Chapter 17
Pedometric Valuation of the Soil Resource

David G. Rossiter, Allan E. Hewitt, and Estelle J. Dominati

> *"[W]hen you can measure what you are speaking about, and express it in numbers, you know something about it; but when you cannot measure it, when you cannot express it in numbers, your knowledge is of a meagre and unsatisfactory kind".*
>
> William Thompson (Lord Kelvin)
> – Lecture to the Institution of Civil Engineers, 3 May 1883

Soil forms the thin skin of the Earth and is the site of many ecological processes, transformations, and fluxes. It forms the substrate for most of the activities that take place at the Earth's surface, including almost all food production and human occupation, and underpins both natural and managed ecosystems. Soils differ in their structure, composition, and ability to function under a use. Soil is a multifunctional resource that affects human well-being both directly (e.g., food provision) and indirectly (e.g., surface and groundwater supplies) and that affects all near-land surface ecological processes. Clearly, soil is "valuable" as that term is understood in common language. The pedometric program as outlined in this book, i.e., the development of "quantitative methods for the study of soil distribution ... as a sustainable resource," should therefore include an attempt to quantify this value. Chapter 1 of the present book lists as the third of four items on the pedometric

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D.G. Rossiter (✉)
School of Integrative Plant Science, Soil & Crop Sciences Section, Cornell University, Ithaca, NY 14850, USA

ISRIC-World Soil Information, PO Box 353, 6700 AJ Wageningen, The Netherlands
e-mail: d.g.rossiter@cornell.edu

A.E. Hewitt
Manaaki Whenua – Landcare Research, PO Box 6940, Lincoln 7640, New Zealand
e-mail: HewittA@landcareresearch.co.nz

E.J. Dominati
Farm Systems and Environment, Land and Environment, AgResearch Grasslands, Palmerston North, New Zealand
e-mail: Estelle.Dominati@agresearch.co.nz

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agenda "evaluating the utility and quality of soil," and it is in this sense that we attempt in this chapter to define and quantify the value of the soil resource. This process is referred to as "valuation."

As the review of Robinson et al. (2014) on the value of the soil resource states, "Common to all valuation is the initial and fundamental question, what is the valuation for? There must be a clearly defined policy objective or management purpose for valuation." In this chapter we consider that the fundamental reason to value the soil resource is to include a fair representation of the multifunctionality of the soil resource in any discussion about the use of natural resources and value-driven trade-offs in resource management discussions. Examples of synchronic and short-term resource management issues are taxation of agricultural lands and fair value in land swaps, e.g., in land consolidation programs. A diachronic longer-term example is the capability of soils to perform under different land uses and different management intensities, compared to one-off uses such as foundation for construction (urbanization) or as a mineable resource.

We proceed as follows. First, we define the concept of value and in what terms it can be measured. Second, we describe pedometric approaches to internal valuation, i.e., comparing soils to each other, such as land indices. Third, we link the concept of external value, i.e., comparing the value of the soil resource to other goods in monetary terms, to FAO-style land evaluation. We then expand the discussion of value by describing the multiple contributions of the soil resource to human well-being via the concepts of natural capital and ecosystem services. Finally, we propose an approach to measuring the value of the soil using these concepts.

17.1 Concepts of Value

The noun "value" has a number of meanings in both common and technical English. The simplest definition is "quality of an object that permits measurability and therefore comparability" (Robertson 2012). In this sense soils can be described by any number of measureable attributes such as effective rooting depth or available water capacity, and pedometrics provides tools for quantifying these. However, in this chapter we consider "value" in the broader sense of the word, namely, value as a suitably defined utility.

The Concise Oxford Dictionary gives the fundamental definition of the term "value" as "worth, desirability, utility, [and] qualities on which these depend," which begs the question "worth, etc. to whom?" If to humans viewed as economic actors, this is economic value, in the sense of the neoclassical economic theory of value. If to humans viewed as social animals, this is sociocultural value. If to ecological systems through interactions between its components, this is ecological value, in the sense of the ecological theory of interactions. If to humans viewed as economic actors through the contribution of natural ecosystems to the value of final economic goods and services, this is contributory value (Ulanowicz 1991), in the sense of the ecological economic theory of value as explained below. The same object will have different values depending on which framework is chosen.

All of these concepts of value are explicitly anthropocentric, which only says that humans are the ones to assign value, including value to other organisms, the ecosystem, or even the planet as a whole. We humans are the only ones in this conversation, and we can choose to include or exclude what we perceive to be value to other actors.

There is also a verb "to value," which has several meanings, including "to appreciate, prize," but in the sense of the noun "value" as used here, it means "to assign a value [the noun] to something."

There are two types of measures of value: internal and external with respect to the resource being assessed. Internal values use a ratio scale, i.e., with a natural zero representing no value, and with units of equal value, to compare two or more objects of the same type, in our case equal areas of soil. The scale is defined so that a given change in a measure attribute of the object represents a specific change in utility. This relation is not necessarily linear. By contrast, external values use a ratio scale to compare a unit area of soil to any other goods. External measures must use a scale that is commensurate with other "valuable" things. The obvious external measure of value is money. McBratney et al. (2014) state that only by placing a monetary value on something is it possible to include that object in an accounting procedure. Hewitt et al. (2015) respond that the importance of soil can be quantified in other terms, e.g., as contributors to ecosystem services provision, for use in policy discussions, without assigning a monetary value. We return later to this discussion.

In the second half of the twentieth century, some economists began to analyze environmental problems in economic terms in order to point out the dependence of human societies on natural ecosystems (de Groot 1992). They stressed that the undervaluation of the contributions of ecosystems to public welfare and economic growth was due in part to the fact that many of the critical nonmarketed contributions of ecosystems underpinning human economies were not adequately quantified in terms that could be compared to economic indicators (Braat and de Groot 2012; Costanza 1997). The resulting discipline of ecological economics (Costanza and Waigner 1991) sees global economies as a subsystem of the larger finite global ecosystem. Ecological economists question the sustainability of the current economy that does not internalize environmental impacts and does not see raw material and energy as finite resources. Ecological economics uses concepts from conventional neoclassical or welfare economics and expands them to include environmental impacts, ecological limits, finite natural resources, and issues of equity and scale as necessary requirements for increasing the sustainability of human activities (Martinez-Alier 2002). These economists have extensively discussed valuation (e.g., Farber et al. 2002; de Groot 2002; Gómez-Baggethun and Ruiz-Pérez 2011). Ecological economics emphasizes the interdependence of economic and social systems. Foundational concepts include natural capital and ecosystem services. These concepts will be extensively used in the following discussion and so are introduced next.

17.2 Ecosystem Approach to Value as Contributions to Well-Being

17.2.1 Natural Capital and Soil Stocks

Traditional methods to value soil, including land indices and FAO-style land evaluation (see below), are restricted to the value of soil as a medium of production, usually for agricultural production within locally important land use systems. The concepts of ecological economics are much broader and correspondingly more complex to apply in practice. In this section we define natural capital, in the following ecosystem services, and then we discuss how to use these concepts to value the soil resource.

Ecological economists define the concept of *natural capital* as “stocks of natural assets … that yield a flow of valuable ecosystem goods or services into the future” (Costanza and Daly 1992). Another definition is “the living and nonliving components of ecosystems – other than people and what they manufacture – that contribute to the generation of goods and services of value for people” (Guerry et al. 2015). The concept of natural capital comes from trying to frame the contribution to the economy of natural resources alongside manufactured capital (factories, buildings, tools), human capital (labor, skills), and social capital (education, culture, knowledge). This is explicitly an anthropocentric viewpoint to illustrate the dependence of human societies on natural ecosystems.

Natural capital can be separated into renewable natural capital (RNC) and nonrenewable natural capital (NNC). Ecological economists hold that sustainable economic activity is based on sustainable income coming from all capital types; this is termed sustainable ecosystem services provision, which requires constant natural capital (Costanza and Daly 1992). In agricultural systems as land use intensity increases, NNC is often reduced (e.g., topsoil is lost; K reserves are depleted), so that ecosystem services flows, and therefore incomes, decrease in the absence of investment in RNC. To keep income constant, total natural capital needs to be maintained, which requires that some of the income coming from nonrenewable resources be reinvested into renewable natural capital, for example, adding mineral nutrients from external fertilizer sources to replace those removed.

Natural capital stocks can be increased or decreased by management. For example, the fertilization strategies in the Brazilian Cerrado (Goedert 1983) use large doses of lime to neutralize the pH-dependent charge of these highly weathered soils dominated by Al; this also reduces P-fixation capacity so that P fertilizers are effective. These effects last for several years, and at large enough doses, the subsoil becomes semipermanently changed. Thus, the natural capital for crop production is enhanced. On the other hand, Noble et al. (2000) present a sobering example of permanent land degradation after 37 years of continuous cropping on an acrisol in northeast Thailand: organic matter decreased dramatically; the exchange complex became almost saturated with Al, which was backed by a large Al reserve from soil minerals. The cation exchange capacity was so reduced that cations released during mineralization of organic matter could not compete with Al and thus were leached out of the reach of plant roots. The natural capital for crop production was severely depleted.

The concept of changing natural capital by management is close to the concepts of McBratney et al. (2014) of *soil condition* relative to a reference state they call the *capability*: "an optimal capacity of the soil to which the current condition of the soil can be compared." Capability can be permanently reduced by erosion, soil sealing, or mining for raw materials. Less permanent changes in natural capital are changes in condition only, which may be reversed.

Soil natural capital stocks include inherent stocks that vary over long timescales (e.g., clay content) and manageable dynamic stocks that vary over short timescales (e.g., soil water content) (Dominati et al. 2010). Soil carbon stocks are familiar for their use in carbon inventory and as proxies for overall soil condition. Soil stocks also include nonmaterial soil properties such as energy (e.g., stored heat) and soil fabric (e.g., total porosity). These directly relate to the mass, energy, and organizational components of soil natural capital identified by Robinson et al. (2009). Topsoil stocks are generally more dynamic than subsoil stocks, since they respond more rapidly to management.

17.2.2 Ecosystem Services

An *ecosystem service* is defined as "the direct and indirect contributions of ecosystems to human wellbeing" (TEEB 2008). This concept is an explicitly anthropocentric viewpoint and focuses on what is important to humans. This is not as restrictive as it might first appear, because there is ample evidence that a healthy ecosystem benefits humans. The interconnectedness of the ecosystem means that some aspects that might at first seem to be unimportant for the provision of benefits fulfilling human needs, e.g., crop production, are in fact necessary components of the provisioning mechanisms. For example, soil organisms form a complex food web; the direct service of, e.g., waste recycling or pest population regulation cannot be isolated from this web, so that the health and biodiversity of the web have value. The concept of ecosystem services has led to a large literature, including an eponymous journal, to examine and extend it.

Ecosystem services can be grouped into three types: provisioning, regulating, and cultural (TEEB 2008; Dominati et al. 2010). These groups contain specific services that can be quantified for a specific land use. Provisioning services include provision of physical support; of food, wood, fiber, and other plant and animal products; and of raw materials. Regulating services include flood mitigation, regulation of greenhouse gases, and control of pest populations. Cultural services include landscape preservation. Figure 17.1 is taken from Dominati et al. (2010) and illustrates the relation between natural capital, ecosystem services, and human needs.

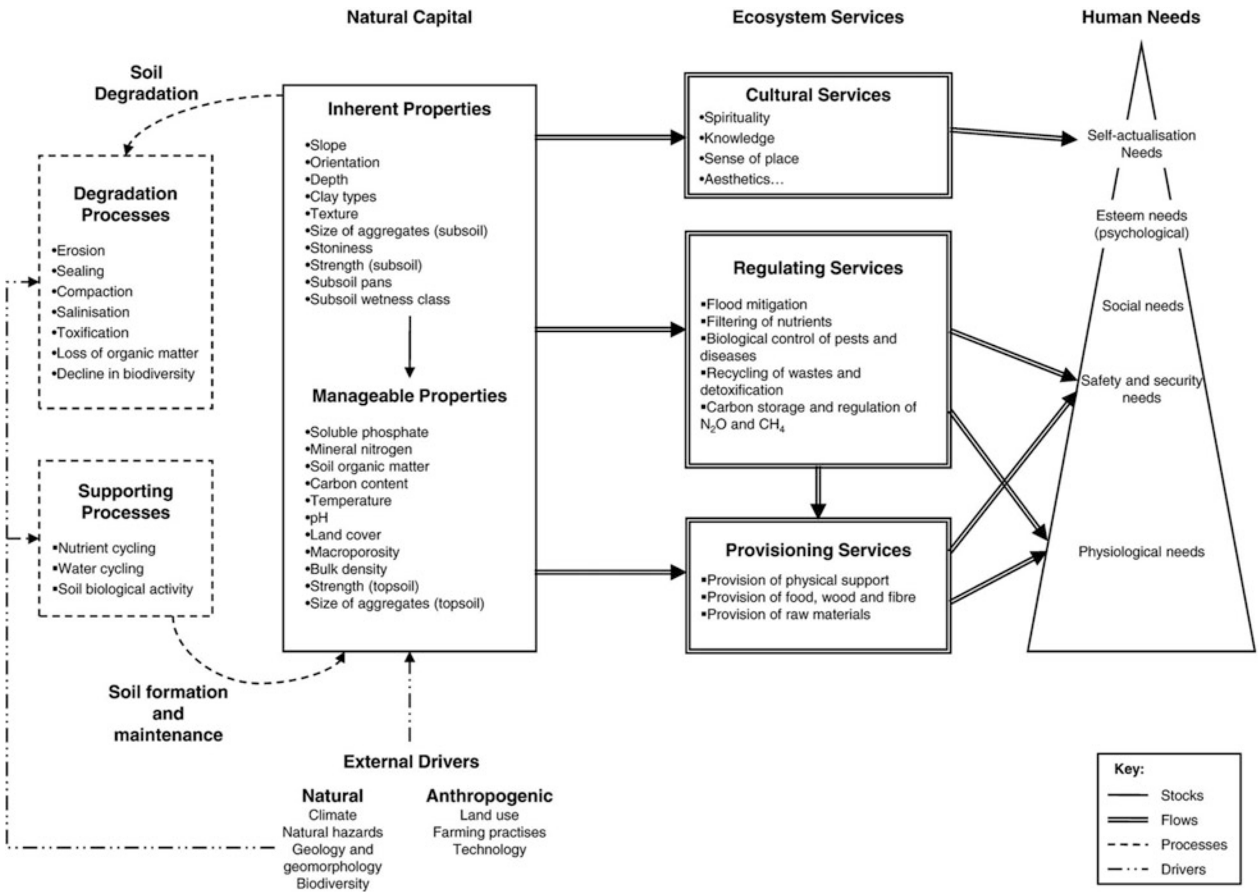

Fig. 17.1 Framework for the provision of ecosystem services from soil natural capital (From Dominati et al. 2010, Figure 2)

Earlier literature such as MEA (2005) and Haygarth and Ritz (2009) also refers to another category: supporting services. These are the ecosystem processes which support the provision of other services. These include primary production, soil formation, and nutrient cycling. These processes, not being directly of use to humans, are not referred to as services any longer to avoid double counting (Boyd and Banzhaf 2007).

The provision of ecosystem services, from a combination of land type and land use, can be quantified using information coming from a range of disciplines from ecology, to soil science, agronomy, or social science, as long as appropriate metrics have been defined for each service. These quantifications can then be used to put economic values on ecosystem services.

Economic valuation in monetary terms of environmental goods and services provides quantified information that can be used in a benefit-cost analysis (BCA). It has been extensively used for resource management and decision-making. The aim is to promote sustainable development by ensuring that policies fully account for the costs and benefits of development proposals on the natural environment. For policy-making in the context of agroecosystems and resource management, the more relevant application of economic valuation is to compare management options or assess an investment in either built (e.g., irrigation) or ecological (e.g., soil conservation) infrastructure that increase natural capital.

BCA is usually based on existing information, e.g., prices, as well as assumptions about future prices, regulations, and discount rates, and thus is not as objective as one would hope. However, failure to include ecosystem services in benefit-cost calculations implicitly assigns them a value of zero. To date this has been the norm, and this lack of accounting is accused of contributing to the depletion of natural capital stocks and increasing environmental problems (MEA 2005).

Traditionally the only soil contributions to ecosystem services that have been valued are those related to food provision, since that can be approximated by a farm budget. The other soil contributions to services provision are either unknown or considered too difficult to value. We propose to include all services in the valuation approach proposed here, as recommended by Dominati et al. (2016). This is intended to provide a holistic representation of the sustainability of land uses and the pivotal role the pedosphere plays in human well-being.

17.3 Soil vs. Land

Our aim is to value the soil resource. However, the soil is only one aspect of land as defined by the FAO (1976): "an area of the earth's surface, the characteristics of which embrace all reasonably stable, or predictably cyclic, attributes of the biosphere vertically above and below this area including those of the atmosphere, the soil and underlying geology, the hydrology, the plant and animal populations, and the results of past and present human activity, to the extent that these attributes exert a significant influence on present and future uses of the land by [humans]." Further, each land area has a definite location, which greatly influences its use and therefore value.

Soils occur at definite landscape positions with combinations of soil-forming factors, notably climate and relief, not independently of these land factors. Therefore, we propose to use the term "soil" to refer to the land resource including those soil-forming factors that are intimately bound to pedogenesis as discussed in Part VI of this book. Further, we accept the concept of USDA soil taxonomy (Soil Survey Staff 1999), which considers soil moisture and temperature regimes as soil characteristics. We exclude certain aspects of land, notably location, legal status, and water resources other than water that naturally affects the soil profile, and details of atmospheric climate that are not reflected in the more general concept of moisture and temperature regimes. For example, access to irrigation water affects land value, but the soil value for irrigated land uses only includes properties such as infiltration rate, water holding capacity, effective rooting depth, and soluble salts.

One method to disaggregate the effect of location from other land characteristics, including soil properties, is hedonic valuation within a single economic and market context (Rosen 1974). The hedonic hypothesis is that the value of an object (e.g., an area of soil) is equivalent to the utility received from its use, as revealed by differential prices associated with different levels of the attributes (e.g., rooting volume, available water capacity) that influence utility. Hedonic valuation estimates differences in value based on a set of attributes, typically in a spatial auto-regressive regression (SAR) formulation (Anselin and Bera 1998). The coefficients of the fitted model are interpreted as the elasticity of value due to changes in that attribute. Samarasinghe and Greenhalgh (2013) used this method to compare land values among 4477 farms in $a \approx 6000$ ha catchment in New Zealand. The type of current land use, distance to towns and roads, administrative unit, and their interactions all affected land value, as was expected. Soil characteristics including profile available water, rooting depth, gravel class, and drainage class also affected land value. Their model is a typical SAR model (Eq. 17.1):

$$\ln \quad V_i = \beta_0 + \beta_S S_i + \beta_A A_i + \beta_C C_i + \beta_T T_i + \beta_G G_i + \beta_L L_i + \lambda W_i u_i + \varepsilon_i \quad (17.1)$$

where V is the value (as measured) at site i, other capital letters represent factors that influence land price (Soil, surface Area, Climate, Topography, Geography, Location), β are the weights to be estimated, W is a spatial weights matrix which accounts for spatial autocorrelation in the model residuals, λ is the strength of the autocorrelation, and ε is the random error ("noise"). The partial derivative for S gives the hedonic value of the chosen soil property. This is most easily estimated when the other factors are constant, but the equation can be solved even if every observation has different combinations of factors. Since we consider soil climate and landscape position to be part of the definition of "soil," we would remove T and C and include them under S in this equation.

17.4 Pedometric Approaches to Internal Valuation

In internal valuation the aim is to establish a ratio scale by which soils used for a given purpose may be compared, rather than to establish a scale by which soils may be compared with other goods. This has historically been motivated by the need for equitable taxation, compensation for taking, and land swaps for agricultural land, not considering location or water resources, but usually considering permanent land improvements such as drainage or irrigation works. These valuations have used some kind of quantitative *land index* on a ratio scale (typically 0–100), which can then be used for direct comparison in land swaps or converted to valuation for taxation by their proportion of the total tax base. These approaches are pedometric in that they use measured soil properties, either on a continuous or classified scale, and relate these on a linear scale, so that a given change in each property has implicitly the same effect on the soil's value to the user at any position in the measurement scale, and in addition are comparable across properties. A land index could include non-soil land characteristics, e.g., climate and water resources, but the ones we examine in this section do not.

A well-known internal valuation is the Storie index from California (Storie 1933, 1978). This assigns points to attributes of the physical profile (factor A), the surface texture (B), the slope (C), and other factors (X) including drainage, acidity, alkalinity, nutrients, actual erosion, and microrelief. These are multiplied to reach a final rating. The Storie index and its derivatives are additive, multiplicative, or maximum-limitation indices, or some combination, as summarized by Riquier (1974). A number of soil properties that are known to influence agricultural production, e.g., rooting depth, stoniness, and presence of salts, are rated on a ratio scale, e.g., 0–100, and these ratings are combined by arithmetic or geometric averaging. This approach is also used in the former USSR, for example, the Ukraine (Starodubtsev et al. 2011), where it is called soil *bonitas*, from the Latin for "goodness"; this was likely derived from the German *Bonitierung* from the same Latin root. These indices have the following forms, depending on whether the combination is (possibly weighted by weights w_i) additive (see formula 17.2a), multiplicative (17.2b), or geometric (17.2c); in all cases the LC_i are the quantitative land characteristics normalized to a 0.1 range, from "worst" to "best." A Storie-like index S is then computed as

$$S = \sum\nolimits_{i=1}^{q} \mathrm{LC}_i \cdot w_i \quad \text{where} \sum\nolimits_{i=1}^{q} w_i = 1 \tag{17.2a}$$

$$S = \prod\nolimits_{i=1}^{q} \mathrm{LC}_i \tag{17.2b}$$

$$S = \sqrt[q]{\prod_{i=1}^{q} \mathrm{LC}_i} \tag{17.2c}$$

In Germany, agricultural and grazing land is compared by the *Landwirtschaftliche Vergleichszahl* (LVZ), i.e., agricultural comparison number, based partly on the *Bodenschätzung*, i.e., soil valuation, based on a law of October 1934 (Rothkegel 1950). This attempts to separate the intrinsic productivity of the soil from other factors that influence agricultural land value, such as field size and location with respect to main farm buildings and markets.

A problem with land indices is how to assign points to attributes singly and collectively. One approach has been to compare agricultural productivity on each soil, compared to some standard assumed to be without limitation. This implicitly assumes a reference land use system that could in principle be applied on all land areas. For example, in the German system all agricultural soils are compared to the most productive soils in Germany, a Chernozem at a reference location (Eickendorf in the Magdeburger Börde, Saxony-Anhalt), which is assigned full marks (100). Lower marks are estimates of the proportional productivity of common field crops under common management practices. Proportional yields are established by the long-term yields of reference crops at experimental plots (*Musterstücke*) spread throughout the country. Not all soils can be tested this way, so the characteristics of these sites are used to construct a point system that can be applied to any soil, based on its measured characteristics. For agriculture, the "agriculture number" (*Ackerzahl*, abbreviation AZ) is based on general soil texture (*Bodenart*), general type of parent material (*Entstehungsart*), and topsoil thickness (*Zustandsstufe*). For grassland, the "pasture number" (*Grünlandzahl*, abbreviation GZ) is based on general soil texture, topsoil thickness, mean annual air temperature, and drainage conditions (*Wasserverhältnisse*). In both cases, the number is supposed to be a proportional yield and thus is considered a fair basis for taxation or land exchange. If all parties accept the point system as fair, it is de facto the correct valuation for its purpose.

Another way to establish a point system is by regression analysis. For example, Olson and Olson (1986) used multiple regression analysis to estimate average maize yields in New York State (USA) using soil and climatic data, maize being a key indicator crop in that state; this work was later extended to Illinois (Olson et al. 2001). Their rationale was that "states [in the USA] need reasonable yield estimates to determine land value using an income capitalization approach to value for land appraisal and taxation ... our method is an improvement over the existing procedures based primarily on the collective judgment of experts." In New York the yield prediction was based on storage of available water (to counteract short droughts at key periods such as silking) and drainage (to avoid damage from surface ponding); the proportional predicted yields based on these were converted to a relative point system via the multiple regression equation. In this case atmospheric climate did not vary much within the study area (adjusted $R^2 = 0.13$ using only in-season rainfall), so most of the successful model fit (adjusted $R^2 = 0.80$) is attributed to the soil.

For geographically distributed observations, e.g., the reference crop yields and soil properties of the previous example, we cannot assume independence of the linear model residuals. Instead, the model formulation is

$$\mathbf{y} = \mathbf{X}\beta + \eta, \eta \sim \mathcal{N}(0, \mathbf{V}) \tag{17.3}$$

where $\mathbf{y}$ is the response vector, $\mathbf{X}$ is the model design matrix, and $\mathbf{V}$ is a positive-definite variance-covariance matrix of the model residuals, estimated by a covariance function of the separation between observations (see Sect. 3.2). This mixed model (fixed effects, i.e., regression parameters β, and random effects η, i.e., covariance of residuals) can be solved by restricted maximum likelihood (REML) (Lark and Cullis 2004).

Land indices are related to so-called soil quality indices, developed over the past 25 years (Karlen et al. 2003; de Paul Obade and Lal 2016) and the more recent concept of *soil health* (Moebius-Clune et al. 2016), which select soil properties though to be related to sustainable agricultural production and create indices based on their values, especially to highlight the effect of changes in management. However, these indices are not intended as measures of value, rather as measures of soil condition.

A modification of these approaches is to use fuzzy variables, rather than single values, to represent soil properties at a location and fuzzy logic using semantic import functions, rather than Boolean combinations, to combine them (Burrough 1989). This procedure results in a possibility of each rating, i.e., the combination does not give a single value, but rather a fuzzy variable. This avoids the problem of crisp thresholds. Several land evaluations since Burrough's paper have used this approach; a typical example is by Bagherzadeh and Gholizadeh (2016). Burrough gave an example of an FAO-style land evaluation for maize in Kenya, using six land qualities, each with asymmetric membership functions:

$$\begin{aligned} \mu_A(x) &= 1; x \geq c \\ \mu_A(x) &= \left(1 + a(x-c)^2\right)^{-1}; x < c \end{aligned} \tag{17.4}$$

where $\mu_A(x)$ is the membership for an area or site A on 0.1 for the value x of the land quality (here, an ordered class number); a controls the shape of the function, in particular the crossover value where the membership is 0.5 (equally possible and not possible); and c is the land quality value at the central concept, i.e., maximum suitability. The individual memberships were combined with a convex combination (weighted sum) operator to obtain an overall suitability rating μ_A:

$$\mu_A = \sum_j w_j \cdot \mu_{A_j}; \sum_j w_j = 1; w_j > 0 \tag{17.5}$$

See Sect. 5.5.1.1 for more details of fuzzy sets and fuzzy logic.

The first difficulty with any land index is the calibration of the scoring method. In many cases this is based only on expert opinion, due to the lack of sufficient experiments or observations. More fundamentally, these approaches do not consider the land use or management system, except in the most general terms. In the New York example, the indicator crop (maize) is typically produced by best practices on each soil (in particular, fertilization is based on soil testing), so that the land use and management system is implicit. The crop is widely grown and accepted by growers as an indication of agricultural land value, even for fields where it is not currently grown. However, even in New York there is a diversity of management (conventional tillage, no-till, residue management, organic amendments) that is not considered in this relation. This is clearly not appropriate in a more diversified environment and in addition only considers one ecosystem service.

These methods are also purely empirical, i.e., from calibration observations of system productivity to either expert judgment or parametric combinations. Beginning in the 1980s, many semi-physical models of the soil-plant-atmosphere system were built, including the Wageningen models WOFOST and its descendants (van Ittersum et al. 2003), CropSyst (Stockle et al. 2003), and the DSSAT suite (Jones et al. 2003). The supposed advantage is that these models can simulate the effects of changing conditions, including management interventions, based on physical principles. In practice they require extensive parameterization for each location and a very large minimum data set and in addition contain many empirical factors within each "physical" process. They are well suited for understanding the soil-plant-atmosphere-management system, e.g., the response of a system to changing conditions, but much less so for prediction to an accuracy that could be used as a measure of value.

17.5 Simple Pedometric Approaches to External Valuation

External valuation attempts to value the soil in relation to other goods. In this section we examine some cases where this can be fairly easily accomplished, if the soil is only considered a mineable or productive resource, i.e., a marketable good. In the following section, we expand the view of external valuation to ecological economics.

17.5.1 External Valuation of the Soil as a Mineable Resource

Soil can be used as a directly marketable raw material, i.e., a resource to be mined for its bulk properties or constituents. Portions of the profile can be removed for potting mix or for landscaping. In addition, constituents can be separated out from the whole soil, for example, gravel for roads, clay for bricks, or bauxite ore for aluminum. Such uses inevitably destroy some or all of the original resource, in favor of some immediate economic benefit. As such, external valuation is straightforward: the market value of the mined resource net of direct costs including transportation and any land reclamation. This is the minimum market value of the soil. The value of leaving the soil in place is much greater because of the soil's contribution to the provision of ecosystem services within an ecosystem for a range of potential uses.

Unless the soil is completely removed to bedrock, the degraded soil, i.e., what is left after mining, still contributes to the provision of ecosystem services but to a lesser extent. Dominati et al. (2014a) quantified and valued the services provided by a soil under pastoral agriculture, before and after erosion. The same could be realized for a soil before and after partial extraction of material: the difference in value of services associated with the removed soil material is an expression of the value of that material, in place, under a use.

If soil materials are relocated, e.g., for landscaping, the relocation process affects the services provided both where the soil is removed (loss of natural capital) and where the material is added (gain of natural capital). If substantial portions of the soil profiles are relocated, they are usually mixed during excavation, transport, and relocation. Soil functions rely on the character of the soil horizon sequence and may be changed when soil is disturbed. For example, the morphology that controls soil water dynamics may be radically changed by mixing, bulking, compaction, and by the effects on soil biological and soil organic matter processes. Relocation changes also include the incorporation of foreign materials, as in bricks and mortar, weeds and pests, and the litter of the Anthropocene. It is possible to compare total soil functional status before and after relocation as was done by Dominati et al. (2014b) in the case of erosion. The net market price should account for the extra functionality from the point of view of the landowner where the relocated soil is placed.

17.5.2 External Valuation by FAO-Style Land Evaluation

Land evaluation is the process of predicting the use potential of land on the basis of its attributes (Rossiter 1996). The FAO land evaluation framework (FAO 1976) and subsequent guidelines (e.g., FAO 1985) were important advances over land indices. One of the most important innovations was to make the land use system central to the evaluation. Thus, there is no single index of value, rather a rating for each actual or projected land use system. These may be combined in various ways to produce a single index.

In the FAO method, the evaluator specifies a set of land use systems ("land utilization types," abbreviation LUTs) that may be imposed on a land area and evaluates the suitability of each area for each use. This can be done in terms of limitations (so-called physical evaluations) or in financial terms, by linking limitations to reduced outputs, increased inputs to compensate, or longer term to production (Rossiter 1995). The soil is valued as part of the land, but not considering location; thus outputs and inputs are costed at the farm gate.

The external valuation procedure is conceptually simple but complicated in practice (Rossiter 1995):

1. Select representative LUTs that are feasible in a given socioeconomic-political context.
2. Describe these by their land use requirements (LUR), together with the financial effects of less-than-optimum LUR (lower yields, higher production costs, or longer time to product).
3. Build a model to evaluate the level of output(s) from each LUT, based on levels of the LUR.
4. Build models to evaluate each LUR from measurable land characteristics (LC).
5. Describe the LC of each land area to be evaluated, the land mapping unit (LMU).
6. For each LUT, apply its LUR models to each LMU.
7. For each LUT, combine the results of the LUR models in the LUT model.
8. Compute the financial balance for each LMU and LUT.
9. Compute financial indicators for each LMU, combining all the LUT results.

These combinations express the external value in whatever financial terms are considered most relevant. This could be the maximum return from any LUT ("highest use"), the average of all relevant LUTs ("versatility") or their standard deviation ("security"), or some weighted combination. Already by 1990 the ALES computer program implemented a sophisticated set of methods for these computations (Rossiter 1990), in which yields, time to production, and input amounts can be tied to severity levels of land qualities within a set of contrasting LUT.

This procedure can be applied to LUT where the return is from annual crops, in which case gross margin is the financial measure. It can also be applied to multiyear LUT, in which case the present value of a discounted cash flow is the financial measure; here a discount rate must be selected. These evaluation results can then be used in further planning procedures such as linear programming to reach a constrained optimum in a planning area (FAO 1993). They can also be combined to give a final value to each LMU. This can simply be the value of the maximally valuable LUT ("best use") or some statistic of the returns for the selected LUT, e.g., average (expected value over time) or standard deviation (as a measure of risk). These are suitable for external valuation in the current context, e.g., for land taxation, land taking, or land reallocation.

This approach is useful for comparing land areas in the short to medium term, and with only one ecosystem service in mind, e.g., food and fiber provision, along with any immediate devaluation of ecosystem services resulting from this primary goal, e.g., monetized externalities. It has several problems that must be addressed before it can be used to value the soil resource.

A first problem is that economic land evaluation is for the short to medium term, but land evaluation for multiple outcomes that preserve the soil resource needs to be long term. A net present value approach rapidly discounts the future.

A second problem is the selection of LUTs. In the short term, this can be based on project objectives or current options within the constraints under which the producer operates (Bouma 2001). However, when considering the value of the soil resource over the long term, it is not possible to anticipate all uses in the face of changing environments, both natural and especially socioeconomic-political; these are known as "option values." For example, an area currently used for crop production, and thus valued for a certain set of LUTs, may later be reassigned for groundwater supply or biodiversity conservation, with completely different LUTs. In addition, new technologies that cannot be anticipated may radically change land use requirements. Examples from the past are LUT based on conservation tillage (minimum disturbance of the soil) or even, if we go back far enough, LUT based on large quantities of manufactured fertilizers. Since the development of the Haber-Bosch process, large extents of sandy soils in northern Europe are now feasible for intensive crop production (which has also led to nitrate pollution of groundwater and streams), which would have been impossible with only organic inputs; this could not have been foreseen prior to 1913.

17.6 Ecological Economic Approaches to External Valuation

This section discusses how to use the concepts of ecosystem services, as provided by natural capital including soil, as the basis for external valuation as an extension of FAO-style land evaluation, as well as fundamental difficulties with this approach.

17.6.1 FAO-Style Land Evaluation and the Ecosystem Service Approach

The value reported by economic land evaluation may be purely financial, e.g., the cash flow to the land user from producing and selling agricultural products. However, it may be economic, including outputs with fictional (nominal) values, for example, the value of preserving an agricultural landscape, estimated by hedonic pricing, and outputs with negative value (externalities), for example, sediment or pollutants delivered off-site. In an economic analysis, shadow prices, representing the opportunity cost to society, may also be used (Huhtala and Marklund 2008).

Two trends emerging from reevaluation of the FAO land evaluation frameworks (FAO 2007) are the recognition of the wider functions and services provided by landscapes and the need for greater stakeholder participation in exploring the balance between economic, environmental, social, and cultural outcomes. Adding an ecosystem services approach to land evaluation would enable the supply of ecosystem services to be directly linked to the performance of a combination of land, land use, and management to deliver outcomes identified by stakeholders. This would provide a more complete picture of the efficiency of use of the natural resources, assist in defining natural ecosystem boundaries, and give quantitative information on the progress toward economic, environmental, social, and cultural outcomes desired by community (Dominati et al. 2016).

Ecosystem services provision cannot be defined abstractly – this is where a key insight of the FAO Framework comes in. It arises from the combination of land type and land use. As a simple example, consider a reserved area for groundwater recharge to an aquifer used for municipal water supply; this is a common practice in the central Netherlands. The main contribution of soil to ecosystem service here is the quantity and quality of precipitation reaching the groundwater table. However, if the same area is reserved for a production forest, the main contribution of soil to ecosystem services is the supply of water and nutrients to growing trees, as well as physical support. A deep sandy low-humus soil might be ideal for the first use: precipitation filters through a well-aerated sand column with no addition of impurities and with little loss due to evapotranspiration. The same soil is unsuited for the second purpose: the water quickly flows through the root zone leading to drought, and there are few nutrients and little nutrient retention.

17.6.2 Conceptual Framework for Valuation by Ecosystem Services

Dominati et al. (2010) proposed a conceptual framework that places soils into the context of wider ecosystems and human well-being, by showing how soils as natural capital are contributing to the provision of ecosystem services under a use. They proposed an approach to fairly represent soils when quantifying and valuing ecosystem services provision under any land use, in particular from agroecosystems. Their framework considers soil as a form of natural capital, i.e., semipermanent stocks, which provide ecosystem services under a use.

The framework consists of five interconnected components: (1) soil as a form of natural capital stocks; (2) the processes of formation, maintenance, and degradation of this natural capital; (3) the external drivers affecting these processes and thereby natural capital; (4) the provisioning, regulating, and cultural ecosystem services flowing from natural capital stocks under a use; and (5) the human needs fulfilled by these ecosystem services. Depending on which land use the soil resource is under, the services and their level of supply will vary.

This framework requires the analyst to specify one or more land use systems and some way to value each ecosystem service if economic valuation is desired. Samarasinghe et al. (2013) review several approaches to this.

Many ecosystem services benefit the land user (on-site services), but some have their primary benefit from off-site services. For example, the ecosystem service of food production, from the natural capital stock of soil nutrients and water, mainly benefits the farmer first. The ecosystem service of regulating stream and groundwater, from the natural capital stock of soil as a filter and buffer, benefits all users of the water, as well as stream ecology. These externalities (from the point of view of the land user) are rarely attributed to the landowner, e.g., by a pollution tax, although they are sometimes controlled by regulations, which limit the profitability of a land use (Huhtala and Marklund 2008).

17.6.3 Problems with Economic Valuation of Soil Contributions to Ecosystem Services

There are two aspects to economic valuation of ecosystem services that present difficulties. The first is assigning a value to a level of service, and the second is transforming this into money within the economy. Baveye (2015), using the term "soil services" for what we call the soil contribution to each ecosystem service, outlines these problems succinctly in regard to soils:

> Whereas the ecosystem services idea has been espoused enthusiastically by soil scientists, etc., very little progress has been made to date on the monetization of soil services. This may be due to the fact that it is not straightforward to assign a price to features or processes one does not understand satisfactorily, or the slow progress might be related more to uncertainty and lack of trust about what financiers might do with prices associated to soil services. Nevertheless, significant pressure is currently exerted on soil scientists by national governments and international agencies to engage actively with the ecosystem services framework. The challenge for soil scientists is either to find ways to monetize soil services meaningfully or to demonstrate convincingly (and relatively rapidly) that there are alternative paths that can be followed to preserve soils without necessarily putting price tags on their services.

A first difficulty is that valuation in terms of money is by nature short term. Methods to account for the time value of money, such as net present value, rapidly discount the future. The soil resource, with the exception of its value as a mineable resource, is meant to last "forever" in human terms. In this sense ecological economists argue that there should be no discounting at all, thereby giving equal value to current and future generations. This is well explained by Robinson et al. (2014) as the distinction between *value* and *price*, perhaps taking their cue from Oscar Wilde ("Nowadays people know the price of everything and the value of nothing"). Although both may be measured in money, "[e]conomic value seeks to identify all the final use and non-use, market and nonmarket values, and will often be unrelated to the price that soil commands as a commodity. This is because price only reflects purchase for a single or limited number of uses, whereas economic value tries to identify a combined value for all uses," i.e., the price plus consumer surplus.

In this respect Robinson et al. (2014) attempted to use prices to estimate the value of the topsoil, by calculating its quasi-replacement price: what it would cost the landowner to replace the topsoil, were it to be lost to, for example, erosion. We use the qualifier "quasi" because this price does not account for rebuilding topsoil structure after the (hypothetical) event of purchasing and spreading topsoil. This sets a floor for the value of the soil in place. In their calculation, to replace 30 cm of English topsoil would cost about \$110 k ha^{-1}. This is not a pedometric calculation, because it is just based on market prices for soil constituents. It does not consider how this is used and performance under that use.

But this is not the most serious problem. Fundamental to all attempts to value ecosystem services is the inescapable fact that markets, including virtual markets revealed by hedonic pricing, contingent valuation, or other indirect neoclassical methods, by definition are set by the present-day population. In so far as they value the future, this can be included, with a suitable discount rate, but humans can only understand relatively short-term outcomes, which is why net present value methods (time value of money) work well for short-term planning and (modestly) delayed gratification. But this does not extend to the services to future generations.

Lastly, Baveye (2015) points out that soil functioning is complex and therefore so are soil contributions to ecosystem services, making it impossible to disaggregate the economic values of ecosystem services to be attributed to specific soil properties. Changes in soil condition (McBratney et al. 2014) can be related to changes in ecosystem services provision and their value under a specific use, as shown by Dominati et al. (2014b). However, those relationships would be different for different land uses, so that any values obtained thus could not be generalized.

Thus, we conclude that a full monetary valuation of the soil resource in terms of its ecosystem services is impossible, in the sense that the future is inherently unknowable. However, it is possible to provide a monetary valuation to decision-makers operating in project (short-term) mode, i.e., a short time frame within which planning occurs. The study of Dominati et al. (2016) does just this for two soils under high-intensity dairy LUT. See Baveye et al. (2013) for an extended discussion of the selection of the time frame for valuation.

Therefore we propose to value the soil resource in terms of its contributions to all ecosystem services which can be envisioned at the site and defer the question as to how these can be expressed in monetary terms, and eventually monetized, to ecological economists. In any case we believe that any such monetization cannot represent the full value of soil over the long term.

17.7 Toward a Pedometric Valuation of the Soil Resource Based on Ecosystem Services

Dominati et al. (2016) argue that land evaluation, thus implicitly soil valuation, needs to be able to inform capability for multiple functions in order to recognize the whole range of ecosystem services provided by landscapes. In this sense, and summarizing the above discussion, we here present the outline of a method to value the soil in terms of its multiple contributions to ecosystem services provision, building on both the stock adequacy approach of Hewitt et al. (2015) and the indicator approach of Calzolari et al. (2016).

A pedometric valuation of the soil resource based on an ecosystem approach must first specify the geographic, political, and socioeconomic context within which it is carried out. This shows explicitly that any conclusions would have to be revisited if the context substantially changes. The first step, selection of LUT, depends completely on context, but so do subsequent steps. For example, if a new dam is to be built, soils upstream must now provide ecosystem services related to water quality and quantity for the reservoir (e.g., prevention of siltation to extend reservoir life or postpone dredging), whereas before this quality was for streams only. The soils now have no role in floodwater regulation downstream of the dam.

In addition, the valuation must be within a defined geographic area with a limited set of soil types. This is because the capability and condition (see below) are normalized to a 0–100 scale and thus require maximum and minimum values of soil properties, which only make sense within a defined area.

The method has the following steps, applied to each soil type to be valued:

1. Select a set of representative actual and potential land utilization types (LUTs), including management, which could be realized on this soil type. This list controls the list of ecosystem services (next step) and so should include enough uses to represent all feasible uses and their services. Note that this is not restricted to uses that are currently feasible, but also those that, should the socio-political-economic-technological context change, become feasible.

   The description of the LUT must be specific as it influences the choice of levels of soil properties required for ecosystem services provision and sustainable management. As in the FAO Framework, land management practices must be included in the definition of LUT.

   For example, high-intensity heavy-animal grazing requires specification of the soil's resistance to treading damage, as this is required for animal health, the health of the soil, and the quality of pasture production, whereas low-intensity grazing has minimal problems with such damage.

   As another example, low- or no-tillage annual field crop systems have less risk of erosion compared to clean-tillage systems. Thus, selection of levels of soil properties for ecosystem services related to surface water quality affected by agricultural runoff will be quite different between these LUTs.
2. For each LUT:

   (a) List the soil-mediated ecosystem services (provisioning, regulating, or cultural) to consider.
   (b) List the critical soil properties, that is, soil natural capital, influencing services provision. These are the *soil stocks*. These may be soil properties measured directly in the field, indirectly by proximal sensing, from laboratory analysis, or derived properties calculated using pedotransfer functions (see Sect. 6). Because of the difficulty of quantifying soil stocks, simpler methods using easily measured indicators may be used to get a first idea of the value of soil stocks (Robinson et al. 2012).

Each service is influenced by one or more soil properties. For example, filtering for groundwater provision is mediated by soil thickness, horizonation, and hydraulic conductivity; these all affect residence time of water in the soil column (Keesstra et al. 2012). It is also affected by the type and quantity of soil organisms.

(c) Determine *soil capability functions* for each service, that is, soil functions delivering levels of that service, in the context of the study area.

   (i) Specify an empirical relation between the identified critical soil properties and the service with pedotransfer functions or process models.
   (ii) For each critical property used in the empirical relation, standardize its range of values to a 0–1 scale relative to its contribution to the service. For example, a soil with rapid saturated hydraulic conductivity will be good (value approaching 1) for flood mitigation, but not good (value approaching 0) for contaminants filtration for groundwater supply. This can be a monotonic, optimal value range or even multimodal curve.
   (iii) Standardize the results of the empirical functions to a continuous 0–1 scale.

(d) For each service, determine the adequacy of the values taken by the soil capability function to the requirements of that land use and management, standardized to a continuous 0–1 scale. This can be done by using agronomic principles or by quantifying with models the provision of each ecosystem service at the extremes of the range of step (4), as well as some intermediate values, and then standardizing these values.
(e) Develop a suitability function, not necessarily linear, to combine the results of the individual services provisions into a single value for ecosystem services provision. This can be evaluated at all values of the soil capability functions.

3. Present suitability ratings for the selected LUTs, both for the individual (step (d)) and overall services (step (e)).

The result is a pedometric suitability rating for the services provision under each LUT carried out on each soil type for which it has been selected. The individual services ratings can then be used directly in multi-criteria evaluations or combined in optimization frameworks; this is beyond our pedometric scope.

Figure 17.2, from Dominati et al. (2016), shows the results of this method for a single LUT (a dairy-based agroecosystem) on two contrasting soil types. This spider diagram shows complete fulfillment of a service at the outer edge (1) and no fulfillment at the center (0). In this diagram there is no attempt to combine the separate services as in step 2(e) of our method; a multi-criteria weighting could be used for this. We clearly see that one soil, Horotiu, has much superior performance for P filtering but much worse than the other soil, Te Kowhai, for net C accumulation; how these are weighted is a social issue.

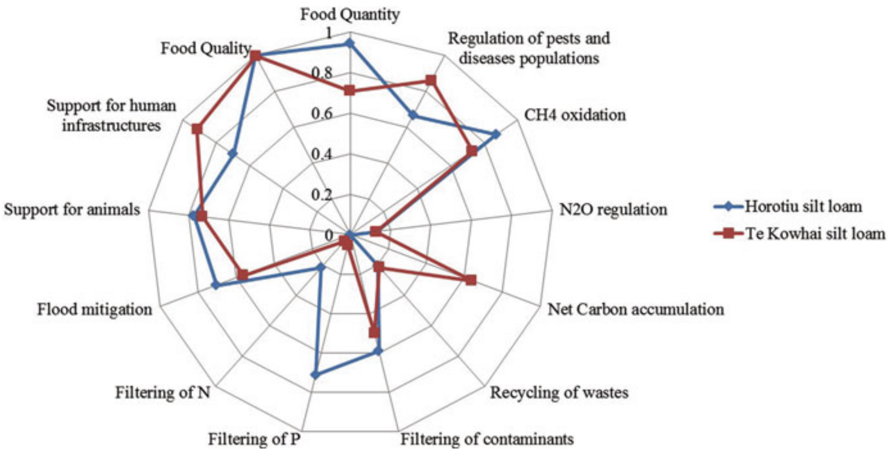

Fig. 17.2 Performance of ecosystem services delivery for two agroecosystems, based on biophysical measures (From Dominati et al. 2016, Figure 2)

17.8 Spatialization

The basic areal unit to be valued is the soil profile (Sect. 8), often called a pedon, in practice some small representative volume, e.g., 1 m^2 surface area to some depth (Hewitt et al. 2015) considered as the point support. The profile comprises a bundle of soil stocks that are the soil properties needed to drive the soil processes that result in soil functions that contribute to the provision of ecosystem services under a use. The profile is considered to represent a grid cell of some desired resolution or a sufficiently homogeneous map unit such as a consociation of a single soil series. The per-profile value can then be multiplied by the number of point support units in a map unit or parcel to give the value of the soil over that area. Conversely, quantities measured over larger land areas, e.g., crop production per hectare, can be normalized to the point support.

However, this concept of the areal unit to be valued assumes that (1) the area of soil to be valued is isolated and (2) ecosystem services are provided vertically (from atmosphere through soil to hydrosphere), so that mapping soil value would be a simple case of valuing at each small area. Of course the soil cover is more or less continuous, so grid cells at any resolution are connected to their neighbors, and many processes operate also horizontally, e.g., throughflow on hillsides and accumulation of materials at seeps and footslopes.

Soil functions may also depend on some minimum contiguous surface area of soils with similar function: a small patch of alluvial soil will hardly provide any floodwater regulation ecosystem service, whereas a floodplain commensurate with maximum overflows will provide this service.

Functions may also depend on some spatial pattern of contrasting soils with different processes. A well-known example is the production function for traditional grazing of upland soils in the wet season and lowland meadows in the dry season. These two must be combined for successful year-round grazing.

This implies that for some ecosystem functions, we cannot evaluate its capability function for a grid cell or map unit in isolation; the adequacy of a functional unit (e.g., toposequence for stream recharge) must be evaluated as a unit and then allocated across the components of the functional unit.

17.9 Communicating Value

The valuation proposed in this chapter goes far beyond what the public naïvely consider to be the value of soil. If they consider it at all, they would most likely consider the relative value for agricultural production of typical adapted crops under typical current management. Many services are not appreciated, e.g., the regulating services for ground and surface water quality, and in addition these are rarely directly valued back to the soil that contributes to them. In a few high-profile cases such as the New York City water supply (Pires 2004), there may be some public awareness that the natural capital providing the services sustainably (e.g., water purification) is an alternate to high-cost built capital providing the same services. Simply reporting that "scientists" or "experts" have used a complicated method (no need to worry about the details) and have valued a soil at so many dollars per hectare is unlikely to satisfy the general public, let alone policy makers. The ecosystem services that were valued must be listed and their importance explained. Jobstvogt et al. (2014) faced a similar problem in communicating the value of another "hidden" resource, the deep sea. They chose the Ecosystem Principles Approach (Townsend et al. 2011), which presents the valuation in terms of general ecosystem principles that (it is hoped) are easily understandable, e.g., "[f]lora and fauna that filter food or nutrients from the water column and maintain a sedimentary lifestyle have a stabilizing effect on the sediment." For soils this would be something like "deep, well-aerated soils with good structure let rainwater infiltrate and pass at moderate speed through to groundwater."

17.10 Conclusions

Our proposed ecosystem services method has certain drawbacks. It is certainly laborious. A large number of LUT must typically be selected, because management options often have a major influence on ecosystem services types and levels of soil properties that affect them. Then, ecosystem services must be listed and their link to soil properties described. This may reveal a lack of understanding of soil processes or lack of computational models to quantify them. Even if this is successful, there may not be sufficient data on these properties in standard soil survey databases.

However, these deficiencies clearly motivate research into actual and potential LUT, ecosystem services provisions, and models. They also show gaps in soil properties in current databases and thus motivate new survey methods and soil resource inventory.

In the ecosystem services approach, we abandoned any attempt to place an external monetary value on the soil resource over the long term, let alone how to monetize soil services. If an external monetary value is needed, a FAO-style economic land evaluation or the approach of Dominati et al. (2016) can be used to compute a minimum value, under current conditions and for the short term; as we argued above, this cannot represent the long-term value of the soil resource. However, it is hoped that our proposed method can be used in each policy-making situation to provide a relative valuation that is commensurate with valuations of other resources, which can be used for decision-making. In any case, simply pointing out that soil has value and quantifying the many services it provides should raise public awareness of this resource, which is "hidden in plain sight."

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