEA, 2015b).

Given that CICES was developed for accounting purposes, it would, in principle, be the most appropriate classification for economic analysis, in particular cost–benefit analysis (see Chapter 3). However, the fact that the 'supporting' services are no longer explicitly recognised in CICES represents a drawback when seeking to identify where soils (and their use and management) make a difference in delivering services, which is an equally important aspect of the economic analysis. Taking the emphasis away from supporting services results in the risk that soil-based services are under-identified and possibly under-valued. Indeed, many soil-based services are seen as supporting services, which underpin provisioning, regulating and cultural services. In addition, the CICES classification defines ecosystem services as requiring a biotic component; this risks ignoring particular soil services, such as the provision of minerals and raw materials and the provision of a carrier or medium for urban settlement and other activities.

A number of writers have reviewed the appropriateness of an ecosystems approach to guide soil resource management. Haygarth and Ritz (2009) provide an overview of soil systems, as they support the provision of land-based services. They use the Millennium Ecosystem Assessment classification to identify 18 ecosystem services and functions that are relevant to soil, and make particular reference to the contribution of soil to a number of key supporting services: nutrient cycling, water cycling and regulation, hydrological processes, gas and climate regulation, and biodiversity and genetic resources. Their assessment also recognises the importance of 'carrier/medium functions' and their association with different land uses, noting that this function in the built environment is commonly associated with soil sealing, which severely limits other functions and services. Given the diversity of services provided, Haygarth and Ritz argue that there is a need to maintain the multi-functionality ofsoils and to avoid critical thresholds beyond which they might be irreversibly degraded. Quantification and monitoring of soil inventories and functions, linked to land use and the services provided, are, they argue, essential prerequisites for efficient management of the soil resource.

Dominati et al. (2010) provide an extensive and insightful review of the literature covering the development of current thinking on ecosystem services and, within this, the treatment of soil. They conclude that the natural capital and ecosystem services of soil often go unrecognised and are generally not well understood, in spite of a good understanding of soil formation and functioning. They emphasise that the part played by soil in particular services, such as erosion or flood control, is not explicitly defined, nor is it linked to any particular soil attribute or function. They note that the provision of 'underground' services is particularly overlooked. In reviewing the literature on soil services, they identify a number of key roles of soil in delivering services, namely fertility, filter and reservoir, structural, climate regulation, biodiversity conservation and resource roles. They also highlight the complexity of soil processes and the tendency to link one particular soil property (e.g. bulk density) to a particular service (e.g. water cycling). In reality, multiple properties (such as carbon content and porosity) are important. For soil to be appropriately considered, they develop a conceptual framework that explicitly links stocks of natural (soil) capital and what they call the inherent — that is static (e.g. slope, texture) — and manageable — that is dynamic (e.g. structure, porosity, bulk density) — properties of soil with the flows of provisioning, regulating and cultural services that in turn meet human needs or wants. They also recognise the positive feedback between stocks and soil formation and the negative feedback between stocks and soil degradation, affected by both natural and human drivers and processes. Thus, they argue that it is essential to consider soil stocks and flows simultaneously in an integrated framework.

Robinson et al. (2013) also apply the concepts of natural capital and ecosystem services to soil, again arguing that the treatment of soil as a supporting service tends to obscure many of the goods and services that it contributes to. Like Dominati et al. (2010), they argue that the focus on flows of ecosystem services tends to under-value the critical role of stocks in the case of soil, because the underlying conceptual models mainly identify soil as delivering supporting services, and these are not included directly in flows of ultimate goods and services. For this reason, Robinson et al. adopt an earth-systems approach to represent the complex interactions between soil stocks and flows of services, focusing on the condition of the former. They develop the concept of an ecosystem services 'supply chain' (see also Barrios, 2007) in which the 'pedosphere' sub-system, comprising the abiotic and biotic properties of soil, interacts with other natural (e.g. hydrological) and human (e.g. land use) sub-systems to provide a stock of natural capital that generates flows of intermediate (e.g. support for plant growth) and final services (e.g. food provision) in the human sphere. They identify and classify a range of soil services, distinguishing between stock-flow provisioning services (such as peat and turf supply) and so-called fund-flow services associated with regulating and cultural services. Like Dominati et al. (2010), they make the point that, to date, there has been a focus on understanding the relationship between single soil properties/functions and the resultant supporting services and final goods (such as compaction and run-off). Or, alternatively stated, soil has to be understood as a complex system made up of components that can be configured differently to simultaneously deliver a range of services at individually varying levels. What is really needed, they argue, is an appreciation of the effects of multiple changes in soil characteristics, as this affects the multi-functionality of soil in the land, and associated trade-offs and synergies. Examples here include the joint and simultaneous effects of soil compaction and erosion on run-off, nutrient cycling and carbon exchange. Echoing Haygarth and Ritz (2009), they argue that more monitoring and quantification of changes in soil stocks are required as a support to the economic valuation of management options.

Finally, soil-related ecosystem services are also discussed under the concept of 'soil security' (Koch et al., 2013). Soil security refers to 'the maintenance and improvement of the world's soil resources so that they can continue to provide food, fibre and fresh water; make major contributions to energy and climate sustainability; and help maintain biodiversity and the overall protection of ecosystem goods and services' (Koch et al., 2013, p. 435). Soil security is described as being at the heart of addressing a number of inter-related global issues: food security, water security, climate change abatement, ecosystem service delivery, biodiversity protection and energy sustainability. They identify the following soil functions, a combination of which address the major societal issues: provision of physical stability and support; nutrient cycling; water retention; storing, filtering, buffering and transforming compounds; and biodiversity and habitat. In order to maintain the visibility of these functions, Koch et al. (2013) emphasise soil as a discrete component, yet a core building block of land. They thus propose a soil-centric approach to policy design that raises awareness of soil degradation and addresses the issue, thus contributing to sustainable development.

Soils and natural capital

Ecosystem services can be considered as an indirect contribution of natural capital to human well-being (ELD Initiative, 2015). Natural capital refers to 'the stock of natural resources that provides flows of valuable goods and services' (WB, 2012, p. 105). Natural resources are 'the naturally occurring assets that provide use benefits through the provision of raw materials and energy used in economic activity (or that may provide such benefits one day) and that are subject primarily to quantitative depletion through human use' (OECD, 2005). EEA (2015b) distinguishes two major components of natural capital: abiotic and biotic:

> Abiotic natural capital comprises subsoil resources (e.g. fossil fuels, minerals, metals) and abiotic flows (e.g. wind and solar energy). Biotic natural capital or ecosystem capital consists of ecosystems, which deliver a wide range of valuable services that are essential for human well-being (EEA, 2015b).

Natural resources are 'subdivided into four categories: mineral and energy resources, soil resources, water resources and biological resources' (OECD, 2005). The World Bank (WB, 2012) also considers the atmosphere as a component of natural capital, as does EEA (2015b). Soil can thus be considered as part of the stock of natural capital (Hawken et al., 1999). Soil is a natural resource or naturally occurring asset because it can be used, along with other types of resources, to produce goods and services that are of value to people. The combined ecosystems and natural capital perspective on soils views the demand for the soil resource as indirect, driven by the demand for final consumption.

In principle, complementing natural capital with other forms of capital (financial, manufactured, human, social) increases its productive capacity (WB, 2012). Depending on the level of sustainability one is willing to accept, components of natural capital can also be substituted by other forms of capital. For example, mineral fertilisers (implying use of manufactured capital) can substitute nutrients that are essential for plant growth and naturally generated by a well-functioning soil. In doses adapted to the context, artificial fertilisers help to maintain soil fertility. However, in excessive doses, they may have negative impacts on the environment (e.g. eutrophication of surface water). Substitution can also happen among different categories of natural resources. For example, soil as a plant-growing substrate is replaced by water in a hydroponics system (i.e. plants growing in water). Generally, however, the extent to which other forms of capital can be substitutes for natural capital is bounded, because people need water, food and air to live, and the demand for water and food will grow as populations and incomes rise (WB, 2012).

The limited degree to which the soil resource can be substituted by other resources highlights the finite or limited nature of the soil resource. Referring to these limits evokes the concepts of planetary boundaries, safe operating space and tipping points, which also refer to land aspects (land use, nitrogen and phosphorus cycles, and soil stability) (Rockström et al., 2009; Galaz et al., 2012; Brook et al., 2013; de Vries et al., 2013; Steffen et al., 2015).

From a natural capital perspective, the value of a stock of natural resources is a measure of the present value of future flows of benefits that they can generate over their lifetime, until they are used up. Some natural resources, such as fisheries and forestry, biologically regenerate; thus, their exploitation can be managed at rates which do not reduce the overall stock. However, soil formation is a slow process that normally takes hundreds of years. In Europe, soil formation by both weathering and dust deposition amounts to around 0.3–1.4 tonnes per hectare per year (t/ha/year) on average (Verheijen et al., 2009). Degradation rates owing to water erosion across the EU-28 are (depending on the model used) estimated to be around 2.46 t/ha/year (Panagos et al., 2015) to 2.76 t/ha/year (Bosco et al., 2015). Thus, given the time scales involved, soil is considered a non-renewable resource. The finite and non-renewable character of the soil resource implies that its supply is limited, in terms of both quantity and quality.

The economic importance of soil as a component of land systems was recognised two centuries ago (Ricardo, 1817). However, the economic role of soil is subsidiary to the role of land resources in general. The possible scope and extent of activities that a given land area can support (i.e. its potential or capacity to deliver valuable goods and services) depends on the different land dimensions: land cover/land use, land management, soil and the natural systems context in which they are placed (i.e. climate, altitude, topography and hydrology). Land with a high potential for service delivery may, however, be in poor condition, meaning that, while it has a good potential to deliver services, it has been degraded (Box 1.2). Important degradation processes include erosion, loss of soil organic matter, compaction, contamination, decline in soil biodiversity, salinisation and sealing (EC, 2006). Thus, it is the current condition or state that defines the quality of the land or soil resource.

Accepting that soil is an essential component of natural capital, and thus a natural resource that is instrumental in delivering valuable goods and services to society, implies that the importance of soil needs to

Box 1.2 Restoring degraded soil in urbanised areas

Afforestation of degraded land and soil in urbanised areas increases its capacity to provide soil-supported ecosystem services. Investment in restoration by afforestation can provide substantial long-term returns for urban communities.

Case: Parque Florestal de Monsanto, Lisbon, Portugal

Photo 1.1 Past creation in 1938 (left) and present-day (right) views of the Parque Florestal de Monsanto

© https://commons.wikimedia.org/wiki/Category:Parque_Florestal_de_Monsanto#/media/File:Parque_Florestal_de_Monsanto_1938_Foto_n%C3%A3o_identificada_1.jpg (left);

© Manuel V. Botelho, Creative Commons Licence: https://commons.wikimedia.org/wiki/File:Parque_de_Monsanto_5557.jpg (right)

More than 1 000 ha of land on the Serra de Monsanto in Lisbon had become severely degraded by soil erosion. After 1934, the bare soil was replanted following a plan made by the architect Keil do Amaral for landscaping, and leisure and sports areas. Today, it is the largest green area in Lisbon, with diverse tree cover and a 50 ha ecological park providing education on environment and wildlife conservation. The park covers about one-eighth of the city and provides a valuable leisure and amenity resource, as well as being an internationally recognised tourist destination.

Source: Fedenatur, n.d.

be recognised. The *valuation* of the soil resource and the flows of goods and services that can be derived from it is an approach that allows such recognition, highlighting that soil is an asset to the economy. Soil (as part of land) can be seen as both a 'productive' and a 'financial' asset (Fairbairn, 2014). As a *productive asset*, land is appreciated primarily for its 'use value' or how it can fulfil human needs, whereas the exchange value and thus capital gains come to the fore when its role as a *financial asset* or *investment* is highlighted. Note that, when focusing on the 'productive' or 'use' value of an asset, the premise that natural capital can be 'turned into' a monetary value is no longer a prerequisite for recognising the value of an asset.

In this report, we focus on the value of soil in fulfilling human needs, even though the two roles cannot be entirely separated. Thus, high-quality soils are considered more valuable because they can supply more or higher quality services, and/or because they are more versatile regarding the portfolio of services they can contribute to. Degradation processes lead to losses in the economic value of soil and remediation processes restore it.

1.2.2 Place- and asset-based multi-level governance

Multi-level governance and sustainability

Decisions on resource use and management, the subject of *governance*, are made in the political arena. Governance, in this report, refers to the exercise of control in the broad sense, and thus concerns the processes by which decisions are made, and power and control are held and exercised, by individuals, private and public organisations, and government (North, 1994). It includes the functioning of institutions and authority (informal and formal, traditional and modern), as these influence the use of (natural) resources in a society or economy. Governance embodies the norms, codes of behaviour and rules that guide human activities and interactions, and that help to resolve conflicts that may arise between competing uses of resources. A key dimension of governance is the allocation of resources and access to them (Biermann et al., 2010).

Land is a resource or asset that can, in principle, be governed from different governance levels, and thus requires a multi-level approach to its governance. At the global level, the United Nations (UN) Rio+20 Summit (UNGA, 2012) called for a land-degradation-neutral world in the context of sustainable development, in particular recognising soil degradation as part of land degradation. The European Union (EU) has also committed to this goal. The EU's 7th Environment Action Programme (EAP) (EU, 2013a) 'aims to ensure that by 2020 land is managed sustainably' (EEA, 2015a):

> Concretely, this commitment requires coordinated governance and integration of environmental considerations (including water management and biodiversity protection) into territorial planning decisions on land use. Land policy targets would also help achieve this goal, and the 7th EAP specifically suggests a target of 'no net land take' by 2050. (EEA, 2015a).

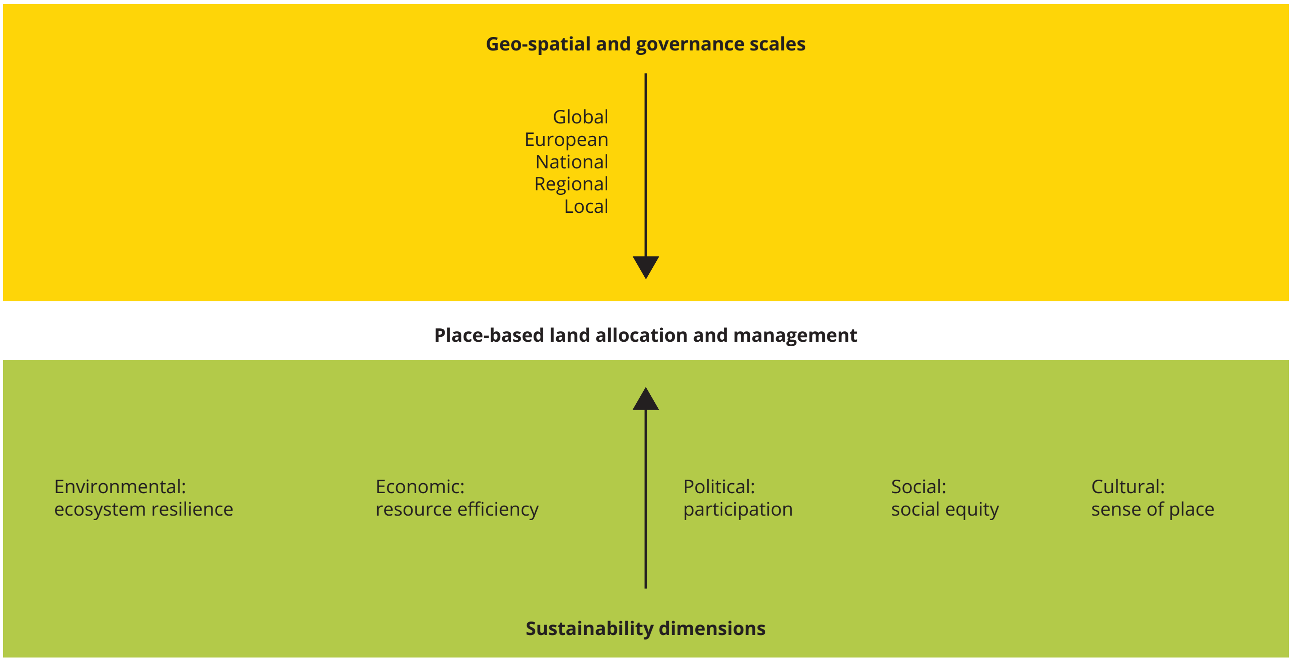
These concerns are also echoed in the adopted sustainable development goals (UNGA, 2015), which, through their implementation at national and sub-national level, are expected to contribute to a place-based approach to land governance.

Place-based land allocation and management allows land functions to be integrated 'across multiple sectors, sustainability dimensions and governance levels (global, EU, national, regional and local)' (Figure 1.2) (EEA, 2015a):

> A place-based approach takes into consideration local specificities and assets while designing and implementing policies to pursue development at different geographical scales (Zaucha and Świątek, 2013). This is the opposite of a sectorial approach, which makes policy integration across different geo-spatial and governance levels cumbersome. Through offering important synergies and coordination mechanisms and enhancing endogenous developmental forces — including territorial cohesion — a place-based approach is conducive to improving policy performance. Interaction between various stakeholders is central to such a cooperative process, and is also a key driver for territorial cohesion, as described in the Territorial Agenda of the EU (EU, 2007a; EEA, 2015a.

Both the global and the EU policy agendas can thus 'set a frame to promote place-based planning and solutions that make the most of an area's inherent features. Land decisions should therefore reflect such solutions, while being adapted to the local conditions and assets, including soil, terrain, climate and communities' knowledge' (EEA, 2015a)

Figure 1.2 Place- and asset-based land governance



Source: EEA, 2015a.

The emphasis on local assets in the place-based approach means this approach is aligned with the asset-based approach, which originates from community planning. To integrate an asset-based approach to planning, a community ideally makes a comprehensive inventory of assets and qualities, derived from physical features and natural resources. Ohm (1999) emphasises that inclusion of 'subjective and less visible assets that are *valued* by people in the community', yet 'may not be discernible to outsiders', is crucial in this approach. Accordingly, protecting and reinforcing what is good for a community is prioritised over the destruction of valued and irreplaceable resources. The asset-based approach is also characterised by applying terms and concepts from the private sector (e.g. asset, services and market-based instruments) to the natural environment (Curtis and Lefroy, 2010). The positive focus and the use of terminology from the private sector are reflected in application of the concepts of natural capital and ecosystem services (Section 1.2.1), along with the valuation approach (Section 1.2.3), to soils.

Emphasising the interests of the community at large is an approach that aligns with taking account of *sustainability dimensions*. The land systems framework (Figure 1.1) highlights different sustainability dimensions: pressures and state reflect changes in the bio-physical/environmental domain; impacts broadly reflect the bio-physical and socio-economic dimensions; responses happen in the political-institutional dimension; and driving forces reflect the socio-economic and environmental domains. However, the place- and asset-based multi-level governance framework (Figure 1.2) recognises two additional sustainability pillars: (1) quality of life and, particularly in relation to land, sense of place, recognising that human well-being comprises more than just material wealth and economic growth (socio-cultural), and (2) participation, which recognises that sustainable development entails the political involvement of all groups or stakeholders in society (political-institutional).

1.2.3 Decision-support tools for soil resource efficiency

Recognising soil as an essential component of natural capital, including its pivotal role in delivering ecosystem services to society, emphasises the importance of soil as a productive asset and provides an entry point for decision-making. The key question on which this report is centred is whether or not the importance of soil is reflected in decision-making, in other words in governance. Given the spatial character of land, such analysis requires, in principle, a spatially explicit approach.

Several analytical tools exist to support decision-making. Samarasinghe et al. (2103) provide details of different techniques that have guided the use and management of the soil resource, commenting on advantages and limitations. They consider cost-benefit analysis, cost-effectiveness analysis, multi-criteria assessment, optimisation model, total factor productivity, general equilibrium models, partial equilibrium models, simulation models and life cycle analysis. Most of these focus on the economic aspect of decision-making. Economic analysis can indeed provide essential supporting information. However, Samarasinghe et al. (2013) also recognise that new frameworks that do not rely solely on economic data are needed to support ecosystem service approaches to decision-making.

In this report, we focus on the *existing body of evidence* that is based on the application of such techniques in the context of urbanised areas (Section 1.3), as opposed to rural ones — we do not consider the full detail of the techniques that are the basis of the evidence. Nevertheless, some of the rationale behind such techniques will be presented in the course of the report, when underlying assumptions are deemed essential to understand or interpret the findings and to link the evidence to the broader analytical framework presented above (sections 1.2.1 and 1.2.2).

Soil valuation

One of the approaches that can be used to recognise the importance of soils is valuation.

Valuation refers, in principle, to the act of deciding how much money something might be sold for, or to the actual amount of money decided on; thus, valuation is about defining a monetary value for soil: an economic valuation.

However, in a natural capital and ecosystem services framework, monetary valuation is considered to be at the apex of progressive steps, namely (from bottom to top) listing the full range of ecosystem services, qualitative review, quantitative review, monetary valuation (TEEB, 2010). Drawing from the notion of 'value pluralism', that is 'the idea that there are multiple values which in principle may be equally correct and fundamental, yet in conflict with each other' (Gómez-Baggethun et al., 2014, p. 7), the EU FP7 OpenNESS Project ([1]) recognises that multiple values are required to capture the *diversity* of demands

([1]) EU FP7 OpenNESS Project: Operationalisation of Natural Capital and Ecosystem Services (http://www.openness-project.eu).

that society and individuals may make of nature. Accordingly, they propose an *integrated valuation* of ecosystem services, which considers three dimensions: ecological, economic (monetary) and socio-cultural. These dimensions are broadly in line with the societal dimensions and goals recognised in the place- and asset-based multi-level governance framework (Figure 1.2). The authors describe such an integrative ecosystem services valuation tentatively as 'the process of synthesising relevant sources of knowledge and information to elicit the various ways in which people conceptualise and appraise ecosystem services values, resulting in different valuation frames that are the basis for informed deliberation, agreement and decision' (Gómez-Baggethun et al., 2014, p. 20). Both monetary and non-monetary methods have to be considered in performing such a valuation.

In this report, we use the spirit of this definition to explore different approaches to valuation in the present-day governance of soils. Valuation is understood in the broadest sense of 'recognising the importance of', and thus does not necessarily refer to accounting approaches, for which a metric would be needed. In line with Defra's (2007) definition for economic valuation, valuation is defined in this report as an attempt to 'elicit public preferences for changes in the state of the environment'. Environmental changes also include changes resulting from policy interventions. Chapter 2 will address whether or not the knowledge base on soils (in urbanised areas) is sufficient to inform ecological-biophysical soil valuation. Given the growing interest in monetary valuation, including in applications to the domain of land (e.g. Samarasinghe et al., 2013; ELD Initiative, 2015), Chapter 3 focuses on the economic, monetary, dimension, in which the main aim is to evaluate whether or not the existing body of evidence on monetary valuation of soils (in urbanised areas) is sufficient to recognise the importance of soil to society. Chapter 3 will also further explore the economic dimension of soil. Chapter 4 explores the extent to which valuation is reflected in governance, and whether or not current governance practice recognises the importance of the soil resource (i.e. soil valuation in its broadest sense). This report does not elaborate on the socio-cultural dimension of valuation (Boxes 1.2 and 1.3). Nevertheless, in taking an asset-based approach (with a focus on productive rather than financial assets) (Section 1.2.2), we are particularly interested in the role valuation may play in protecting and maintaining the soil resource for human, public benefits.

Box 1.3 The socio-cultural dimension of soil valuation

Community access to urban soil resources that support community gardens provides a focus for social development and education. The services provided by soil in urban environments offer opportunities for cultural transformation.

Case: Prinzessinnengarten, Kreuzberg, Berlin (Germany)

Photo 1.2 Views of the Prinzessinnengarten

Nomadic Green rented 6 000 m^2 of vacant brownfield space and installed soil in boxes to create a community garden. The aim was to demonstrate that there was a demand for a space for social and environmental commitment. Collaboration with schools and kindergartens has developed this into a successful educational project. Social media has been used to create wider international awareness of this project, attracting film makers and tourists and contributing to the local economy. Putting the soil resource in boxes makes it possible to move the garden to a new location, if needed.

Source: Prinzessinnengarten, 2016.

Linking decisions to the soil resource

In order to apply the ecosystem services concept to decisions in which soil natural capital is a key consideration, it is essential to frame the link between the decision and the ecosystem services relevant to the decision-making (Samarasinghe et al., 2013).

Such framing requires that relevant analytical questions are considered:

- Where/how do soils and their use and management make a difference in delivering services (i.e. what is the marginal value of the soil resource in delivering services)?
- What are the (degradation) costs and benefits, and how are they distributed?
- Placing excess demand (for benefits) on natural resources (including soil) causes a decline in soil performance. How can this demand be managed optimally (Figure 4.1)?

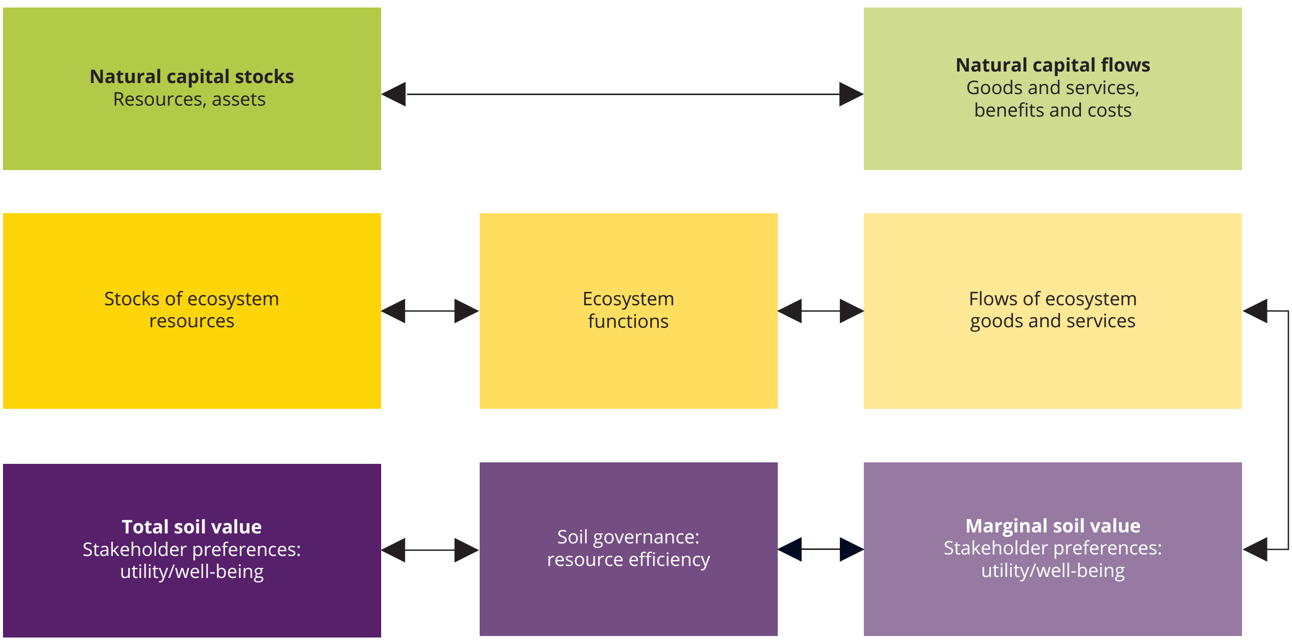
An answer to the first two questions will be sought in Chapters 2 to 4, while the third question will be addressed in Chapter 4. The natural capital and ecosystems concepts as discussed above (Section 1.2.1) provide the background against which soil can be valued and governance options can be evaluated (Figure 1.3). While the total value of the soil stock is difficult to estimate, there have been attempts 'to value marginal changes in the contribution of soils to ecosystem services' (Samarasinghe et al., 2013, p. 13) (see also Section 3.1.3).

1.3 Urbanised areas

In this report, we apply the proposed analytical framework for soil resource efficiency to urbanised areas, which, in essence, include core urban areas and the adjacent peri-urban areas.

There is a continuum between the dense built-up areas of core urban zones and the rural environment, with an associated gradient in the density of occupation and the proportion of an area occupied by artificial structures. The *urban core* is characterised by continuous artificial structures (so-called 'grey infrastructure'), interspersed with 'green infrastructure', within which soil is strongly modified and substantially sealed. Outside this core there is increasing discontinuity in the urban landscape or fabric, and the variety of land cover types increases to include areas of agricultural, forested and semi-natural vegetation. This is the peri-urban area, within which soil is less modified and sealed. This is of particular importance with respect to soil, because the peri-urban area extends over much larger areas than core urban zones. The peri-urban area thus contains the majority of the soil resources within urbanised areas as a whole (the core urban and peri-urban areas combined).

Figure 1.3 Natural capital and ecosystems: a framework for soil valuation and governance



Peri-urban areas are areas of transition (Errington, 1994). They are also areas of complexity where rural areas are transformed into continuous urban fabric owing to pressures arising in adjacent urban areas (Antrop, 2004). As such, these could be areas where the environment is considered to be at its most vulnerable. Their heterogeneity and the diversity of land use and management within them, combined with economic and demographic pressures, lead to contested access to land and related natural resources, including soil. The outer edges of peri-urban areas — where the variety of land cover/use narrows to agriculture, forestry and semi-natural areas — is not distinct and is typically star-shaped, with 'fingers' of discontinuous urban fabric and a greater scarcity of structures extending outwards along major transport routes.

This low-density, dispersed expansion of urban areas is commonly referred to as urban sprawl. Urban sprawl is a phenomenon that can be visually perceived in the landscape as a scattering of urban development or as solitary buildings and comprises three dimensions: (1) the size of the built-up area, (2) the dispersion of the built-up area and (3) the size of the built-up area per inhabitant or job (or the utilisation density) — the higher these three measures, the greater the urban sprawl (EEA [forthcoming] — EEA Report 'Urban sprawl in Europe').

The governance of urbanised areas does not fit an institutional structure based on an urban–rural dichotomy, and is complicated by overlapping urban, rural and regional interests (Allen, 2003; EEA 2015e). Nevertheless, this report does not focus on rural areas, except when considering the processes and implications of the progressive extension of urbanised areas into rural ones.

This report focuses on urbanised areas because:

- urbanised areas, particularly core urban areas, have the highest population densities; thus, more people can benefit from the services derived from soils, and these people are equally exposed to the disservices or risks resulting from unsustainable soil use and management;
- peri-urban areas are very dynamic in nature, and are, compared with core urban and rural areas, more likely to present conflicting demands on soil (Box 1.4).

Where competition for the soil resource is high, exploring potential synergies and trade-offs between various service options is essential. Land resource efficiency — originally an economic concept — therefore becomes a question of use and management options subject to decision-making and thus governance.

Box 1.4 Competition for the soil resource: differences between and dependencies of core urban, peri-urban and rural areas

Urban areas are not independent of rural ones. With a population of around 500 million people in an area of 4.4 million square kilometres, in 2012 each person in the EU-27 had 8 818 m^2. For the same year and area, the mean cropland (including arable land and permanent crops) area per person was 2 178 m^2, while the mean area with artificial cover per person was 406 m^2 (Eurostat, 2012). However, about one half of the area with artificial cover is estimated to be sealed and has to be regarded as having limited soil functions (EC, 2012a).

This means that most of the food supply and largely also the drinking water supply in cities comes from peri-urban and rural areas. Urban waste, production residues and sewage sludge from the cleaning of sewage water are exported from urban areas and deposited in peri-urban areas. On the other hand, the expansion of peri-urban areas is heavily dependent on the construction of roads, creating increased mobility (EEA, 2006).

This results in competition between urban and peri-urban areas in particular, but rural areas are also involved (EEA, 2015d). Therefore, supply and export from peri-urban and rural areas to core urban areas are essential for the functioning of urban soil use. This means that urban soil use has to be managed in view of its dependency on contributions from outside the city. This includes not only regional but also national and international relationships, a view that has not been sufficiently considered.

2 The soil resource in urbanised areas

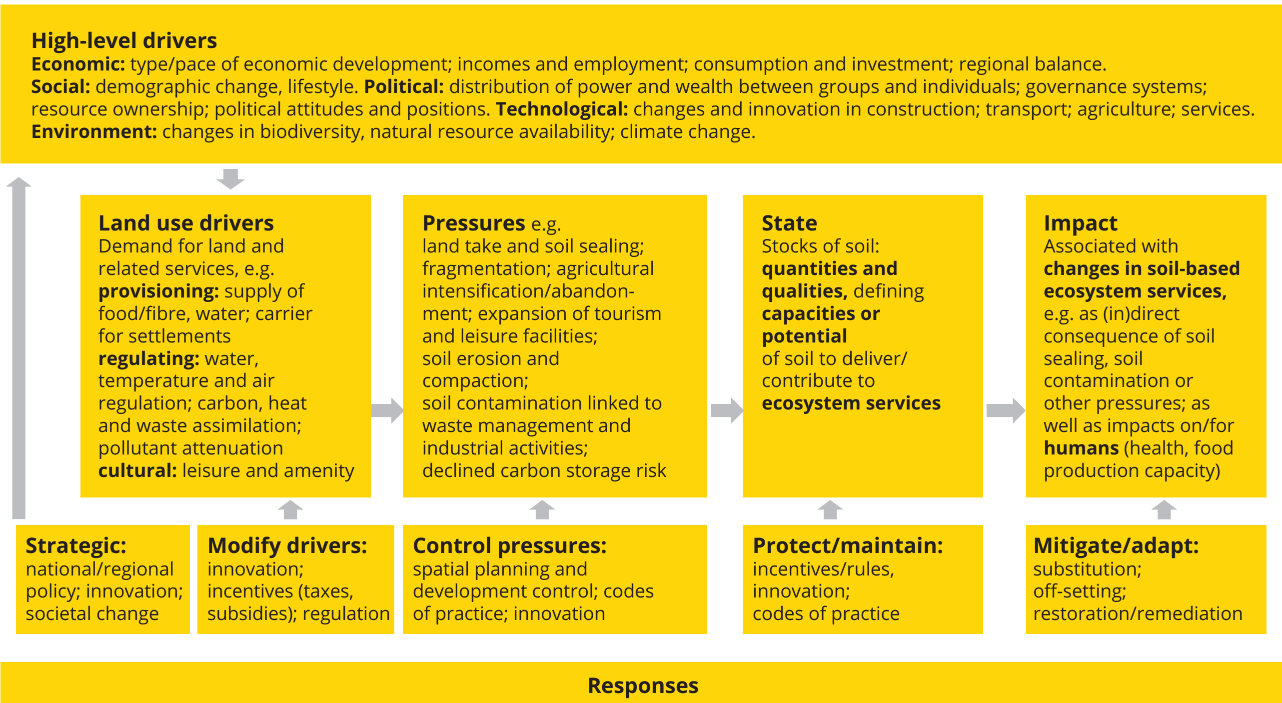
This chapter gives more detail on the processes that soil is subject to in an urbanised setting, and unavoidably focuses on how human activities shape soils. It recognises that the knowledge base on soils in core urban areas is different from that in rural areas, and highlights the need to build a body of knowledge to help understand the importance of urban soils to society. Thus, this chapter will identify the knowledge requirements and gaps for asset-based soil governance in urbanised areas.

Figure 2.1 elaborates on the DPSIR framework applied to the land system (Figure 1.1), focusing specifically on the soil component. It illustrates how high-level drivers influence land-use drivers that create pressures on soil via the activities of stakeholders (private citizens, private and public organisations) that change its state. Such changes impact on the services that soil delivers, and any alterations create responses. These responses may be strategic and intended to affect drivers of land use and management, or may be more tactically focused on modifying direct drivers and pressures acting on soil. Responses will not be dealt with in this chapter, but rather in Chapter 4.

2.1 High-level drivers of soil resource dynamics

Towns were founded thousands of years ago, evidence of which can be found in Greek and Roman cities in Europe. In most cases, in the past, cities covered relatively small areas, but this changed in the 18th century with the onset of industrialisation and the accompanying technological changes, when towns started expanding. Rapid development and growth of urban areas and increasing populations in urban areas ensued. Thus, most soils in present-day urban areas have a relatively short history of human impact and of development in an urban environment.

Figure 2.1 DPSIR representation of soil resource dynamics in urbanised areas



Likewise, urban system dynamics are a relatively young phenomenon.

Society is always changing; changes in demography, the structure of households and income are key drivers that influence the demand of society for resources (EEA, 2015d).

Size and structure of the population (*demography*) have been shown to affect the extent of urbanised areas: generally, the larger the population, the more space is required to accommodate people. Population growth has achieved its limit in most European countries, owing to a low birth rate (Bini et al., 2010). However, urban areas are not homogeneous across Europe: there is rapid and strong expansion in some regions, while there is on-going decline in other regions.

Migration within and between countries already results in growth stagnation and shrinkage of some cities (Bini et al., 2010). In shrinking cities, a large number of houses remain vacant (Bini et al., 2010), and many sites previously used for housing, commerce and industry are abandoned. Nevertheless, urban sprawl — a soil resource-intensive phenomenon (see Section 1.3) — has continued to increase in regions where population decline is apparent (Hoymann, 2011). In economically prosperous regions, urban areas are still growing and there is demand for sites to be transformed into urban areas, both of which are driven by immigration (from outside Europe) and internal migration between European countries and regions. Since 1985, Europe has increasingly been a receiving region for immigrants, however, with an unequal distribution and intensity across countries (Muñoz de Bustillo and Antón, 2015). In addition, high internal migration levels have contributed to urban growth, sometimes following the sprawl pattern (Bontje, 2001).

At the same time, the *structure of households* has changed. Since the 1960s, the size of homes has increased steadily and the number of households has grown owing to the increased number of single-parent and single households (including elderly people), with a dwindling number of persons per household as a consequence (Boardman et al., 2005; Bini et al., 2010). The average household size in the EU-25 declined from 3.3 persons in 1960 to 2.4 in 2003, implying a much faster growth in the number of households than in the population size (EC, 2008). The most common household type in the EU-28 in 2014 was a single person living alone, representing about one-third of all households (Eurostat, 2014). The size of private households influences resource demand, in particular for housing. Accordingly, the average personal living space has increased; for example, in England, this parameter rose from 38 m^2 per person in 1991 to 43 m^2 in 1996 and 44 m^2 in 2001 (Boardman et al., 2005).

Gross domestic product (GDP) per capita expresses the productivity per inhabitant in a given region and increases proportionately with *income*. Income affects consumption patterns, including those of housing and cars, both of which affect the use of land (through the development of residential areas and road infrastructure). GDP has been shown to stimulate urban sprawl in individual countries (Barbero-Sierra et al., 2013), as well as across Europe (Bosker and Marlet, 2006) and the world (Bertaud and Malpezzi, 2003). According to the Organisation for Economic Co-operation and Development (OECD)'s projection (Kharas, 2010), worldwide, there will be three billion (3 000 million) new middle-class consumers by 2030. This rise in new consumers will be a driver for world economic growth, but will exacerbate the demand for natural resources and food, with additional pressure put on the soil resource. At the same time, lifestyle changes owing to increasing environmental awareness (e.g. sharing initiatives versus private ownership) and increasing attention to healthy lifestyles are expected to create shifts in consumption patterns, with positive effects on resource use and the environment in general.

Climate change is also affecting the soil resource: in particular, regulating services, such as soil as a carbon sink or soil as a medium that regulates floods and temperature, will be affected by climate change; climate also directly influences provisioning services such as water supply and biomass production (Section 2.4.2).

Drivers that constitute measures of governance control (e.g. subsidies, taxes) are outlined in Section 4.4.

2.2 Land-use drivers and multi-purpose land use

As a consequence of the varying trends in population growth, structure and habits, we can expect strong regional differences in the demand for soils for urban use in Europe. Particular population groups, such as children, single people, elderly people and migrants from rural areas, may also make different demands of soil. Such demands include both *quantity* (amount) and *quality* (type) of soil. One of the ways to describe soil types in a functional sense is to refer to the different soil functions and the different potential uses of soil, and thus the services that can be delivered.

Trends in land ownership and management have often lead to a *single main purpose* for land, to the exclusion of others, often in response to dominant markets or policy drivers. For example, restrictions on development and the scarcity of land available for development make it more costly to allocate land for urban green space. In addition, production-oriented farm subsidies and, more recently, strengthened agricultural prices encourage intensive agriculture at the expense of biodiversity.

By contrast, *multi-purpose* land use involves multiple activities within the same space, combining, for example, built structures, farming and nature conservation, housing and gardens for biodiversity, forestry and carbon sequestration. In this respect, land use produces 'multiple outputs and, by virtue of this, may contribute to several societal objectives at once' (OECD, 2001; De Groot, 2006). The multi-functionality of land is key to this idea, and it is particularly relevant for peri-urban areas, as these are characterised by a mosaic of land uses in transition. A large urban population enhances the potential value of multi-functional unsealed land in urbanised areas, both within and outside the core urban area (Box 2.1).

With respect to peri-urban areas, Zasada (2011) identifies a growing 'urban-centred' appreciation of the goods and services provided by, and dependent upon, agriculturally managed land. While land area for agriculture is limited and reducing, there is increased interest in and competing demand for the remaining open space in peri-urban areas to be used for recreation, nature conservation and intensive farming (Davoudi and Stead, 2002). Furthermore, urban populations place considerable value on 'landscape amenities, supported by a heterogeneous and small farm structure, punctuated with natural elements' (Arriaza et al., 2004; Kaplan et al., 2006; Zasada, 2011). Outdoor recreation is particularly important, and there is an increasing demand for educational, social and caring functions (Di Iacovo and O'Connor, 2009). The growing interest of urban consumers in local-area produce, including organic produce, also favours multi-functional farming in peri-urban areas (Van Huylenbroeck et al., 2005). In this respect, multi-functional 'peri-urban agriculture' can provide a potentially complementary component of strategies to safeguard agricultural land, limit urban sprawl and reduce the likelihood of dereliction on the urban fringe (Zasada, 2011). Multi-functional landscapes, and the communities that relate to them, are also likely to be more resilient and sustainable in the longer term (Banks and Marsden, 2000). However, much depends on market and policy incentives, and the ability and willingness of 'locally embedded' famers to respond (Wilson, 2007).

The suitability of land for one or more uses/purposes is defined by land characteristics (e.g. soil, climate, topography). Current land use and management also determine possible future land uses. However, soil is the critical component of the multi-functional character of land (Section 2.4) and thus of multi-purpose land use. Recognising its role (e.g. in food production, amenity provision and flood regulation) reduces pressures on soil outside urbanised areas. The challenge is to recognise the value of multiple flows of services and build these into policy instruments (including market-based ones) that reward the delivery of services, or penalise their loss.

The natural capital and ecosystem services approach helps to appreciate the multi-functional role of soil and its ability to support a range of different benefits

Box 2.1 Multi-purpose land use and green infrastructure

The concept of multi-functionality is relevant to the concept of green infrastructure (GI), namely a way to work with nature to provide ecological, economic and social benefits (such as air quality, temperature regulation, noise reduction, flood protection and recreational areas) to the (urban) population (EC, 2013). The EU Working Group on the GI Strategy argues that GI also 'promotes integrated spatial planning by identifying multi-functional zones and incorporating habitat restoration measures into land-use plans and policies' (EC, 2012b, p. 1).

GI is perceived to 'benefit human populations and contribute to a more sustainable economy based on healthy ecosystems delivering multiple benefits and functions' (EC, 2012b, p. 1). The value of green space to manage storm water, to counter the heat island effect and to restore ecological functions in urban areas has indeed been recognised (Brown et al., 2012). Thus, GI is also seen as a means of acknowledging and increasing the economic value of ecosystem services and of creating incentives for local stakeholders and communities to deliver them (EC, 2012b). In addition, GI delivers social and cultural benefits by providing open spaces and recreation facilities, supporting urban–rural connections and a general sense of community, and strengthening the societal and cultural link with nature and biodiversity (EC, 2012b).

The ecosystems approach provides a useful framework for classifying the multiple benefits of GI in ways that appeal to policymakers and other stakeholders (EEA, 2011) (see also Section 3.3.1).

Box 2.2 Mapping land use in urbanised areas

In practice, it is more common to map land cover than land use. As an alternative, a hybrid of both can be produced, as is the case in the Corine Land Cover ([2]) and the Urban Atlas ([3]) maps.

The most detailed harmonised mapping of land cover/land use in 'functional urban areas' (FUAs) in Europe is currently provided by the Urban Atlas. FUAs have been defined as having more than 100 000 and 50 000 inhabitants (as defined by the European Commission's Urban Audit) for the reference years 2006 and 2012, respectively. Urban Atlas 2012 is being produced for 695 FUAs, including 301 existing and 394 new FUAs. The mapping distinguishes 17 classes in the core urban area with a minimum mapping unit of 0.25 ha, and 10 classes in the urban fringe/peri-urban area with a minimum mapping unit of 1 ha. The higher minimum mapping unit and the higher number of land cover/use classes used in the Urban Atlas reflect that urbanised areas show a much higher diversity of land use types than rural ones.

However, at regional or local level, a stronger differentiation between land use types may be needed. Examples are the STABIS (Statistisches Informationssystem zur Bodennutzung) classification system as recommended by the Arbeitskreis Stadtböden der Deutschen Bodenkundlichen Gesellschaft (BRBS, 1989) and the ATKIS (Amtliches Topographisch-Kartographisches Informationssystem) system of the German state North Rhine-Westphalia, which maps contaminated land (LANUV NRW, 2007). Targeted mapping systems, such as STABIS and ATKIS, allow distinct land use types to be linked to very specific soil features/characteristics.

simultaneously to different stakeholders. For the most part, the degree of multi-functionality of soil and the range of ecosystem services provided largely depends on the way that land is used and managed (Box 2.2). Likewise, the different claims, serving one or multiple purposes, will subsequently have effects on land/soil use and management.

2.3 Pressures on the soil resource

2.3.1 Soil sealing

A set of soil profile degradation processes (Wood et al., 2005; Lehmann, 2010) are linked to construction and development of the built environment. The soil profile may be truncated by the removal of topsoil, buried by material such as gravel or rubble placed on the soil surface, or degraded by an admixture of imported materials and soil during excavation and construction activities. Most importantly, the soil surface may be *sealed* by structures and paved surfaces (Photo 2.1). Despite using the soil's function as a medium for construction in these cases, soil sealing strongly reduces the availability of other soil functions owing to disrupted water, nutrient and biological cycles. In addition, extreme surface soil compaction makes soils impermeable. Soil sealing thus results in a loss of soil functioning that is close to irreversible (EC, 2012a).

Local patterns of soil sealing are not uniform. The degree to which an area is affected by soil sealing is related not only to the different types of land use, but also to the time it took a place to be transformed into an urban area and to the urban planning. For example, the cities of Sofia (Bulgaria) and Helsinki (Finland) differ greatly in their sealing patterns (EEA, 2011). Accordingly, the number of unsealed patches of land within an urbanised area varies strongly with the land and soil function use objectives (Dahlmann et al., 2001). Peri-urban areas have lower population densities than core urban areas. Nevertheless, they also suffer from the effect of sealing. Numerous 'grey' infrastructure elements, such as roads, fragment an area into smaller parts. Land fragmentation not only affects the biosphere by challenging species movement and exchange, but also affects the geosphere by impeding ecosystem processes such as material flow.

2.3.2 Soil erosion and compaction

Soil excavation and transport are common in both core urban and peri-urban areas. Excavating soils for use elsewhere can be interpreted as a form of deep, human-induced erosion (Burghardt, 2011). Generally, this type of erosion affects urbanised more than rural areas. The effects of construction and related human activities will diminish with distance to 'grey'

([2]) http://www.eea.europa.eu/publications/COR0-landcover; http://land.copernicus.eu/pan-european/corine-land-cover/view.
([3]) http://land.copernicus.eu/local/urban-atlas/view; http://www.eea.europa.eu/data-and-maps/data/urban-atlas.

Photo 2.1 An alternative to complete sealing of a parking lot

© Geertrui Louwagie

infrastructure, so that intensity gradients can be observed. Soil also gets compacted as heavy equipment spreads tipped material. This compaction not only occurs in the surface layer, but also extends to greater depths (Burghardt, 2000, 2007).

2.3.3 Risk of declined carbon storage

The combined effects of the expansion of impermeable surfaces and of excavation and degradation of soils during construction may result in low carbon storage. Unsealed land in urban areas can hold as much organic carbon as rural land and almost twice as much carbon as carbonate. At the same time, several factors suggest that carbon storage in urban areas can potentially be increased: 'intensive management of urban soils can result in higher carbon reserves than similar soils in rural areas' (Brown et al., 2012, p. 173). Thus, carbon storage in urban soils is an important function that needs to be managed (Box 2.3). If not managed properly, for example by making spatial planning decisions that favour soil sealing, low carbon storage may become an issue and thus a potential pressure.

2.3.4 Soil contamination

Contamination represents another important pressure in urbanised contexts. Contaminants can be both inorganic (e.g. lead (Pb), copper (Cu), zinc (Zn)) and organic (e.g. polycyclic aromatic hydrocarbons (PAHs), polychlorinated biphenyls (PCBs), dioxins) in nature. Sources are commonly point or local sources, such as former industrial plants (as evidenced on some brownfield sites), which are well-known emission sources of potentially toxic elements such as Pb, Cu and Zn. The burning of fossil fuels and traffic are other local sources in urbanised contexts: Pb, Cu and Zn are commonly associated with traffic (Biasioli and Ajmone-Marsan, 2007), whereas motor vehicle exhausts have been indicated as major sources of PAHs (Morillo et al., 2007). However, long-distance diffuse pollution is also affecting urbanised environments. Analysing a series of organic and inorganic contaminants in urban parks of Torino (Italy), Biasioli and Ajmone-Marsan (2007) found that historical gardens that were located far from the main emission sources had the most contaminated soils, indicating a long-term deposition of airborne particulate matter rich in pollutants.

Overall, where land/soil use and management represent the pathway via which pressure is exerted on the soil resource, for example through soil sealing and soil erosion, additional attributes of land use (e.g. emissions resulting from industrial use) have to be taken into account where soil contamination induced by both local and diffuse sources is concerned.

2.4 The state of soils in urbanised areas

2.4.1 Urbanised areas as a new environment for soil functions (supply)

One of the main differences between urban and rural soils is the fact that the former are more strongly influenced by settlement and ensuing anthropogenic activities, including grey infrastructure development (roads, telecommunications networks, supply pipelines, etc.) and industrial activity. As a result, urban soils are characterised by strong temporal and spatial heterogeneity (Effland and Pouyat, 1997; Morel et al., 2005).

- First, the use of soils in urban contexts changes a lot with time — which may entail risk for residents — such as a change from industrial use to residential, public or recreational use.
- Second, the many inputs of exogenous materials and the mixing of these (often) anthropogenic materials with the original soil material leads to a high spatial variability; generally, the closer to the core of the urban area, the more artificial the soil material.

Considering that *soil formation* typically involves three processes — weathering, transport and accumulation — natural weathering plays a minor role under urban conditions, with transport and accumulation the predominant processes (Morel et al., 2005).

Soil-forming factors and resulting soil characteristics and functions

In urbanised areas, the factors steering soil formation and development (i.e. substrate and parent material, climate, topography, vegetation and fauna, humans and time) are essentially the same as those influencing natural ecosystems. Considering the *time factor*, 'older cities show the most modified soils, as they are generally constructed on their own waste materials that have accumulated over years. For example, cities like Paris (France) and Moscow (Russia) are built on several metres of anthropogenic materials that hold remains of former human activities and materials' (Morel et al., 2005, p. 203). However, the *human factor* is, overall, the more defining one, as it involves 'extremely rapid transformation cycles in comparison with those dominant under natural conditions' (Morel et al., 2005, p. 202). Human activity also results in conditions different from those resulting from the other soil-forming factors.

In urbanised areas, natural and man-made soil material is commonly excavated, transported and tipped. The material can be uniform or reflect various origins, and can appear stratified or homogenised. It often consists of or contains human-made materials: rubble from constructions and demolition; waste from households, municipalities and industry; production residues; ashes and residues from heating and energy production (Zikeli et al., 2002); slag from iron and metal processing; sludge from cleaning processes; diverse sorted natural and man-made material; mining spillage; dredging material from deepening harbours and rivers; composts; and soils treated by soil (mechanical, heating, biological) remediation (Hiller and Burghardt, 1992). Intrusion of solid particles (e.g. atmospheric dust), liquids and gases can also contribute to this mixture, commonly called *substrate*. New substrates may also contain pollutants; in fact, core urban areas may even play a key role in concentrating particular pollutants (such as Pb, Zn and Cu) within their borders (Biasioli et al., 2006).

The wide spectrum of substrates used in urbanised areas results in a high diversity of urban soil properties over small areas (Bullock and Gregory, 1991). Urban soils are generally expected to have one or more of the following soil characteristics: an increased stone/gravel and sand content (resulting from predominantly sandy and stony construction material import), a high pH (often as the result of alkalising products, such as rubble or slag, mixed in the soil), a high organic matter content and a high carbon (C) to nitrogen (N) ratio (because of the presence of organic material that is low in N, especially in industrial soil material, owing to contamination with oily wastes) (Burghardt, 1997, 2000; Morel et al., 2005).

The diverse range of substrates and their properties also result in changes in the soil's capacity to hold *water* (storage and permeability of water) and *air* (air content and diffusion); conditions can be further affected by soil compaction and soil sealing (Section 2.5.1). Soil hydrological properties are also affected by changes in the distance between the soil surface and the water table caused by the removal and tipping of soil material, and changes in the water table owing to changes in abstraction regimes. Particularly intensive drainage in core urban areas splits the original wide natural river catchment area into numerous artificial areas. The unsaturated (or vadose) zone is affected by the differences in porosity of new, artificial soil material, by inhibited gas exchange as a consequence of soil compaction and sealing, and by the development of methane due to decomposition of some organic wastes (only under anaerobic conditions, e.g. as a consequence of waterlogging).

Another urban phenomenon is the *increase in air temperature* (Mount and Hernandez, 2002), which can increase physical, chemical and biological weathering processes in soils (as also observed under natural conditions).

In particular, substrate and climate, and their resulting properties (including the effects on the dynamics of water and air in the soil) will influence *vegetation and fauna*. The variability of soils in urbanised areas, along with a high variability of environmental components, results in numerous ecological niches. Therefore, species richness in urbanised areas can be high (Schadek et al., 2009; Meffert and Dziock, 2012). Furthermore, vegetation and, to a lesser degree, fauna supply the soil with organic matter (especially in green spaces, parks, gardens, etc.). Through evapotranspiration, vegetation has effects on soil drainage and soil temperature. Mosses and lichen can initiate the weathering of sealed surfaces, although this capacity is reduced under polluted conditions. Animals will contribute to turbation (mixing) and loosening of soils, and can also create large pores for water infiltration and soil aeration (Dornauf and Burghardt, 2000). In addition to the human activities mentioned above, *humans* also contribute to the bio- and pedosphere through gardening and landscaping. The overall artificial management of urban soils also augments natural processes of carbon capture and storage (Box 2.3).

2.4.2 Urban demand for soil-based services

Following the above, soils modified by humans and new substrates in urbanised areas are subject to conditions that are different from those in a rural, more natural, context. As a consequence, soils formed in an urbanised environment will have different properties (Section 2.4.1) and so potentially different *soil functions*.

Within urban and peri-urban areas, soils are mainly used, although not necessarily recognised, to serve diverse human activities and needs, including as carriers for infrastructure at or above the soil surface (e.g. buildings, roads, railways) and underground (e.g. parking areas, pipes); as sources (e.g. landfill material from excavation) and sinks (e.g. for dredged sediments) of raw materials; for food production; for recreational activities (e.g. parks); and in connection with rituals (e.g. cemeteries) and cultural heritage (e.g. archaeological sites open to the public) (Morel et al., 2005).

However, using land for urban purposes requires that soils be *suitable* to serve those human needs (demand), namely that they have the appropriate characteristics and exhibit the set of functions required (supply). This aligns with the concept of balancing demand and supply, as proposed as part of the land system (Figure 1.1). A generic classification of the soil-based services particularly relevant to urbanised areas and geared to monetary soil valuation is provided in Table 3.2. Some of these are discussed in more detail below.

Supporting services

Soil provides a range of 'supporting' services, such as the cycling of nutrients, water and atmospheric gases, and provides habitats for biodiversity, both above and below ground. These support the functioning of ecosystems (usually classified by habitat type) and the range of services that these provide. The ability of soil to provide this support is dependent on its rate of formation and retention. In fact, soil formation is itself a supporting service.

Provisioning: carrier/medium for human activities

To serve as a construction medium and to withstand the physical stress applied, soils should have the appropriate strength or stability, which is largely influenced by particle size, moisture content and plasticity of the colloidal/clay fraction (Brady and Weil, 1999). Plasticity (i.e. the degree to which a substance can be changed into a new shape, here specifically referring to a state between solid and liquid) of the colloidal fraction relates to a soil's potential for gradual vertical subsidence or settlement. In the context of construction, soils rich in certain colloidal silicate clays and micas are not easily compacted into a stable base for roads and foundations (Brady and Weil, 1999). Furthermore, some clays, particularly smectites, swell when wet and shrink when dry; soils rich in such clays expand, and subsequently shrink and crack easily. Such movements can be sufficient to crack building foundations or burst pipelines, for example (Brady and Weil, 1999). Thus, fine sandy, coarse silty and organic soils are not suitable for construction purposes.

Using land as a burial ground requires good soil aeration. Without sufficient aeration, the decay of corpses will be slowed down or stopped. Therefore, cemeteries are commonly located on soils that are well drained and not influenced by groundwater.

Provisioning: water supply

Many urban areas cover catchment areas for drinking water abstraction (Banitz et al., 2000). High-quality groundwater is a very precious resource that is highly protected. Thus, contamination by inorganic and organic pollutants, pesticides and nitrate must be inhibited. The soil that the percolating water is passing through must either be free of pollutants, pesticides and nitrate or reduce their solubility. Reducing the solubility of heavy metals happens under non-acid soil conditions (Arnz et al., 2000; Bolan et al., 2003). In many urban soils, this is achieved owing to the high carbonate content of rubble. Such soils have a high acid-neutralising capacity, which keeps the pH stable at a neutral to slightly alkaline level (Hiller, 2000). Furthermore, the concentration of pollutants in the percolating water is influenced by the soil structure. A dual soil-pore system composed of bigger pores percolated by water and smaller pores active only through diffusion reduces the concentration

Box 2.3 Carbon storage in an urban context

Carbon (C) storage in urban soils takes two forms: organic carbon and inorganic carbon (in the form of carbonate minerals) (Renforth et al., 2011). Organic carbon enters soil from vegetation as it does in rural systems, with inputs from leaf litter and composts, as well as through root processes and associated microbiological activity. The amount of organic carbon stored in urban soils for a variety of land uses to varying depths (1 m or less) ranges from 20–160 t/ha (Renforth et al., 2011) to 175 t/ha in rural soils (Howard et al., 1995).

The formation of carbonate minerals (pedogenic carbonates) through precipitation from the soil solution in urban soils is widely observed (Renforth et al., 2009; Rawlins et al., 2011; Washbourne et al., 2012). This concerns predominantly calcium carbonates; their formation depends on the availability of calcium, which is readily available from the products of demolition of concrete structures or from mortars. When a building is demolished, current practice is to recover as much material as possible for reuse, but inevitably some material (especially the fine-grained fraction derived from concrete, brick or masonry crushing) enters the soil. This material contains calcium hydroxide and poorly crystalline calcium silicates derived from cement, and these react with carbonate in solution to form the calcium carbonate mineral calcite.

A number of papers have investigated the calcium carbonate content of urban soils. The top metre of soil typically contains up to 20 % calcium carbonate, equivalent to 2 % inorganic carbon (Renforth et al., 2009; Washbourne et al., 2012). These proportions are equivalent to 300 t C/ha. The formation of soil carbonate minerals extends throughout the full depth of artificial soils (which can be several metres), giving a carbon capture function that potentially extends much deeper than the accumulation of organic carbon in rural soils. Carbonate formation to 10 cm depth equivalent to the removal of 20 t C/ha annually was measured by Washbourne et al. (2015). Importantly, there is some evidence that organic carbon is stabilised as carbonate through natural mineralisation reactions (Manning et al., 2013).

Calcium carbonate minerals formed by pedogenic processes in urban soils can be distinguished from natural carbonates of geological origin (such as limestone aggregates) using carbon and oxygen isotope analyses (Renforth et al., 2009).

Combining organic and inorganic carbon, urban soils have the potential to contain 320–460 t C/ha. This important function needs to be considered in planning and management of urban soils. It is a very dynamic process, given that its development follows the demolition of a building, the date of which provides a 'time zero' for the accumulation of inorganic carbon.

of pollutants (Baedjer and Burghardt, 2000). From this viewpoint, soils with a low pollutant content, a high acid-neutralisation capacity (i.e. an alkaline (carbonate) geochemical barrier in urban topsoils) and a soil structure favourable for reducing the pollutant concentration in percolating water may be suitable for water abstraction from groundwater in urbanised areas. Nevertheless, all potentially contaminated land needs to be subject to a specific risk assessment before any use can be approved. The approach taken varies from one country to another. However, methods developed for agricultural soils may need to be adapted for assessing and managing the risk of transfer of pollutants to the food chain in urban systems (Morel et al., 2005).

The provision of both domestic and industrial water supply can be affected by soil degradation. Storage of water in the soil depends on available pore space. Therefore, compaction or the removal of soil by erosion (including the loss of organic material) reduces the available water storage capacity. Factors that restrict infiltration (e.g. surface compaction and surface sealing) disrupt groundwater recharge.

Provisioning: mineral and organic material extraction

Soil is a source of materials, including peat and aggregates (strictly, sand and gravel are geological resources, but their exploitation often requires the removal of soil). These direct soil-based services are valuable for construction in urbanised areas but simultaneously degrade the soil resource for future use. Even where post-extraction restoration of soil is practised, there is likely to be some net degradation of the soil resource and a loss of soil-related services during the extraction period, which may extend over several years.

The demand for turf grass in urbanised areas is high and can support business and employment. Favourable soil for turf production is fine textured and well drained and, as such, is also valuable for higher value food production. However, turf production is an extractive process that moves soil from one location to other dispersed ones, resulting in an overall loss of agricultural and/or horticultural potential. More positively, conservation of topsoil during construction is recognised as good practice that can save substantial costs (Defra, 2009a) and that is incorporated into quality systems for the construction industry with product standards (e.g. ISO 9001:2008/BS EN ISO 14001:2004 and OHSAS 18001:2007).

Provisioning: biomass (food, feed, fibre, fuel) production

Agriculture (both crop production and animal husbandry) and horticulture mostly occur in peri-urban areas. Crop production requires soils that are well drained, have a low pollutant content and high fertility, maintained by sufficient levels of organic matter. Requirements for horticulture are similar; however, horticulture can be performed in greenhouses, using either the natural soil on which they are located or a substrate, the quality of which is independent of the local soils.

Regulating: water, temperature and air-quality regulation

Soils have a high capacity to store or retain water, which will be evaporated on the soil surface and transpired by plants. Soils differ in water storage capacity, in the capillary rise of water from the groundwater and in the depth plants need to be rooted to take up water for transpiration (Brady and Weil, 1999). Precipitation flowing through the soil will, with a time lag, reach the groundwater and eventually the rivers (Photo 2.2). In core urban areas, and also partly in peri-urban areas, water is collected in storm water sewers, which discharge the precipitation with only limited delay into the river. High amounts of storm water thus create flood peaks and flooding in lower lying parts of urbanised areas, particularly in alluvial plains (Konrad, 2003). Thus, storm water should be kept as long as possible in the city, meaning that a sufficient amount of the urban(ised) area should be kept available for storm water infiltration to control run-off and flood generation downstream. As the soil's pores provide the actual storage medium, to perform their flood control and regulation function well, soils should have a sufficient surface infiltration rate (strongly influenced by soil physical structure and the absence/presence of soil sealing), should not be compacted down to the groundwater table and should not be polluted (e.g. by saline water) (Terhorst et al., 1999).

Water is the main sink for heat: evaporation makes water the main consumer of heat and so also the cooling medium of heat (Depietri et al., 2012). Soils are thus also a heat-regulating medium, a function that is particularly relevant in urban(ised) contexts. Plants differ in rooting depth (shallow with grass, deep with trees). For heat mitigation, it is essential to have enough and a good distribution of soils with a high water storage capacity and suitable plant cover. Alternatively, technical solutions such as irrigation can be helpful. Green roofs are equally suitable for extending the area contributing to water storage and heat mitigation (EC, 2013).

Photo 2.2 The role of soil in flood regulation

Urban vegetation provides many benefits in urban areas, such as air cleaning, temperature reduction, giving shade, noise reduction or groundwater lowering, and urban vegetation depends on soil (Morgan, 2007; Perino et al., 2011). Hence, we need to encourage interest in developing so-called green infrastructure (Box 2.4), the components of which are well distributed over (particularly) the core urban area. To enable such development, enough unsealed soil must be available for plant growth. Besides air quality, the development and health of urban green infrastructure depends on the biomass production potential of soils. Soils should be of a high enough quality to supply grass, herbs, bushes and trees with sufficient water and nutrients, and should not be compacted for deep-rooted trees (Brady and Weil, 1999). Cemeteries are important elements in the green infrastructure network; today, many of them have been converted into urban parks.

Regulating: carbon, heat and waste assimilation (sink)

Urban soils have a wide spectrum of carbon contents, and act as a sink for both organic and inorganic carbon (Box 2.3).

The heat sink function of soils is used in a practical way in mechanical heat pumps.

Waste disposal usually happens in peri-urban areas. Waste disposal heaps should be covered by soil layers of low permeability that are thick enough to store the precipitation water and minimise water percolation. However, there is a legacy of historic urban waste disposal within most industrialised urban areas, often predating regulations that are designed to prevent leakage of gases and liquids from waste.

Sewage sludge is usually applied to soils in peri-urban areas. For safety reasons, that is to avoid contamination via sludge application, and to dilute the high nutrient content of the sewage sludge, soils on which the sludge is applied should be loamy and have a high water storage capacity. Their pH should be in a range such that heavy metals are not mobilised. Overall, the requirements as set out in the EU Sewage Sludge Directive (EU, 1986) should be complied with.

Regulating: pollutant attenuation

Organic matter plays an important role in the retention of so-called urban metals (Cu, Pb and Zn), which either enter soil from the atmosphere, dust or various human activities, or are directly added to soil with the addition of organic amendments (e.g. those derived from urban wastes or sewage sludge) (Madrid et al., 2004). Pollutants can thus undergo transformations in the soil during mineralisation or immobilisation, processes that can be related to organic matter breakdown and in which soil organisms play a major role. Nevertheless, urban soils exhibit a range of retention capacities for various organic components, and their alkaline reaction often limits the mobility of heavy metals (Morel et al., 2005).

Box 2.4 Soil contributes to services delivered by green infrastructure

Gómez-Baggethun and Barton (2013) consider the classification and valuation of ecosystem services for urban and peri-urban areas. The authors include all 'green and blue spaces' in urbanised areas (e.g. parks, cemeteries, yards and gardens, urban allotments, urban forests, wetlands, rivers, lakes and ponds) that are often portrayed as 'green infrastructure' (GI). The ecosystems approach can indeed provide a useful framework for classifying the multiple benefits of GI in ways that appeal to policymakers and other stakeholders (EEA, 2011). Drawing on research literature, they constructed a framework for classifying urban ecosystem services. In virtually all of these, to varying degrees, the service classes they identified depend on the support services provided by soils, although, as already noted and concluded by Dominati et al. (2010), it is difficult to robustly apportion service flows to particular soil attributes.

Food supply from urban and peri-urban agriculture is an important provisioning service in which soil properties differentiate the type and value of output. Regulation services are particularly important in the urban environment: water flow/water retention regulation and run-off mitigation are components of the hydrological cycle through which soil properties, along with land cover, affect infiltration, surface water movement, groundwater storage and flood probability. The soil–water–plant system regulates urban temperature through evaporative cooling, and soil provides a heat sink for heat pumps used for heating or cooling buildings. Noise abatement, air purification and buffering against extreme events such as storms, heat waves and floods are important services provided by urban vegetation, with urban vegetation dependent on soil functions and properties. The assimilation of urban and agricultural wastes by soil-supported flora and microbial fauna is another important supporting service provided by urban and peri-urban soils. At a global rather than a local level, urban ecosystems can help to reduce greenhouse gas emissions through sequestration by vegetation and soils, especially high-carbon peats; this is a regulating service. Pollination by insects and birds is part of a complex flora–fauna interaction, in which soil plays a critical support role. With respect to cultural services, green space in urbanised areas can improve public health, well-being and social cohesion, and can increase the appreciation of the importance of the wider contribution of urban ecosystems.

Cultural: amenity and recreation

The human urban population in all stages of life needs space and functioning soils for numerous outdoor activities. In addition, recreation sites, such as footpaths, seating areas, lawns, urban parks and forests, should not be polluted or produce dust. However, sites altered by humans (e.g. barbecue sites) are more likely to be contaminated. Sites for artificial sport and leisure facilities, such as hard-ground football pitches and tennis courts, should also be well drained. Outdoor entertainment facilities, such as leisure parks, race courses and horse-riding stables, are more commonly located in peri-urban areas. The main requirements for such land use are soils that are well drained, have a sufficient bearing capacity and have low dust release and low pollutant contents.

Cultural: archive for heritage

Urban soils contain archaeological and historical layers that document a town/city's history (Box 2.5). Each cultural, social or economic period produces its own deposits and layer. To maintain/preserve these cultural archives, the soil conditions must be managed properly, as soil degradation can adversely affect the cultural heritage archives in soil. Buried artefacts may be damaged by soil erosion, the uncovering of fragile remains, excessive surface loadings, construction processes and surface sealing. Peat soils in particular, because of their properties, contain valuable records of past environments and cultural activities, which can be lost through erosion or removal of the organic layer. Surface sealing may help to preserve buried artefacts if it creates anaerobic conditions. However, surface sealing may also change the historic land use of an area and prevent investigation of past cultural heritage. Importantly, excavation of heritage sites or monuments should be performed only under the control of experts (Jones, 2007).

2.5 The impact of urbanised soil use and management

2.5.1 Impacts on ecosystems

Soil sealing and local and/or diffuse soil contamination are pressures specific to urbanised contexts (Section 2.3); low carbon storage could become a pressure if carbon storage is not properly managed. Other disadvantages of urbanised soil use are soil compaction, soil structure degradation by soil disturbance, increased stone content, and strong acidification (from the non-liming of former urban parks or chemical factories) on some sites or extreme alkalinity (from slag) on other sites. These pressures result in soils that no longer function in the way they would under (semi-)natural conditions.

Soil sealing

Sealing interrupts the contact between the pedosphere and the atmosphere and thus changes the gas, water and material fluxes (Burghardt et al., 2004). The soil sealing pattern heavily influences water infiltration and preferential flows (Arnold and Gibbons, 1996): rainwater cannot directly infiltrate an impervious area; thus, it either drains beside the sealed area (e.g. along a small road) or is discharged and enters the sewer system (e.g. around big buildings or parking areas). Sediment and dust particles follow the same pathway. Soil sealing also decreases plant and soil evapotranspiration. If water infiltration is not facilitated at the edges of completely sealed larger areas, groundwater recharge may decrease; if infiltration is facilitated, surface water may break through to the groundwater, with an increased risk of groundwater pollution as a consequence (Burghardt et al., 2004). This so-called barrier effect at the edges of impervious areas may lead to erosion in adjacent areas. Sealing can also lead to a reduction in the capacity of the soil to act as a carbon sink (Scalenghe and Ajmone-Marsan, 2009).

In addition to affecting evapotranspiration, soil sealing also influences the albedo (or reflection coefficient) of surfaces, which may lead to increased temperatures above sealed areas (micro-scale) and within urbanised areas at large (meso-scale) (Burghardt et al., 2004). Furthermore, the permeability of the materials used to seal areas affects, to a large degree, the potential of sealed areas to host life, and can thus have an effect on biodiversity (Scalenghe and Ajmone-Marsan, 2009). Thus, sealing may also result in ecosystem fragmentation, with effects on habitats and species. By contrast, the deliberate introduction and maintenance of green infrastructure can counterbalance biodiversity impacts.

Nevertheless, despite the impact that soil sealing has on the soil resource, the quantification of its influence has been limited so far (Scalenghe and Ajmone-Marsan, 2009).

Soil contamination

Since coming into existence, urban areas have been areas of waste production that are also well drained, and thus prone to dust formation. Waste, dust and pollutants are deposited on soils and will move into soils (Thornton, 1991; Craul, 1992). Soils can store pollutants as solutes, bound to minerals and organic matter or precipitated as solid compounds. However, storing contaminants has possible trade-offs: a high capacity to immobilise contaminants makes

Box 2.5 Soil as an archive of urban heritage

With the growth of cities around the globe, soils are more and more influenced by the urban and industrial environment. Technosoils are soils that include all kinds of materials made, exposed or transported by human activity that otherwise would not occur under natural conditions at that specific location of the Earth's surface. The urban soil is exposed to pavement, buildings and pollution. The profile development is often very limited, except for archaeological dumps.

The soil shown in Photo 2.3 is from under the city of Hilversum (the Netherlands). At around 73 cm, the original soil, a sandy soil from a former heathland, is visible. This soil was exposed to prolonged waste water infiltration that was channelled to these heathlands in the past (since 1860), when it was still at the edge of the border of the city. This caused pollution with, among others, heavy metals and polyaromatic hydrocarbons. Between about 50 and 73 cm in the profile, artefacts are visible. This layer consists mainly of poorly degradable to non-degradable materials from a former waste dump that was placed on top of the former infiltration fields.

The top half metre of the profile is the result of the expansion of Hilversum in the early 20th century, when the soil was covered with construction sand and pavements, streets and houses. Between 2003 and 2006, the soils in this part of the city were remediated. The profile is an archive of past uses and different phases in society. It reflects the development of environmental awareness and the perception of community health (Mantel, 2015).

Photo 2.3 Profile of an urban soil

contaminants unavailable to plants or humans and may prevent groundwater contamination, while the same retention of contaminants may be harmful for the soil biota. Soil thus performs the functions of a pollution sink and source in parallel. Nevertheless, a sufficient area of uncovered/unsealed soil within urbanised areas is required to allow the exchange and transformation dynamics described above. Most worryingly, soil contamination represents a health hazard (i.e. has the potential to cause disease). Whether or not this hazard turns into a health risk (i.e. the extent to which the potential may be realised) is strongly dependent on humans' contact with soils and their exposure to contaminants, and thus on how they use the soil.

Other soil disturbances

Loss of the archaeological, as well as the geological, soil record as a consequence of soil excavation and other disturbances for construction (housing, grid infrastructure, etc.) can be considered an ecosystem impact. However, this impact is also directly relevant to humans, as it is relevant to a community's or society's past.

2.5.2 Impacts on humans

Impacts on humans are related to health and disasters, such as floods and landslides, but equally economic impacts are also relevant to humans (e.g. costs related to health, insurance costs, soil cleaning costs and costs for the maintenance of green areas, as well as costs related to people moving away from the city in search of better environmental quality) (Burghardt et al., 2004).

The most sensitive groups of inhabitants in urbanised areas (children and elderly people) deserve particular attention and protection. The main problems these groups face are pollutants (Järup, 2003), dust (Ritz and Wilhelm, 2008) and extreme temperatures (NRDC, 2011). Some of these are directly related to the soil condition.

Soil contamination

Increased concentrations (including those above national legislative limits) of contaminants in European urbanised areas compared with rural ones have raised concerns about adverse health effects. For example, some types of PAHs are potential carcinogens (Morillo et al., 2007), while some of the organic and inorganic pollutants found in urban parks are highly toxic (Biasioli and Ajmone-Marsan, 2007). Of the potentially toxic elements (commonly called heavy metals), Cu, Pb and Zn have been recognised as having a particular urban signature (Madrid et al., 2004; Biasioli et al., 2007); they are commonly found in European urban parks (Madrid et al., 2006). Urban soils can also be a source of potentially toxic elements when small particles — such as $PM_{2.5}$ and PM_{10}, in which potentially toxic elements are accumulated — are removed from the surface layer and suspended in the atmosphere (Ajmone-Marsan et al., 2008; Sialelli et al., 2011). The main pollutant sources for children are the soils in and around their homes (e.g. enrichment of metals from paints), the soils of playgrounds, schoolyards and gardens, and unconsolidated and paved walkways and streets. Dust, contaminated or not, is particularly dangerous for children, as they are generally close to the ground. The impact of children's involuntary soil ingestion during outdoor activities depends on the soil adherence to the skin, which tends to be limited to particles with a diameter up to 50 micrometres (μm) (roughly the clay and silt fractions) (Madrid et al., 2008). On playgrounds, children are also potentially exposed to the faeces of cats, dogs and foxes (Rokicki et al., 2007). Humans are also accumulators of pollutants, particularly owing to their long life expectancies and increasingly long life durations (Casino, 2011).

Furthermore, pollutants in soil can restrict vegetable cultivation as practised in many home and allotment gardens, as contact with soils is often intensive. Humans will also be affected by eating their harvest (Alloway, 2004; Kabala et al., 2009). Compared with gardens, the frequency of soil contact in park areas is expected to be lower. However, public parks in particular represent a critical environmental compartment in relation to health risks, as the pathways between sources and receptors (the public and their pets) of pollutants are shortest in these areas (Biasioli and Ajmone-Marsan, 2007). Nevertheless, the availability and human bio-accessibility of potentially toxic contaminants will ultimately determine whether an identified health hazard in an urbanised context becomes a health risk (Madrid et al., 2008; Poggio et al., 2008).

Soil sealing

Finally, core urban areas suffer from heat islands (Mount and Hernandez, 2002), exacerbated by soil sealing. High air temperature causes very serious health problems, particularly for elderly people but also for children.

2.6 Evaluating the knowledge base on ecological soil valuation: functional urban soil mapping

2.6.1 Soil functions and service terminology adapted to urbanised settings

The description of soil functions and derived services in Section 2.4.2 has given us a fair idea of how soil

— of the required quality and in sufficient amounts — can make a difference in urbanised settings (see key question in Section 1.2.3). The analysis has revealed that the demands on soil in an urbanised context are different from those in a rural one, and that some soil-based services are more relevant than others. Likewise, the urban soil-forming environment is very particular, affecting soil characteristics and ultimately the supply of soil-based services. Therefore, soil function and service descriptions have to be adapted to urbanised contexts (Box 2.6).

2.6.2 Need for urban soil function mapping

Given the spatial character of land, it is equally important that relevant soil information be mapped. Based on the information gained by soil mapping, soil analyses and soil evaluation, soil information systems are established. Such systems ensure that soil information is available and ready to be used in practice (Schneider, 2002).

In order to assess the contribution of soils to the functioning of urbanised areas, soils along with their functions and related services need to be mapped and monitored. However, soils in core urban areas are spatially complex and generally poorly mapped (Effland and Pouyat, 1997). In many European countries, soils were not mapped in core urban areas during the era of the soil survey (e.g. Belgium, Denmark, United Kingdom), but geochemical elements and compounds have been mapped in several European urban areas since

Box 2.6 Adapting soil function terminology to an urbanised context

Case: the city of Berlin (Germany)

The city of Berlin (Senate Department of Urban Development and the Environment, 2013) uses the following criteria for the development of soil functions:

- regional rareness of a soil association;
- special characteristics of the natural environment;
- near-natural (or undisturbed) quality;
- exchange frequency of the groundwater;
- nutrient storage capacity;
- nutrient supply;
- water supply;
- infiltration capacity;
- binding capacity for heavy metals and other pollutants.

The *soil functions* distinguished by the city of Berlin (Senate Department of Urban Development and the Environment, 2013) are:

- habitat functions for rare and near-natural plant communities;
- yield function for cultivated plants;
- buffering and filtration function;
- regulatory function for the water balance;
- archival function for natural history;
- efficiency of soils in the fulfilment of the natural soil functions and the archival function.

Photo 2.4 Recognising the importance of soil in built-up areas

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then (e.g. in Czech Republic, Estonia, Germany, Greece, Italy, Lithuania, Norway, Serbia, Slovenia, Spain, Sweden, United Kingdom) (Johnson et al., 2011). Nevertheless, some European towns/cities have invested in comprehensive urban soil surveys (Box 2.7). There are also examples of large-scale soil mapping (1:6 000) from the United States (Effland and Pouyat, 1997).

Box 2.7 Soil information systems in urban areas

Case: soil maps of German towns/cities

A set of maps of urban soil associations and their properties exist for **Berlin** at a scale of 1:50 000 (Senate Department of Urban Development and the Environment, 2013). The properties included are sealing of surfaces, potential for de-sealing, lead and cadmium content in soil and plants, soil characteristics, radioactivity in soils, soil functions, groundwater and groundwater temperature. Urban soil characteristics can differ considerably over distances of just a few metres. Therefore, the scale of 1:50 000 gives good guidance for urban district planning, but cannot cover the real local situation.

More detailed information is available from the urban soil map of **Stuttgart** at a scale of 1:20 000 (Holland, 1995). Nevertheless, the wide diversity of soil-forming substrates, in particular, calls for intensive and high-scale mapping of urban soils. Thus, soil maps at a scale of 1:5 000 are recommended to cover the need of urban administration. This scale was used, for example, in the urban soil maps of Oberhausen Brücktorviertel and of Krefeld (Schraps et al., 2000).

Mapping instructions for urban soils were developed and applied in Germany (Arbeitskreis Stadtböden der Deutschen Bodenkundlichen Gesellschaft, 1997; Ad-Hoc-Arbeitsgruppe Boden, 2005). In some instances, thematic maps to guide decision-making on urban water management, heat mitigation or land planning in general may be more meaningful.
For example, recommendations for mapping soil contamination also exist (Kuylaars and Barkowski, 2000, for the city of Duisburg; LANUV NRW, 2007). Instructions for surveying the soil for storm water infiltration are also available (Terhorst et al., 1999). An instruction was also proposed for monitoring soil sealing, focusing on practical measures such as de-sealing or assessing the potential for changing the surface cover from impermeable to permeable (e.g. of roads, parking areas, etc.) (Burghardt et al., 2004).

3 Valuation of soil in urbanised areas: the monetary perspective

3.1 Economic dimensions of soil governance

Decision-support tools are part of the proposed analytical framework for soil resource efficiency (Section 1.2.3). This chapter elaborates on economic tools, including those that are used for monetary valuation, and expands on some of the theoretical foundations. It also presents a generic approach to responding to two of the analytical questions supporting the application of the ecosystem services concept to decisions in which soil natural capital is a key consideration:

- Where/how do soils and their use and management make a difference in delivering services (i.e. what is the marginal value of the soil resource in delivering services)?
- What are the (degradation) costs and benefits, and how are they distributed?

3.1.1 Economic decision-support techniques

Economics as a discipline purports to guide public choice about the allocation of limited resources among alternative uses in order to maximise some measure of utility or welfare from a societal viewpoint (Boardman et al., 2006). It can also help to inform choice of policy instruments to implement preferred choices (Perman et al., 2011). There are three main economic decision-support tools that the results presented in this report are based on: cost–benefit analysis, economic impact assessment and cost-effectiveness analysis. All three techniques have a monetary character.

Cost–benefit analysis

Cost–benefit analysis (CBA) involves the appraisal of projects from the perspective of society as a whole rather than from the perspective of those responsible for decisions on projects (i.e. public versus private interests). It is concerned with developing systematic ways of analysing costs and benefits when market prices do not reflect social costs and benefits (Stiglitz, 2000). CBA identifies and, using a variety of estimation methods/techniques (Figure 3.1), puts values on the 'extra' benefits and costs associated with a development activity in order to judge whether or not it is worthwhile from the point of view of the public interest (i.e. total societal welfare).

> CBA requires that all benefits and costs can be given values on a common scale or denominator (e.g. monetary values). This allows for direct comparison of the costs and benefits associated with a project. However, while projects will often contain a number of costs and benefits that can be relatively easily identified and quantified, it is likely to be difficult to quantify all aspects of that project. (CBA Builder, 2011).

For example, no direct market value or appropriate market value may exist, that is the price may not reflect the actual value of a good, commodity or asset. In those cases, shadow pricing ([4]) can be used, and allows wide-ranging 'external' effects to be included beyond those that accrue to the immediate decision-maker. CBA is used when choices and discretion can be exercised and represents a tool that can be used beyond compliance with some prescribed minimum standard or output that must be met.

The determination of 'extra', or 'the margin', rests on a comparison of the 'with' and 'without' project interventions (i.e. alternative spending options). These interventions are mapped out over a future planning horizon, expressed in present monetary values using a social discount rate that reflects society's preference for consumption now or later. Two points are particularly important here: the choice of a 'counterfactual' with which an intervention is being compared, and an assessment about the extent to which an intervention really results in extra costs and extra benefits that would not otherwise

([4]) 'Shadow pricing is a proxy value of a good, often defined by what an individual must give up to gain an extra unit of that good. The value of a good resulting from a project measured with shadow pricing may, however, differ from the value of that or similar goods measured with market prices (occurrence of market failure)' (CBA Builder, 2011).

occur (referred to as the additionality/displacement issue) (HMT, 2011). CBA would indicate whether, for example, actions to protect soil are likely to deliver more benefits than costs, and are therefore economically efficient.

For the purpose of a monetary valuation of natural capital, CBA is most commonly used, as it relies on the comprehensive monetisation of all goods and services that are considered relevant to the decision/question (Samarasinghe et al., 2013).

Cost-effectiveness analysis

Cost-effectiveness analysis (CEA) compares mutually exclusive development alternatives in terms of the ratio of their costs and a single quantified, but not monetised, effectiveness measure (e.g. EUR/ha of decontaminated land) (Perman et al., 2011). It is used to determine the least costly option for achieving a non-discretionary target outcome, such as a binding legal obligation that is pre-determined and agreed (e.g. compliance with an adopted European directive). It can also inform the selection of options that will maximise benefits for a given cost. It is not a welfare-maximising technique in the way that CBA is, but it is useful for selecting cost-effective programmes of measures to deliver prescribed policy targets associated, for example, with implementation of the Industrial Emissions Directive (EU, 2010), the Water Framework Directive (EU, 2000), the achievement of target soil conservation or restoration measures, or policy options to remove public health risk from contaminated land.

CEA can be used as an alternative to CBA. The technique compares the relative costs of the outcomes (effects) of two or more courses of action (normally social or environmental). CEA is most useful when there are obstacles to conducting CBA, the most common of which is the previously mentioned difficulty of quantifying (monetising) all costs and benefits.

Economic impact assessment

Economic impact assessment (EIA) explores the economic impacts of interventions in terms of resultant changes in key economic indicators such as incomes (and their distribution), expenditures, gross value added and employment (Armstrong and Taylor, 2000). It considers the distribution and scale of impacts, including direct and indirect 'multiplier' effects focusing on impacts at the local and regional rather than national scale. It can be used for judging overall value against economic development objectives and for comparative assessment of interventions. EIA analyses can supplement CBA and CEA, for example to shed light on the local effects of regulations or to inform investment decisions (Box 3.2).

Box 3.1 Economic decision support and policy evaluation

Cost-effectiveness analysis and economic impact assessment are directly relevant to policy evaluation. CEA is suitable for comparing alternative options for reaching a defined outcome (e.g. alternative options to reach a policy target), whereas economic impact assessment can be used to assess the type, magnitude and distribution of impacts of policy interventions on the local and national economy.

Evaluating the relevance, effectiveness, efficiency and coherence of existing policies to account for the land resource is the subject of a separate EEA report: 'Evaluation of EU policies and their direct and in direct impact on land take and land degradation' (forthcoming). Likewise, policy analysis can also shed light on whether or not the importance of soils is reflected in the current body of policies. A similar reasoning is applied in Chapter 4, which explores the extent to which valuation is reflected in governance.

Box 3.2 Combining site-specific and regional economy valuation

Vandermeulen et al. (2011) propose that the site or project level (assessing the direct benefits and costs) should be distinguished from the regional economy level (assessing the indirect value for the development of the region). CBA is commonly used to do the assessment at the project level, whereas multiplier analysis can be used to look at the effects of the investment on competitiveness at the regional economy level. The combination of both methods enables the user to identify '(1) whether benefits outweigh costs at project level, and (2) whether the marginal multiplier effects on the region outweigh the project level costs' (Vandermeulen et al., 2011, p. 200). Thus, 'economic valuation can be used to convince stakeholders of a certain choice in landscape planning in urban regions' and 'can help policymakers to balance several issues' (Vandermeulen et al., 2011, p. 203).

3.1.2 *Economic efficiency as a governance principle for soil use and management*

The economic evidence presented in this report (Chapter 3) takes a *normative* utilitarian perspective. A *utilitarian* objective is to use soil so that social welfare is maximised over a manageable future planning period of time, perhaps 100 years. Given that soil formation is slow (in Europe about 0.3–1.4 t/ha/year on average), a period of 100 years will be too short to ensure that use and formation rates are matched; however, meaningful economic analysis over longer periods is not possible.

From an economist's perspective, the *value of soil stocks* at a point in time is determined by the present value of the future flows of services, discounted at the social discount rate. Thus, a change in the stock value is indicated by a change in the present value of flows of future services delivered. For example, the reduction in carbon content of soils associated with emissions to the atmosphere results in costs borne by society measured at the social cost of carbon. The present value sum of these emissions indicates a decline in the value of soil carbon stocks between any two points in time. A similar approach can be used in principle for the value of soil as a medium for food production. It is noted, however, that some degraded stocks and associated services may be *substituted* by other 'replacement' inputs, such as artificial fertilisers (see 'Soils and natural capital' in Section 1.2.1). This does not reduce the loss of stock value. Rather, it substitutes the natural functions of soil at an additional cost. The justification and feasibility of this substitution depends on how essential a soil attribute is to a particular service, and the ease and cost of substitution. In the case of soil sealing, the asset and service value of soils for agriculture are lost. There may be opportunities for offsetting and substitution elsewhere, but at a cost.

Economists tend to focus on processes of *change* rather than a *steady state*. For policy purposes, the concern is with the assessment of the extra (marginal) benefits and costs attributable to an intervention. This is fortunate, because it removes the need to determine the total value of soil stocks, a task that is virtually impossible. It is possible and important, however, to focus on changes in the condition of stocks of soils (e.g. the degree of erosion, compaction, organic matter decline, sealing) and the likely implications for flows of soil services. However, this does not remove the need to monitor and maintain *critical soil stock levels* (both quantity and quality), especially where these may fall below critical thresholds (Dominati et al., 2010). Likewise, Braat and de Groot (2012, p. 12) contend that it is key 'to identify, quantify and valuate all changes in ecosystem services', not just those resulting from changes in welfare (as dictated by the marginal value approach). Farley (2012) adds that, if ecosystem services are considered essential, marginal analysis and monetary valuation are inappropriate tools in the vicinity of ecological thresholds, considering that small changes can lead to enormous impacts under such conditions.

Criteria/conditions for *economic efficiency* in soil use and management (within urbanised areas) are proposed as follows (see also 'Soils and natural capital' in Section 1.2.1):

- The *marginal (extra) benefits* of a particular soil use or management practice are greater than or equal to the associated extra marginal costs. Net benefit (benefit minus costs) here is valued in terms of combined private and public benefits and costs associated with soil use, expressed as present values discounted at the social discount rate and aggregated over the appropriate time horizon.

- Soil is allocated among alternative uses *to maximise the present social value* of the future aggregate net benefit attributable to soil, with future benefits and costs valued and discounted as above. Any reallocation reduces overall social and economic welfare.

- The optimum use of stocks of non-renewable soil resources over successive periods of time (e.g. years), and their associated rate of degradation, is that which maximises the present value of net benefits as defined above. This *optimum allocation between periods* depends on the opening stocks of soil, net benefits of soil use in each time period, the rate and associated costs of degradation, the social discount rate, and other factors such as technologies that affect soil use and performance.

- The (weakly) sustainable use of finite, non-renewable stocks of natural capital such as soil requires that the rents (profits) derived from their use are reinvested in other forms of capital to substitute for the degraded stocks and the services provided by them. The *degree of replacement* depends on the essentialness of the soil service and the feasibility and cost of substitution.

- Decisions to protect, conserve or restore soils are also subject to the above criteria.

Following the above, soil use and management interventions should focus on:

- the *essentialness* (or added/marginal value) of soil in delivering flows of related ecosystem services;

- the *non-substitutability* of soils in delivering these services; and
- *resilience* (i.e. reducing environmental risks and ecological scarcities), considering that soil is a non-renewable and finite resource.

3.1.3 The monetary value of soils

The marginal economic value of soils

The soil resource is a subset of a complex land system (Foresight Land Use Futures Project, 2010), along with other characteristics such as topography, altitude, hydrology, living systems and, critically, geographical location. In turn, land as natural capital is combined with other types of capital such as physical infrastructure, technology and human knowledge to provide a range of benefits for people and communities.

In this respect, land and its attributes have the potential to add value and, given that it is a finite resource, choices are required on how best to use it. The guiding principle, as referred to earlier, is to allocate areas of land to those uses that have the *highest positive marginal social value* (measured as far as possible in euros per hectare): the greatest value added. The term 'social' here refers to the value that society places on the net benefits provided by land, recognising that its use has wide-ranging impacts, both positive and negative, for more than just those who own or occupy land for a specific purpose.

For the purposes here, it is the extent to which *variations in the attributes of soil affect the added value* of a particular piece of land, in particular uses, that is of interest.

Using the ecosystems framework, it is clear that *the relative importance of soil* supporting services for

Box 3.3 Land prices as a proxy for soil valuation?

If soil properties are important for particular land uses, in theory they should account for variation in land market prices. Land market prices should reflect the present value of the expected rents (profits) from land over a reasonable time period of use, say 50 years or so (UN, 2011), discounted to reflect time preference. To some degree this is evident in agricultural land markets, especially where the particular soil quality is favourable for high-value production (e.g. certain vegetables, vineyards).

An analysis of the relationships between land prices, taxes and land use (EEA, 2010) concluded that' environmental quality is not relevant for price determination if it is not important for the current or intended use of land'. This report notes that the price drivers for agricultural and urban land are different; it concludes that land productivity (which is related to soil type and condition) strongly affects agricultural land prices, but identifies no drivers for urban land prices that are soil related. It concludes that' the root problem is that the societal value of open space [and other environmental services] is not reflected in its market value'. In another study, it was concluded that variation in the ability of the soil to support construction appears to have little impact on land values, 'reflecting the high premium on development land in comparison with other land uses and the relatively low impact of different soils on costs of development' (ADAS, 2006).

However, land productivity is not always reflected in agricultural commodity prices either (EEA, 2010). According to Swinnen et al. (2009), agricultural productivity does not appear to be a major factor influencing land prices, as productivity today is dependent not only on soil quality but also on other inputs and investments, such as the use of agricultural chemicals. Plantinga et al. (2002) on the other hand provides evidence that the most productive agricultural areas in the USA face the least threat of being converted to other land uses, as high productivity means that their value as agricultural land is higher than it would be for other land-use options.

Case: land prices in England (United Kingdom)

In England, for example, prices in 2012 ranged from about GBP 20 000/ha for prime arable land (typically Agricultural Land Classifications (ALC) Grades 1 and 2) to GBP 8 000/ha for poor-quality grassland (Grades 4 and 5) (SmithsGore, 2013). Broadly, these price differences reflect ALC grades in which soil type and condition are important factors. However, poor-quality farmland in many urbanised areas commands much higher prices than high-quality agricultural land in rural areas that are distant from the urban core, suggesting soil attributes are relatively less important in urban land prices. Farmland that has been awarded consent for housing development in these areas in England can increase in market value to well over GBP 3 million/ha, irrespective of soil type in the higher economic growth areas of the south east of England (Foresight Land Use Futures Project, 2010). These severe price differences reflect a high degree of disconnection and distortion in land markets, partly owing to a failure to internalise the real costs and benefits of different land uses in market prices, including failure to account for soil-based services where these are particularly valuable.

provisioning, regulating and cultural services *vary according to land use* (Table 3.1). Soil, as an essential medium for agriculture, has a strong influence on the type and performance of crop production. Indeed, soil quality can be a factor affecting agricultural land market prices, reflecting potential value added. The full range of services provided by soil is less important in urban land use, where they mainly provide a platform function. They are, however, a particularly important and defining attribute for natural habitats, including those present in the urban core, as well as the peri-urban areas.

A further observation can be made on the major service flows by which soil adds value. For the most part, the provisioning services associated with agricultural production and a medium for urban development are likely to be reflected in the *market prices for land*, because the commodities they produce — food and built structures — are themselves traded both locally and globally. However, given the relative importance of these provisioning services, soil often makes more of a difference to prices for agricultural land than it does to prices for land for housing development, despite the fact that the consideration of soil in farmland prices may also depend on the importance given to other inputs and investments (Box 3.3).

Another observation can be made: many of the regulating, supporting and cultural services associated with land use are *public goods* for which market prices do not exist. Their values are not automatically included in the decisions of private land users and in land market values. Changes in these 'external' services or *externalities*, both positive and negative, owing to impacts on soil associated with land use may go unnoticed and unvalued. This indicates a *failure of land markets* to 'internalise' the 'external' benefits (e.g. failing to reward private land users for public goods) and the costs (e.g. failing to penalise private land users for public 'bads') of soil-related changes in ecosystem services (market failure). It also indicates a potential institutional failure where systems of governance that control land use do not formally recognise these external effects (*governance failure)*.

From a societal viewpoint, because of failure to identify and value non-market goods and reward their production, their supply is less than if markets were working properly. Similarly, failure of markets to value public 'bads', such as pollution from contaminated sites or sediment transfer to surface waters and transport surfaces, means that their incidence is greater than it would be if the costs of pollution were borne, for example, by those responsible, namely polluters.

The sealing of soil surfaces associated with infrastructure and housing development can generate run-off and flooding that can cause damage to third parties without redress. In the absence of control measures, those affected by off-site flooding have no entitlement to be protected from floods generated by newly developed land. Sealing of the soil surface may also alter the micro-climate adversely owing to a reduction in evaporative cooling. These real external costs, and the rights to protection or compensation that they might imply, are not usually the subject of the transaction when land is sold for development. This represents a failure of the market system that needs to be corrected by policy intervention while balancing the extra costs of the intervention against the extra benefits of remedying market and institutional failure.

The total economic value of soils

The total economic value of natural resources is often classified into two main types: those associated with use and those with non-use. This approach, often referred to as total economic value (TEV), can be applied to the soil resource (Figure 3.1) and is modified here to incorporate a classification of soil-based ecosystem services.

Most of the benefits generated by soil are associated with their *direct use* (mainly provisioning services with a consumptive character, but also regulating

Table 3.1 Indicative relative importance of soil attributes to added value of major land uses

Ecosystem services	Land use		
	Conservation	Agriculture	Urban
Provisioning	Low	High	Low (biomass)–high (carrier) ([a])
Regulating	High	Moderate	Low–high
Supporting	High	High	Moderate
Cultural	High	Moderate	Moderate–high ([a])

Note: ([a]) There is a perception that the local food production and cultural heritage (soil as archaeological archive) services are increasingly appreciated in urbanised areas, which should, in theory, be reflected in monetary valuations of the soil resource.

and cultural services that have a non-consumptive character) or *indirect use* (regulating, supporting and cultural services) (ELD Initiative, 2015). Some services that could be of direct use now, but are left to use in the future, get option value (provisioning, regulating and cultural services) (ELD Initiative, 2015). By not irreversibly changing soil now (by maintaining or preserving the potential to deliver services), there is also an option value to use it differently at a later date. The preservation of the best and most versatile agricultural soil that could support future farming when needs might be greater is an example. The option value concept incorporates a risk aversion premium arising from uncertainty about future demand and/or supply of the resource. Other services, however, are associated with *non-use value*, because the environmental good is considered intrinsically valuable in itself (existence or intrinsic value beyond human interest) or because it can be left for future generations (legacy or bequest value).

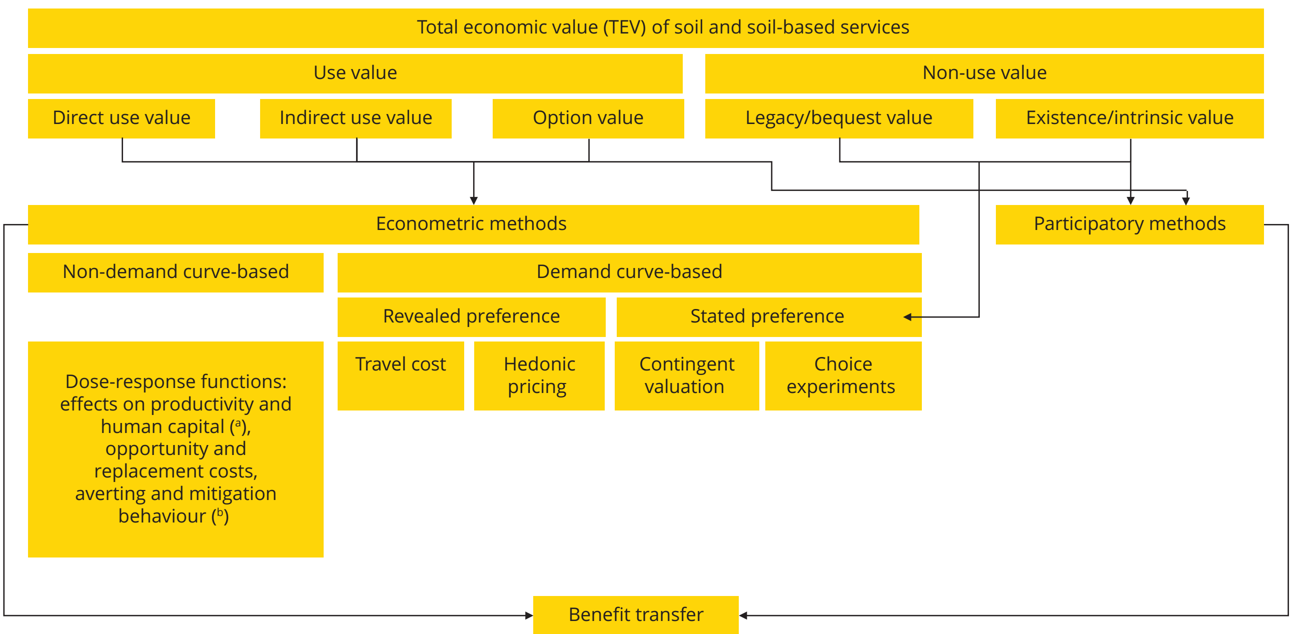
Methods for monetary valuation of natural capital/the environment

'Economic valuation attempts to elicit *public preferences* for changes in the state of the environment in monetary terms' (Defra, 2007, p. 12). The concept of multi-functionality is consistent with the ecosystem services approach and the valuation of diverse services from land, whether marketed or non-marketed, and whether associated with use or non-use. More specifically for the purpose here, the ecosystem services approach can help to identify the changes in goods and services provided by land that are attributable to changes in the quality of soil. Furthermore, the approach can link the flows of goods and services to stakeholder preferences.

Some supporting services provided by soil add value to marketed goods and services, such as the contribution of fertile soils to agricultural output. Many service flows attributable to soils, however, contribute to non-traded public goods. For this reason, valuations should involve a broad assessment of soil contributions and a range of techniques that broadly fall into two main groups: econometric and participatory/deliberative (Defra, 2007).

Econometric methods used can measure either the costs/benefits of environmental change (non-demand curve-based methods) or welfare changes resulting from environmental change (demand curve-based methods). Although non-demand curve-based methods are relevant in the environmental valuation toolbox, the resulting values are not well rooted in economic

Figure 3.1 Total economic value of soil between categories



Note: ([a]) Productivity is based on market prices (adjusted to remove the effects of taxes and subsidies and non-competitive market practices) and human capital is based on earnings forgone.

([b]) Averting behaviour is based on damage costs avoided, replacement costs or preventative/defensive expenditure, and mitigating behaviour is based on mitigation or substitute costs.

Source: Adapted from Millennium Ecosystem Assessment, 2005; Samarasinghe et al., 2013; ELD Initiative, 2015.

theory. In reality, the non-demand curve-based methods 'do not provide strict measures of economic values, which are based on peoples' willingness to pay 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 value of these ecosystems or services' (Ecosystem Valuation, 2000). Effects on productivity methods are particularly relevant to the assessment of direct benefits use (mainly related to provisioning services). An example is the extra yield obtained as a result of the soil fertility value at adjusted market price. Averting and mitigation behaviour methods are particularly relevant to the assessment of indirect use benefits (mainly related to regulating services). Examples are the costs of avoiding soil compaction in construction, or the cost of building flood defences to mitigate increased run-off from agricultural soil.

Demand curve-based methods, on the other hand, produce measures of welfare, such as willingness to pay — either to secure a benefit or to prevent a loss — or willingness to accept compensation — either to forgo a benefit or to tolerate a loss. The demand curve-based methods use revealed (derived from actual or surrogate markets) or stated preferences (derived from hypothetical markets) to observe changes in ecosystem services (Samarasinghe et al., 2013). The methods that rely on revealed preferences are particularly relevant to the assessment of indirect use benefits (mainly related to regulating, supporting and cultural services). Such methods can be used when preferences are elicited through park and nature conservation site visits (travel cost method) or appreciation of aesthetics reflected in house prices (hedonic pricing), for example. Stated preferences express willingness to pay for a particular good or service. Such methods are particularly relevant for indirect use, option and non-use benefits (mainly related to cultural services). Expressed willingness to pay for soil conservation programmes would be a relevant example here.

In contrast to predominantly quantitative methods, deliberative participatory methods elicit preferences and non-monetary values through discourse and exchange among various stakeholder groups (HMT, 2007). Applied to land use, they also help to understand the processes of decision-making and governance, using a mixture of numbers, narratives and metaphors, to support individuals and groups as they seek to determine and achieve desirable and socially just outcomes. Consistent with the principles of sustainable land use, deliberative approaches treat land (and by implication soil) in its entirety, operating at the whole landscape scale with citizens exercising both private and collective rights on the use of land to achieve overall social and cultural well-being. Their use assumes a participatory democracy in which land use decisions are based on agreed, typically more local, criteria and preferences rather than predominantly economic ones (Marshall, 2005). While deliberative methods have increasingly been used to inform choices on land use, especially involving scenario analysis, the link to soil quality and management is mainly one of association (rather than a cause–effect relationship).

The expense of carrying out surveys to elicit values for the wide range of services delivered by land has encouraged the use of benefit transfer methods, whereby values derived from completed studies are 'transferred' for use elsewhere (Bateman et al., 2009). However, given that elicited values are often strongly place-based (i.e. related to a specific context or site), applying benefit transfer is less straightforward than it may at first seem.

How the different methods are linked to the concept of total economic value is illustrated in Figure 3.1. All valuations, and indeed the decisions that follow, are liable to bias of one form or another, suggesting the need for quality control (Söderqvist and Soutukorva, 2009).

3.1.4 Where do soils and their use and management make a difference in generating benefits to society?

Soil stocks, described in terms of their *quantities* and *qualities*, as well as their contextual attributes such as altitude, topography, climate, hydrology and location, have the potential to support a range of benefits to people via ecosystem services. The use of soil in the pursuit of benefits can, however, lead to its degradation, resulting in loss of benefits or increased costs.

Drawing on the sources reviewed in Section 1.2.1 and adopting the frameworks commonly used in ecosystems assessment, Table 3.2 outlines a generic classification for soil-related ecosystem services under the headings of supporting, provisioning, regulating and cultural services, together with examples of services, benefits to people and the particular contribution of the soil resource, with respect to both their quantity and quality. It also provides examples of approaches to the valuation of soil-related services (see Sections 1.2.3 and 3.1.3). Changes in the quantity and quality of the soil resource (i.e. change in stocks of soil) have consequences for flows of services and benefits. Table 3.2 thus provides a first indication of how the analytical question (Section 3.1) 'where and how do soils and their use and management make a difference in delivering services?' can be answered.

Table 3.2 Generic classification and valuation of ecosystem services with particular reference to soils

Ecosystem service	Examples of services	Contribution of the soil resource, whereby a change in soil condition affects services and benefits	Approaches to monetary valuation of soil-related services, usually as a subset of the valuation of final goods and services
Supporting other processes and services	Soil formation Nutrient cycling Water and gas cycling Habitats/refuge Biodiversity	Context and facilitator of natural processes, including support to (above- and below-ground) biodiversity (habitats, species, genes)	Not directly valued
Provisioning of material goods and services	Biomass production (food, feed, fibre, fuel) Mineral and organic material extraction Genetic resources Water supplies Platform/carrier ([a])	Effect on land use suitability/ capability/productivity in agriculture and forestry Water-holding capacity Water quality Provision of usable space for development: with load-bearing capability, stability, resilience	Yields effects valued at market prices of agricultural and forestry commodities Soil quality effects on farmland values Cost of substitution of soil contribution to production, water supply, carrying capacity Costs of avoiding losses/risks associated with loss of soil services (e.g. insurance, additional engineering costs)
Regulating natural/ ecosystem processes	Air quality Flood/drought control Erosion control Carbon storage Water purification Temperature regulation Waste assimilation Pollution attenuation Pest/disease control	Control of run-off, drainage, erosion, diffuse pollution and sedimentation Drought management Carbon sequestration Water filtration, storage and nutrient recycling Temperature management Waste assimilation (sewage sludge/composts)	Avoided urban flood damage/ flood defence costs Economic value of carbon storage Savings in water supply regulation and treatment costs Alternative waste disposal or treatment costs Avoidance of erosion, pollution, sediment transport and deposition costs Property prices and asset values
Cultural, provision of non-material services and goods	Heritage Landscape Amenity Recreation Social relations Interaction with nature	Preservation of archaeological artefacts Supporting landscapes, habitats and biodiversity (e.g. grasslands, peatlands) Supporting recreation, countryside access	Cost of substituting or maintaining soil services that underpin provision of cultural services Proportion of willingness to pay that is attributable to maintenance/protection of soil quality (e.g. in habitat creation and maintenance, heritage preservation, green space, countryside recreation)

Note: ([a]) Carrier functions, such as the (non-soil-engaging) spatial and activity load-bearing capacity of soil are important especially for human settlement and associated infrastructure (e.g. for underground distribution networks: drinking water, waste water, district heating, power, fibre optics, etc.). Carrier functions that provide material benefit (rather than non-material cultural benefits) are included as provisioning benefits here.

3.1.5 *What are the soil degradation costs and how are they distributed?*

Much of the economic analysis of soils is concerned with soil conservation and avoiding the costs of degradation. Typically, a distinction is drawn between *on-site* and *off-site* costs and between *internal or private* and *external or public* costs (Table 3.3).

On-site soil costs and benefits accrue at the location of the degradation processes. They are mainly (but not exclusively) borne by those causing degradation and, as such, are mainly 'private' costs to the individuals or organisation that had entitlement to use soils as part of their land property rights. Off-site costs accrue elsewhere and are borne by third parties, usually without compensation, and are therefore mainly public/societal costs. Some on-site costs may be public costs that accrue to other users, such as walkers in the countryside (with limited and subsidiary entitlement to use land for access), whose non-market benefits are reduced without compensation (assuming they cannot go elsewhere to derive the same net benefit). The majority of on-site private costs can be valued using market *prices*. The valuation of off-site, public costs may involve a mix of market and non-market prices. It is noted, however, that on-site costs may include some non-market costs, such as the stress or loss of amenity or reputation caused by a degradation process, such as a serious soil erosion or pollution incident on private land.

From an economic viewpoint, the cost of soil degradation and loss of soil quality is the sum of on-site and off-site costs (with some adjustments to market prices to remove the effect of taxes and subsidies where they apply).

Calculating the costs and benefits and analysing whether they accrue to private actors or the society at large (public) provides an indication of how the analytical question (Section 3.1) 'what are the (degradation) costs and benefits and how are they distributed?' can be answered.

Table 3.3 Generic classification of costs of soil degradation and the basis for valuation ([a])

Spatial extent	Ecosystems perspective	Economic perspective	Basis for valuation
On site: private	Loss of provisioning services (e.g. yield loss, input substitution in agricultural production) Loss of on-site regulation and cultural services owing to erosion, contamination, on-site flooding Loss of private property values	Mainly 'private' costs borne by individuals and organisations Some non-market costs borne by site users	Mainly market prices to reflect changes in benefits and costs to users, including loss of property values (e.g. from reduced rental income)
On site: public	Loss of site specific cultural services (e.g. recreational, habitat) Reduced carbon sequestration	Losses and costs mainly associated with loss of site 'user' benefits, or loss of specific soil quality attributes affecting welfare, such as valued habitat or strategically important agricultural soils	Non-market prices for uncompensated site users Development/usage tax and/or offset costs Willingness to pay for non-user benefits (e.g. owing to lack of access) Option value for strategic assets
Off site: public	Loss of regulation services (e.g. flood control, greenhouse gas regulation, water quality) Off-site displacement effects: relocation of displaced activities and burdens	Public costs borne by society at large: damage costs incurred Mitigation costs/defensive expenditure Mitigation costs and/or off-site defensive expenditure (e.g. sediment traps, additional flood defences)	Combination of market and non-market prices Non-monetary (intangible) impacts
Total	Combined value of provisioning, regulating and cultural services	Combined value of private and public costs	Mix of market prices (adjusted for taxes and subsidies) and non-market prices Non-monetary (intangible) impacts

Note: ([a]) Actions to achieve soil enhancement or offset degradation are designed to have the opposite effects, namely to mitigate loss of services and associated value.

3.2 Site-specific valuation of soil in European urbanised areas

This chapter looks at cases in the literature in which CBA (see 'Cost–benefit analysis' in Section 3.1.1) has been applied to urbanised areas.

3.2.1 Soil degradation costs in the European Union

A comprehensive review of the economic costs of soil degradation was carried out by Gorlach et al. (2004) for the European Commission. The study included the eight processes of degradation referred to in the Thematic Strategy for Soil Protection (EC, 2006). It identified over 60 studies that quantified the economic impact of soil degradation, the majority from Australia and North America. Generally, the review concluded that there is limited evidence from Europe (the current report indicates that this is still largely the case). There was limited reference to soils in urbanised areas, with a strong focus on agricultural land, including on erosion-induced yield losses and the consequent on-site costs. Relatively few data were related to urban land use, and these were mainly confined to estimating the remediation costs of contaminated land. Erosion was the most widely covered degradation process including impacts on flooding and sedimentation. The review concluded that the effect of soil degradation on indirect use values and especially on ecosystem services had not received attention.

For the purpose of cost estimation, Gorlach et al. (2004) classify *costs* according to damage suffered and the costs of mitigation, simultaneously distinguishing between private on-site and social off-site costs. They used this approach to derive estimates of the costs of erosion, contamination and salinisation from a meta-analysis of available evidence.

Soil erosion

For erosion (Table 3.4), a central estimate for average annual costs in Europe was EUR2003 122/ha, of which 9% was on-site costs and 91% was off-site costs. This estimate was used to estimate the annual total costs of soil erosion in Europe at EUR2003 9 496 million.

Soil contamination

For soil contamination, mainly drawing on a case study (Box 3.4) extrapolated up to the regional scale, the total annual cost of soil contamination was estimated at EUR2003 24.9 billion, mainly represented by damage and remediation costs (Table 3.5).

Table 3.4 Estimates of the annual cost of soil erosion in Europe (EUR/ha)

	On-site costs		Off-site costs		Total
	Production losses/ damage	Mitigation costs	Damage costs (a)	Mitigation	
Upper estimate	11	29	169	26	235
Central estimate	8	3	86	26	122
Lower estimate	0.50	0	21	0	22

Note: ([a]) Includes damage to surface waters (loss of fisheries, siltation, nutrient enrichment, etc.) from sediment transfer.

Source: Gorlach et al., 2004.

Table 3.5 Estimated total costs of soil contamination in Europe (million EUR2003)

Estimate	On-site costs		Off-site costs		Total
	Impact monitoring costs	Mitigation/clean up	Damage costs	Mitigation	
High	96	2 187	1 183	482	5 049
Central	192	6 658	17 126	965	24 941 (GBP$_{2010}$ 20 billion)
Low	289	41 234	207 615	1 447	250 585
	0.8%	26.7%	68.7%	3.9%	100.0%

Source: Gorlach et al., 2004.

Box 3.4 Case study: cost of soil contamination in a semi-urban area in France

Case studies were used to demonstrate particular degradation processes, including one relating to an urbanised area. The costs of contamination were estimated for a 'semi-urban' area in France (105 persons/km^2) that had been highly modified by industrial activity, containing decommissioned industrial plants and significant agricultural activity. The focus was on local soil contamination, located on the industrial plant site (c. 30 ha) and in the surroundings. This industrial activity affected soils on site and in the vicinity, through emissions to the atmosphere and water. It has also had significant socio-economic impacts. Table 3.6 shows the breakdown of an estimated total annual cost of EUR_{2003} 5.7 million. The total costs of the off-site measures outweigh on-site costs by a factor of five. Off-site social costs account for about 80 % of total costs. Gorlach et al. (2004) point out that the costs do not include those for any remediation completed before plant closure. They also point out that local authorities and public agencies met the bulk of expenditures associated with off-site costs. A private investor intended to remediate this site with the help of public subsidies.

Table 3.6 Estimated annual equivalent costs of contamination of soil (on and off site) due to industrial activity on 30 ha on a semi-urban industrial area in France

Private costs	Mitigation costs ([a])	Off-site social damage costs	Off-site damage avoidance cost	Cost of the loss of non-use values
Reclamation of the site within the redevelopment project, performed by private investor Monitoring impact	Demolition of contaminated buildings Soil decontamination treatment Acquisition of contaminated land and re-fitting forests	Human health impact (costs of disease, lost work days, etc.) Agricultural impact (lost income) Urban impact (decrease in housing prices)	Hydraulic pumping in the aquifer to limit propagation of the plume Survey of groundwater quality Decontamination of school yards	Loss of non-use value for citizens
Included in mitigation costs	EUR_{2003} 947 800/year	EUR_{2003} 4 429 647/year	EUR_{2003} 312 400/year	Not estimated

Note: ([a]) Mitigation costs may be met from private or public funds or both.

Source: Gorlach et al., 2004.

Box 3.5 Case study: costs and benefits of soil remediation in the Netherlands

The economic case for remediation of local contamination at brownfield sites is not clear at the national level; a cost–benefit study for the Netherlands (Van Wezel et al., 2008) found that any balances were within the range of uncertainties so that there was no clear margin of benefit value over costs. If, however, remediation options are assessed for individual sites, it is expected that a net benefit can be shown for many sites on a site-specific basis, depending on the nature of these sites, the contamination present and the receptors at risk from the contamination. This is also illustrated in the case study of northern France (Box 3.4).

Overall, the study by Gorlach et al. (2004) of soil degradation in the EU concluded that the private on-site costs of soil degradation are 'not a major cause of concern in most cases', meaning that generally the financial cost to land owners/users of causing soil degradation are small relative to other costs incurred from soil use and the profit from its use. Overall, off-site costs typically exceed on-site costs by more than a factor of seven, except for contamination where on-site costs may account for a greater share of total costs. The study notes that the majority of the costs of soil degradation are not felt by those causing them, but rather by those in other locations, who bear the consequences without compensation. It also notes that off-site costs are subject to much greater uncertainty in their estimation than on-site costs.

3.2.2 Soil benefits and degradation costs in England and Wales (United Kingdom)

Set in the context of formulating a strategy to safeguard future soils in England (Defra, 2009b), ADAS (2006) reviewed evidence for the 'monetary' *benefits* of soils services, broadly but not explicitly set in an ecosystem framework. Focus was placed on (1) carbon storage and sequestration, (2) water storage and flow mediation, (3) nutrient cycling and crop production, (4) supporting construction, (5) natural attenuation of pollution and contamination, (6) archaeological and landscape heritage protection and (7) support of ecological habitat and biodiversity. Most attention was given to the management of soils in the farmed landscape, while recognising that the costs are borne by adjacent urban communities and society at large. In some cases, soil benefits were framed in terms of losses owing to soil degradation. Examples of benefits and costs are given where available. For example, for arable and/or horticultural land uses, estimated carbon storage losses ranged from about GBP 100/ha/year on upland peats to over GBP 800/ha/year (in 2006 prices) (GBP 0.68 = EUR 1 in 2006). The costs of deviating from best practices on farmland, which are largely associated with soil erosion, compaction and surface run-off of contaminated water, were found to range between GBP 20/ha/year and well over GBP 300/ha/year, depending on context and type of degradation. ADAS (2006) concluded that urbanisation greatly alters the ability of the soil to provide economic benefits in terms of water storage and flood abatement, attenuation of pollutants and contaminants, and support of biodiversity, but the monetary values remain unknown. They concluded that knowledge and data to confidently predict soil-related economic costs and benefits arising during urbanisation at a local or national level have yet to be drawn together.

Picking up on this challenge, and focusing on the processes of degradation at the national scale for England and Wales, Graves et al. (2012) used the ecosystems framework to assess the impact of changes in soil quality on soil-related supporting services, provisioning, regulating and cultural services. For this purpose, combinations of land use and soil type defined 'soilscapes', each with a particular vulnerability to soil degradation. In the England and Wales context, degradation focused on erosion, compaction, loss of organic matter, decline of soil biodiversity, diffuse contamination and soil sealing. Spatially specific land use/cover data (urban, agriculture (arable and grassland), forestry and woodland, 'wildscape') and soils data (clay, silt, sand, organic matter content) were combined to estimate the probability and intensity of different types of degradation for given 'soilscapes' and the consequences for ecosystem services and 'final goods' using a range of valuation methods. Key indicators of *soil condition and changes* were identified. Costing algorithms were constructed for each degradation process and applied across the various soilscapes to estimate the average cost per hectare at risk and total economic cost (Table 3.7). Mitigation costs may be met from private or public funds or both.

Table 3.7 Estimated annual economic costs of soil degradation in England and Wales by degradation process and soil ecosystem services

GBP million per year, 2010	Ecosystem service								
Degradation process	**Provisioning**	**Regulating**				**Cultural**	**Total**		
	Agricultural produce	**Flooding**	**Water quality**	**Greenhouse gases**	**Other**		**Range**	**Central estimate**	**%**
Erosion	30–50	46–80	55–62	8–10	–	?	139–187	165	13
Compaction	180–220	120–200	60–80	30–40	–	?	390–540	481	39
Soil organic carbon content	2	?	?	360–700	–	?	362–702	558	45
Diffuse contamination	?	?	?	?	25([a])	?	25	25	2
Soil biota loss	?	?	?	?	–	?			
Sealing	?	?	?	?	–	?			
Total	**212–270**	**166–280**	**115–142**	**398–750**	**25**	**?**	**916–1454**	**1129**	
%	20 %	19 %	11 %	49 %	2 %				100

Note: ([a]) Cost of regulation to protect soils from contamination.

? Estimates not available at national scale.

Source: Graves et al., 2012.

The estimated total annual cost of soil degradation in England and Wales ranged between GBP 900 million and GBP 1 400 million, with a central estimate of GBP 1 200 million (GBP 0.85 = EUR 1 in 2010). Of the total quantified annual soil degradation costs, about 45 % were associated with loss of organic content of soils, 39 % with compaction and 13 % with erosion, indicating the *main processes of impact*.

Of the estimated annual costs of soil degradation, 20% were associated with a loss of provisioning services linked with agricultural production, both reduced output and increased costs. The remaining 80% of total annual degradation costs were associated with a loss of regulating services, of which about 30% were linked to flooding and water quality and almost 50% were linked to greenhouse gas emissions.

While the larger rural area, compared with that covered by urbanised areas, means that the bulk of degradation may occur in the former, significant impacts of degradation are expected from agricultural and forestry activities in *urbanised* and particularly *peri-urban areas*, where the proximity of residential, commercial, industrial, transport and other economic activities increases the risk that degradation and consequent reduction in ecosystem services will cause harm to water quality and increase run-off and sedimentation.

Selecting particular elements of concern for urbanised areas, over 33% of total *erosion* costs were associated with the removal of sediment from rivers and drainage systems of which the bulk (30% of total) related to clearance of urban drainage channels.

With respect to soil compaction, estimated costs of *flood damage*, mainly to urban property owing to run-off-induced flooding, accounted for 35% of all compaction costs and 65 % of all off-site costs. A catchment flood management tool (Hess et al., 2010) was used to assess the impact of changes in soil condition on relative changes in the mean depth of run-off from land for given rainfall return period events (Hess et al., 2010). This showed that soil degradation, particularly associated with compaction, could be responsible for an increase in the depth of run-off of between 3 % and 10 % for a 75-year event across a range of soilscapes. Currently, the estimated total annual costs of flood damage in England and Wales are about GBP 1 400 million, with a further GBP 1 000 million spent on flood risk management: an annual total of GBP 2 400 million (Pitt, 2008; Environment Agency, 2009a, 2009b). Currently about 550 000 households are at serious risk of flooding (annual probability of flooding above 1.3 %) and a further five million properties exposed to moderate to low probability of flooding (annual probability between 0.5 % and 1.3 %).

A recent study states that 'data [in the EU-27] that establish a direct link between the changes in soil-water-retention capacity and its impact on water-related services in different sectors (e.g. agriculture, water distribution, industries using water in their production process, and tourism) are scarce, scattered and site-specific, which makes the estimation of economic impacts very challenging' (BIO Intelligence Service, 2014, p. 121). However, considering England and Wales as an example, and assuming simplistically that changes in soil condition affect run-off for all return period rainfall events, and are associated with an equivalent change in the probability of flooding at the catchment scale, soil degradation could be responsible for between GBP 72 million (3 %) and GBP 240 million (10 %) of flood damage and risk management costs per year, with a central estimate of GBP 168 million/year (based on a 7 % increase in flood risk). It is noted that the approach adopted here specifically attributes changes in flooding to changes in soil condition (for given land uses). Some agricultural land uses are known to be associated with more rapid run-off than others, as referred to earlier. The estimate of GBP 168 million is similar to the estimate by the Environment Agency (2002) that associated about 14% of flood events and direct annual flood damage costs (GBP 1 400 million) with run-off from farmland, producing an estimate of annual cost from agriculture of about GBP 200 million per year (and GBP 336 million if all GBP 2 400 million flood risk management costs are assumed). The costs of flood damage will be mainly generated in urbanised areas, although the proportion is not known. Furthermore, these costs will in part be generated by inappropriate soil management.

Sealing of soil surfaces is of particular concern in the urban environment (Wood et al., 2005; Scalenghe and Ajmone-Marsan, 2009), curtailing services that otherwise would be provided, especially but not exclusively linked to hydrological regulation. About 11 % of the surface area in England and Wales is classed as 'developed' land, characterised by a built environment (Foresight Land Use Futures Project, 2010). Of this, about 3 % is actually built on, with the rest being mainly gardens and open spaces. While recent development has mainly occurred in pre-used 'brownfield' areas, there has, however, been a tendency towards increased sealing within existing urbanised areas, especially for parking areas.

It is not possible to determine the proportion of the current flood damage costs and flood risk management costs that could be saved by reducing the extent of soil sealing. However, the Environment Agency (2007) estimated that two-thirds of the 55 000 homes and 8 000 businesses affected by the flooding of summer 2007 were flooded because drains, culverts, sewers and ditches were overwhelmed, equivalent to insurance claims of about GBP 2 400 million. A proportion of this could be attributed to excessive surface sealing, which renders the drainage system inadequate during extreme events.

There is currently insufficient knowledge to assess the potential significant economic impacts of soil sealing at the local or national scale. This is a valid area for research, especially regarding the hydrological functions of soils, sustainable urban drainage systems (Box 3.6) and development regulation, and in the context of predicted climate change.

The preceding assessment confirmed the difficulty, indicated by earlier reviews and analyses, of deriving complete and reliable estimates of the benefits provided by soil and how these change according to soil condition. Three main areas of uncertainty arise: (1) 'identifying' biophysical relationships between soil properties, soil functions and the 'performance' of soil in particular applications, (2) 'valuing' the diverse range of market and non-market benefits and costs attributable to soil in different applications and (3) assessing the 'dynamics' of changes in soil properties, as these affect changes in the value of services, especially under conditions of climate change. There is particular uncertainty about the costs of urban soil sealing. Moreover, where degradation occurs in farmed land, many of the impacts are felt in urbanised areas.

3.2.3 Carbon-rich soils under climate change

Some soil services can almost be perceived as final goods, such that valuation is more straightforward. This is the case for lowland peatlands, which are present in some urbanised areas in Europe (e.g. Cambridgeshire, United Kingdom; the Netherlands). Here, intensive agriculture has led to widespread degradation of peat soil, resulting in carbon loss and eventually reduced

Box 3.6 Soil-supported urban drainage

Sustainable urban drainage systems (SUDS) avoid soil sealing and allow continuing use of the water management services provided by soil that lower flood risk and aid groundwater recharge.

Case: sustainable urban drainage, Cambridge (United Kingdom)

Photo 3.1 Sustainable urban drainage systems in Cambridge (United Kingdom)

© Royal HaskoningDHV (http://www.susdrain.org/case-studies/case_studies/lamb_drove_residential_suds_scheme_cambourne.html)

The Lamb Drove SUDS Residential Scheme mimics the natural drainage processes of soil surfaces before development. Roof waters drain to water butts for garden use, and excess water from roofs, roads and paths runs to permeable paving overlying underground crushed-stone soakaways. These drain to swales, detention basins and wetlands/ponds. These measures also contribute to the provision of green space, visual amenity and wildlife on the site. Capital costs were marginally lower than conventional drainage. Subsequent monitoring also revealed significant water quality and habitat benefits.

Source: Cambridgeshire County Council, 2012.

agricultural productivity. At the same time, there is a call to restore peatlands to wetland habitats, providing a range of services to adjacent urban communities. In this context, soil is a defining attribute of land. Comparing continued agricultural use with wetland restoration, Graves and Morris (2013) derived steady-state estimates of the annual net benefits for 2012 of about GBP 150/ha (GBP 0.81 = EUR 1 in 2012), rising to between GBP 330/ha and over GBP 1 000/ha in 2080, depending on the climate change scenario at current agricultural commodity prices and carbon prices based on the cost of abatement. Switching from arable farming to peatland restoration and extensive wet grassland gave an estimated present value benefit of about GBP 40 000/ha over the period to 2080, applying the social discount rate and allowing for real increases in both carbon and agricultural prices. This excludes estimates of non-carbon wetland benefits, such as biodiversity and flood risk management that were estimated at GBP 304/ha and GBP 407/ha, respectively, for United Kingdom inland freshwater wetlands (Morris and Camino, 2011). It also excludes the option value of conserving rather than degrading an agricultural asset for future (food security) needs if required (valued at an extra GBP 4 000 to GBP 11 000/ha).

3.2.4 *Soil-related benefits of green infrastructure projects*

The soil-related services of green infrastructure (GI) are outlined in Box 2.4.

Naumann et al. (2011) reviewed design, implementation and cost elements of GI projects. In their understanding, GI comprises the network of semi-natural and artificial green spaces in rural, urbanised and coastal areas. They found that *benefits* were described qualitatively in 77 projects, while 31 lacked information about benefits and only 19 (15 % of projects) provided any quantitative evidence of benefits. It is clear from six of the exemplar projects that soil features strongly as a general environmental component to which ecosystem services are attributed, such as products from the land, carbon sequestration and storage, provision of habitats and green space. There remains, however, no formal assessment of the specific contribution of soils in economic terms, even though soil protection features as a key intervention in two cases, including in a project in the Gallecs region (Spain) (Naumann et al., 2011).

> The objective of the project was to protect the region 'Gallecs' from urban and industrial pressures and subsequent environmental degradation. The aim was to strengthen the area's function as a 'biological interface', i.e. a buffer zone between the urban fringe and the countryside beyond. Improved environmental conditions were to result in a higher quality of life for the inhabitants of the areas on the outskirts of Barcelona. An integrated approach was developed with a view to achieving sustainable land-use in the area. The strategic plan comprised a series of actions to control and manage urban sprawl, as well as to mitigate its detrimental impact on the environment. Activities included initiatives in the following areas: the restoration of natural habitats, sustainable agricultural and forest management, the use of renewable energy, and environmental education. (Naumann et al., 2011, p. 11)

The European Commission's (EC, 2012c) review of multi-functionality of GI considers changes in the socio-economic value of the ecosystem services provided, some of which relate specifically to soils (e.g. value of carbon storage, value of reductions in property damage associated with flooding in the absence of water retention measures, and value of semi-natural habitats). While providing a useful review of the benefits of final goods, it appears there is little evidence to support quantitative assessment of the value of soil properties and differences therein.

From the above cases, it appears that the role of soil remains under-identified, while the benefits associated with the provision of GI are identified and measured in broad terms.

3.3 Regional economic impact assessment of soil-related services

There have been a limited number of studies on the economic impacts of changes in soil quality in urbanised areas. More recently, however, increasing reference, albeit often implicit, has been made to soils with respect to GI (see also sections 2.2 and 2.4.2), much of which is set in an urban context.

3.3.1 *The contribution of green infrastructure to economic growth*

The appraisal of GI commonly applies economic impact assessment to justify the use of public funds and to attract external regional development funds. There is considerable interest in investing in nature as a means of promoting economic development through direct provision of GI and related services and the boost it provides to other sectors of the regional economy (EBRD, 2011; EC, 2012b). Reviews have confirmed the important links between the natural environment and-macro-economy (Reveill et al., 2012; Gore et al., 2013).

Photo 3.2 Cemeteries as part of green infrastructure

© Geertrui Louwagie

Gore et al. (2013) explore the extent to which GI (as a dependent variable) affects economic growth, reviewing evidence from the literature and using five international case studies to provide context. GI is taken to mean 'a planned approach to the delivery of nature in the city in order to provide benefits to residents', namely street trees, roadside verges, pathways, gardens, green roofs, allotments, woodlands and community forests, parks, cemeteries, rivers, canals and wetlands (Photo 3.2). Economic growth is defined as a change in GDP, conventionally measured in terms of gross value added (value of final goods after accounting for costs, incomes and expenditures) and employment.

Using an ecosystems framework, Gore et al. (2013) developed six logic chains or pathways to examine the link between GI and economic impact. As they explain, the chains are illustrative rather than exhaustive of the complex relationships involved. They point out that the chains do not operate independently, but are strongly interactive. They used the logic chains to ascertain evidence from research and practitioner literature on the impacts of GI both on its own and alongside other development or renewal activities in the urban space. They considered the impacts of GI within the predominantly urban zone and in the adjoining peri-urban areas. They associate GI with *economic impact* through inward investment (including increased property valuations), visitor spending, environment management benefits or cost savings, health improvement, market sales and employment generation (Table 3.8). These pathways are in turn associated with selected ecosystem service flows (classified into provisioning, regulating and cultural) that subsequently impact on measures of economic activity (e.g. premises occupation, business start-ups), gross value added and employment.

The review points to issues with attributing benefits to GI, independent of other development interventions, and confirms that GI effects are additional to what would occur anyway and do not merely displace economic activity and growth elsewhere. It is also clear that GI-related benefits are particularly context and scale dependent, such that they are difficult to generalise. Rather than creating new economic growth, some effects of GI may be confined to displacing economic activity and value added that would otherwise occur elsewhere in the economy. The environmental management savings and health benefits are probably the main source of net gain at the national economy level. Other impacts may involve a large degree of displacement within a country.

Soils do not feature explicitly in Gore et al. (2013)'s review of GI. While it is clear that some soil-based activities, such as restoration and land works, were carried out, the particular contributions of soil per se are not distinguished. For example, the costs of land works are identified as part of investment costs. There is, however, considerable reference to soil- and land-based services such as flood risk management, carbon sequestration, habitats and biodiversity. For the purpose of eliciting a connection between soil and economic impact, Table 3.8 extrapolates from the Gore et al. (2013) review to consider how and to what extent

Table 3.8 The role of soil (within land use and management) in supporting the link between green infrastructure, ecosystem services and economic development

Logic chain link between GI and economy	Ecosystem services provided by GI	Major soil function associated with GI	Degree of influence of soils on GI services (high, moderate, low) and assessment of economic impact
Inward investment	Cultural services: • place making • increased appeal ([a]) • access to open space • increased asset value • links to built heritage (canals, docklands, mines)	Soil as a platform/ location for open green space, infrastructure and services	**Low** Minor impact on total construction costs of GI and associated development, except where soil properties are not favourable (e.g. presence of expansive clay minerals leading to damage to structures and buried infrastructure) Low impacts on asset values (although subsidence and contamination may affect development options and costs) Soil services can be engineered or substituted, albeit at a cost Soil imports to parks, gardens and recreational centres, although at a cost and with displacement elsewhere
Visitor spending	Increased attractiveness Increased spending Links to built heritage (canals, docklands, mines)	Soil as a platform/ location for green space, supporting biodiversity/habitats/ landscapes Urban parks and gardens	**Low**, depending on context Soil is important for specialist habitat/ biodiversity provision (e.g. urban wetlands, nature conservation, areas of semi-natural grassland) Soil is vulnerable to visitor pressure, requiring expenditure on protection (e.g. from compaction)
Environmental management benefits or cost savings ([b])	Regulation: • improved air quality (trapping of pollution, dust) • alleviation of heat stress/urban heat island effects via evaporative cooling and via support for plants, transpiration • control of run-off with reduction of flood risks ([c]) • waste management • carbon sequestration • urban habitat targets • reduced pressure at meso- to macro-scale	Soil support to nutrient cycling, surface and groundwater quality, urban surface and groundwater supply/ drainage, flood control, protection of urban buildings and archaeology	**Moderate to high** for selected services (depending on intensity, location and scale) Benefits of increased provision of services or savings in costs of alternative equivalent provision (e.g. savings in flood defence expenditure) High where soil-related services provide multiple benefits associated with hazard reduction and service gain, urban waste removal and restoration/ delivery or nature conservation targets Estimates based on willingness to pay indicate welfare benefits rather than economic impacts Reduced pressures on urban edge owing to proximity and containment of GI
Health Improvement	Improved physical and mental health Reduced health costs Improved work productivity Increased security and cohesion	Soil as a platform/ location for biodiversity/ habitats/landscapes Public rights of way: recreation and transport Private urban food production (citizens and firms) Public/private parks and gardens	**Low** Soil affects suitability and accessibility for particular outdoor activities Soil limitations overcome by engineered responses (e.g. modified or artificial surfaces to substitute natural ones) Estimates based on willingness to pay indicate welfare benefits rather than economic impacts Removal of public health risks associated with contaminated lands

Table 3.8 The role of soil (within land use and management) in supporting the link between green infrastructure, ecosystem services and economic development (cont.)

Logic chain link between GI and economy	Ecosystem services provided by GI	Major soil function associated with GI	Degree of influence of soils on GI services (high, moderate, low) and assessment of economic impact
Market sales	Goods and services from GI (e.g. foods; local materials for artefact production, for example arts and crafts; sports/ recreational goods and services; educational and caring services) Employment and expenditure	Soil supporting production of produce: defining type and yields of local food and related produce, regional produce	**Low to moderate** Soil as an input into production functions, adding value through increased crop/ornamentals producing yields and quality Soil limitations overcome by substitution and/or additional cost, drainage/irrigation, soil improvers Employment and incomes generated through soil management required for production of marketed goods and services
Employment generation	Direct employment through creation, operation and maintenance Indirect employment through extra spending and associated multiplier effects	Soil influence on processes and cost of GI construction (especially for brownfield/ restoration sites) Employment related to regulation services, especially drainage Effect on surface and ground conditions affecting cultural services, especially recreational use	**Low to moderate** Moderate when restoration of brownfield sites involves contaminated soil Low to moderate for creation, operation and maintenance of natural surfaces for recreation, nature conservation, gardens including landscape, park and garden contractors and support services and suppliers

Note: ([a]) Increased appeal, attracting more residents, visitors and businesses, with additional spending on local goods and services.

([b]) Cost savings providing that GI investment is at a lower cost than alternative interventions, and as a result of reduced expenditure on additional repairs and maintenance owing to extreme events making resources available for other, more productive, purposes. The environmental management savings and health benefits are probably the main source of net gain at the national economy level. Other impacts may involve a large degree of displacement within a country.

([c]) To avoid floods in lower-lying urban areas and in river flood plains, fees have to be paid in some German federal states, at least if storm water does not infiltrate near buildings. Facilitating storm water infiltration near buildings helps to save on construction costs, as the diameter of storm water sewage pipes can be reduced, bringing economic benefits to the land owner.

Source: Developed from Gore et al., 2013.

soil and soil quality in urbanised areas might make a difference to GI and hence economic growth.

Just as the links between GI and the economy are shown to be mainly of association (rather than cause and effect), the same is true of soil attributes as a subset of GI. The most significant interactions between soil quality, GI-based ecosystem services and economic growth in the context of GI relate to:

- regulating services that provide environmental management savings, mainly flood risk management, carbon sequestration and remediation of contaminated soil;
- cultural services associated with access to and enjoyment of green spaces, and provision of soil-dependent habitats;
- provisioning of local produce from gardens and allotments.

While the overall efficacy of GI is not strictly dependent on soil quality, the latter can affect the type of GI development and service provision. However, where soil limits GI potential, depending on scale, actions can be taken to remove soil constraints by improvement or substitution; but, since these come at a cost, this will reduce overall gross value added.

3.4 Monetary valuation of soils: assessment of the knowledge base for soil governance

This section assesses what soil valuation can contribute to asset-based governance by taking stock of the existing knowledge and comments on/criticises its use for governance purposes.

The examples provided in Chapter 3 give only a fragmented view of the implementation of soil valuation to urbanised contexts, but illustrate the derivation of values for soil-related ecosystem services. Few examples attribute benefits to soil properties and soil condition directly. A number of studies, however, identify the costs associated with the degradation of services, mostly in the extended urbanised area. Many provide examples of the benefits or costs associated with land use and management practices known to be associated with differences in soil attributes. The methods involve a range of estimation techniques.

3.4.1 Methodological challenges

Uncertainty and thresholds

Environmental valuation of the soil component (e.g. Graves et al., 2012; see Sections 3.2.2 and 3.3) confirmed the difficulty, indicated by earlier reviews and analyses, of deriving complete and reliable estimates of the benefits provided by soil and how these change according to soil condition. Three main areas of uncertainty arise: (1) 'identifying' biophysical relationships between soil properties, soil functions and the 'performance' of soil in particular applications, (2) 'valuing' the diverse range of market and non-market benefits and costs attributable to soil in different applications and (3) assessing the 'dynamics' of changes in soil properties, as these affect changes in the value of services, especially under conditions of climate change. There is qualitative understanding of some of the roles of soil in support of cultural services (e.g. preservation of archaeological artefacts), but others are less well identified and defined (e.g. education, nature conservation) and in all cases their valuation is difficult.

Decisions affecting land systems, including on soil, must allow for the inherent complexity of ecosystems and uncertainty about their response to change. In some cases, even small changes may set in motion a chain of events that leads to a large and possibly irreversible change in outcomes — a threshold regime change effect (Limberg et al., 2002). The rapid decline of wetland habitats that previously appeared resilient (Scheffer and Carpenter, 2003) demonstrates this phenomenon. It is important that economic appraisal of land use and related soil management options explicitly consider uncertainty, and that guidance is available on how best to deal with this (HMT, 2011). The uncertainty associated with climate change heightens the need for estimating uncertainties in the assessment of soil resilience and stability to future change and the effect on asset values (Stern, 2006). Where uncertainty and the potential impacts of soil degradation are high, a precautionary approach may be justified to secure 'safe, minimum standards' of service and welfare. This is in line with Farley's (2012) viewpoint that, if ecosystem services are considered essential, 'marginal analysis and monetary valuation are inappropriate tools in the vicinity of ecological thresholds', considering that small changes can lead to enormous impacts under such conditions. Such considerations assume particular importance in urbanised areas, where there are manifestly large pressures on the soil resource.

Uncertainty about the possible future social and economic costs of soil loss and degradation has justified a precautionary 'target'- or objective-based approach to soil resource management in many countries. This includes, for example, targets or policy commitments on the avoidance of loss of prime agricultural soils, the curtailment of peat extraction and a 'zero/no net land take' for urban development.

Limitations to the approaches used

Although CBA (see 'Cost–benefit analysis' in Section 3.1.1) is the most appropriate tool for exploring the value of the services provided by soil, its application is limited by several difficulties. First, it is difficult to separate the essential attributes of natural soil that generate value. The value of soil is inexorably linked to land use: in some cases, soil makes a major contribution that is difficult to substitute; in other cases, it does not, and its contribution is readily substitutable albeit at a cost. Second, it is difficult to define the appropriate counterfactual for CBA, that is the situation that would prevail in the absence of interventions to preserve or protect soil, whether this be the type of land use (e.g. urban or agriculture) or the type of land management practices and the related extent of soil degradation within a given land use and under given land management practices. Third, while an ecosystems approach provides the best means for systematically identifying the particular contribution of soil to land-based services, it is imperfect and the contribution of soil cannot be completely separated from that of other land components, such as water and biodiversity, making apportionment uncertain. Fourth, the major service flows by which soil adds value to overall 'final' goods and services include some for which there are financial transactions (e.g. food production) but also many others that are not traded directly (e.g. regulating and cultural services, including control of flooding, contribution to heat management and carbon sequestration). In addition, soil has non-use values, representing the potential for future services, including the value of conserving it for future generations (legacy value) and intrinsic non-anthropogenic value (i.e. preservation of intrinsic soil properties and biota not associated with use, e.g. in pedo-stratigraphic records or markers), while the environmental valuation framework is based on human preference (related to use) revelation. Therefore, challenges exist in monetising costs and benefits, especially from cultural services and for indirect and potential but unused services, as well as in assigning non-use legacy and intrinsic values.

The methods used to value environmental stocks and flows of services (or lack thereof) have indeed many limitations, as elaborated in Smith et al. (2011):

- Data requirements are great, while data may in reality be scarce and collecting them is costly.

- Valuation also involves many assumptions concerning both physical relationships (e.g. the link between soil characteristics and flood regulation (Photo 2.2)) and the values (be it in physical or monetary terms) attributed to the goods and services derived from natural capital.

- As noted already, the issue of values becomes particularly critical when markets for the goods and services in question are non-existent, poorly developed or imperfect (e.g. the value of soil storing geological and archaeological heritage, allowing long-term information transfer).

- Some of the valuation techniques themselves are flawed, and may even be subject to inherent biases. Information bias in the contingent valuation approach is a common example: values attributed to natural assets will depend on the information provided by the researcher.

Finally, and perhaps most importantly, 'when ecosystems are near critical thresholds and ecosystem change is irreversible, money values do not help as regulatory mechanism' (Braat and de Groot, 2012, p. 12).

Smith et al. (2011) reflect that:

- Instead of looking at environmental problems in a narrow economic sense and using monetary valuation, it could be argued that the problems and solutions lie in the political arena, and this is where decisions concerning environmental issues are ultimately made. Changes to politics and institutions may be more important in changing decision-making priorities than the use of environmental valuation methods. (Smith et al., 2011, pp. 33–34).

- The models used in environmental economics to inform decision-makers are based on simplified assumptions. Critical use of the model results, for example by incorporating the complexity of the social and ecological context, should lead to better-informed policies overall.

3.4.2 Is the knowledge generated relevant to and sufficient for asset-based governance?

Adopting an asset-based and place-based approach to governance requires recognition of the fact that there are several other levels of governance that are relevant to the site in question. This multi-level aspect is also reflected in the different monetary valuation examples presented. Most importantly, there is a distinction to be made between the project- or site-level valuations on the one hand and the regional-level valuation on

Box 3.7 System of compensation measures

In Romania, the system of cross-selling accounts on the basis of eco-points can be considered a 'bank of ecological compensation'. Developers must demonstrate that their compensation measures are equal to the value of the soil functions lost, in accordance with the national law on nature conservation. If compensatory measure eco-points are not sufficient, they can be purchased at formally authorised agencies. The agencies are holders of eco-compensation accounts, sell eco-points and are responsible for compensatory measures.

Source: Ludlow et al., 2013.

the other hand. Nevertheless, both valuation levels can be successfully combined to a consistent valuation exercise, as equally illustrated in the GI application. Regardless of the level of valuation, the context-specific character of valuation results makes transferring values from one context to the other (using the benefit transfer method) problematic.

Where soil is a critical defining attribute of the benefits of land use, monetary valuation is challenging but possible. Where its contribution is more subtle and diffuse, monetary valuation is much more difficult or even inappropriate. The important point is to determine where soils and soil management make a difference, reflecting the essentialness of the soil service to the benefit concerned. At the same time, monetary soil valuation should not be disconnected from a broader assessment framework, such as ecosystem services or natural capital, including specific applications of that broad framework, such as GI or natural water retention. Nevertheless, the examples above show that soil is often not (sufficiently) considered in monetary valuation efforts of ecosystem services.

Where it is feasible to apply soil valuation through a market mechanism, this provides an opportunity to internalise costs that would otherwise be borne by society (as unintended side effects or externalities of a production or consumption process). As opposed to 'hidden cost is value lost' in the case of externalities, valuation in such a case means a reduction of the burden on society, and maintaining economic efficiency in resource allocation and management. Relying on a market mechanism to recognise the value of soils is especially important when soil protection through legislation or some other policy measure may not exist or may fail because of poor implementation.

In this report, we focus on soil as a productive asset, that is, we appreciate it primarily for its use value or how it can fulfil human needs. The most basic attribute of land (of which soil is a part) — location — gives land a unique character, but also severely restricts its substitutability as a marketable good or its exchange value (Alexander, 2014). This may reduce the risk of the soil resource being handled as a commodity, disconnected from the area where it is sourced. If compensation measures for nature protection and restoration are in addition based on soil functions and derived services, such risk is further reduced (Box 3.7). Despite its limited exchange value, the importance of location to the value of land also implies that land, and therefore soil, is prone to being regarded as a financial or investment asset. However, speculation in land-property markets can have severe social and economic consequences, as these demand public intervention, and thus explicit recognition of the use value of soil (Alexander, 2014). Whether current governance practice reflects the use or the financial value of soil will be explored in Chapter 4.

4 Implications for governance: towards soil resource efficiency in urbanised areas

Building on the above, this chapter explores the concept of governance as it applies to soil as an essential component of natural capital in urbanised areas. It considers the institutional arrangements that affect its use and management, and it explores possible policy instruments to protect soil in urbanised areas. It also evaluates whether or not the importance of soil is recognised, that is whether or not the value of soil is reflected in current governance practice.

4.1 Optimising the balance between demand and supply in urbanised areas

The number of ways in which soils are used in urbanised areas is much higher than in rural areas. In rural areas, principles and experiences of soil conservation (i.e. to protect, maintain and improve soils for use) have a long tradition. They predominantly aim to protect, maintain and improve soil fertility on farmland. Habitat protection for natural reserve development and establishing water protection zones for water abstraction have complemented these objectives.

In urbanised areas, the demand for soil-related services is much more diverse. This means that the package of soil conservation measures has to satisfy very different demands and should be adapted to the conditions of urbanised areas (Hoogveld et al., 2004). However, current policy targets and ways to achieve them (actions) are not in line with the peculiarities and amount of soil resource demand in urbanised areas, and are not consistent. This calls for special soil conservation measures or a particular soil conservation package in urbanised areas (Hoogveld et al., 2004).

Photo 4.1 Alternation of built-up and green space in the urban area

© Geertrui Louwagie

The borders of urbanised areas are often indeterminate and overlapping with those of rural areas. The core urban area, and peri-urban areas and zones at the rural edge form a whole system requiring integrated management, including of the soil resource. Commonly, however, the core urban area and its hinterlands fall within separate administrative and political structures leading to a lack of congruence of governance across different scales. Therefore, there is a requirement for existing political and administrative structures to work in tandem when designing and implementing policy on soil in urbanised areas.

Principles of sustainable soil governance that apply outside the urban fabric, such as for agriculture, natural habitats, woodlands, etc., also apply in urbanised areas. The characteristics of peri-urban areas as zones of transition place particular stresses on systems of soil governance, such that special measures might be required to avoid inefficient outcomes. The relative scarcity of accessible soil resources in core urban areas also indicates that special soil conservation measures may be required within them.

Soil use and management is, for the most part, subsumed within land use and management. There are two broad approaches to the governance of soils in urbanised areas:

- Soil is *considered a key determinant* of land use suitability and productivity, and governance concerns allocation of soils to the best, most efficient uses (e.g. the soil resource as a determinant of agricultural land grade and capability, or peat soil as a carbon store). The main focus is to avoid loss of essential, *non-substitutable* and hence highly valued soil services associated with agricultural soils and natural habitats.

- Soil is *not considered a major determinant* of land use suitability and productivity, and governance concerns the management of the soil resource, usually to achieve subsidiary outcomes or the avoidance of negative consequences associated with their inappropriate use (e.g. avoidance of sediment generation by soil erosion or excess surface water run-off to the urban fabric due to soil sealing).

As elaborated on in Chapter 2, soils in urbanised areas are highly affected by human activities and are also subject to their spatial and temporal variability. In addition, the many and diverse services that can be derived from the soil resource are in high demand in the urbanised space. Particularly in the urban fringe — the transition zone between the core urban and rural zones — competition for the multi-functional soil resource is likely. The diverse range of services that soil contributes to, however, means that the benefits and costs of soil use and management are often widely distributed across the geophysical landscape, affecting a wide range of stakeholders in a variety of ways.

Recalling the three analytical questions that were introduced in Section 1.2.3 and building on the answer to the first question 'where/how do soils and their use and management make a difference in delivering services (i.e. what is the marginal value of the soil resource in delivering services)?', which was elaborated on in Chapter 3, this section attempts to provide partial answers to the second and third questions:

- What are the (degradation) costs and benefits, and how are they distributed?

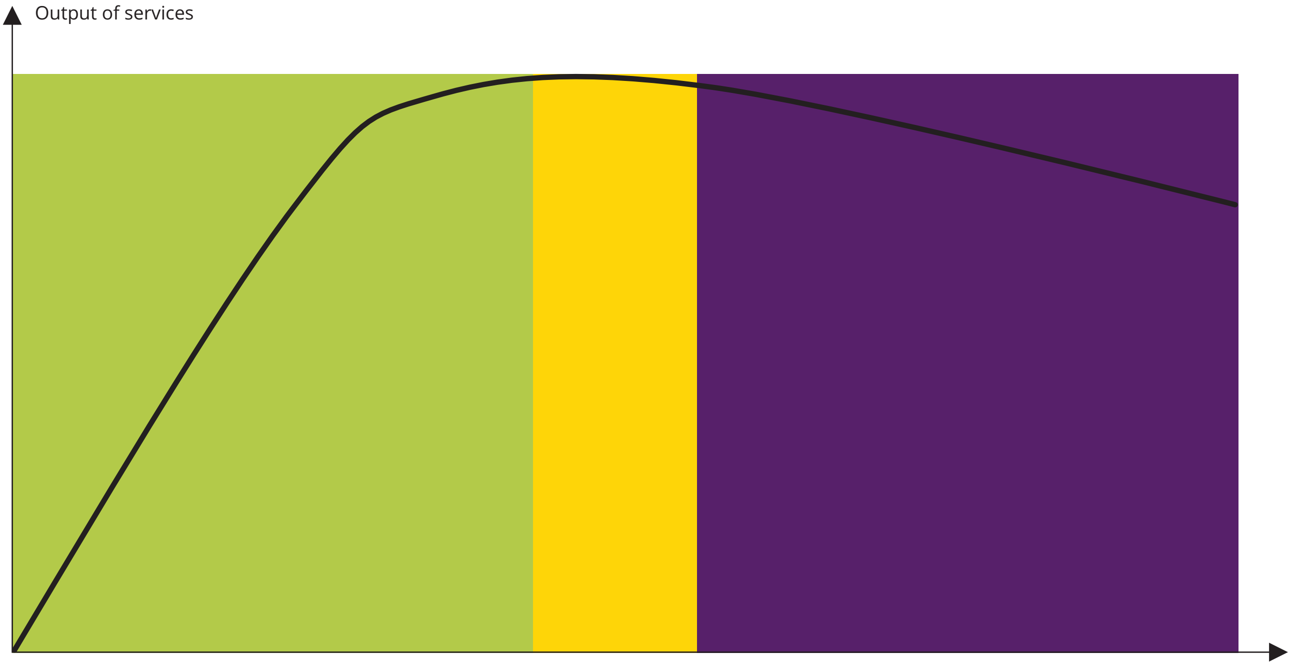
- Placing excess demand (for benefits) on natural resources (including soil) causes a decline in soil performance. How can this demand be managed optimally' (Figure 4.1)?

4.2 Soil, property rights and stakeholders

A particular issue in the governance of soil is the *spatial and temporal distribution of costs and benefits* between communities. As discussed above (Section 3.2) the benefits of soil 'exploitation' accrue mainly on site to the land owner or to others with property rights, whereas the costs of soil degradation are widespread, ranging from a global to a local level. In addition, one generation may exploit soil, while degrading it and reducing the benefits that can be enjoyed from its use by future generations. Associated with property rights are duties: to a great extent, the governance of soil, including in urbanised areas, concerns the definition and enforcement of these duties so that excessive costs do not compromise the common good for all stakeholders, both private and public.

The ecosystem framework can help to clarify the links between flows of soil-related services and their value to stakeholders, providing a perspective to inform governance. According to Reed et al. (2009), *stakeholders* here are individuals, private and public groups or organisations (present and future) with an interest in, and who derive potential benefit or loss from, a change in soil-supported ecosystem services. They may also be differentiated in line with the degree to which they can influence service flows, through property rights, entitlements and the control of resources. *Property rights* define entitlements (rights) and responsibilities (duties) with respect to the use of a given property or resource that provides some benefit or income (Bromley, 1991) and are protected by the

Figure 4.1 Optimisation of soil resource management



Note: Green, resource use within system capacity; yellow, resource use close to upper system performance limit and likely to degrade the resource if continued; purple, resource use causing degradation.

Source: Re-drawn from Kibblewhite et al., 2008.

state. The definition and operation of property rights are key elements of governance: to work properly, property rights regimes must be comprehensive and specific, exclusive to the owner, transferable and secure from involuntary seizure.

Soil is a subset of land and the property rights associated with soil are normally integrated with the property rights associated with a piece of land. The rights to beneficial soil-based services may be vested in more than one individual stakeholder. A tenant farmer may hold rights to agricultural production but not to hunting (these may be retained by the land owner), and public rights of access may exist. Arguably, some property rights are removed by new legal requirements placed on land owners, such as those requiring conservation of habitats or landscape character.

As mentioned above, property rights are claims to benefit or income streams in which institutions have a role to play. However, property rights regimes often fail to recognise and value the 'external' effects of land and soil use, whether positive or negative, with consequences for welfare. Environmental externalities that are not incorporated in formal entitlements and transactions indicate a failure of the institutions of governance. From a societal perspective, the inefficient and damaging use of soil constitutes a failure of governance. For this reason, restriction or covenants on land and soil use may be imposed on developers to guard against uncompensated effects, such as stipulations for sustainable urban drainage, a minimum provision of open space or possibly soil conservation. Alternatively, financial incentives may be given. For example, in Europe, farmers may be 'compensated' (e.g. under agri-environment schemes) for maintaining quality agricultural soils as a strategic natural resource or for providing public access where the benefits to urban populations of agriculturally managed land may otherwise go unrewarded and under provided. Section 4.4 provides additional examples of 'corrective' instruments.

'Entitlements to benefit' are *not absolute*, but rather derived according to dominant societal preferences and priorities, and these *vary spatially and temporally* (Tawney, 1948). Historically, property regimes have given priority to dominant provisioning services such as farming or housing, evident, for example, in the recognition of agricultural land tenure or development consent (Bromley and Hodge, 1990). As other land and soil services, such as floodwater regulation, carbon storage or cultural heritage, become more important, new property regimes will be required to reflect changing priorities (Beedell et al., 2011). Increasingly, land and soil in urbanised areas may be subject to conditions of use that secure a wide range of ecosystem services to a wide range of stakeholders in response to changing preferences and priorities (e.g. Rawlins and Morris, 2010). This tends to realign governance systems away from a single purpose towards multiple purposes, including a broader community of stakeholders in the process of governance and its outputs.

4.3 Distribution of soil costs and benefits

As explained earlier, soil, as a component of natural capital, supports a diverse range of ecosystem services, many of which take the form of unpriced, non-market public goods such as carbon storage or flood control, which are not subject to formal property rights and entitlements. For example, users of soil may have no obligation to maintain soil organic carbon stocks or to control run-off from soil by restricting sealing for the benefit of society at large, unless specific covenants are placed on them to do so. Nor may members of society have entitlement to such benefits associated with the use of soil. Users of soil, whether farmers or property developers, are unlikely to provide such 'external' benefits to third parties without compensation. Hence, the associated public goods are under-provided from the view point of societal welfare.

Conversely, users of soil are inclined to pass on external costs associated with the uncompensated 'off-site' negative impacts of soil use, such as carbon loss or run-off due to soil sealing that can cause flooding in the sites of third parties. Hence, public 'bads' are over-provided from a societal welfare point of view. The implication here is that there is a potential conflict between the use of soil for private gain and for public good. Good governance of soil requires an optimum balance between the two whereby, using a variety of means, the externalities of soil use are 'internalised' into decisions about its use and management. For the most part, this involves encouraging public goods and discouraging public 'bads' in order to achieve agreed economic and socially efficient outcomes, where agreement here is a product of good governance.

Thus, inefficient use of soil, either because of its misallocation between uses or because of its degradation in its particular use, is essentially a problem of governance, indicative of failure to adequately recognise the economic and social worth of soil and build this into systems of resource management and decision-making by private and

Box 4.1 Participatory approach to soil valuation

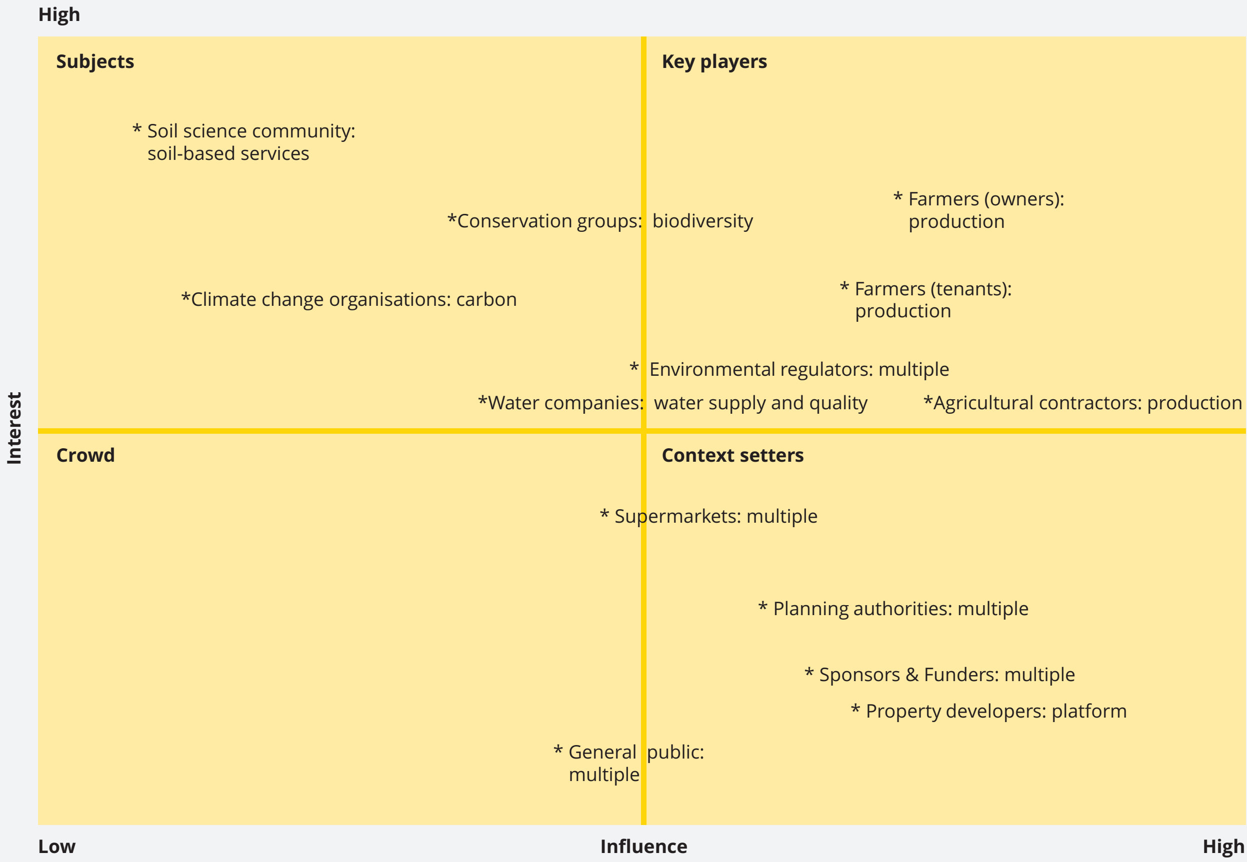
A variety of methods can be applied to identify, differentiate between and investigate relationships between stakeholders, in order to elicit information about stakeholders' interests in particular ecosystem services and goods (Reed et al., 2009). One method that is useful as a starting point and is explained in the paper is to construct interest/influence dimensions that place stakeholders on a matrix according to their relative interest in a phenomenon, such as loss of green areas or flood risk management, and the extent to which they influence or exert control over it. Stakeholders can be classified as 'key players', 'context setters', 'subjects' or 'the crowd' (e.g. Eden and Ackermann, 1998). This can help to specify how stakeholders might be engaged to achieve specific outcomes.

Key players are stakeholders who should be actively engaged because they have significant interest in and/or influence over a particular set of outcomes. Influence is typically exerted through some legal entitlement or property right. Context setters are highly influential, but may have little direct interest in outcomes and, because of this, they may be a significant risk to achieving an optimum outcome, requiring their monitoring and management. Subjects have a strong interest in outcomes but low influence of which ones emerge. They lack the capacity for direct impact on the governance process, although they may become influential through alliances with other stakeholders. Land development projects often find they include the marginal stakeholders that they want to empower. The crowd are stakeholders who have little interest in or influence over desired outcomes; they are seen as not worth engaging with (Reed et al., 2009).

Analytical insights can be enhanced by adding further attributes to the stakeholders, such as details of the type or expression of interest (such as campaigning) or of the means by which they exert influence (such as ownership of resources). A formal stakeholder analysis for soil-supported services in urbanised areas would inform a better understanding of urban soil governance. It might, for example, be applied to the governance of land with a high-quality soil resource for which there are contested outcomes (e.g. as a platform for extension of continuous urban fabric, support for urban food production, construction of artificial vegetated surfaces for amenity and leisure and/or conservation of soil-dwelling animals).

Figure 4.2 provides an indicative map of stakeholder interests in and influence on the sustainable management of the soil resource based on our understanding of the literature.

Figure 4.2 An indicative map of stakeholder interests and their influence on the sustainable management of soil resources in urbanised areas



The management of contaminated land (e.g. CLARINET, 2002; Cundy et al., 2013) provides an example of an arena in which the importance of establishing governance that is inclusive of many different stakeholders has been recognised is. An 'integrated approach', that includes early and continued engagement with the full range of stakeholders in sites of local soil contamination, and effective communication with them, is widely recognised as good and effective practice (see CityChlor (2013) for a recent appreciation).

public actors. Failure of land and soil governance has particular resonance in urbanised areas, where land use change is shaped by more intense dynamics of economic development and demography. Two broad *urbanised contexts* are apparent: (1) where economic development and population growth increase competition for land with allocation often to the highest bidder; and (2) where economic development and population are in decline and land is under-utilised and abandoned. Both of these situations have the potential for economic inefficiency in the use of soil.

Physical and temporal separation of the impacts of on-site benefits and off-site costs encourages degradation (see Table 3.3) associated with inefficient use of soil by those who benefit, at the expense of those who bear the off-site costs. This tendency is of particular importance in peri-urban areas, where there is the potential for greater interaction between parties pursuing competing services and goods from multi-functional land and soil (e.g. residential development versus agriculture production or biodiversity conservation). In an economic sense, environmental risk associated with soil use and management is the probability of a soil degradation process causing an unacceptable level of harm to a receptor (e.g. other environmental compartments, humans). The risks that arise from soil use and management are often greater for other parties than those using the soil directly. Thus, the proximity of a larger population that can potentially be affected in urbanised areas increases the risks relative to those in the non-urban (i.e. rural) areas with lower population densities.

From a governance perspective, it is important to understand the underlying trade-offs and synergies between ecosystem services associated with soil use and management, their relative values and how they (i.e. the different benefits and costs) are distributed among different parties and over different spatial and temporal scales. In addition, the scale over which costs and benefits arise from soil use and management are variable between services. The costs of reduced carbon sequestration by soil, or the benefits of maintained levels thereof, are distributed globally, while the costs and benefits relating to the water cycle occur at a catchment scale and the benefits derived from using soil as a platform are local.

The distinctions between on-site and off-site costs, and between private and public costs and benefits of soil degradation, are critical from a governance perspective. Evidence to date (see Section 3.2) suggests that the on-site benefits of soil use often exceed the on-site costs of degradation, and that the on-site 'private' benefits of soil conservation may be insufficient to recover the 'private' costs involved. Moreover, the costs of substitution for losses of on-site services appear to be low relative to the on-site benefits that such substitution supports, because of technical innovations (e.g. fertiliser input to maintain yield levels despite decline of organic matter). This results in limited incentives for soil users to conserve soil. Off-site costs, however, often considerably exceed off-site benefits. Indeed, they may in some cases exceed all of the private and public benefits of soil use. Hence, there is justification for interventions to conserve soils in the public interest and design policy instruments to that aim.

Governance needs to consider the scale over which off-site costs occur. This varies from global to local, as different territories and regimes may be impacted, to greater and lesser extents, thereby affecting the optimal choice of intervention. Costs of reduced carbon sequestration have impacts globally. Increased run-off and reduced groundwater recharge arising from soil sealing have impacts at the catchment scale, as does higher sediment generation from soil erosion. Local soil contamination can impact on adjacent communities and local water resources, while diffuse contamination may restrict options for food production. Some on-site costs may be public costs that accrue to other users, such as walkers in the countryside (with limited and subsidiary entitlement to use land for access), whose non-market benefits are reduced without compensation (assuming they cannot go elsewhere to derive the same net benefit).

4.4 Instruments to protect the soil resource

Most natural resource and environmental policy is concerned with identifying the benefits and costs of resource use and, where external costs are not internalised into on-site ones, designing appropriate interventions to enhance or protect social welfare. This may involve public or private instruments or a mixture thereof. Table 4.1 summarises the main types of instruments, illustrated with generic examples. Examples of instruments that are particularly relevant to controlling soil contamination and soil sealing in urbanised areas are provided in Section 4.4.6.

Table 4.1 Types of policy and private instruments relevant to soil protection in urbanised areas

Intervention	Public/government				Private
Instrument	Regulatory	Economic		Awareness raising	
		Price-based	Market-based		
Process to control external impacts	Government *enforces* environmental law; command and control approach	Government provides *incentives* for behavioural changes by *modifying existing markets*	Government provides *incentives* for behavioural changes by *creating new markets*	Government *raises awareness* among others by providing grants and credits (investment support)	Private, *self-motivated* change of behaviour, possibly inspired by public awareness-raising initiatives
Participation	Mandatory	Voluntary	Voluntary	Voluntary	Voluntary
Examples	Soil protection legislation	Fiscal intervention (tax on use); subsidies; compensatory offsets	Tradable development certificates; eco-labelling	Outreach and education; innovation	Adoption of sustainable soil management practices

Sources: Based on EU, 2004, 2008a, 2010, 2013b; EC, 2011a; Louwagie et al., 2011; Samarasinghe et al., 2013.

As opposed to regulations, private actors' participation in economic instruments is voluntary, taken up at the discretion of land managers. Economic instruments can involve an array of financial incentives (such as agri-environment payments), induced by potential market advantage, or possibly as a result of the threat of formal regulation.

4.4.1 Public regulatory instruments

Land-use regulations

Land-use regulations include prohibitions, covenants or consents placed on land use or land use change. These restrictions are usually supported by statutory regulations and steered through land use planning. They typically control the development of greenfield sites and/or give priority to brownfield sites. This includes bans or restrictions on development in the urban fringe, on high-grade agricultural land, on designated habitats and where development could result in unacceptable on-site or off-site risks, such as development in the floodplain or on hill slopes.

Regulatory consent for the development of land that is designated as 'contaminated land' because it presents an unacceptable risk to humans or other receptors will commonly be conditional on completion of agreed remedial actions. Contaminated land that is not intended for development, but nonetheless still presents unacceptable risks will normally be subject to remediation under an enforcement action by a statutory regulator.

Compulsory land purchase is another instrument affecting land use. It will normally be accompanied by compensation to owners/occupiers by a government agency and aims to facilitate a particular land use and associated land-based services, supported by powers of compulsory acquisition. Examples include the purchase of land for the transport and storage of flood water or for the restoration/creation of habitats for biodiversity. In some cases, purchased land may be returned to previous owners/occupiers under leaseback arrangements or easements, subject to some modified or constrained entitlement or property right, such as the right to flood. Compulsory acquisitions are often supported by land swaps, especially where public ownership of land is high. In some situations, severely degraded and contaminated land and soils may transfer at 'peppercorn' rates to authorities taking responsibility for restoration.

Land management regulations

Non-discretionary requirements are provided, for given types of land use, with 'general binding' rules and covenants on land management practices in order to protect soil quality and control potential on-site and off-site risks. Where land is developed for potentially polluting activities, statutory regulations (e.g. under the Industrial Emissions Directive: EU, 2010) will apply to ensure that soil is not contaminated.

4.4.2 Public economic price-based instruments

Taxes

The potential for using economic instruments to optimise land use has been explored (EEA, 2010), supported by the observation that natural factors, including soil type and condition, are not reflected in urban land prices, indicating market failure. This study reported that the outcome of a debate in Germany and the Netherlands on the use of *taxes*, for example, was that, while economic instruments can be used to supplement traditional land-use planning, they are not a replacement for it. The focus of this study was on land use as distinct from land management within a defined use; the potential for applying taxes or other economic instruments to promote beneficial land management (e.g. to encourage the use of porous in place of impervious hard surfaces) could also be considered.

Programmes for urban renewal and brownfield restoration and/or redevelopment are often supported by national government grants and regional funds. *Development taxes* (compensation charges) imposed on development projects that convert land from agriculture use are used in some countries (Box 4.2).

Box 4.2 Development taxes

In the Czech Republic and Slovakia, for example, special permits obtained from local, regional or national authorities (depending on the size of the land take) are required for conversion of land from agricultural to urban use. Charges levied in EUR/m^2 of land taken vary according to agricultural land class. This has generated a revenue stream for the regulating authorities but has reportedly had little impact on the demand for land for conversion.

In Schleswig-Holstein (Germany), land take charges range from EUR 1–5/m^2, with higher charges for land that is sealed.

Subsidies

Subsidies or *compensatory payments* allow for the adoption of land-based ecosystem services with particular reference to soil. These relate to payments made for the adoption of land and management practices that provide (usually off-site) benefits to third parties, such as retention of soil carbon, control of run-off that could generate flooding, protection and enhancement of biodiversity, and long-term strategic protection of agricultural soils, especially under conditions of climate change (Box 4.3). These arrangements, linked to 'voluntary' instruments, may be supported by capital grants for soil conservation in vulnerable catchments or annual payments under Rural Development Programmes' agri-environment schemes. In the latter case, farmers are rewarded for 'beyond compliance' actions that serve the public interest, such as conversion of arable to grassland, woodland planting or measures to control run-off from farmland. These interventions help to maintain the integrity of the soil resource and services.

Compensatory offsets

Compensatory offsets have been introduced by some countries and provide a compensation system that offsets losses of soils, biodiversity and habitats in one place by like-for-like reinstatement elsewhere (Box 4.4).

4.4.3 Public economic market-based instruments

Tradable development certificates

Tradable development certificates are similar in principle to tradable abatement permits (e.g. carbon credits). Municipalities are issued with an initial stock of land-use

Box 4.3 Payments for ecosystem services

Payments for ecosystem services (PESs) are a potential means of converting the non-market values of land-based ecosystem services into real incentives for land managers. PESs involve voluntary transactions to exchange well-defined environmental services between service buyers and service sellers (Wunder et al., 2008). Most PES schemes operate through specific land uses and management protocols capable of producing the required environmental service (OECD, 2010), such as the establishment of urban parks or run-off interceptors, rather than focusing on the achievement of specific outcomes such as urban biodiversity or particular public health improvements, and subsequent rewards for them, which cannot be guaranteed.

While some PES schemes are financed by users for commercial benefit, such as water companies wishing to secure water supply and quality, most are funded by governments providing public goods such as biodiversity, flood control and public access to green space and the countryside. The PES concept is now widely applied in the provision of watershed services (Tognetti et al., 2005) and habitat services (Landell-Mills and Porras, 2002). EU-funded agri-environment schemes are PESs that reward farmers for additional environmental services beyond compliance with minimum standards of environmental protection. PES schemes can help to mobilise the development of GI and the promotion of land management options that support multi-functional land use in urbanised areas (North-West Investment Forum, 2013).

The potential of using PES schemes as instruments for affecting land use has been highlighted (EEA, 2010), with the first step towards their introduction suggested as being 'to create a system that allows for the comparison of land values including the physical and environmental qualities [that determine potential for supply of ecosystem services, which would necessarily include soil type and condition].'

Box 4.4 Compensation accounts

In Germany, for example, an eco-accounting system is managed by an apex compensation agency and appointed subsidiary local organisation under the National Nature Conservation Act. Developers must acquire certificates to confirm offsetting of losses of conservation areas, farmland and forestry over the long term. Developers can, for example, undertake compensation measures themselves or pay a compensation fee to a responsible organisation, such as a local government authority, that manages offsetting. Developers can 'deseal' and 'green' proposed new developments in order to reduce the sealed 'footprint', and then offset incremental effects by reducing sealing on other existing developed areas or by enhancing existing green areas.

Prokop et al. (2011) reported on the proposals for the Vienna region for a Landscape Compensation Account to implement offsetting requirements for relatively large-scale projects, such as airport development, identified by environmental impact assessments. Experience with compensation arrangements has been mixed, partly because of difficulties in defining the equivalency, suitability and effectiveness of compensation measures, and in ensuring that offsets are fully implemented.

Photo 4.2 Recreation on sealed soil. With the pavement breaking up: time for enjoying the soil functions as well?

change permits. They can then either use or trade them depending on need, the cost of measures to reduce land take (i.e. abatement) and market values of permits. The approach has similar advantages and disadvantages to tradable permits in other applications, such as emissions to atmosphere and water. It has the potential, at least in theory, to provide flexibility and dynamic incentives to support efficient allocation of land use and management, especially at the regional scale. In practice, however, administration and transaction costs can be high, given the need for an overall regulatory framework. Furthermore, the initial setting of limits and allocation of permits are often contested and, for a variety of reasons — including speculative accumulation of land and the potential benefit that retained permits might bestow — the extent of trading may be limited. A variant of tradable permits involves a fixed quantity of permits being auctioned. The rationale underlying this variant and a system of fully tradable permits is the belief that using markets is the most efficient way to allocate scarce resources.

Labelling

Labelling is an instrument that is used, for example, in combination with certification schemes for organic or ecological produce.

4.4.4 Public awareness-raising instruments

Financing instruments: grants and credits from public funds

Innovation, on the one hand, and outreach and education (to promote adoption of alternative practice), on the other hand, are the two main areas to which public financing applies. Investment in innovative technology and its adoption is, for example, supported by EU research funding. Furthermore, a number of measures to mitigate, remediate and remove impacts on soils resulting from development (including soil contamination) and other soil uses in urbanised areas have been developed by EU research programmes such as REVIT, CLARINET and CABERNET. Sometimes, both goals (innovation and outreach) are served under the same funding vehicle/instrument (Box 4.5).

Other awareness-raising instruments

The European Commission (EC, 2012a) published soil sealing guidelines to raise awareness and promote positive action against soil sealing and its negative effects for the environment.

Box 4.5 Natural Capital Financing Facility

The Natural Capital Financing Facility (the 'Facility' or 'NCFF') is a financial instrument blending European Investment Bank (EIB) funding with European Commission (EC) financing funded by the Programme for the Environment and Climate Action (LIFE programme).

The NCFF shall contribute to meeting the LIFE objectives, in particular for the priority areas 'nature and biodiversity' under LIFE Environment and 'climate change adaptation' under LIFE Climate Action by providing financial solutions to bankable projects which are revenue-generating or cost-saving, and promote the conservation, restoration, management and enhancement of natural capital for biodiversity and climate adaptation benefits. This includes ecosystem-based solutions to challenges related to land, soil, forestry, agriculture, water and waste.

The primary objective of the NCFF is to develop a pipeline of projects, testing different financing options in order to identify the most suitable approach. The overall objective is to provide a proof of concept demonstrating to the market, financiers and investors, the attractiveness of such operations, thereby developing a sustainable flow of capital from the private sector towards the financing natural capital and achieving scale (EIB, n.d.).

The focus will be on projects that are 'at an advanced stage of development and have the potential to be replicated within the EU'. The financing facility is particularly targeting projects on:

- 'green infrastructure (e.g. green roofs, green walls, ecosystem-based rainwater collection/water reuse systems, flood protection and erosion control);
- payment for ecosystem services (e.g. programmes to protect and enhance forestry, biodiversity, to reduce water or soil pollution);
- biodiversity offsets/compensation beyond legal requirements (e.g. compensation pools for on-site and off-site compensation projects);
- pro-biodiversity and adaptation businesses (e.g. sustainable forestry, agriculture, aquaculture, eco-tourism)' (EIB, n.d.).

Stepping up financing in these areas is considered as one of the conditions for achieving the EU's 2020 biodiversity goals. Technical assistance in project preparation, implementation and monitoring will accompany the financing.

The NCCF will be managed by the EIB. Both public and private entities can be beneficiaries.

Source: EIB, n.d.

4.4.5 Private interventions

Self-motivated adoption of sustainable soil management practices by individuals and private organisations are part of a broad ethical commitment to sustainability, respect for the natural world and 'living within environmental limits'. They include the adoption of best practices to minimise or avoid loss of soil functions, such as conformity with planning and operations guidance for construction agents and developers and with guidance for farmers on soil protection. Some of these voluntary actions may nevertheless be inspired by public awareness-raising initiatives (Box 4.6).

4.4.6 Addressing soil degradation processes in urbanised areas

As mentioned earlier, soil sealing and local soil contamination are the two major soil degradation processes that affect soils in urbanised areas. At country level, a variety of instruments are used to control land take by the construction of artificial structures and consequent/concomitant soil sealing, and to deal with new and historic local soil contamination. Monitoring and reduction of sealing is already a main policy target in some urban areas (Technical Soil Protection Committee (NL), 2009; EC, 2012a). In addition, soil contamination is already receiving targeted responses: the threshold values for the direct intake of pollutants of the Federal Soil Protection and Contaminated Sites Ordinance of Germany allow a higher contamination status in park areas, recreational facilities and plots of land used for industrial and commercial purposes than in playgrounds and residential areas (BRD, 1999).

Commentaries and summaries of the possible performances of different public instruments for protecting soil in urbanised areas from *sealing* and from new *local soil contamination* are presented in Tables 4.2 and 4.3.

Box 4.6 Private action inspired by public awareness raising

Following a review of the consequences of over compaction of soils in the construction process (WSP, 2006), Defra (2009a) introduced a construction code of practice for the sustainable use of soils on construction sites. The latter was preceded by a policy impact assessment (Defra, 2008) that identified potential savings to site developers of 5–80 times the initial costs due to reduced soil import/export costs, landscape scheme failure and soil stripping and waste, in addition to any 'off-site' benefits.

The wide adoption of such self-motivated voluntary initiatives, however, requires an increased awareness of the importance of soil amongst the 'context setters', especially developers and their agents, as well as the general public.

Table 4.2 Indicative performances of different public instruments for protecting soil from sealing

Type of policy instrument	Mandatory regulations	Economic price-based	Economic market-based	Awareness raising (innovation)
Effectiveness	High (e.g. regulations on construction type and extent)	Low to moderate (e.g. tax on sealed areas; subsidy for not sealing areas or porous surfaces, depends on tax rates relative to soil de-sealing cost)	Low to moderate: moderate for participants, low for others	High (e.g. introduction of new surfacing materials; improved architectural design)
Motivation for further improvement	Low	Moderate to high, incentive to avoid tax	Moderate to high for participants	High if reduces costs/ increases value, and linked to development consent
Economic efficiency	Not known, but possibly low if 'one size fits all', and monitoring cost is high	High, in theory because of an efficient allocation of resources through avoiding a 'one size fits all' contexts approach	Medium, if targeted and proportionate	Moderate to high, depending on innovation success and eventual reduced dependency on subsidy
Administrative efficiency (extent of transaction costs)	Low, as inspection is required for all sites	Medium, but possibly low if combined with existing fiscal administration. Usually requires a 'regulatory setting'	Medium (may be burdensome for smaller firms)	High, as costs are potentially shared and recovered through wide application. Possibly funded through tax
Acceptability to different stakeholders	Regulator preference is to ensure minimum standard. Smaller firms and private citizens may bear disproportionate costs and resist	Smaller firms and private citizens may bear disproportionate costs and resist	Acceptable to all if truly voluntary, but may be resisted by small and marginal operators	High acceptability to all
What are the main risks/ assumptions relating to use?	Economic impact is not disproportionate; technical basis for regulation not overly complex	Market processes ensure efficient distribution of benefits and costs, changing behaviour	Benefits of participating exceed the costs for many actors	Successful innovation is feasible and adopted
What actions could be taken to improve the instrument's performance?	Cost-efficient means for inspecting and reporting (e.g. remote sensing)	Economic analysis to tune incentives and optimise net benefits	Support to leaders in industry and communities	Ensure user needs are well defined and inform objectives

Table 4.3 Indicative performances of different public instruments for protection from new local soil contamination

Type of policy instrument	Mandatory regulations	Economic price-based	Economic market-based	Awareness raising (innovation)
Effectiveness	High (e.g. IED ([a]) permit not released until remediation is completed)	Low: cannot guarantee environmental outcome	Low, at discretion of polluter	High, including if linked to permitting
Motivation for further improvement	High, as costs of non-compliance are large	Variable (e.g. depending on tax rates)	Low to moderate (must be embedded in corporate responsibility)	High if reduces costs/ increases land values
Economic efficiency	Expected to be high, as total costs of new contamination are potentially high	High, in theory, but, for example, requires that tax rates are set at the marginal cost of pollution, which is generally not known	Low	High if innovation successful
Administrative efficiency (extent of transaction costs)	Medium, as costs of inspection may be high	Low, in theory, but requires regulatory framework	Low	High, as costs are potentially shared and recovered through wide application
Acceptability to different stakeholders	Smaller firms and private citizens may bear disproportionate costs and resist	Moderate, suits market-based approach, but less favoured by regulators	Acceptable to all if truly voluntary, but less by small and marginal operators	High acceptability to all
What are the main risks/ assumptions relating to use?	Economic impact is not disproportionate; technical basis for regulation not overly complex	Market processes will ensure efficient distribution of benefits and costs. Tax rates change behaviour	Benefits of participating exceed the costs for many actors	Successful innovation is feasible
What actions could be taken to improve the instrument's performance?	Cost-efficient means for inspecting and reporting	Economic analysis to tune incentives and optimise net benefits	Support to leaders in industry and communities	Ensure user needs are well defined and inform objectives

Note: ([a]) IED, Industrial Emissions Directive (EU, 2010).

In protecting soil from both sealing and new local soil contamination, financing — with an emphasis on innovation — is identified as a desirable instrument because of its potential to provide increased economic efficiency and a high net benefit when successful, and because of its acceptability. Mandatory regulation is identified as the most effective instrument to control both soil sealing and new local contamination; its economic efficiency will depend on adopting a technical design that allows for different situations and circumstances so that the intensity of regulation is matched to the extent and risk of soil damage; its administrative efficiency will increase if innovative monitoring and reporting technology can be applied. Fiscal incentives and other voluntary initiatives are identified as potentially less appropriate for controlling new soil contamination because they may allow non-compliance with the application of the polluter pays principle. However, fiscal incentives and other voluntary initiatives appear more promising for limiting and reversing soil compaction and sealing. Economic instruments that use subsidies, taxes or tradable sealing permits could help limit the extent of soil sealing, including by directing it away from higher value soils. In all cases, interventions may result in disproportionate costs to small firms and individual citizens, which may justify targeted public funding.

An important consideration when reviewing the choice of instruments is that soil degradation, including soil sealing and contamination, is not a 'uniform' pollutant like carbon emissions. The impacts of soil sealing (Box 4.7) and contamination (Box 4.8) tend to be *very context specific* (e.g. the value of losses of services from soil sealing of a hectare in one location may not be the same as that for another hectare in a different

Box 4.7 Awareness-raising incentives to reduce land take

Land take — a measure of how much land covered by agriculture, forests and semi-natural land, wetlands and water is converted to land cover for urban (including the creation of green urban areas over previously undeveloped land), commercial, industrial, infrastructure, mining or construction purposes — is a phenomenon that happens at the urban fringe (peri-urban zone) (EEA, 2013). In addition to the above solution that builds on the soil information base, so-called greenfields can also be preserved by alternative guidance to the use of space.

Case: a policy target

Germany's National Sustainable Development Strategy (2002) includes the following objectives:

- By 2020, land take for new housing and transport developments is to be limited to 30 ha/day ([5]).
- The German approach is a twin-track strategy, comprising:
    - further strengthening of inner urban development;
    - limiting new land take on the urban fringe.
- Implementation with a mix of instruments, such as:
    - giving priority to inner urban development;
    - revitalising the inner cities;
    - space-saving housing developments with low levels of traffic;
    - enhancing the productivity of land;
    - land recycling;
    - taking soil qualities into account; and
    - safeguarding open spaces.

The 30 ha objective is addressed primarily to the federal states (regional and sub-regional planning) and local authorities (development planning). The federal government supports the federal states' efforts through legislation (spatial planning law, urban development law); financial assistance and research programmes; and information (Ludlow et al., 2013, p. 172).

To counter-balance land take, trans-regional trade with area certificates ('Flächenzertifikate') was set up: all towns and municipalities together can only consume non-artificial land within the limits of the 30 ha/day target (Forschungs- und Entwicklungsvorhaben des Umweltbundesamtes, 2013). In 2013, 15 local governments were participating in the trade mechanism; the initiative has spread in the meantime (for an update of the participating municipalities, see Planspiel Flächenhandel, 2013).

Case: information and exchange platform for reuse of brownfields

In Switzerland, the Federal Office for the Environment, together with partners, set up a website serving as an information and exchange platform for the reuse of brownfields ([6]). It targets owners or site developers, transient users, tenants, investors and federal, cantonal and communal authorities.

The information is continuously adapted and supplemented, and currently covers topics such as location and area of the abandoned site; state of the site and buildings; planning status; guidance on reconversion of brownfield sites, including financing instruments available; legal requirements for reuse; and management of contaminated sites. However, the database does not contain any explicit information on soil status. Brownfield reuse is an initiative that, indirectly, is intended to avoid additional land take (on so-called 'greenfields'). Ludlow et al. (2013)

([5]) The objective is very ambitious: between 1997 and 2000, consumption was 129 ha/day; consumption was reduced to 87 ha/day for the period 2007–2010, and is currently estimated at 75 ha/day (Planspiel Flächenhandel, 2013).

([6]) www.areale.ch (in German); www.friches.ch (in French).

geographical context). Compared with regulatory measures, price-based (in particular fiscal) and market-based instruments can be more easily designed to allow for non-uniform, context-specific impacts of soil degradation (although the EU subsidiarity and proportionality principles also allow for context specificity in implementing regulatory measures). In principle, they can allow for context-specific variations in both the value of degradation abatement and the costs of soil degradation abatement, both of which vary spatially, between types of development and developers, and over time. There is a requirement to match appropriate instruments to the scale and the scale-dependency of impacts. Tradable permits for carbon sequestration by soil may be appropriately implemented at a continental scale, as their impact is global, while a catchment scale for implementation may be appropriate for controlling impacts on the water cycle.

In most situations, a mix of policy interventions, set within a regulatory framework that seeks to protect soils in the public interest, will be required. Making an informed choice of the mix of policies needs to be predicated on a sound understanding of the functions and values of soils.

4.5 Integrating the natural capital concept in governance

Chapters 2 and 3 have shown that the knowledge base on soil valuation in urbanised contexts, be it of ecological or monetary nature, is fragmented at best. An assessment in this chapter has revealed that the body of evidence — with all its limitations — may also not always be suited to support asset-based governance (i.e. soil information may not always be available or, if available, may not necessarily be used in the suite of policy instruments currently applied). Drawing from existing practice (some of which has been touched upon earlier), this section presents possibilities of acknowledging soil as a 'productive' asset (i.e. highlighting its potential to fulfil human needs), drawing from its pivotal role in natural capital. In the following sections, three aspects are highlighted: soil functions (ecological focus), soil-based ecosystem services (implying a biotic component) (monetary focus) and the non-biotic component of soil.

4.5.1 The role of urban soil function maps in spatial planning

'Unlike other tangible investment assets, speculation in land-property markets can have severe social and economic consequences that demand public intervention. [...] To work, land-property markets must take on modified forms of governance' (Alexander, 2014, p. 539). With these statements, the author refers to the role of spatial planning systems, namely the methods the public sector uses to influence the distribution of people and (economic) activities in spaces of various scales.

Box 4.8 Regulatory instruments in dealing with soil contamination

Case: transparent soil information in land transactions

Collating information relevant to land trading (e.g. on local soil contamination) and making that information available publicly and in a transparent manner can help to avoid the externalisation of (soil degradation) costs to society.

In Flanders (Belgium), the Public Waste Agency of Flanders (OVAM) is directly involved in the removal of waste products and soil remediation. OVAM seeks to integrate its clean-up activities into broader redevelopment projects, in order to create added value. Good communication with all parties involved is considered crucial in this process. The selling of 'used land' is, therefore, guaranteed and transparent, which may indirectly lead to no net land take.

Soil evaluation is mandatory when there are indications of soil contamination. A descriptive soil investigation (trying to find out about the dispersion of the contamination and its future evolution) and a remediation project must be worked out. Anyone looking to trade/sell land must possess a soil status/certificate.

In Finland, an example of good practice is found in appropriate environmental communication: when someone is selling or renting a polluted site, he/she is obliged to tell the new owner or tenant what kind of activities have been carried out on the site and if any pollution has or may have occurred. If this requirement is ignored, the buyer has the right to demand that the agreement be cancelled, that the price be lowered or that the seller covers the damages. In this way, the taxpayers are not required to invest money to protect so-called 'innocent buyers' as is done in other countries.

Source: Ludlow et al., 2013.

Box 4.9 Integration of soil information in environmental planning

Driven by the EU INSPIRE Directive (EU, 2007b) and in response to the Environmental Planning Act, Dutch public organisations joined forces to develop a common spatial data infrastructure for environmental information. PDOK ('Publieke dienstverlening op kaart') ([7]) — literally translated as 'public services on a map' — is a geo-information distribution portal. It acts as a distribution node for key geo-registers (buildings, topography, etc.) and environmental information. The service providers deliver the data with consistent quality (availability, usability, reliability); most data are publicly accessible (one exception, for example, being the cadastre data).

Soil-related data are also covered, although to a limited extent. Soil information (along with geomorphological and surface water characteristics) is included in the delineation of biogeographical regions. Higher level (supra-municipal, i.e. beyond municipal level, often at provincial level) restrictions related to the Soil Protection Act ('Wet bodembescherming', addressing contaminated sites) are also included in the map. Land cover/land use is included as a separate data layer. However, the INSPIRE-compliant soil map is currently not included in the PDOK facility.

The initiative involves the Ministry of Infrastructure & Environment (owner of the system), the Ministry of Economic Affairs, the National Road Authority, the Cadastre, Land Registry, the Mapping Agency of the Netherlands, and the National Spatial Data Infrastructure executive committee (Geonovum, which has a coordinating role in the implementation of the INSPIRE Directive).

Source: van der Vegt, 2014.

However, in most countries, little direct account has so far been taken of the need to consider the soil resource in the governance of land in urbanised areas (Burghardt et al., 2004; Ludlow et al., 2013) (Box 4.9). Be that as it may, soil makes an important contribution to natural capital, including the flows of essential goods and services that can be derived.

Returning to the specific role of soils, designating soil value and, on that basis, reducing consumption of valuable soils, are possible on the basis of urban soil function maps (Box 2.6). Constructions can be locally planned in such a way that the consumption of soils of high value is minimised, and that the use and management of soils overall is adapted to the local situation and demands. This enables a system to be introduced that integrates economic aspects in the urban planning process by valuing the soil as determined by the services it delivers to residents (Wolff, 2007).

Also in the urban fringe, where competition between mutually exclusive economic activities and uses can be high, evaluation of the soil resource can be a useful and necessary tool to integrate into spatial planning (Box 4.10).

4.5.2 Appropriate integration frames for applying an ecosystem services-based approach

Chapter 3 has shown that monetary valuation of soil may actually be most meaningful in connection with a bigger (investment) project (e.g. payment of ecosystem services schemes, GI, the conversion of an abandoned or under-used brownfield to a productive site, large-scale land restoration through afforestation) (Box 4.11). Chapters 3 and 4 have also shown that GI provides a very relevant and feasible framework for both ecological and monetary soil valuation. The concept of GI, developed in spatial planning practice (Sandström, 2002), provides a direct connection with public policy instruments — a link that is very welcome when discussing governance.

4.5.3 From a linear to a circular economy: soil as a resource versus soil as waste

As opposed to the previous examples, this approach does not deal with *in situ* soil but rather focuses on the related material and waste streams. Generally, it also highlights the non-biotic component of natural capital, which has received limited attention so far.

Some data on excavated soil classified as both contaminated and non-contaminated waste, exist in European waste statistics. However, the data only exist for 2010 and 2012, and are not well harmonised across European countries. In 2012, 397 million tonnes of waste soil were generated across all economic activities and households in the EU-27, of which 96% came from the construction sector (Eurostat, 2015). The amount of waste soil generated ranges from 0 kg/capita in Greece and Malta to 12 300 kg/capita in Luxembourg.

([7]) https://www.pdok.nl/en.

Box 4.10 Protecting high-quality agricultural soil from urban development

High-quality agricultural soil within urbanised areas can be protected by effective spatial planning. This ensures that the future of soil for agriculture is maintained and contributes to the general prevention of urban sprawl.

Case: protection of agricultural soil in the Czech Republic

Photo 4.3 Agricultural land in Lidice (Czech Republic)

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In Prague (Czech Republic), urban development and the consequent loss of good-quality agricultural soil is minimised by controlling development on open land, focusing residential housing development on existing developments and giving priority to the reuse of brownfield sites. Under Act No 334/1992 on the Protection of Agricultural Land Resources, high-quality soil in the outer city belt is protected by limiting exemption from the 'agricultural fund' and defraying penalties for the exemption. Key to this process is soil survey and evaluation.

Source: Lexer et al., 2010.

Box 4.11 Green infrastructure as an investment project

Case: green cycle belt of Bruges (Belgium)

Vandermeulen et al. (2011) advocate that monetary valuation can help 'justify policy's support for and investment in green space' in urban areas (Vandermeulen et al., 2011, p. 198). To that aim they have created a model that can be used to put the value of GI investments into monetary terms, evaluating the GI project both at site and regional scale. By using cost–benefit and multiplier analyses, the net present value of the GI project could be estimated. Integrating the site-specific and regional valuation enables the user to identify '(1) whether benefits outweigh costs at project level, and (2) whether the marginal multiplier effects on the region outweigh the project level costs' (Vandermeulen et al., 2011, p. 200). The results show the total economic value of the GI project, which, in addition to the classical use and non-use value, includes the investment value. The investment value is 'the value generated by the creation of the GI and can cover the costs for purchasing land, the costs for designing and constructing the GI or the income generated through the start-up and exploitation of the GI' (Vandermeulen et al., 2011, p. 199). Nevertheless, the authors acknowledge that many of the values generated by GI are not economic in nature (e.g. landscape improvement), and would require alternative methods, although using a common scale or denominator.

This indicates not only differences in generation of waste soil data, but also in reporting or in classification of soils as waste. Better data may, however, exist at national, but more likely at municipal, level.

Reuse and recycling of soil from construction and demolition

> Construction and demolition waste is one of the most significant waste streams in the EU, accounting for approximately 750 million tonnes per year. It consists of numerous materials, including concrete, bricks, gypsum, wood, glass, metals, plastic, solvents, asbestos and excavated soil, many of which can be recycled (EC, 2015).

The European Commission is currently commissioning a study on Member States' management of construction and demolition waste ([8]). The data generated are expected to shed light on quantities of soil transported/traded within the construction sector.

Analysis of voluntary reporting data following the European List of Waste (ELW; Commission Decision 2000/532/EC) classification indicated that 'soil and stones containing dangerous substances' (originating from construction and demolition activities ([9])) constituted the single largest category of exported (shipped) hazardous waste in 2007 (Fischer et al., 2012), corresponding to nearly 687 kilotonnes of contaminated soil (19 Member States of the EU-27). This soil was shipped either to be cleaned or to be disposed of in a landfill. Generally, the contamination level was not homogeneous, and this waste category included soil of which the actual contamination level was unknown.

However, reporting on waste import/export only applies to hazardous waste, in compliance with Annex III of the Waste Framework Directive (EU, 2008b). Accordingly, data on unexcavated (*in situ*) contaminated soil, and uncontaminated soil and other naturally occurring material excavated in the course of construction activities but planned to be used on site, are not included in the ELW reporting.

The construction industry has long operated under a linear economy following a 'take-make-consume-and-dispose' model, which is based on the assumption that resources are abundant, available and cheap to dispose of (Dickinson and Allen, 2015). By comparison, a circular (resource-efficient and low-carbon) economy seeks to preserve the value of the product/resource for as long as possible and virtually eliminates waste. 'A transition to a circular economy shifts the focus to reusing, repairing, refurbishing and recycling existing materials and products; what used to be regarded as 'waste' can be turned into a resource' (EC, 2014).

To apply the circular economy concept to soil, it is important to understand the pressures that the construction industry exerts on the soil resource, as well as the subsequent impact on soil (Dickinson and Allen, 2015), namely:

([8]) http://ec.europa.eu/environment/waste/studies/mixed_waste.htm#intro.
([9]) Construction and demolition waste constituted 23% of the total hazardous waste shipped across national borders, and was predominantly made up of polluted soil, contaminated wood and asbestos.

- creation of impervious surfaces, effectively resulting in soil sealing, which implies breaking the connection between the pedosphere and other spheres, likely resulting in reduced soil functioning overall;
- excessive trafficking on soil, which leads to soil compaction and a loss of soil structure;
- spillages of chemicals and/or fuels in soils, resulting in soil contamination;
- poor storage and handling of soil (including mixing of soil types with different characteristics and thus quality), resulting in a loss of nutrients, soil structure, and geotechnical properties;
- disposal of superfluous (often fertile soil that is not suitable for construction purposes for stability reasons) in landfills.

In the context of waste, the waste hierarchy is one of the key principles applied to increase resource efficiency. Priority is given to waste prevention, followed by preparation for reuse, recycling and other recovery, with disposal as the least desirable option. Three case studies illustrate this approach (Boxes 4.12–4.14).

Box 4.12 Public policy instruments to manage excavated soil as an asset

Case: Flanders (Belgium)

The Soil Remediation Decree (1995) provides Flanders (Belgium) with a powerful instrument for controlling soil contamination. Since 2004, the Decree has included a provision for reusing and recycling excavated soil. Initially, the focus was solely on the prevention of new soil contamination. More recently, the potential for using excavated soil as a substitute for primary minerals has been acknowledged.

The regulation for the reuse and recycling of excavated soil is based on 'the stand-still principle', meaning that 'there cannot be any deterioration in the current environmental condition and any increase in health or environmental risks must be avoided' (Dedecker et al., 2015, p. 1). In order to meet the predefined targets, the regulation imposes a soil quality survey. It also sets the conditions for the use of the excavated soil: the higher the contamination level, the more restricted the use.

Acceptable contaminant levels depend on threshold (based on the background level in the soil and soil remediation standards, which differ according to land use type), which are set in the Soil Decree. Non-contaminated soil can be reused freely as 'soil' (60–75 % in the period 2009–2013). Somewhat contaminated soil can be reused as 'soil' if the land on which it will be used is more contaminated, or it can be reused for specific building purposes (13–22 % in the period 2009–2013). If the contamination exceeds specific levels, the soil is considered waste and cannot be reused (8–15 % in the period 2009–2013).

However, the Flemish government promotes recycling of contaminated soils by imposing high environment taxes on the disposal of soil that can be treated (by biological, physico-chemical or thermal means) ([10]) in a landfill. Only soil that cannot be treated, can be disposed of in a landfill at lower environment tax levels.

The potential for excavated soil to be a substitute for primary minerals is derived from the 'monitoring system for a sustainable surface mineral resources policy' (Dedecker et al., 2015, p. 2). The monitoring system gives an overview of material market developments on an annual basis (imported primary materials, primary materials supplied by the domestic market, and secondary materials). Thus, in 2011, about half of the total demand of surface mineral materials was covered by secondary materials (alternatives) such as excavated soil and recycled granulates. Out of a total of 9–15 million tonnes of excavated soil (2012–2013), about 7–13 million tonnes is reused or recycled in construction projects; about 1.3 million tonnes cannot be reused/recycled because of the poor mechanical quality (strength or stability), and is commonly used for the restoration of abandoned quarries; less than 5 % is disposed of in landfills.

The Flemish legal framework on the use of excavated soil thus defines rules and methods that ensure human health, environmental protection and resource efficiency. Legal protection of the different actors (liability) involved is guaranteed by a traceability procedure. Nevertheless, reliable basic data on the total needs for primary mineral resources and the potential of possible alternatives to substitute primary minerals are needed to ensure the step towards resource efficiency.

Source: Dedecker et al., 2015.

([10]) About 800 kilotonnes of soil is treated on a yearly basis (2009–2013).

Box 4.13 Initiatives towards the sustainable use of excavated soil

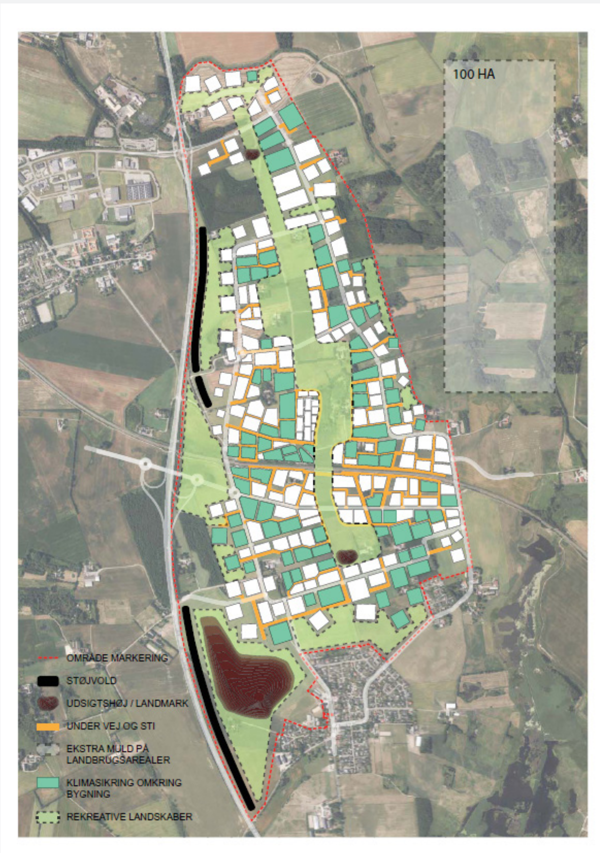
Case: Denmark

In most urban areas, construction activities often generate large quantities of soil, which are disposed of in ways that are not always beneficial to the environment. In the Copenhagen area, construction projects generate some 10 million tonnes of excavated soil per year. This soil is traditionally used for land-reclamation and noise barriers, or transported to distant locations where disposal is cheap. This causes excess transport emissions and a potential hazard to land and water resources. In recognition of this, new knowledge and tools (www.jordhaandtering.dk) have been generated, including a new website (www.jordbasen.dk) for soil-dating, which will increase transparency in the soil market. Changing the stakeholders' mind-set is a slow process, but examples of successful projects that are profitable may speed things up. Incentives, regulation and soil banks are already known to make it happen, for instance in the Netherlands.

Urban planning

The new city of Vinge is a 370-hectare development project in Denmark with an expected 20 000 residents and 4 000 jobs. At an early planning stage, it was decided that all soil produced from construction works should be reused locally, and an analysis of options was made, together with a strategy for local reuse of the soil (based on soil quality assessments). The analysis showed that the development of Vinge would result in the excavation and removal of approximately 2 million m^3 of soil. The analysis, however, also found that it was possible to use 3 million m^3 of soil in local projects, for example noise barriers, recreational landscapes, climate adaptation or the substitution of primary mineral resources in roads or other construction works (Figure 4.3). In order to evaluate the sustainability of the soil strategy, the environmental, social and economic impacts of the suggested options for reuse were rated. This showed that a 9 000 tonne CO_2-equivalent reduction could be obtained, plus financial savings of EUR 25-30 million.

Figure 4.3 Plan for the new city of Vinge



Source: Frederikssund Kommune (Municipality of Frederikssund).

Figure 4.4 Modelling of temporary soil storage in function of local needs (City of Copenhagen)

Source: NIRAS.

Mitigation of climate change

In an urban development project at Hillerød, analyses have shown that large areas are at risk of flooding as a consequence of increased precipitation. To mitigate this risk, a master landscape plan based on hydraulic models was made. The new landscape will be built using soil from the construction works in the urban area and from the surrounding landscape. By doing this analysis at an early stage of the planning process, mitigation of climate change by reusing soil is almost free of cost.

Landscaping for recreation and better public health

In a construction project in the city of Copenhagen, the municipality wanted excavated soil to be reused close to the construction site. A study of options for local reuse was made and a local former railway area, currently a public park, was found suitable for temporary soil storage. In order to reduce the negative effects on nearby residents, the temporary soil storage will be equipped with recreational features such as mountain bike trails, jogging paths and interval training paths, contributing to liveability and a healthy environment (Figure 4.4). When construction works are completed, soil from the temporary storage will be removed for reuse around the new building. Reuse of soil in this project saves a projected EUR 150 000, while the projected total building costs amount to EUR 8 million.

Creating value for builders

The integration of strategies for soil management at an early stage in the planning of construction projects provides more options and opportunities for the creation of alternative solutions at various stages of the project. Most of the benefits (saved primary resources, reused soil, reduced transport and economic gain) in the examples mentioned here are related to early planning and strategic thinking. Even though builders and constructors in general have an environmental approach, the 'value for money' factor has a significant impact on builders' decisions, especially so for public builders, who manage taxpayers' money. A sustainable approach to soil management, however, creates a broader value, not only for the investor but also for society, the climate and the environment. This is the whole idea behind the mantra: 'From waste to a resource'!

Facts of the Danish initiative

In 2013, the Capital Region of Copenhagen initiated a three year project to increase resource efficiency by transforming excavated soil from waste-status into a sustainable resource, preferably used locally in construction projects, roads, landscaping and mitigation of climate change, for instance. With regional funding contributing 50 % of a total budget of EUR 1.5 million, nine promising ideas and technologies have been tested and developed in nine individual public-private partnerships, including more than 45 municipalities, authorities, public and private builders, and organisations. The results are freely available and include websites, tools, guidelines, examples for inspiration and more.

The Danish Association of Construction Clients, the engineering companies SWECO Denmark and NIRAS, and The Danish Natural Environment Portal have been lead partners in the initiative and are currently involved in the implementation of the results. For more information, see: www.jordhaandtering.dk.

Box 4.14 Soil management in airport construction

The soil resource and materials for creating soil can be reused and recovered on site during construction. Reusing soil in construction projects conserves the soil resource and avoids the costs of off-site disposal and imported materials.

Case: London Heathrow Terminal 5 (United Kingdom)

Photo 4.4 London Heathrow Terminal 5

A strategic approach to soil management during construction of a new airport terminal avoided the need for off-site disposal of excavated materials and supported the creation of a new landscape. Requirements for different soil types were specified (e.g. texture, permeability, stone content, soil organic matter content, pH). An inventory of soil-forming materials was made (e.g. clay from tunnel excavation, river dredgings, stripped topsoil). Trials established the viability of meeting the use specifications. Careful formation and management of stockpiles avoided damage to soil prior to use. In total, 140 000 m^3 of topsoil and subsoil was created and used on site.

Source: O'Hare and Price, n.d.

5 Soil as an asset to the green economy

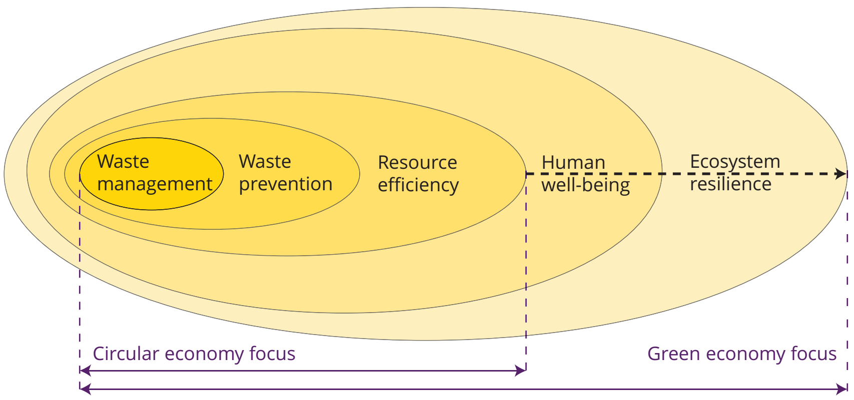
'The green economy is seen by the EU, UN and OECD as a strategic approach to the systemic challenges of global environmental degradation, natural resource degradation, natural resource security, employment and competitiveness' (EEA, 2015c, p. 152). Within the context of sustainable development, a green economy is one that (1) reduces environmental risks and ecological scarcities (ecosystem resilience), (2) improves the efficiency of the use of (natural) resources within the economy (resource efficiency) and (3) enhances human well-being and fair burden-sharing (social equity) (UNEP, 2011; EEA, 2012). The green economy concept thus emphasises economic development that is resource efficient, equitable across society and in line with environmental limits.

The perspective of the green economy also provides a framework for integrated and coherent policy action (EEA, 2015c). Figure 5.1 illustrates how policy priorities can be nested and integrated across the sustainability dimensions. It also shows that the focus of the green economy goes beyond the circular economy, which concentrates on optimising material resource flows by cutting waste to as close to zero as possible. Applied to the soil resource, the circular economy viewpoint may make sense where soil is used as a material (e.g. in the construction sector). However, appreciating soil as a productive asset requires acknowledgement of the full spectrum of uses that humans can make of soil to fulfil their needs.

The soil's use value is enhanced where population densities are higher, namely in the core urban and urban fringe zones. Recognising its use value requires identifying both the abiotic and the biotic characteristics of the soil resource, including its central role in delivering valuable ecosystem services to society. Adopting an asset-based approach to soil governance demands ecosystem-based management. From a resource efficiency perspective, it is thus essential that the ecosystem services dimension of soil is consistently reflected in policy action. Logically, in relation to soil governance, the resource efficiency domain has to be stretched beyond a focus on the circular economy, so that it coincides with the green economy perspective.

Originally an economic concept, resource efficiency refers to a level of resource use and management at which costs and benefits are balanced, representing the optimal level of resource use in production and consumption activities. However, information is usually lacking to define this optimal level, and cost effectiveness may give more useful practical guidance.

Figure 5.1 The green economy as an integrating framework for soil resource efficiency



Source: Adapted from EEA, 2015c (focusing on policies related to material use).

Cost effectiveness means that either an outcome with given resources is maximised or a given outcome with minimum resources is achieved. Applied to land, and considering resource use as a socio-economic (rather than a purely financial) cost to society, cost effectiveness could be understood as either achieving the maximum level of land services (e.g. flood regulation, carbon capture) possible with the land resources available or obtaining a given level of services with the lowest amount of resources possible (i.e. by limiting the input of the land resource itself).

The second option corresponds to a situation in which targets — often expressed as land use and/or land management limitations — have been proposed or agreed through political processes. At global level, this resonates with the UN Rio+20 goal of a land-degradation-neutral world in the context of sustainable development (UNGA, 2012). At EU level, the 7th EAP aims to ensure that, by 2020, land is managed sustainably, and specifically suggests an objective of 'no net land take by 2050' (EU, 2013a). This policy initiative also ties in with targets of the EU 2020 Biodiversity Strategy, in particular to 'restore at least 15% of degraded ecosystems by 2020' (EC, 2011b). These concerns are also echoed in the Sustainable Development Goals (UNGA, 2015). The biological nature of the resources involved calls for ecosystem-based management to implement these broad policy initiatives at national to local level (cf. asset- and place-based multi-level land governance framework).

The first option corresponds to increased productivity of the land resource, or increased economic returns (i.e. more euros per land area). When soil is primarily used as a medium for human-made structures, land prices are generally a poor reflection of the multiple use values of the soil resource. This points to a lack of internalisation of the real costs and benefits of different uses in market prices, including a failure to allow for soil-based services. When the focus of soil resource use is on biomass production, increased land resource productivity implies an intensification of service 'extraction' through technological innovations and/or by intensifying the use of additional (manufactured) inputs (e.g. water, (in)organic fertilisers). This commonly results in increased environmental impacts overall (e.g. uncontrolled and excessive use of fertilisers and pesticides leading to diffuse soil and water pollution), leading to indirect costs to the wider community. These costs are not reflected in the costs borne by those with property rights, namely those who are getting the benefit or income from using the soil resource, an indication that the externalities are not internalised back to the users of the soil resource.

Soil is a component of the overall land system, and as such an essential component of natural capital from which humans derive a range of valuable services, although the contribution of the soil resource is often indirect and somewhat obscured. The systemic character of the resource efficiency challenge implies that there are interconnections and interdependencies between the resource use systems (EEA, 2015c). These interdependencies in using the mineral and energy, soil, water and biological resources underlines the importance of coherent approaches to resource efficiency. Indeed, focusing on resource efficiency for a single resource may result in unintended outcomes for other resources; for example, bioenergy production to increase energy security may result in pressures on the finite and non-renewable soil resource.

Taking account of synergies and trade-offs within single and across multiple resource use systems thus calls for governance actions that are not only integrated but also coherent across policy sectors. Optimising the use of the soil resource (i.e. the net value of future benefits and costs attributable to the soil resource) in the development of new urban areas, and the restoration and renaissance of existing degraded urban areas may thus refer to remediation of soil degraded by contamination, compaction and other damage, but may equally include the explicit incorporation of soil in GI projects. In that sense, the Green Infrastructure Strategy (EC, 2013) appears to be a promising instrument for valuing the ecological function of green areas — in line with the EU Biodiversity Strategy (EC, 2011b) and its restoration targets — and for valuing the soil's water retention capacity, in line with natural water-retention measures proposed as part of the Water blueprint (EC, 2012c). Urban planning and design have important roles to play; key factors of influence include the density of development (compactness, fragmentation) and the capacity of cities to infiltrate water (presence of green areas, permeable surfaces, capacity of urban drainage systems) (BIO Intelligence Service, 2014).

However, various knowledge gaps need to be bridged to live up to the spirit of a green economy. This report has identified various gaps in recognising the importance of soil in urbanised areas, that is in taking soil 'at face value'. These knowledge gaps give indications for a 'roadmap' that shows how to support soil resource efficiency in urbanised areas through integrated and coherent policy action:

- soil function and soil-based service description adapted to an urbanised context, taking account of the natural system limits (ecosystem resilience);

- a description of the impact of a dysfunctional soil resource on the environment (ecosystem resilience) and humans (human well-being);
- geo-spatial mapping — taking account of the spatial variability of the soil resource, which is particularly prominent in urbanised areas — of soil characteristics and functions relevant to urbanised areas, and of dysfunctions where resource limits have been exceeded;
- customisation of the information to professionals who are considered important actors in integrating soil information into governance practice, requiring:
  - adaptation to the needs of professionals dealing with spatial (including urban) planning and urban design;
  - awareness raising among professionals (spatial planners, urban designers) on the use value of the soil resource and its limited and non-renewable nature, as well as on the systemic and interconnected character of natural resource use in general;
- use of ecosystem-based soil information to support decision-making by:
  - emphasising the non-substitutable roles of soil in the land system;
  - taking an integrated and qualitative valuation approach based on an informed narrative, set within the relevant spatial and temporal context to highlight the soil-related benefits and costs of conserving the soil resource.

Various instruments are already being applied to support improved governance of the soil resource in urbanised areas, including regulation, economic incentives or public funding and awareness-raising measures. However, current governance of the soil resource in urbanised zones does not consistently consider (the importance of) soil explicitly, including:

- internalisation of costs and benefits via property rights regimes (public policy instruments);
- internalisation of society costs via the application of soil valuation through a market mechanism; relying on a market mechanism to recognise the value of soils is especially important when soil protection or conservation through legislation or some other policy measure may not exist or fail because of poor implementation.

However, overall, raising awareness of the productive asset value of the soil resource (across the spectrum of possible uses) is fundamental in driving the necessary societal behavioural changes and incorporating soil into the transition agenda towards a green economy.

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