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Review Paper

Soil quality – A critical review

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A R T I C L E I N F O

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A B S T R A C T

Sampling and analysis or visual examination of soil to assess its status and use potential is widely practiced from plot to national scales. However, the choice of relevant soil attributes and interpretation of measurements are not straightforward, because of the complexity and site-specificity of soils, legacy effects of previous land use, and trade-offs between ecosystem services. Here we review soil quality and related concepts, in terms of definition, assessment approaches, and indicator selection and interpretation. We identify the most frequently used soil quality indicators under agricultural land use. We find that explicit evaluation of soil quality with respect to specific soil threats, soil functions and ecosystem services has rarely been implemented, and few approaches provide clear interpretation schemes of measured indicator values. This limits their adoption by land managers as well as policy. We also consider novel indicators that address currently neglected though important soil properties and processes, and we list the crucial steps in the development of a soil quality assessment procedure that is scientifically sound and supports management and policy decisions that account for the multi-functionality of soil. This requires the involvement of the pertinent actors, stakeholders and end-users to a much larger degree than practiced to date.

1. Introduction

Soil quality is one of the three components of environmental quality, besides water and air quality (Andrews et al., 2002). Water and air quality are defined mainly by their degree of pollution that impacts directly on human and animal consumption and health, or on natural ecosystems (Carter et al., 1997; Davidson, 2000). In contrast, soil quality is not limited to the degree of soil pollution, but is commonly defined much more broadly as “the capacity of a soil to function within ecosystem and land-use boundaries to sustain biological productivity, maintain environmental quality, and promote plant and animal health” (Doran and Parkin, 1994, 1996). As Doran and Parkin (1994) state explicitly, animal health includes human health.

This definition reflects the complexity and site-specificity of the belowground part of terrestrial ecosystems as well as the many linkages between soil functions and soil-based ecosystem services. Indeed, soil quality is more complex than the quality of air and water, not only because soil constitutes solid, liquid and gaseous phases, but also because soils can be used for a larger variety of purposes (Nortcliff, 2002). This multi-functionality of soils is also addressed when soil quality is defined from an environmental perspective as “the capacity of the soil to promote the growth of plants, protect watersheds by regulating the infiltration and partitioning of precipitation, and prevent water and air pollution by buffering potential pollutants such as agricultural chemicals, organic wastes, and industrial chemicals” (National Research Council, 1993 as cited in Sims et al. (1997)). Soil quality can be assessed both for agro-ecosystems where the main, though not exclusive ecosystem service is productivity, and for natural ecosystems where major aims are maintenance of environmental quality and biodiversity conservation. Given the scope and readership of this journal, the “non-ecological functions” of soil *sensu* Blum (2005), such as the physical basis of human activities, source of raw materials, and geogenic and cultural heritage, are beyond the scope of this review.

Extrinsic factors such as parent material, climate, topography and

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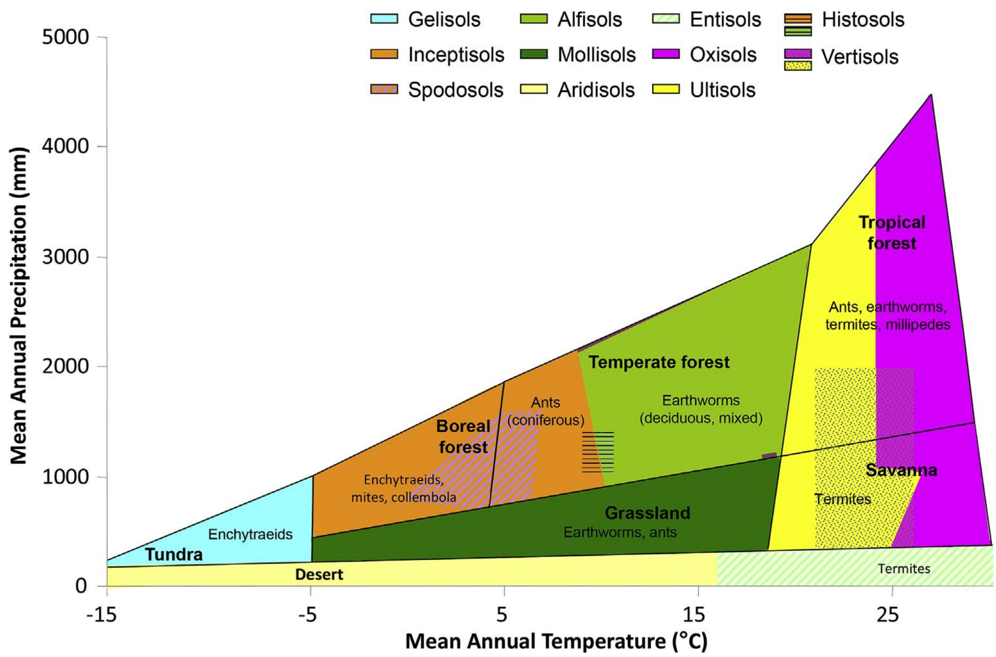

Fig. 1. Abiotic and biotic factors constituting soil quality in the soils of the world (modified from Brussaard (2012)). Reproduced with permission from Oxford University Press (www.oup.com).

hydrology may influence potential values of soil properties to such a degree (Fig. 1) that it is impossible to establish universal target values, at least not in absolute terms. Soil quality assessment thus needs to include baseline or reference values in order to enable identification of management effects. Soils often react slowly to changes in land use and management, and for that reason it can be more difficult to detect changes in soil quality before non-reversible damage has occurred than for the quality of water and air (Nortcliff, 2002). Therefore, an important component of soil quality assessment is the identification of a set of sensitive soil attributes that reflect the capacity of a soil to function and can be used as indicators of soil quality. Because management usually has only limited short-term effects on inherent properties such as texture and mineralogy, other indicators, including biological ones, are needed. The distinction between inherent (static) and manageable (dynamic) attributes, however, is not absolute and also context-dependent (Schwilch et al., 2016). For example, stoniness as an inherent property is nevertheless manageable, e.g. by removal of stones from an area to facilitate tillage and to build separating walls between fields, or by addition of gravel and stones to improve friability, to accelerate soil warming in spring or decrease evaporation. Soil management by humans has even given rise to separate classes in the soil taxonomic system, such as Plaggic anthrosols, the plaggen soils of northwestern Europe (e.g., Blume and Leinweber (2004)), and Terric anthrosols, the Amazonian Dark Earths, also known as Terra Preta de Índio (Glaser and Birk, 2012).

The history of the concept of soil quality shows that it is rooted in two different approaches that either put more emphasis on the inherent soil properties or on the effects of human management. The oldest mention in the scientific literature is by Mausel (1971) who defined soil quality as “the ability of soils to yield corn, soybeans and wheat under conditions of high-level management. The choice of these crops to reflect soil quality in Illinois is due to their overwhelming agricultural economic dominance.” This definition emphasises agricultural production and is linked to land evaluation (see below). A similar description was provided by SSSA (1987; cited in Doran and Parkin, 1994) as the “inherent attributes of soils that are inferred from soil characteristics or indirect observations”. This definition is comparable to the more recent term soil capability, defined as the intrinsic capacity of a soil to contribute to ecosystem services, including biomass production (Bouma et al., 2017). The emphasis on inherent, more static soil properties was closely connected to soil taxonomy. It also took management for granted (“under conditions of high-level management”), without specifying those conditions. Larson and Pierce (1991) expressed uneasiness with the focus on agricultural productivity and proposed to disconnect soil quality from productivity. Doran and Parkin (1994) observed that definitions of soil quality included the capacity of soils to function sustainably, but likewise considered the focus on production to be too restrictive. They wanted a definition of soil quality to stress the main issues of concern regarding soil use. Besides productivity, they therefore included the ability of soils to contribute to environmental quality and to promote plant, animal and human health in their definition as cited above.

The concept of soil quality by Doran and Parkin (1994) was heavily criticized in a series of papers (Letey et al., 2003; Sojka and Upchurch, 1999; Sojka et al., 2003). That criticism contained various elements. First, these authors claimed that the concept of soil quality could transform soil science from a value-neutral science into a value system and even referred to soil quality as promoting ideas of a politically correct soil. Second, they expressed discontent with the idea of a universal soil quality index, to which they referred as institutionalizing soil quality. Third, they criticized the concept because of its bias towards certain soil types as a consequence of the focus on intrinsic properties. And finally, they criticized the definition because in its original form it puts too much emphasis and value on a limited number of annual crops that provide cheap food and that are heavily subsidized. Their proposal to replace the term soil quality management by the term quality soil management did not find support, but their criticisms did influence the further development of an operational concept of soil quality, in which management has become the central issue: agricultural productivity does not hold a privileged position any longer, trade-offs are explicitly recognized at the expense of a universally applicable index, and the role of soil scientists in relation to societal stakeholders who manage soils (farmers, owners of land for nature conservation, policy makers, etc.) has changed. A particular recommendation of Sojka and co-authors was to speak of soil use rather than soil functions, so that the responsibility to maintain the quality of the soil can be clearly assigned to the user of the soil. Soil quality assessment then provides the scientific tools for evaluation of the management of soil resources, considering also the societal demands of the various benefits that soils, if managed well, can provide to humankind. The valuation of soil quality hence becomes connected to the valuation of the ecosystem services provided by soils. A further benefit of such a soil quality concept is that it raises awareness and enhances communication between stakeholders regarding the importance of soil resources (Karlen et al., 2001). Recently, there has been renewed interest in this educational aspect, either by focusing more on visual soil assessment (Ball et al., 2013) or by proposing interactive soil quality assessment tools, such as LandPKS (https://www.landpotential.org/) and the app currently being developed in the EU Horizon-2020 project 'Interactive Soil Quality Assessment in Europe and China for Agricultural Productivity and Environmental Resilience (iSQAPER - http://www.isqaper-project.eu/).

In this paper, we aim to critically review soil quality publications and assessment tools, especially with respect to soil quality indicators, in terms of commonalities, meaningful differences and omissions. To this end, the relevant definitions and terminologies are introduced in section 2, followed by an overview of approaches to soil quality assessment in section 3. The focus of this review is on analytical measurements. The most important approaches using visual soil evaluation in the field are only briefly presented, since visual soil assessments have been reviewed recently (Emmet-Booth et al., 2016). In section 4, the choice of soil quality indicators is discussed in-depth with respect to requirements of indicators and methods to select a minimum dataset. A compilation of the most frequently proposed indicators is followed by paragraphs on novel soil quality indicators with potential added value and on the interpretation of indicator values, including the potential aggregation into an operational soil quality index and its disadvantages. In the conclusions (section 5), we propose the crucial steps to be taken for successful soil quality assessment and analyze to what extent these have been implemented so far. Finally, fostering soil quality is considered in the wider context of enhancing environmental quality, embedded in an interactive process of co-creation of knowledge by scientists and other actors in urgent transitions towards sustainable use and management of natural resources (section 6).

2. Concepts related to soil assessment

2.1. Soil fertility, land quality, soil capability, soil quality and soil health

Various forms of soil assessment are encapsulated in different concepts. Apart from mining minerals, the main interest in soil has traditionally been in its potential for agricultural production. Assessments of the suitability of soil for crop growth may have been made even before the evidence of written records. Documentation can be found in ancient Chinese books such as "Yugong" and "Zhouli", written during the Xia (2070–1600 BCE) and Zhou (1048–256 BCE) dynasty, respectively (Harrison et al., 2010), and in the work of Roman authors such as Columella (Warkentin, 1995). Ethnopedology also provides several examples of indigenous soil classifications that focus on indicators that allow judgement of the suitability of particular soils for various crops (e.g., Barrera-Bassols and Zinck, 2003). The suitability of soil for agricultural production is captured in the concept of *soil fertility*, originating from the German literature on "Bodenfruchtbarkeit" that is predominantly aligned to crop yields (Patzel et al., 2000). Accordingly, the FAO describes soil fertility as "the ability of the soil to supply essential plant nutrients and soil water in adequate amounts and proportions for plant growth and reproduction in the absence of toxic substances which may inhibit plant growth" (www.fao.org). Mäder et al. (2002) extend that scope in proposing that a fertile soil "provides essential nutrients for crop plant growth, supports a diverse and active biotic community, exhibits a typical soil structure and allows for an undisturbed decomposition". Nevertheless, the concept of soil fertility is generally operationalized chemically and partly physically in terms of the provision to crops of nutrients and water only.

To address physical and/or biological characteristics of soil, other concepts are more commonly used. One of the earliest is *land quality*, which integrates characteristics of soil, water, climate, topography and vegetation (Carter et al., 1997; Dumanski and Pieri, 2000) in the context of land evaluation, which aims to assess the use potential of land, based on its attributes (Rossiter, 1996). An early comprehensive elaboration of the concept is the FAO Framework for Land Evaluation (FAO, 1976). Soil survey is part of land quality assessment for land evaluation. It is done once or only repeated over large time intervals, relying heavily on field observations, supplemented with very few measured parameters (Huber et al., 2001). Land evaluation anticipates decisions on the optimal allocation of land for various uses and is, hence, the first step to sustainable land management. In countries with low population densities, the main purpose of land evaluation in the past was to identify fertile land for agricultural production, whereas in more densely populated regions such as Europe it was more targeted at identifying deficient factors in agriculture that could be remedied, in particular by manuring (van Diepen et al., 1991). However, land evaluation has also been used as part of a strategy to assess broader land use options (van Latesteijn, 1995). Similarly, *soil capability*, i.e. the intrinsic capacity of a soil to contribute to ecosystem services (Bouma et al., 2017), provides a neutral assessment of what soils can do and how their potential can be reached.

Since Mausel (1971) introduced the term *soil quality*, it has sometimes been used in the context of land quality and land evaluation (e.g. Eswaran et al., 1997). Whereas land quality and land evaluation primarily address the inherent soil properties that do not change easily and are often assessed for the entire profile, soil quality is more focused on the dynamic soil properties that can be strongly influenced by management and are mainly monitored in the surface horizon (0–25 cm) of the soil (Karlen et al., 2003). However, when studying direct impacts of soil quality on water quality it is imperative that inherent soil properties in deeper parts of the soil profile are included in the assessment.

Typically, the concept of soil quality is considered to transcend the productivity of soils (Larson and Pierce, 1991; Parr et al., 1992) to explicitly include the interactions between humans and soil, and to encompass ecosystem sustainability as the basis for the benefits that humans derive from soils as well as the intrinsic values of soil as being irreplaceable and unique (Carter et al., 1997). The term soil quality in this broader sense was already used by Warkentin and Fletcher (1977). Recently, soil quality assessment is increasingly incorporated in land evaluation, as land evaluation procedures are now used in many different ways and for a range of purposes, including sustainable land management (Hurni et al., 2015), environmental risk assessments, monitoring of environmental change (Sonneveld et al., 2010) and land restoration (Schwilch et al., 2012). In the land-potential knowledge system LandPKS, general management options are based on long-term land potential (depending on climate, topography and inherent soil properties) and can be modified according to weather conditions and dynamic soil properties (Herrick et al., 2016). The integration of soil quality and land evaluation goes as far as developing soil natural capital accounting systems, stressing the importance of soils for human well-being (Robinson et al., 2017).

In a program to assess and monitor soil quality in Canada (Acton and Gregorich, 1995), the term soil quality was used interchangeably with *soil health* and, in spite of the wider context in which it was presented, defined primarily from an agricultural perspective as "the soil's fitness to support crop growth without becoming degraded or otherwise harming the environment". The term soil health originates from the observation that soil quality influences the health of animals and humans via the quality of crops (e.g. Warkentin, 1995). Indeed, linkages to plant health are common, as in the case of disease-suppressive soils (Almario et al., 2014). Soil health has also been illustrated via the analogy to the health of an organism or a community (Doran and Parkin, 1994; Larson and Pierce, 1991).

The debate about soil quality vs. soil health arose quickly after the concept of soil quality was criticized in the 1990s. In contrast to soil quality, soil health would "capture the ecological attributes of the soil which have implications beyond its quality or capacity to produce a particular crop. These attributes are chiefly those associated with the soil biota; its biodiversity, its food web structure, its activity and the range of functions it performs" (Pankhurst et al., 1997). These authors further consider "that the term soil health encompasses the living and dynamic nature of soil, and that this differentiates it from soil quality". They therefore "adopt the view that although the concepts of soil quality and soil health overlap to a major degree and that in many instances the two terms are used synonymously (….), soil quality focuses more on the soil's capacity to meet defined human needs such as the growth of a particular crop, whilst soil health focuses more on the soil's continued capacity to sustain plant growth and maintain its functions". Meanwhile, the debate subsided and partly changed focus. For example, Moebius-Clune et al. (2016) consider that soil quality includes both inherent and dynamic soil properties, and that soil health is equivalent to dynamic soil quality. The differential usage may also link to the observation of Romig et al. (1996), that, whereas soil quality is the preferred term of researchers, soil health is often preferred by farmers.

The differences between land quality and soil quality observed by Karlen et al. (2003) and between soil quality and soil health observed by Pankhurst et al. (1997) and Moebius-Clune et al. (2016) can be summarized in a transition in focus from land quality to soil quality and soil health going from inherent to dynamic soil properties. The website of the Natural Resources Conservation Service, USA (http://www.nrcs.usda.gov/wps/portal/nrcs/main/soils/health/) states that "soil health, also referred to as soil quality, is defined as the continued capacity of soil to function as a vital living ecosystem that sustains plants, animals, and humans". We conclude that the distinction between soil quality and soil health developed from a matter of principle to a matter of preference and we therefore consider the terms equivalent. We further express this by explicitly including the soil biota/biodiversity and related soil functions and soil-based ecosystem services in Figs. 1–3.

Like in land quality assessment and land evaluation, approaches to soil quality and soil health go beyond the reductionist approach of measuring (indicators of) soil properties and processes. Although such measurements remain important from a practical perspective (Kibblewhite et al., 2008a), the concepts of soil quality and soil health also include the capacity for emergent system properties such as the self-organization of soils, e.g. feedbacks between soil organisms and soil structure (Lavelle et al., 2006), and the adaptability to changing conditions.

2.2. Linking soil quality to soil functions and ecosystem services

Ecosystem services are defined as "the benefits which humans derive from ecosystems" (Costanza et al., 1997). With the early concept developed by Doran and Safley (1997), soil quality was addressing not only one ecosystem service such as provision of food, but also trying to represent and balance the multi-functionality of soil. This has recently been further embedded in the development of "functional land management", which assesses both the benefits and trade-offs of a multi-functional system for managing soil-based ecosystem services in agriculture (Schulte et al., 2014) and a wider range of land uses (Coyle et al., 2016).

Among scientists, the concept of ecosystem services is often used in connection with the concept of soil functions. 'Function' is, however, variably used as a synonym for 1) process, 2) function*ing*, 3) role and 4) service (Baveye et al., 2016; Glenk et al., 2012). Therefore, Schwilch et al. (2016) advise against using the term, but Baveye et al. (2016) note that function "in a narrow and well-defined context (…) has been used in connection with soils for over 50 years, and has served as a conceptual foundation for an appreciable body of research and significant policy making, at least in Europe" (e.g., the Soil Thematic Strategy of the European Commission, 2006). Therefore, we concur with Baveye et al. (2016) that "it makes sense to try to retain both "function" and "service" terminologies, as long as they can be articulated (…) with respect to soil properties and processes". In their seminal paper reconstructing how the notion that nature meets, or gets in the way, of the needs of people has pervaded concepts and theory in ecology vs. soil science, Baveye et al. (2016) argue that mainstream ecology, by its emphasis on organisms, tended to neglect the soil, in particular the non-

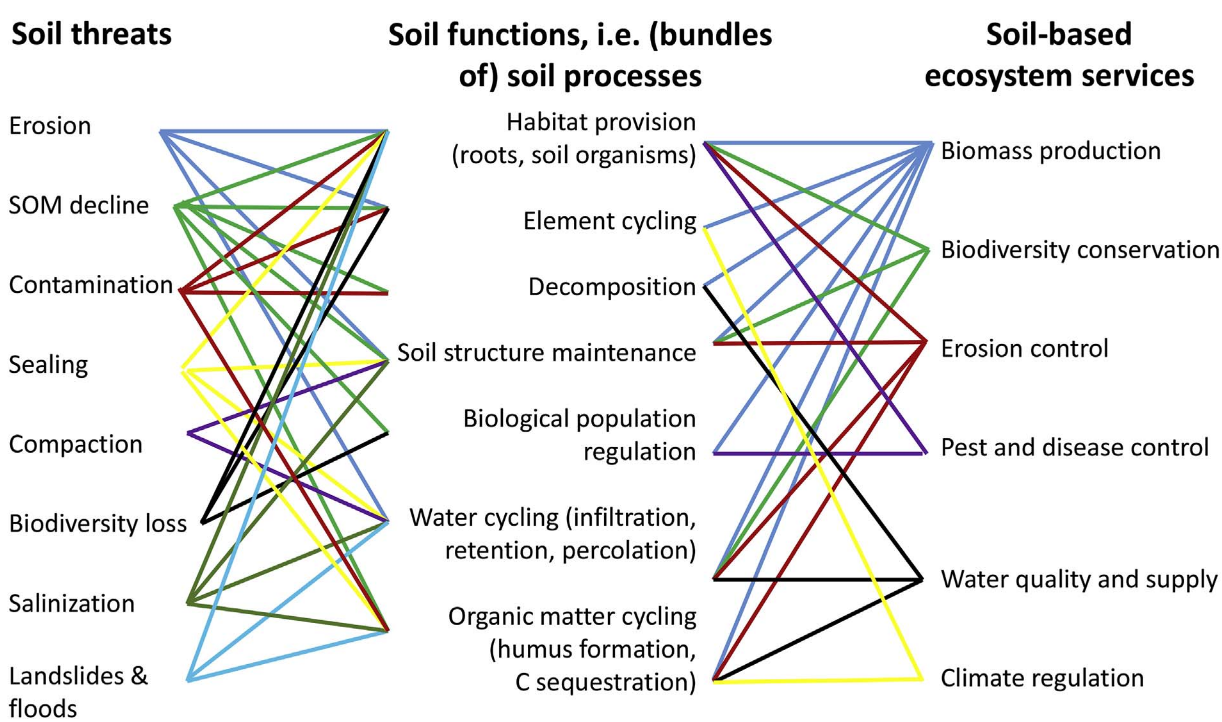

Fig. 2. Linkages between soil threats, soil functions and soil-based ecosystem services. Further developed from the scheme presented by Kibblewhite et al. (2008a) and modified by Brussaard (2012).

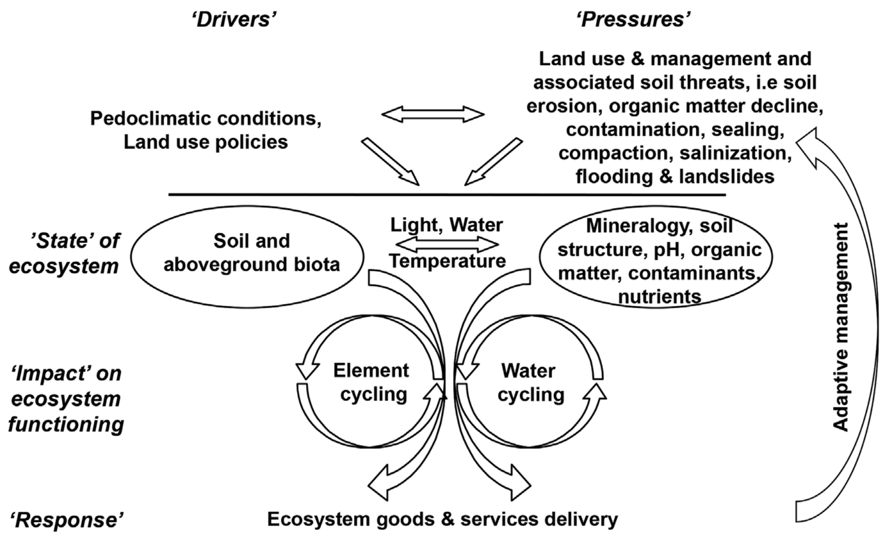

Fig. 3. The Driver-Pressure-State-Impact-Response framework applied to soil. Modified from Brussaard et al. (2007). Permission for reproduction granted by Elsevier.

living soil, whereas mainstream soil science tended to avoid the term ecosystem, emphasizing the importance of soil properties and processes in landscape terms. In accordance with Glenk et al. (2012), we define soil functions as (bundles of) soil processes that underpin the delivery of ecosystem services. This definition will suffice for all practical purposes related to manageable soil functions, which can be used to address the gap between "what is" and "what can be", based on soil capability, i.e. "what soils can do" (Bouma et al., 2017), which is, in the context of this review, what living soils can do. Complementary to this bottom-up approach, soil functions can be used in a top-down approach when identifying the gap between what is currently measured in soil assessment schemes and what should be measured in view of assessing the soil functions that are impacted by, or to be managed in view of current and upcoming policies (van Leeuwen et al., 2017), possibly through the use of environmental accounting systems increasingly adopted by policymakers, such as the soil natural capital accounting system proposed by Robinson et al. (2017).

Just as ecosystem services are influenced by (bundles of) soil processes, the latter are in turn affected by soil threats. The EU Soil Thematic Strategy identified the main threats to soil quality in Europe as soil erosion, organic matter decline, contamination, sealing, compaction, soil biodiversity loss, salinization, flooding and landslides (EuropeanCommission, 2002; Montanarella, 2002). Soil threats have been emphasized in order to inform risk assessment exercises indicating (geographical) areas where soil functioning is potentially hampered (van Beek et al., 2010). Different schemes linking soil-based ecosystem services and soil functions have been developed (Haygarth and Ritz, 2009; Kibblewhite et al., 2008a; Tóth et al., 2013), but none of them includes soil threats. The scheme presented by Kibblewhite et al. (2008a) and modified by Brussaard (2012) was developed as a conceptual basis for the iSQAPER project, including soil threats as affecting the various soil functions and associated ecosystem services (Fig. 2). The soil functions in Fig. 2 equate almost entirely to the "intermediate services" defined by Bennett et al. (2010), which are similar to the soil processes presented by Schwilch et al. (2016). The ecosystem services in this scheme can be seen as a soil-related sub-set of the ecosystem services mentioned in the Common International Classification of Ecosystem Services (CICES - http://biodiversity.europa.eu/maes/common-international-classification-of-ecosystem-services-cices-classification-version-4.3), currently elaborated in the Mapping and Assessment of Soil Ecosystems and their Services (MAES-Soil) Pilot project (https://webgate.ec.europa.eu/fpfis/wikis/display/MAESSoil/MAES+Soil+Pilot).

It has been argued that soil quality can indeed only be assessed in relation to one or several soil functions, ecosystem services or soil threats (e.g. Baveye et al., 2016; Bouma, 2014; Sojka and Upchurch, 1999; Volchko et al., 2013). Therefore, clear definitions of these terms as well as firmly established associations with soil quality indicators are the basis of any functional soil quality concept.

As soil quality plays a role in decision-making in the face of soil threats, the DPSIR (driver–pressure–state–impact–response) framework (EuropeanEnvironmentAgency, 1998) has frequently been adopted for use in EU policy to support decision-making and as a means to bridge the science-policy gap (Tscherning et al., 2012). Applying the DPSIR framework to soil (Fig. 3), "drivers" are pedoclimatic conditions and land use policies, while "pressures" are land use and management and the associated soil threats. Pressures and drivers and their variabilities and interactions determine the "state" of the soil, with subsequent "impact" on soil and ecosystem functioning, and the "response" in terms of the delivery of ecosystem goods and services. Subsequent adaptive management may be re-active to observed deterioration of soil functioning or pro-active to reach transitions to newly desired soil functioning. To assess any changes in the status of soil quality, assessment tools are needed, and these are the subject of sections 3 and 4.

3. Approaches to soil quality assessment

A plethora of soil quality assessment and monitoring tools have become available since the 1990s. Here, we give an overview of the main developments in different countries, before addressing aspects of soil quality indicators in more depth in section 4.

3.1. Analytical approaches to soil quality

National assessments of soil quality are often based primarily on analytical approaches (Table 1). One of the earliest national programs to assess and monitor soil quality was started in Canada in 1988 (Acton and Gregorich, 1995), using benchmark sites to assess changes in soil quality over time, especially in relation to the soil threats erosion, compaction, organic matter loss, acidification and salinization (Wang et al., 1997). While the Canadian soil quality monitoring program as such was not consistently continued, the data are still partly used in the assessment of agri-environmental indicators that cover soil, water and air quality (Clearwater et al., 2016). At a coarser scale, a GIS-based approach to characterize primarily inherent soil quality was presented by Macdonald et al. (1998).

Two major soil quality assessment approaches focusing at the plot scale were developed in the USA (Table 1). The Soil Management

Table 1
Major soil quality assessment approaches based on analytical indicators according to geographic origin (North America, Europe, China): Objectives, target group, scale, interpretation approach, website.

Country	Reference(s)	Name (if any)	Objectives	Target group (assumed)	Spatial scale	Interpretation	Website (if any)
Canada	(Acton and Gregorich, 1995; Wang et al., 1997)	National soil quality monitoring program	Assess status of and trends in soil health	Not stated (policy)	23 benchmark sites (5–10 ha each) across Canada	Mainly trend analysis	
	(Macdonald et al., 1998)		Assess inherent soil quality and susceptibility to change	Not stated (policy)	Regional and national	Rating procedures with respect to 4 soil functions	
USA	(Andrews et al., 2004; Karlen et al., 2001; Wienhold et al., 2004; Wienhold et al., 2009)	Soil management assessment framework (SMAF)	Evaluate management practices, educate about soil quality	Land managers, advisors, general public	Plot scale	Scoring curves, additive index	http://www.soilquality.org
	(Idowu et al., 2008; Moebius-Clune et al., 2016)	Cornell Soil Health Test	Assess soil health, address soil degradation, increase productivity	Farmers	Plot scale	Scoring curves, overall score	http://soilhealth.cals.cornell.edu
Australia	(Gonzalez-Quiñones et al., 2015)	Soil Quality Website	Benchmark sites, soil quality monitoring and education	Farmers	National and regional	Target values; threshold values wherever possible	http://soilquality.org.au/
New Zealand	(Lilburne et al., 2004; Schipper and Sparling, 2000; Sparling and Schipper, 2002; Sparling et al., 2004)	"500 soils project", soil indicator assessment (Sindi)	Assess soil quality for environmental reporting	Government; Sindi: regional council staff, landowners	511 sites across New Zealand, x soil types, 10 land uses	Comparative (compared to database) or according to target ranges	https://sindi.landcareresearch.co.nz
France	(Antoni et al., 2007; Arrouays et al., 2002; Arrouays et al., 2003; Martin et al., 1998)	Observatoire de la Qualité des Sols (OQS), Réseau des mesures de la qualité des sols (RMQS)	Assess soil quality for environmental protection, food security and sustainable management practices	Not stated (policy)	11 sites (OQS) 2000 sites (RMQS)	Mainly trend analysis	
UK	(Loveland and Thompson, 2002; Merrington, 2006)		Assess soil function of environmental interaction	Policy	National	Trigger values	
Ireland	(Bondi et al., 2017)	Soil quality assessment research project (SQARE)	Assessment of soil functions	Farmers	Plot (38 farms)		https://www.teagasc.ie/environment/soil/research/square/
The Netherlands	(Wattel-Koekkoek et al., 2012)	National Soil Quality Monitoring Network	Assess soil quality and land-use effects	Not stated (policy)	200 locations	Target values	
EU	(Huber et al., 2001)	European Soil Monitoring and Assessment framework	Provide objective, reliable and comparable information at European level	Policy			
	(Huber et al., 2008; Kibblewhite et al., 2008b; Stolte et al., 2016)	ENVASSO, RECARE	Assess soil degradation	Policy			http://esdac.jrc.ec.europa.eu/projects/envasso http://www.recare-project.eu/

Assessment Framework (SMAF) developed at the Soil Quality Institute (Andrews and Carroll, 2001; Andrews et al., 2004; Karlen et al., 2001; Wienhold et al., 2004, 2009) is rather unique in its flexibility in the selection of indicators. Based on a clear definition of the main ecosystem service(s) or management objective(s) to be addressed, a set of indicators is selected out of 81 potential indicators using selection rules. The user can disregard or alter the proposed minimum dataset as desired, although that limits comparability between sites. The interpretation of an indicator value is based on scoring curves and an additive soil quality index can be derived. The Cornell Soil Health Test (Idowu et al., 2008; Moebius-Clune et al., 2016) is much more standardized and targeted directly at land users, offering various soil health testing packages for farmers, landscape managers and others, and supplying them with management advice together with the results.

In New Zealand, a nationwide survey of seven soil quality indicators at 511 sites aimed at establishing benchmark values across all major soil types and land-uses (Lilburne et al., 2002, 2004; Sparling and Schipper, 2004; Sparling and Schipper, 2002). Based on these data, an online tool called Sindi (soil indicator assessment) was developed (Lilburne et al., 2002) that allows the comparison of measurements of soil properties in a given soil type with the information in the database.

In Australia, a consortium of public and private partners provides fact sheets and regional, soil type-specific critical threshold values of a range of soil quality indicators for impact on agricultural production, supplemented by land use-specific distributions of measured indicator values (Gonzalez-Quiñones et al., 2015). Hence, individual farmers can compare their own data for every indicator with the range of values known for similar circumstances in the region. Supplementary general information is also provided that can be used to modify management for environmental goals such as carbon sequestration and minimizing nutrient losses to the environment.

In Europe, many national approaches to soil quality assessment were developed. Those focusing on soil biodiversity rather than on general soil quality were reviewed by Pulleman et al. (2012). The French “soil quality observatory” was started in 1986 and included 11 sites (Martin et al., 1998). The more recent soil quality monitoring system (RMQS) program is based on a 16 × 16 km grid of the French territory and feeds into the French Information System on soils (Antoni et al., 2007; Arrouays et al., 2003). In the UK, the first approach to soil quality monitoring (Loveland and Thompson, 2002) had a focus on forestry and semi-natural soils. After further elaboration, a minimum dataset of only seven measurements was proposed (Merrington, 2006). In addition, Countryside Survey has been monitoring a few soil properties such as pH, soil organic carbon and some aspects of soil biodiversity (Black et al., 2003) since 1978 (http://www.countrysidesurvey.org.uk). In Ireland, recent work on the assessment of soil functions at grassland farms combines a full soil profile description and visual soil assessment with determination of a suite of analytical indicators (Bondi et al., 2017). In The Netherlands, a set of indicators for soil ecosystem services developed by RIVM (National Institute for Public Health and the Environment) was used in two five-year measurement cycles in 200 sites of the Dutch soil quality monitoring network (Wattel-Koekkoek et al., 2012). Target values and ranges for agronomic land use are based on median values of the monitoring network and on judgement of a group of soil experts. Also in the Netherlands, a large Public Private Partnership ‘Sustainable Soil’ is developing a soil quality assessment system in which a set of soil chemical, physical and biological indicators is related to target values and ranges for integral advice on soil management (www.beterbodembeheer.nl).

Given the plethora of soil monitoring programs in Europe, a common European soil monitoring framework was proposed (Huber et al., 2001), which was based as much as possible on existing monitoring activities. Subsequently, the EU-FP6 project ENVASSO (ENVironmental ASsessment of Soil for mOnitoring) aimed at defining and documenting a soil monitoring system for implementation in support of a European Soil Framework Directive (Kibblewhite et al., 2008b), focused on the assessment of soil threats, which however never materialized. Nevertheless, three priority indicators for each soil threat (Huber et al., 2008) were identified, and this list was further revised and amended by the EU-FP7 project RECARE (Preventing and Remediating Degradation of Soils in Europe through Land Care) as shown in Supplementary Table 1.

The history of soil quality assessment in China was reviewed for an international readership by Teng et al. (2014). Due to increasing pressure to maintain and improve soil quality in China, the Chinese government in 2008 established the China Soil Quality Standardisation & Technology Committee (SAC/TC 404) that has been responsible for formulating and modifying soil quality standards in China, including terminology, indicators, criteria, soil sampling methods, analytical methods, standards for soil quality assessment, and remediation of contaminated soils (Chen et al., 2011). By 2010, 141 soil quality-related standards had been set up, partly adopted from ISO.

The flexible and context-specific approach to soil quality assessment of the SMAF as described above has inspired several recent studies that apply multivariate statistical methods to select the most relevant indicators, often based on assumed but not assessed connections between indicators and soil functions, and utilize scoring functions to arrive at a soil quality index geared to the specific conditions (Armenise et al., 2013; Askari and Holden, 2015; Congreves et al., 2015; de Paul Obade and Lal, 2016; Lima et al., 2013; Swanepoel et al., 2014; Tesfahunegn, 2014; Velasquez et al., 2007). The drawback of such flexible approaches lies in the limited comparability between studies, even more than between different applications of the SMAF.

The compilation of major soil quality assessment approaches in Table 1 shows the variation in objectives, target groups (though often not explicitly stated) and spatial scales. Most of these approaches remain at the plot/field/site scale. Recently developed sensor-based approaches show promise to expand soil quality assessment to the landscape level (e.g. Vågen et al., 2013). Importantly, explicit evaluation of soil quality with respect to specific soil threats, functions and ecosystem services has rarely been implemented, and few approaches provide clear interpretation schemes of measured indicator values. This limits their adoption by land managers as well as policy.

3.2. Visual assessment approaches to soil quality

The above approaches to soil quality assessment typically require analytical laboratory facilities. Approaches targeting farmers and stressing the educational aspect benefit from more empirical, qualitative indicators that can be easily assessed in the field, deliver immediate results, and facilitate communication between farmers and scientists (Beare et al., 1997).

In the Wisconsin Soil Health Program, for example, a soil health score card was developed that collects farmers' observations on soil and plants, and includes a few questions on animal health and water quality (Romig et al., 1996). In Europe, the GROW Observatory (http://growobservatory.org/) was established in 2016, which is developing simple tools to support soil management for farmers and soil stakeholders, such as simple field-based assessments and educational tools. Visual soil assessment (VSA) approaches have been developed in different parts of the world (Table 2). Most of these methods target mainly soil structure, sometimes in relation to productivity (Abdollahi et al., 2015; Mueller et al., 2013). The methods vary in material and time requirements, with spade methods being generally faster to perform than profile methods and thus being more suitable for farmers (Boizard et al., 2005). The method developed by Peerlkamp (1959), which was used in the Netherlands for 40 years, has recently been improved by simplification of the scoring scheme and inclusion of a visual key (Ball et al., 2007; Guimaraes et al., 2011) to further support the use of the method by non-experts of soil science. Straightforward interpretation is certainly an asset of visual soil quality assessment, but visual soil assessment alone cannot evaluate the status of ecosystem services driven

Table 2
Comparison of major visual soil assessment methods (X signifies required material or performed observations).

Country	Australia	France	Australia	UK	New Zealand	Brazil/UK	Germany
Reference	McKenzie (2001)	Roger-Estrade et al. (2004)	McGarry (2006)	Ball et al. (2007)	Shepherd et al. (2008)	Guimaraes et al. (2011)	Mueller et al. (2014)
Stated objectives (assessment of …)	soil structure, suitability for root growth	soil structure	land degradation	soil structure	soil quality	soil structure	soil properties with respect to yield potential
Method name	SOILpak	Profil cultural	VS-Fast	Peerlkamp	VSA	VESS[a]	M-SQR[b]
Principle	spade	trench	spade	spade	spade	spade	pit
Material							
spade	X	X	X	X	X	X	X
plastic basin					X		
hard square board	X				X		
plastic bag or sheet				X	X	X	
knife	X			X	X	X	X
auger							X
water bottle					X		
tape measure or ruler			X	X	X	X	X
Time needed (min)	25–90	60–180	?	5–15	25	5–15	10–40
General observations							
soil layers, A-horizon			X				X
surface crusting or cover			X		X		
surface ponding					X		X
slope							X
soil erosion					X		
Soil physical properties							
soil texture			X		X		X
soil structure	X	X	X	X	X	X	X
soil consistence	X		X				
aggregate size distrib.			X	X	X	X	X
aggregate shape	X						
slaking/dispersion			X				
soil porosity	X			X	X	X	
soil colour	X		X		X		
soil mottles (no., colour)					X		
available water							X
water infiltration			X				
Soil chemical properties							
soil pH			X				
labile organic C			X				
Soil biological properties							
earthworms (no., size)			X		X		
potential rooting depth					X		X
root development	X		X	X		X	

[a] Visual evaluation of soil structure.
[b] Muencheberg Soil Quality Rating.

by biological and chemical soil processes (Ball et al., 2017). Because visual soil assessment provides different information than laboratory approaches (Emmet-Booth et al., 2016) the combination of both would be advantageous (Pulido Moncada et al., 2014). Ultimately, the increased use of visual soil assessment is considered to be important in yield gap analysis and land management programs (McKenzie et al., 2015).

4. Soil quality indicators

4.1. Requirements for soil quality indicators

Various requirements for soil quality indicators have been identified in some (but by far not all) approaches to assessing soil quality (Table 3). All publications that list such requirements mention at least one conceptual condition such as that a chosen indicator must be related to a given soil threat, function or ecosystem service and be relevant. However, this is not of great use if soil quality assessment is not targeting a specific soil threat, function or ecosystem service.

Of the practical requirements, ease of sampling and measurement is almost always mentioned, and reliability and cost are also considered important. Practical considerations such as the disadvantage of indicators requiring undisturbed samples often play an important role in discarding otherwise suitable soil quality indicators (Idowu et al., 2008), which is a serious limitation from a scientific perspective. Where the measurement of a specific soil indicator is considered too expensive, too difficult or not possible (e.g. bulk density, due to the stoniness of the soil), pedotransfer functions may provide a proxy value through the measurement of other properties, for example carbon and texture for bulk density (Reidy et al., 2016). The application of pedotransfer functions was already considered useful in early soil quality publications (Doran and Parkin, 1996; Doran and Safley, 1997; Larson and Pierce, 1994) and has again been advocated more recently (Bone et al., 2010), especially for complex soil properties such as hydrologic characteristics (Saxton and Rawls, 2006; Toth et al., 2015). However, the inaccuracy of pedotransfer functions needs to be clearly stated.

Sensitivity to changes in management is mentioned frequently (Table 3), but there may be trade-offs with robustness to seasonal variation. Regarding the interpretation of the obtained values, comparability to data from other sampling campaigns is often desired. However, some indicators such as organic carbon (or soil organic matter) content and pH are often measured, whereas others such as bulk density or earthworm diversity are rarely assessed (Morvan et al., 2008). Moreover, the requirement to have clear (absolute) interpretation schemes for a given indicator is mentioned in only half of the publications (Table 3), even though assessment of soil quality cannot be

Table 3
Considerations and criteria for soil quality indicators mentioned in various publications.

Criteria and considerations		Larson and Pierce (1994)	Doran and Parkin (1996)	Macdonald et al. (1998)	Burger & Kelting (1999)[a]	Southorn and Cattle (2000)	Nortcliff (2002)	Merrington (2006)	Idowu et al. (2008)	Ritz et al. (2009)	West et al. (2010)	Oberholzer et al. (2012)	Bone et al. (2014)
Conceptual	Related to soil function and/or ecosystem processes;		x		x		x	x	x	x	x	x	
	Relevance, representation of key variables controlling soil quality, correlated to long-term response, allow evaluation of assessment criteria	x			x			x				x	x
	Significance at the appropriate scale			x		x							
	Integrate soil physical, chemical, biological properties		x	x									
	Allow estimation of soil properties or functions which are more difficult to measure directly		x										x
Practical	Ease of sampling and measurement (simplicity, practicality, single or repeated sampling and measurement, provide information in short timeframe)	x	x	x	x	x	x	x	x	x	x		x
	High throughput of analysis, wide applicability									x			x
	Amount of soil needed									x			
	Sample storage before analysis									x			
	Reliability and reproducibility of measurement	x				x	x		x	x		x	x
	Existence of a standard method of estimation (standard operating procedure)					x				x			
	Availability of reference material for quality control									x			
	Cost (sampling, hardware, analysis, labour)	x			x	x		x	x	x	x		x
Sensitivity	Spatial variation						x						
	Temporal variation (not influenced by short-term weather patterns)		x			x	x				x		
	Sensitivity to changes in management, or land use, response to perturbation as well as corrective measures	x	x	x	x	x	x	x	x		x	x	
Interpretation	Comparability with routine sampling and monitoring programs (context data available); part of standard tests; baseline available		x		x	x	x	x	x	x			
	Ease of interpretation, interpretation criteria available			x		x	x	x				x	x
	Archivability, capable of continuous assessment				x					x			
	Mappable trend indicators					x							
	Generic or diagnostic value			x		x							
	Not redundant			x									

[a] As cited in Bone et al. (2010).

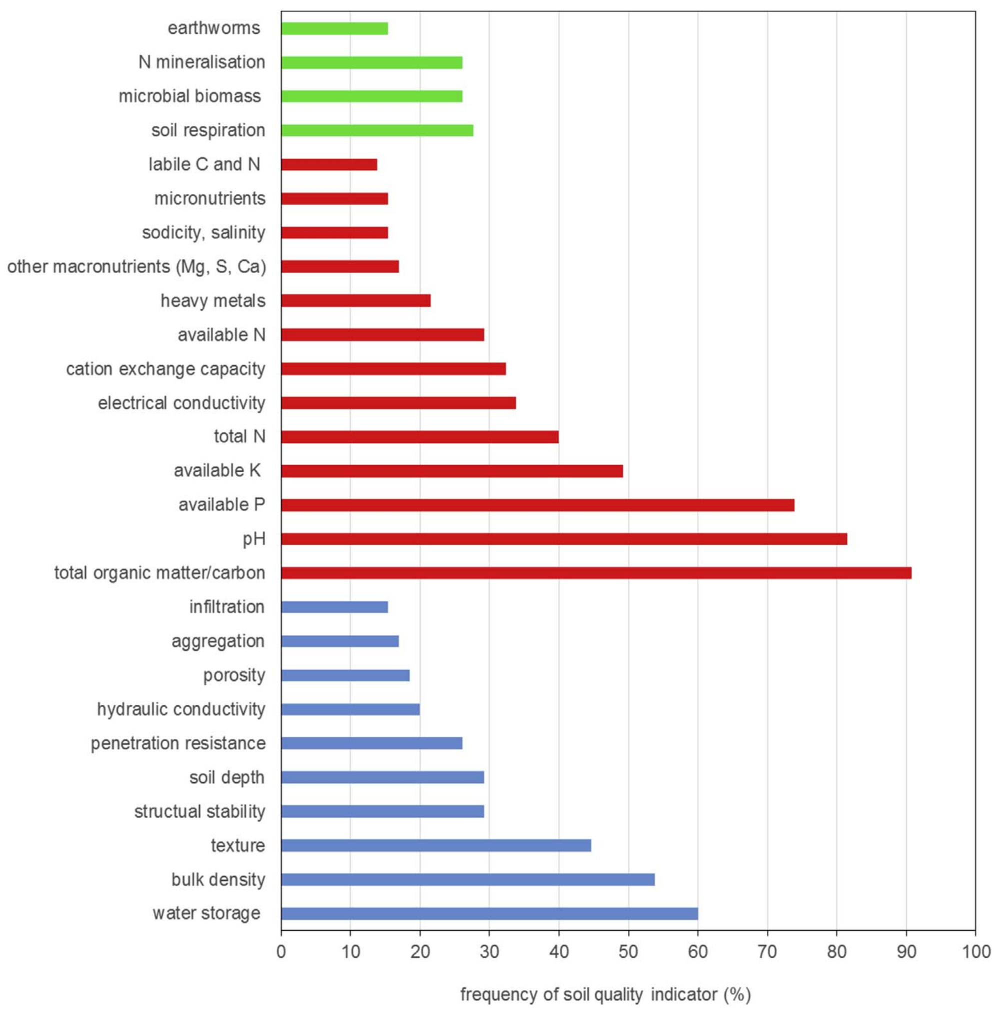

Fig. 4. Frequency of different indicators (min. 10%) in all reviewed soil quality assessment approaches (n = 65). Soil biological, chemical and physical indicators shown in green, red and blue, respectively. For further details on indicators see Supplementary Table 3. Publications dealing exclusively with forest soils (e.g. Schoenholtz et al., 2000; Zhang, 1992) or focusing on biological indicators only, without also looking at chemical and/or physical indicators (Filip, 2002; Parisi et al., 2005; Ritz et al., 2009), were not included in this compilation. If the same authors proposed the same set of indicators in more than one publication, then only the first was considered. In two publications (Andrews et al., 2002; Biswas et al., 2017), two different sets of indicator were proposed. Thus, the total number of reviewed publications was 62 while the total number of indicator sets was 65.

put into practice without it.

Finally, indications to what extent soil quality indicators actually fulfill the requirements listed in Table 3 are often missing but would be needed to make informed choices in soil quality assessment programs.

4.2. Methods for selecting a minimum dataset

Increasing the number of indicators can increase collinearity as well as the complexity of the relationships between indicators and management options. Moreover, costs of measurements easily become prohibitive, especially if detailed soil biological parameters are included (O'Sullivan et al., 2017). For these reasons, the number of soil quality indicators that is actually analyzed on a given set of samples needs to be reduced to a minimum dataset.

In the first proposed minimum datasets, this selection was based on expert judgement (e.g. Doran and Parkin, 1994). Subsequently, statistical data reduction by multivariate techniques such as principal component analysis (PCA), redundancy analysis (RDA) and discriminant analysis (e.g. Andrews and Carroll, 2001; Lima et al., 2013; Schipper and Sparling, 2000; Shukla et al., 2006), and multiple regression (Kosmas et al., 2014) became more common. After this initial data reduction, simple or multiple correlation analysis can further decrease the number of indicators (Andrews and Carroll, 2001; Kosmas et al., 2014), sometimes followed by the use of expert judgement for choosing only one out of two or more highly correlated soil properties (Sparling and Schipper, 2002). With these techniques, the number of indicators finally selected typically ranges between 6 and 8. Because soil properties that are relevant for soil functioning but do not show much variation in a given study will not be included in the minimum dataset, validation of the minimum dataset is important, for example by testing its relation to predefined and independently measured management goals (Andrews and Carroll, 2001).

A participatory approach of selecting soil biological indicators from a long list of potential indicators was presented by Ritz et al. (2009). Potential indicators were scored by scientists and end-users in a “logical-sieve” approach, which allowed several iterations. The different requirements for an indicator (Table 3) were weighted: reproducibility was considered absolutely essential, whereas the existence of a standard protocol had the lowest weight. A modified version of this method was applied by Stone et al. (2016a) to establish the top 10 biodiversity indicators of soil quality (defined as the ability to perform key soil processes) across the agricultural area of European member states for use in future monitoring.

Finally, the most important soil quality indicators can also be inferred from participatory conceptualization of how complex systems function. For example, Troldborg et al. (2013) and Aalders et al. (2011) established a Bayesian Belief network defining which factors are most influential in determining the risk of compaction and erosion, respectively.

Hence, the selection of a minimum dataset derived from a larger set of soil quality indicators is a necessary step in soil quality assessments because of financial and time limitations and to avoid collinearity. Methodological transparency is imperative to allow wide application of minimum dataset selection.

Table 4
Soil biological indicators, methodologies, related main soil functions, and advantages/disadvantages at different scales. Table compiled from (Bastida et al., 2008; Blagodatskaya and Kuzyakov, 2013; Bloem et al., 2009; Bouchez et al., 2016; Brussaard, 2012; Brussaard et al., 2004; Cardoso et al., 2013; de Groot et al., 2014; Gil-Sotres et al., 2005; Lehman et al., 2015; Neher, 2001; Nielsen and Winding, 2002; Orgiazzi et al., 2015; Parisi et al., 2005; Rocca et al., 2015; Saleh-Lakha et al., 2005; Schloter et al., 2018; Stenberg, 1999; Torsvik and Ovreas, 2002; Trasar-Cepeda et al., 2008; Visser and Parkinson, 1992; Watzinger, 2015).

Indicator	Methodology	Main soil functions	Main pros	Main cons
Individual, population and community level				
Presence, richness, abundance of individual soil organisms (for details see Supplementary Table 6)	Traditional handsorting and microscopic methods; molecular quantitation (qPCR).	Element, organic matter and water cycling, biological population regulation, soil structure maintenance	Taxonomic and functional level.	Not always linked directly with functions. Difficult to apply to fauna, e.g. protozoa, mites and collembola.
Microbial biomass and fungal biomass, fungal:bacteria ratio	Direct counting, chloroform fumigation extraction, SIR, PLFA, molecular quantitation.	Element and organic matter cycling, decomposition, soil structure maintenance	Sensitive and well related with other soil quality indicators.	Spatially variable, difficult interpretation, contradictory results. Unclear direct link to functionality.
Indices based on faunal communities (e.g. Maturity Index, Enrichment Index, Channel Index, Structural Index for nematodes)	Counting and identification of specific groups of organisms	Element and organic matter cycling, biological population regulation, decomposition	Sensitive. Taxonomic and functional level.	Time-consuming and costly. Specialist required for morphological identification.
Community composition	Manual counting and identification	Element and organic matter cycling, biological population regulation, habitat provision, decomposition, soil structure maintenance	Division in functional groups can give an indication of functions.	Time-consuming, expertise required. Not indicative of active biota.
	PLFA		Correlated with other measurements. Good indicator of active microbial biomass. Integrated information on the microbial community.	Time-consuming. No direct link with functions. Coarse resolution.
	Fingerprinting methods (e.g. DGEE, T-RFLP, A-RISA, ARDRA, TGGE), microarrays		Greater phylogenetic resolution.	No direct link with function. Difficult comparison between studies due to great variety in methods. Difficulties to extract and amplify DNA.
	Sequencing (metabarcoding)		Detailed view of diversity. Enormous amounts of data. Detects less abundant organisms. Permits discovery of new diversity.	Taxonomic genes no direct link with functions. Difficulties to extract and amplify DNA. Costly. Problems related with handling of large datasets and analyses. Dependent on libraries. No standard methodology.
	Community Level Physiological Profiling (Biolog™, MicroResp™)	Element and organic matter cycling, decomposition, habitat provision	Insight into functionality of the community. MicroResp™ closer to *in situ* conditions, shorter time of measurements.	Many replicates needed because of variability.
Ecosystem level				
Soil respiration, nitrogen mineralization, denitrification, nitrification	CO_2 evolution, N_2O emission, NO_3 produced.	Element, organic matter and water cycling, decomposition, habitat provision	Sensitive and ecologically relevant.	Highly variable and fluctuating. Relatively laborious.
Potentially mineralizable nitrogen	Anaerobic incubation.		Good correlation with MB and total soil N.	Relatively laborious.
Metabolic quotient (qCO_2), microbial quotient (MicrC/SoilC)			Sensitive, simple and inexpensive.	Difficult interpretation: confounds disturbance with stress.
DNA and protein synthesis.	Thymidine and leucine DNA incorporation.		Reflection of active microbial biomass.	No standardized procedure.
Enzymatic activities	Extraction of enzymes in the soil and incubation with various substrates.	Element and organic matter cycling, decomposition,, biological population regulation	Closely related to important soil quality parameters. Very sensitive. Simple and inexpensive methods.	Standard procedure not available. Contradictory results, complex behaviour and variable for each enzymes. Potential activity.
Functional genes and transcripts	FISH, Microarrays, *meta*-transcriptomic, qPCR, metagenome analysis.		Closer link to functionality. FISH and microarrays can give an idea of active microorganisms. High sensitivity and throughput.	Restricted to known gene sequences. Genes and transcripts might not be expressed. Difficulties linked with RNA extraction. Costly.
Metabolomics and metaproteomics	Assessment and quantitation of metabolites and proteins in the soil.	Element and organic matter cycling, decomposition,, biological population regulation, soil structure maintenance	Closer link to functionality.	Field in development. Difficult extraction of metabolites and proteins.
Stable isotope probing	Incorporation of ^{13}C- or ^{15}N-labelled substrates into DNA, RNA, PLFA, proteins	Element and organic matter cycling, decomposition	Permit to establish link between biodiversity and functions. Allow *in situ* analysis of active microbial population.	Field in development. Time involved in the assimilation of the substrates.

4.3. Frequently proposed soil quality indicators

To identify the most frequently proposed (combinations of) soil quality indicators, we summarized 62 publications (Supplementary Table 2) in which 65 minimum datasets of measured soil properties have been proposed. Due to the plethora of methods and terms, a certain aggregation of measured indicators into categories was required, e.g. aggregate stability, shear strength, tilth and friability, structure, consistence and slake test were merged in a category called structural stability (Supplementary Table 3). We included both peer-reviewed journal articles on soil quality assessment approaches and reports on national monitoring programs, aiming at global coverage. Considering that soil quality assessment includes many steps, from the definition of objectives via the selection of indicators to the interpretation of obtained indicator values, we only included studies that address more than one of these steps and thus have a certain conceptual and generalizable nature. Consequently, studies that are entirely limited to the comparison of a set of indicators in different management systems were excluded. Even though we may have missed some publications, especially from national assessment schemes, we noted that increasing the number of evaluated datasets from 45 to 65 during the compilation hardly changed the outcome. Therefore, we are confident that our evaluation shows a valid picture of which soil quality indicators are most used.

Total organic matter/carbon and pH are the most frequently proposed soil quality indicators (Fig. 4), followed by available phosphorus, various indicators of water storage and bulk density (all mentioned in > 50% of reviewed indicator sets). Texture, available potassium and total nitrogen are also frequently used (> 40%). The average number of proposed indicators is 11 (Supplementary Tables 4 and 5), which is probably more than is feasible from a practical as well as a financial viewpoint under most circumstances. Therefore, a trend towards smaller indicator sets in recent years can be seen. However, the development of novel indicators, which can be applied on a high number of samples in a fast and cheap way, could change the picture in the future.

In most publications, at least one indicator of each category (physical, chemical and biological) is included. These categories are typically represented automatically when all soil functions or soil-based ecosystem services are addressed. However, soil biological indicators were missing from 40% of the reviewed minimum datasets.

Soil physical indicators, especially those related to water storage, were frequently proposed in the early assessment schemes and again in the last 5 years, while they were less common in between (Supplementary Table 4). Among the soil chemical indicators, soil organic carbon content, pH, available P and K, total N, electrical conductivity, cation exchange capacity, and mineral N were proposed more often than all other indicators. Likewise, soil respiration, microbial biomass, N mineralization and earthworm density were more frequent among the biological indicators than the other 10 indicators that have been proposed at least once (Supplementary Table 5).

The explicit mentioning of extrinsic factors (Supplementary Table 5) such as climate, management or site data is surprisingly rare. In particular, yield, plant nutrient status and other measures of ecosystem services are very often not included. This means that soil quality assessment is typically not explicitly linked to ecosystem services or soil threats. An example of how to establish linkages between soil properties, soil functions and ecosystem services via correlations can be found in van Eekeren et al. (2010). Recent publications advocate indicators that are applicable to several soil processes (Bone et al., 2010). In Lima et al. (2013), for example, earthworms serve as indicators for both water and nutrient cycling. However, many of the other publications lack a clear conceptual and/or mechanistic relationship between indicators and soil functions and ecosystem services.

4.4. Novel soil quality indicators

Adoption of additional or novel soil quality indicators into minimum datasets is of interest if they have clear added value from the perspective of the management goals for a particular situation. Recent developments in soil science, especially in soil biology, but also in spectroscopy and other fields, hold promise for future soil quality assessment schemes. Below, we briefly review these developments, from biological and biochemical indicators to data capture and high-throughput approaches that have the potential to change soil quality assessment approaches quite substantially.

Soil organisms play a central role in soil functioning (Supplementary Table 6). Therefore, adding biological and biochemical indicators can greatly improve soil quality assessments (Barrios, 2007). Moreover, the assessment of biological indicators of soil quality is required to connect abiotic soil properties to (changes in) soil functions in terms of biochemical and biophysical transformations and (potential) aboveground vegetation performance (Lehman et al., 2015). Nevertheless, soil biological indicators are still underrepresented in soil quality assessments and mostly limited to black-box measurements such as microbial biomass and soil respiration (Fig. 4, Table 4). Despite clear potential, more specific indicators such as those based on nematodes (Stone et al., 2016b), (micro) arthropods (Rüdisser et al., 2015) or a suite of soil biota (Velasquez et al., 2007) have rarely been suggested, possibly because they require specific knowledge and skills. This situation is unfortunate because soil biota are considered the most sensitive indicators of soil quality due to their high responsiveness to changes in environmental conditions (Bastida et al., 2008; Bone et al., 2010; Kibblewhite et al., 2008a; Nielsen and Winding, 2002). In particular, there is an urgent need for indicators of soil–borne diseases (Kyselková et al., 2014; Liu et al., 2016; Trivedi et al., 2017). In this context, soil suppressiveness, defined as the property of a soil to naturally reduce plant disease incidence (Hornby, 1983), is of interest. Specific soil suppressiveness is the result of the presence of specific antagonists to pathogens, while general soil suppressiveness is based on the collective capacity of soil and plant microbiomes to act complementarily against pathogens (Schlatter et al., 2017). Both combined are governing soil suppressiveness as a whole (Yadav et al., 2015). Several soil abiotic and biotic parameters have been suggested to underlie suppressiveness, such as soil pH, specific cations such as Mg and K, soil total N content, microbial biomass and activity, diversity and structure of microbial communities and specific microbial taxa in the case of specific suppressiveness (Janvier et al., 2007; Wu et al., 2015), but without validation.

Recent rapid developments in soil biology have prompted the feasibility of indicators based on genotypic and phenotypic community diversity (Hartmann et al., 2015; Kumari et al., 2017; Nielsen and Winding, 2002; Ritz et al., 2009). Molecular methods focusing on DNA and RNA hold great potential to perform faster, cheaper and more informative measurements of soil biota and soil processes than conventional methods (Bouchez et al., 2016). Consequently, they may yield novel indicators that could substitute or complement existing biological and biochemical soil quality indicators in regular monitoring programs (Hartmann et al., 2015; Hermans et al., 2017). In the participatory approach used by Stone et al. (2016a), seven out of ten selected indicators were indeed based on molecular methods, with 'molecular bacteria and archaea diversity' on top. In addition, recent data analysis approaches such as network analysis, structural equation modelling and machine learning could facilitate the establishment of links between indicators and functions (Allan et al., 2015; Creamer et al., 2016). For example, Karimi et al. (2017) proposed microbial networks as integrated indicators of environmental quality that can overcome the lack of sensitivity and specificity of taxonomic diversity indicators. However, the prediction of process rates from the presence and quantity of genes and transcripts is yet to be clearly established (Rocca et al., 2015). Results gathered with these molecular techniques are also faced with biases introduced by sample contamination, PCR reaction, choice of primers and OTU definition and taxonomic assignment techniques (Abdelfattah et al., 2017; Hugerth and Andersson, 2017; Schloter et al., 2018). The analysis of the "big data" generated with sequencing also poses a serious challenge in terms of time, computing capacities and interpretation, since a large proportion of soil organisms yet remains to be characterized in taxonomic and functional terms (Schloter et al., 2018; Bouchez et al., 2016). Other molecular techniques such as metabolomics (Vestergaard et al., 2017) and metaproteomics (Simon and Daniel, 2011) may yield potentially suitable soil quality indicators because the measurements are directly linked to ecosystem processes (Bouchez et al., 2016). These technologies have benefits but are limited in their application by the difficulty to extract metabolites and proteins from soil and to choose representative samples (Bouchez et al., 2016). Stable Isotope Probing (SIP) in conjunction with phospholipid fatty acid analysis (PLFA) and DNA probing could also help to link soil biodiversity to soil processes (Wang et al., 2015; Watzinger, 2015). Finally, for a meaningful integration of indicators based on molecular methods into soil quality assessments, standardized techniques and a reference system are still lacking and will have to be established (Bouchez et al., 2016).

Although total soil organic matter is ubiquitous as a soil quality indicator (Fig. 4), changes in response to management and land use are difficult to detect since the total pool is large (Haynes, 2005). Moreover, due to the structural and functional heterogeneity of total soil organic matter, its relevance in soil processes is not unequivocal. Therefore, qualitative information on soil organic matter may be more informative in soil quality assessments. Pools of soil organic matter such as labile or active carbon are typically more sensitive to disturbance than total soil organic matter and can give a better indication about soil processes (Gregorich et al., 1994). Suggestions to measure this fraction include: particulate organic matter (Cambardella and Elliott, 1992), permanganate-oxidizable carbon (Weil et al., 2003), hot water-extractable carbon (Ghani et al., 2003) and water-soluble carbon, also called dissolved organic carbon (Filep et al., 2015). Despite their sensitivity to management and strong correlations to other parameters that are more difficult to measure, their relationship with soil processes is not well understood, partly because it is not clear which part of the organic matter they represent. Other methods to characterize (quality and quantity) of total soil organic matter such as thermal and spectroscopic methods are rapidly developing (Clemente et al., 2012; Derenne and Quénéa, 2015; Mouazen et al., 2016) and hold promise for soil quality assessments.

Additionally, soil sensing approaches such as spectroscopic techniques, e.g. near-infrared spectroscopy and remote sensing, offer the opportunity to measure various soil chemical, physical and biological parameters in a fast and inexpensive way (e.g. Cecillon et al., 2009; Gandariasbeitia et al., 2017; Kinoshita et al., 2012; Paz-Kagan et al., 2014). Sensors can be used directly in the field or in the laboratory (McKenzie et al., 2003), and commercial providers increasingly offer spectroscopy-based analyses (e.g. www.soilcares.com, www.eurofins.com). Combining laboratory-based visible and near-infrared spectroscopy with *in situ* measurements such as electrical conductivity and penetration resistance may be particularly useful (Veum et al., 2017). Spectroscopic techniques, however, also face limitations that hamper their routine use in soil quality assessment. First, when applied to the soil surface in the field, information is gained only about the first millimeters of the soil. Second, sample characteristics such as moisture content, particle size distribution and roughness of the soil surface can influence the outcome of the analysis (Baveye and Laba, 2015; Stenberg et al., 2010). Third, a calibration step is used to relate the spectral information to soil characteristics (Gandariasbeitia et al., 2017) and the prediction is as good as the calibration data set. Several studies showed that calibration efficiency varies between studies and parameters considered (Islam et al., 2003); Kinoshita et al., 2012). Through their nature, spectroscopic estimates are always less precise than traditional analytical methods (Islam et al., 2003). Creation of freely-available databases that can be used for proper calibration and prediction of soil properties are essential for realizing the full potential of these techniques. These databases should involve both NIR spectra and results from wet chemistry and biological methods.

X-ray tomography is another non-destructive technique that can be used for soil structural analysis and can shed light on processes integrating soil physical and biological properties (Helliwell et al., 2013). It avoids some drawbacks of spectroscopic techniques, namely the fact that it scans a 3D image of the soil instead of only scanning its surface. Nevertheless, this technique is still a long way from routine application for soil quality assessment.

Such novel indicators potentially allow a more detailed assessment of soil processes. At the same time, some of the techniques may be developed into high-throughput soil analysis to shed light on the spatial and temporal variability of soil parameters and determine soil quality across different scales for application in precision agriculture, monitoring programs and life cycle assessments (Ge et al., 2011; Viscarra Rossel et al., 2017). The rapid evolution of these techniques and the decreasing costs associated with them will facilitate this development. However, the practical operability of these indicators by different stakeholders needs to be taken into account. The various limitations described above still seriously hamper application of such novel indicators in routine soil quality assessments. In addition, the absence of standard operating procedures (SOPs) and accepted threshold values, especially for molecular methods, make the comparison and the interpretation of the results challenging (Callahan et al., 2016). The final and most important limitation to the interpretation of these novel soil quality indicators is the lack of functional linkages with soil processes and management implications.

Although use of novel indicators directly by farmers would be an advantage, most farmers are willing to send samples to the laboratory as long as the analyzed indicators are meaningful and responsive to management (Bouchez et al., 2016). For policy makers operating or setting up soil quality monitoring schemes, the introduction of novel indicators would also be aided by relating them to existing ones that may be phased out when performance (or cost-efficiency) of novel indicators is superior. At the moment, however, most novel soil quality indicators still belong to the research domain, and many technological, practical and interpretation related issues need to be overcome.

4.5. Interpretation of indicator values

An indicator is only useful if its value can be unequivocally interpreted and reference values are available. Reference values for a given indicator could be either those of a native soil, which may however not be suitable for agricultural production, or of a soil with maximum production and/or environmental performance (Doran and Parkin, 1994). In the Netherlands, for example, ten reference soils for good soil biological quality were selected out of 285 sites that had been monitored for over ten years (Rutgers et al., 2008). These reference soils represent specific combinations of soil type and land-use (e.g. arable land on clay soil). Soil quality indicators at a given site could thus be compared to those at the reference site as well as to the mean value, and 5% and 95% percentiles of all sites under a given land-use, with the percentiles given as a means to express the frequency distribution. An important drawback of this approach is that the reference may not be at an optimum in all parameters (Rutgers et al., 2012).

Acceptable values for an indicator can also be defined as those at which there is no loss or significant impairment of functioning

Table 5
Example of weighting of soil functions and associated indicators (Lima et al., 2013).

Soil function	Weight	Indicator level 1	Weight	Indicator level 2	Weight
Water infiltration, storage and supply	0.33	Available water	0.25		
		Mean weight diameter	0.25		
		Earthworms	0.25		
		Correlated indicators	0.25	Soil organic matter	0.50
				Bulk density	0.50
Nutrient storage, supply and cycling	0.33	Available water	0.25		
		Earthworms	0.25		
		Soil organic matter	0.25		
		Micronutrients	0.25	Manganese	0.33
				Copper	0.33
				Zn	0.33
Sustain biological activity	0.33	Soil organic matter	0.50		
		Earthworms	0.50		

(Loveland and Thompson, 2002). In the context of pollution, thresholds of contamination are often used (Chen, 1999). Likewise, Arshad and Martin (2002) list threshold levels for soil quality indicators, but this is rarely found in other publications on soil quality assessment. For plant nutrients, most agricultural advisory services use thresholds of available reserves below which plant production may become nutrient-limited, while maximum values are related to the risk of losses (Allen et al., 2006; Schoumans et al., 2014). Indicator thresholds for other soil functions are absent from most soil quality assessment approaches.

A more advanced way to evaluate soil quality indicators is the establishment of standard non-linear scoring functions, which typically have the shapes i) more is better, ii) optimum range, iii) less is better, or iv) undesirable range, with i-iii being most common in soil science. The shape of such curves is established based on a combination of literature values and expert judgement (Andrews et al., 2004). When scoring curves are based on regional data, such as in the Cornell Soil Health Assessment (Moebius-Clune et al., 2016), then scores are relative to measured values in the respective region. Each indicator measurement is transformed to a value between 0 and 1 (or 0 and 100) using a scoring algorithm (Karlen and Stott, 1994), with a score of 0 being the poorest (lower threshold) and a score of 1 (or 100) the best (upper threshold). The baseline value equals the midpoint between threshold values. Validation of scoring curves is possible if datasets with measurements of the given soil quality indicator and a related soil process are available.

Obviously, acceptable target ranges of soil quality indicators need to be soil- and land use-specific, and they depend not only on targeted soil functions, but also on both spatial and temporal scale of soil quality assessments, with regional target ranges typically being narrower than national ones (Lilburne et al., 2004; Wienhold et al., 2009). In addition, acceptable ranges of a soil quality indicator for one property or process are often highly dependent on the value of another soil property or process, e.g. dependence of microbial biomass or soil organic carbon on soil texture (Candinas et al., 2002; Johannes et al., 2017).

It has been claimed that the interpretation of soil quality indicators, i.e. the establishment of target or workable ranges, will always remain contentious, which is partly due to a lack of data, partly due to the curvilinear pattern that many indicators follow and partly because the use of expert judgement is contentious itself (Merrington, 2006). A comparative approach in which indicator values or scores of a given sampling point are put in relation to other sampling points may be the most intuitive and flexible basis for interpretation, since it gives a relative assessment (e.g. top 25%) and allows continuing evolution of the system. This approach is being implemented in the iSQAPER project, where the variation in soil quality indicator values within pedo-climatic zones is determined. Ranges are defined for specific land uses (e.g. arable land, grassland), and benchmark scores based on relative frequency are given. This approach may also introduce modular extensions of indicators that are only relevant in specific contexts, where stakeholders can relate to them. Decision trees based on environmental conditions, management systems and relevance of ecosystem services can guide the selection of specific indicators.

4.6. Deriving a soil quality index and alternatives

Many studies on soil quality have searched for a way to aggregate the information obtained for each soil quality indicator into a single soil quality index, even though this was deemed impossible by Sojka and Upchurch (1999). For example, Velasquez et al. (2007) summed the contributions of each of five sub-indicators (hydraulic properties, chemical fertility, aggregation, organic matter and biodiversity) to derive the general indicator of soil quality (GISQ). In the SMAF, an additive index yields a number between 1 and 10 (Andrews et al., 2004). However, if assessed soil functions or ecosystem services rank very differently in importance, then some kind of weighting is mandatory.

For example, in the recent Canadian monitoring of soil quality within the agri-environmental indicator assessment, a soil quality compound index is calculated as the weighted average of the performance indices for erosion, soil organic carbon content, trace elements and soil salinization (Clearwater et al., 2016). Another example is the multi-objective approach based on principles of systems engineering proposed by Karlen and Stott (1994). The main soil functions are weighted according to their importance for the overall goal in soil quality management at a given site, and an overall rating of soil quality with respect to the predefined goal is obtained by summing the weighted soil functions. An exemplary application of this approach can be found in Lima et al. (2013), who used SIMOQS (Sistema de Monitoramento da Qualidade do Solo) software developed in Brazil to calculate a soil quality index (Table 5).

Visual soil assessments are also often summarized in an overall soil quality rating (McGarry, 2006; Mueller et al., 2014; Shepherd et al., 2008). Typically, the scores for the different indicators are summed up, with some weighting applied. In the Muencheberg Soil Quality Rating, the weighted sum of the basic indicators is multiplied with values for hazard indicators such as contamination, acidification and flooding (Mueller et al., 2014).

Instead of deriving an overall soil quality index, colour coding for different indicators alone or aggregated according to soil functions is more meaningful. For example, in the outputs the Cornell soil health test, in Sindi, and in the Australian soil quality monitoring framework a traffic light system of 3–5 colours indicates low, adequate or excessive values for a given indicator. Other graphical presentations such as amoeba diagrams (or spider diagrams) can likewise convey more information on trade-offs and synergies than a single number or index (Rutgers et al., 2009, 2012).

The ultimate purpose of a soil quality index is to inform farmers and other land managers about the effect of soil management on soil functionality. An aggregated presentation of the outcome of soil quality assessments, especially by graphical means, can indeed be useful also for educational purposes and for communicating to society as a whole the consequences that human decisions can have on soil-based ecosystem services.

4.7. Stakeholder involvement

Because the reviewed literature is often not clear (enough) on who were the main developers and who are the main end users of the soil quality assessment schemes (Table 1, Table 2), we asked (by e-mail) 17 scientists who stood at the cradle of such schemes, or can currently act as spokespersons for them, to answer the following questions:

1 Who were the three main stakeholders, in order of importance, who were *involved in the development* of the soil quality assessment scheme?
2 Who are the three main stakeholders, in order of importance, *using* the soil quality assessment scheme?
3 Can you guide us to published or internet-accessible information (if any) on the extent of use and on user feedback?

We received answers from 11 countries: Australia (2 programs), Brazil, Canada, China, England, France, Germany, the Netherlands, New Zealand, Scotland and USA. The main *developers* of soil quality assessment schemes turned out to be scientists (8x) and government agencies (3x), while farmer organizations were top-ranked only once. The second position was taken by a mix of scientists (3x), (regional) government agencies (3x) and agricultural advisors (2x). Third positions were filled in only 5x, with various stakeholders. When it comes to *end users*, government agencies and consultants/agricultural advisors are top-ranked (each 4x), and farmers 2x. In second position are scientists (4x), (regional) authorities (3x), farmers/land managers (2x) and students (1x). Hence, not unexpectedly, scientists play a leading role in the development of soil quality assessment schemes. Remarkably, however, farmers/land managers, consultants/agricultural advisors and other stakeholders usually play an insignificant role in development, whereas they turn out to be important end users of the schemes. Quantitative data on the use of the assessment schemes is available in only four cases and user feedback data are equally scarce.

5. Conclusions

Our review has revealed how soil quality assessment has changed through time (Fig. 5) in terms of objectives, tools and methods, and overall approach. A number of steps are to be taken in soil quality assessment (Fig. 6), elements of which are addressed to very different degrees in the large number of approaches that have been developed during the past three decades and reviewed in this article. An elementary start is a clear definition of the **objectives**, i.e. whether soil assessment is meant as a basis for management recommendations, seen as an educational tool, or as part of a monitoring program. Likewise, **target users** should be named and involved from the beginning in order to increase adoption of the developed assessment approach. Such approach has been taken in the Horizon 2020 project LANDMARK, where the assessment of soil functions and indicators has in the first place been derived through stakeholder workshops (http://landmark2020.eu/work-package/work-package-1/). The application of stakeholder-based assessment requires different tools for different knowledge. For example, visual soil assessment tools are targeted at farmers for understanding the status of soil structure in the field, whereas more detailed knowledge on productivity requires laboratory measurements, which are, e.g., offered to farmers in the Cornell soil health assessment (Moebius-Clune et al., 2016) and by recently developed commercial soil testing services based on spectroscopic methods (see section 4).

The **selection of soil quality indicators** needs to be based on mechanistic linkages between indicators and soil functions or ecosystem services that have sometimes been proposed (Creamer et al., 2016) but rarely established firmly through experimental validation (e.g. van Eekeren et al., 2010). A clear definition of the targeted soil function(s) will determine the soil depth that is to be evaluated, since some soil functions are mainly related to the topsoil, whereas others are related to the entire soil profile. An asset of a novel soil quality framework would be the possibility to choose indicators based on the targeted soil threats, soil functions and ecosystem services, which is deemed possible by using the logical-sieve method (Stone et al., 2016a). Conceptually, soil threats, functions and ecosystem services are all linked (Fig. 2), and concepts focusing on either of these can thus be reconciled, if it is recognized that the targeted soil function or ecosystem service and associated choice of indicators are scale-dependent (Norton et al., 2016; Schulte et al., 2015). (Multi-) functionality should clearly be integrated in future approaches to soil quality, such as that of functional land management (Schulte et al., 2015) applied in the LANDMARK project.

The possibility to choose between **substitute or proxy indicators** (Fig. 6) would be highly beneficial but is so far rarely offered. The use of

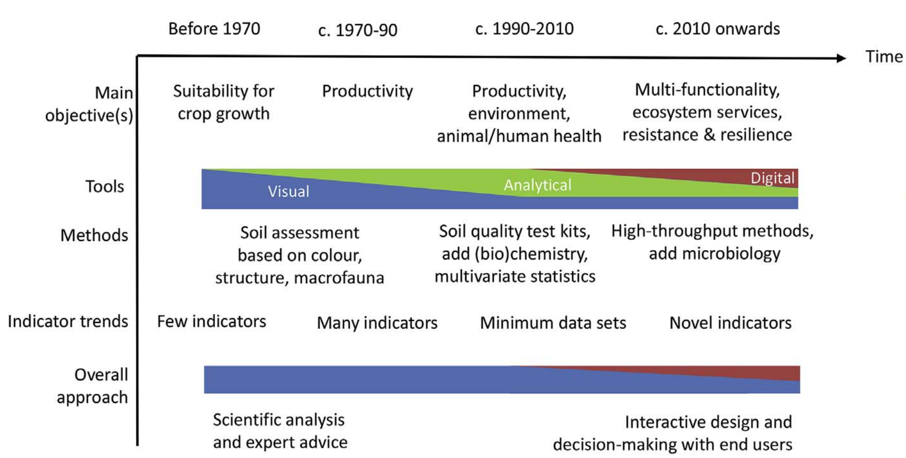

Fig. 5. Main objectives, tools and approaches of soil quality assessment through history.

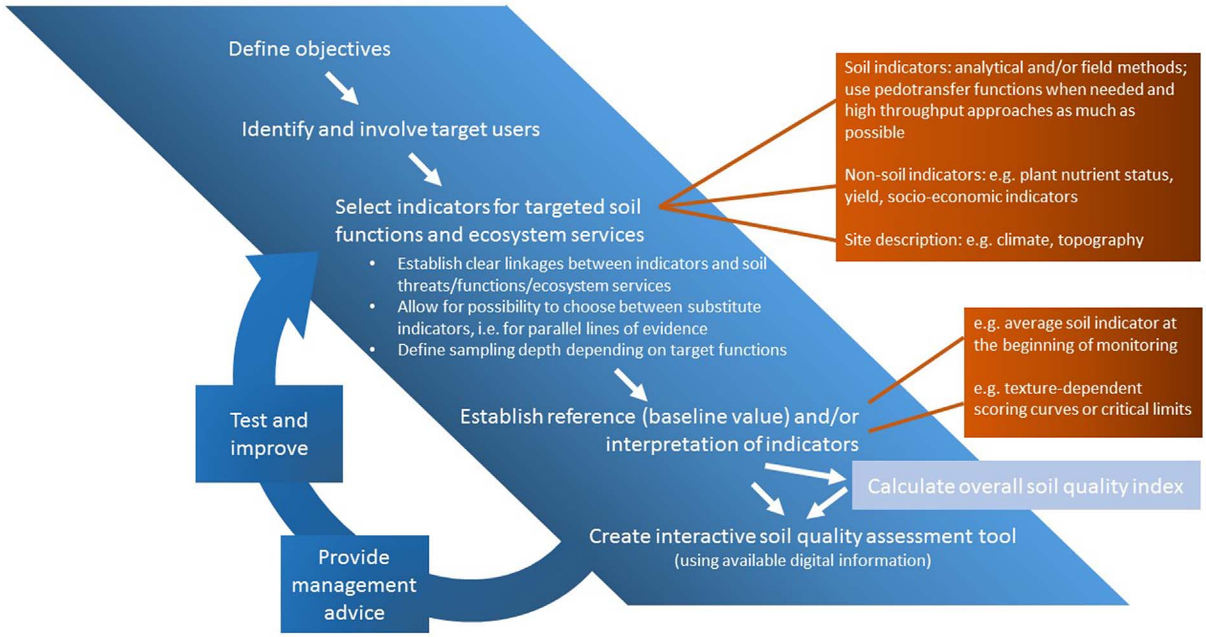

Fig. 6. Main steps in the development of a soil quality assessment approach.

parallel independent lines of evidence in ecological risk assessment (Rutgers and Jensen, 2011) and the inclusion of both qualitative and quantitative information in classical land evaluation (Sonneveld et al., 2010) could be models for that. Besides soil indicators, whether obtained using field assessments, analytical methods, high-throughput approaches or pedotransfer functions, also non-soil factors such as climatic and site conditions and non-soil indicators such as plant performance and aboveground biodiversity, landscape and socio-economic indicators (e.g. Culman et al., 2010; Jackson et al., 2012) should be considered.

The **interpretation** of the values of the proposed soil quality indicators needs to be well-defined. If no system for interpretation is provided, the indicators cannot be used in practice. For many soil properties, texture-dependent scoring curves need to be developed, which is possibly one of the greatest challenges. The increased availability of digital soil maps and soil survey data such as the LUCAS soil data available from the Joint Research Centre (http://esdac.jrc.ec.europa.eu/content/lucas-2009-topsoil-data) or global soil grids in 250 M (https://soilgrids.org/#/?zoom=2&layer=geonode:taxnwrb_250m) provides an opportunity to establish such scoring curves or target values more easily from frequency distributions of a given soil property. However, if soils in a region are badly managed or were so in the past, such a frequency distribution may not include the optimum state. In this case, the principle of identifying reference sites with acknowledged good soil quality (Rutgers et al., 2008, 2012) would be more suitable, or could be combined with the scoring curve approach. Reference or threshold values are required both to use soil quality indicators to their full potential and to translate the interpretation into appropriate management and policy advice. The assessment of the (dis) agreement of results obtained from different lines of evidence (e.g. sets of indicators based on physical, chemical or biological parameters; see e.g. Velasquez et al., 2007) can be adopted from mathematical procedures developed in ecological risk assessment (Karlen et al., 2001; Rutgers and Jensen, 2011).

An overall **soil quality index** is often desired but actually not very meaningful, since soil quality is best assessed in relation to specific soil functions. Rather than calculating an overall index, a graphical representation of how well a given soil fulfils its various functions is much more effective in communicating with stakeholders, target users and the general public. In practice, different sets of soil quality indicators will be used with different weightings, depending on the set of soil threats and ecosystem services at stake according to the "stakeholders".

Future soil quality assessment and monitoring can benefit from recent technological developments such as the SoilInfo App (http://www.isric.org/explore/soilinfo), mobile data capture including photographs and big-data approaches which are both used in the proposed LandPKS tool (www.landpotential.org), and high-throughput soil analysis approaches, such as visual and near-infrared spectroscopy. Future tools promise to be truly **interactive**, such as the soil quality assessment tool (SQAPP) that is being developed within the EU iSQAPER project.

Finally, soil quality assessment can become effective to improve the state of our soils only with inclusion of **management or policy advice**.

6. Outlook

Science plays an important part in the search, under prevailing pedo-climatic conditions (Fig. 1), for indicators of the structural and process aspects of soil functioning that mediate the delivery of soil-based ecosystem services deemed important by actors and other stakeholders who exert(ed) pressures on the soil through land use and soil threats. The key terms here are 'actors' and 'stakeholders'. Terms such as 'soil function', 'ecosystem service' and, indeed, 'soil quality', are boundary concepts, i.e. concepts that enable researchers from different disciplines, policy-makers, and other stakeholders to develop a common language and integrate and derive knowledge relevant to their field (Schleyer et al., 2017). Beyond scientists, those who have an immediate stake in soil quality are land managers, i.e. farmers, managers of nature conservation areas, roadsides, banks of waterways and urban green areas, and the public at large. As soil quality management is also about societal negotiation in the face of unavoidable trade-offs between various soil uses, the very development of soil quality indicator schemes will benefit from the involvement of actors and other stakeholders with a view to implement adaptive land use and management (Barrios et al., 2006, 2012).

Although, clearly, soil quality is not merely a natural science topic, in most of the reviewed assessment schemes farmers/land managers did not play a leading role. We suggest that intimate involvement of end users is a major point of attention, but it may still not lead to full implementation of the results. For example, in the Illinois Soil Quality

Initiative, where farmers were involved in the development of soil quality assessment schemes, they were constrained in the necessary implementation of the results by socio-economic factors (Wander et al., 2002). Clearly, other actors play an important part. Industries that ultimately also depend on the soil, will be (come) important actors, too, such as food, fibre and fuel industries, and electricity production, manufacturing and fashion industries (Davies, 2017). Their interest is in sustained resource supply, which is at stake because of ongoing loss of soil functionality and increased variability in harvests and water supply associated with global climate change, partly induced by unsustainable land use and management. Land managers, industries and, indeed, investors and insurance companies and the public sector at large are increasingly aware of the associated monetary and societal costs and, *vice versa*, they understand the urgency of adaptive land management and re-design in the framework of food systems (Foresight, 2011) and a fossil-free and circular economy (Rockström et al., 2016).

To be part of such urgent transitions, soil scientists are challenged to engage as 'honest brokers' of knowledge who increase the decision space of actors (Pielke, 2007). This engagement of soil (quality) researchers should take into account the following points:

First, we should consider (fostering) soil quality an integral part of (enhancing) environmental quality in general, as argued by Döring et al. (2015). We should not consider soil quality in isolation, but as part of quality assessment and adaptation of systems, e.g. of agricultural systems such as mainstream *vs.* integrated *vs.* conservation agriculture (Stavi et al., 2016) or mainstream *vs.* integrated *vs.* organic agriculture (Mäder et al., 2002; Seufert and Ramankutty, 2017). This requires engagement with farmers of different philosophies from purely organic to industrialized, and with other players in food systems.

Second, we should recognize that the radical changes in agricultural practices, summarized as 'smart farming' (Walter et al., 2017), require novel soil quality assessment tools, both in de-intensifying mainstream agriculture and in intensifying ecological agriculture (Struik et al., 2014).

Third, our focus should not just be on informing adaptive land management in existing agricultural systems, but also on fundamental system re-design, summarized as regenerative agriculture (Rhodes, 2017), in the framework of the circular economy.

Fourth, engaging with societal goals such as the UN Sustainable Development Goals (SDGs) is not only important in itself, but strategic in stressing the importance of soil (quality) knowledge for society (Bouma, 2014). In turn, monitoring progress towards the SDGs will require soil quality monitoring too, e.g. through the UNCCD Land Degradation Neutrality goals and associated reporting mechanism (Akhtar-Schuster et al., 2017).

Finally, awareness of the power relationships in the context of scientific support to stakeholders is essential. Generally, existing institutions and power relations resist innovation. Hence, the challenge is to associate with initiatives and policies that can create a greater space for innovation and system re-design and strengthen actors' influence from lower up to higher levels (Giller et al., 2008).

The engagement we make a plea for may require painstaking efforts, from gradual but consistent improvements within existing legislative frameworks (e.g. Ockleford et al., 2017; Römbke et al., 2016) to developing fundamental alternatives to current land use practices (e.g. Montgomery, 2017; Rhodes, 2017). Such engagement will at the same time require unquestionable scientific independence in the co-creation of knowledge (Mauser et al., 2013). We suggest that such engagement is necessary for the improvement of existing schemes and the development of novel schemes for assessment and monitoring of soil quality, as well as for the evaluation of their use and usefulness for all actors involved.

Authorship statement

EKB, PM and LB conceptualized the manuscript and EKB, GB and LB wrote it. ZB, RC, GDD, RdG, LF, VG, TWK, PM, MP, WS and JWvG contributed suggestions and text on specific components of the manuscript.

Conflicts of interest

None.

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Appendix A. Supplementary data

Supplementary data related to this article can be found at http://dx.doi.org/10.1016/j.soilbio.2018.01.030.

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