Exploring the case for a national-scale soil conservation and soil condition framework for evaluating and reporting on environmental and land use policies

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Abstract

It has long been realised that the conservation of soil capital and ecosystem services are of paramount importance, resulting in a growing case for a change in attitude and policy making in respect of soils. Current UK and EU approaches are risk-based and focused on measures to manage and remediate the adverse impact of current policies and practices directed at maximising productivity and profit, rather than one of resource conservation. Increasing soil loss and degradation is evidence that current policy is not working and a new approach is needed. In the UK there is governmental ambition to progress towards natural capital led land use policies~~,~~ but, in the absence of a framework to determine the relative condition of the soil resource, the delivery of sustainable soil conservation policies will continue to be inhibited. Common Standards Monitoring (CSM) is an established monitoring and management framework (based on ecosystem structure, function and process) and has been effectively deployed for almost two decades by the UK Government for the monitoring and reporting of key biological and earth science natural capital and ecosystem services from ‘field’ to local, regional and national levels to the European Commission. It is argued that a CSM for soils could be developed for the UK’s soil resources as well as for those elsewhere, and would be able to deliver a conservation rather than the current risk-based approach. It is capable of~~-~~ accommodating the complexities and variation in soil types and functions, and potentially being practical and cost effective in its implementation.

Key Words: soil resources, common standards monitoring, natural capital, ecosystem services, land degradation

Introduction

The importance of the ~~soil~~ natural capital of soil and ecosystem services for the wellbeing and prosperity of past civilisations and contemporary society has been well articulated (Leopold, 1949; Richter & Markewitz, 2001; Diamond, 2005; Montgomery, 2012). The scale of land and soil degradation affecting soil capital and the provisioning of food, fibre and other essential services is longstanding and global in its extent (Jacks & Whyte, 1938; Lowdermilk, 1953). Whilst climate and other drivers can be important factors, land use and management practice are considered to have been the main agents; driven by humans, their population growth, socio-economic and political aspirations, and cultural attitudes (Leopold, 1949; Stocking & Murnaghan, 2001; Stika, 2016). In 1935, the US Government was the first to attempt to address the agricultural degradation of soil capital and services with the formation of a dedicated national Soil Conservation Service (SCS) (Helms et al, 1996; Natural Resources Conservation Service, 2016). On a global scale, this was followed nearly sixty years later by the formulation of land use and soil conservation planning and management policy objectives of Chapters 10 and 14 of the Rio Summit Agenda 21 (United Nations Environment Programme, 1992), and more recently 'The World Soil Charter' (FAO, 2015). Despite all of the 178 attending governments being in agreement to the outcomes of the Rio Summit, the impact of the past intentions seems to have been minimal given the continuing extent of land degradation and the potential to increase further with climate change (Montanarella *et al*., 2016).

Leopold (1949) and Hyams (1952), and more recently other commentators (Conford, 1988; Stocking & Muranghan, 2001; Montgomery, 2012; Stika, 2016), have suggested that a fundamentally different approach to land and soil use is required to maintain local and global natural soil capital and ecosystem services. A simple scan of the current scientific literature about soils and their use, not unsurprisingly because of the need to feed the growing human population and concomitant commercial aspirations, indicates it is largely focused on the measures to maintain productivity and profitability despite there has been a growing literature about the importance of soil ecosystems. At our cost, the consequence of simply mitigating the adverse effects of the current production led system will be to perpetuate current thinking and policy making, without fundamentally changing how we view our finite land and soil resources.

What sort of system this should be and the how changes and risks should be measured can only be answered by having an appropriate framework in place (Brouwer & Crabtree, 1999; Prager, Helming & Hagedorn, 2011). The purpose of this paper is therefore to explore the case for applying an existing ecosystem-based approach to soil resource conservation and management to direct future UK and EU policy making.

Soils in the European Policy Context

In the European context, the current farming-based system arose because of the significant increase in population due to industrialisation and urbanisation, and the need to strive for self-sufficiency in food supply during and following World Wars I and II. The manifestation of the current adverse environmental and soil related impacts on soil capital and ecosystem services in the UK and mainland Europe are accepted to be a result of the introduction and widespread use of chemicals (fertilisers and pesticides) and developments in mechanisation since the 1960s and 1970s (Parliamentary Office of Science and Technology, 2006; Haygarth & Ritz, 2009; Prager *et al.*, 2011; Graves *et al.*, 2015; Smith *et al.*, 2015; Environment Audit Committee, 2016a & b; Kibblewhite *et al*., 2016).

Whilst there are continuing land degradation issues such as loss of soil carbon, soil erosion and compaction across Europe and the UK (Kibblewhite *et al.*, 2016), there remains no specific soil conservation framework for land and soil use policy, despite an attempt to establish one (Commission of the European Communities, 2006). Hence, soil related policy remains largely secondary and consequential, being the result of other environmental objectives (particularly biodiversity, air and water quality). For example, compliance with the EU Common Agricultural Policy (CAP) legislation, specifically the obligation to maintain ‘good agricultural and environmental condition’ (GAEC) ~~in order~~ to receive agricultural subsidies is mitigation driven and aimed at minimising impact on the environment and soils, whilst maintaining land use and production practices (Louwagie *et al*., 2011; Prager, Helming & Hagedorn, 2011; Prager *et al*., 2011; Prosperi *et al*., 2011; Verspecht *et al*., 2011). It is within this context that UK and EU policy has been developed and exercised with both environmental and economic adverse consequences (Posthumus *et al*., 2011; Graves *et al*., 2015).

Framework for Future UK Policies

Policy Development

From the late 1990s, government ~~policy~~ policy-making in the UK became more explicitly evidence informed (Davies, 2004). This shift in the process theoretically extended to those policies which are land-use and soil-based. However, until very recently with the development of natural capital and ecosystem services concepts, UK Government policy and vision (in respect of safe-guarding the UK’s soil natural capital) gave little consideration ~~for~~ to the capacity (health) of soil resources to function. Here, the policy drivers were primarily concerned with maintaining profitable returns from agricultural production, reducing pollution and the legacies of contaminated land (particularly in relation to human health) (DEFRA, 2009). However, The Natural Choice White Paper (UK Government, 2011), the UK National Ecosystem Assessment (2011), and the Natural Capital Committee (2013) together reset the UK Government’s vision for future policy making by introducing soil in the context of critical natural capital which supports crucial health and social, economic and environmental ecosystem services. This value-based thinking has the potential to direct future advice and content of government policy making for land use and soils, but not how it should happen.

Concerns have been expressed to the UK House of Commons' Environmental Audit Committee (Environmental Audit Committee, 2016a) that land use policies and visions for the environment and for profitable farming in the forthcoming Department for Environment, Food and Rural Affairs' (DEFRA) Environment Plan (see UK Government, 2016) are in danger of continuing to be treated as separate entities despite emanating from the same government department. It is evident from the published departmental plan for 2015-2020 that the UK government's strategic approach continues to be management focused on reducing the risks of current land use policies and practices on soils. This suggests that the transition to a holistic natural capital and ecosystem services thinking is not taking place. Hence, the UK Government has yet to resolve the conflicting tensions between their environmental objectives and those for food and farming in its forthcoming 25-year environmental plans (Environmental Audit Committee, 2016b). Some in the UK see it being resolved by a policy of designating 'spare land' (i.e. partitioning of land use) for environmental functions (e.g. pollution and flood control, biodiversity), from those allocated for intensive food and fibre production (Garnett & Godfray, 2012; Fairbank *et al.*, 2013). Others consider alternative production systems will be necessary (Smith, 2013). In such a policy framework, it would seem that greater intensification and efficiency measures are envisaged so that UK agricultural profitability and competitiveness are at least maintained (Barnes & Thomson, 2013; Fish, Winter & Lobley, 2014; DEFRA, 2017). Meanwhile, the EU's CAP 'greening' subsidy policy aims to off-set the damage to and degradation of soils and their ecosystems (European Union, 2013). How this approach is supposed to resolve the conflicts between sustainability and production will remain uncertain until there is a soil conservation framework in place for their proper evaluation.

As pointed out by the UK Natural Capital Committee (2013), if there is to be progress there needs to be an evaluative framework in place whereby the assets can be defined and changes and risks to the natural capital and ecosystem services can be measured. The recent review of soil capital and soil health put to the UK House of Commons' Environmental Audit Committee (Environment Audit Committee, 2016a) established that for both the environmental and farming aspects of land use and soils there are no frameworks fit for purpose currently in place. This raises the question of what basis could a newly focused plan and policy on soil natural capital and ecosystem services be formulated and assessed for *both* the environment and farming together?

Frameworks & Indicators

Frameworks provide the basic structures for concepts or systems, whereas indicators are the parameters or values, which describe states and fluxes within the concepts and systems, and their frameworks (OECD, 1994 & 1999).

The current framework for assessing and monitoring soil conservation is determined by current EU policy (European Union, 2013). There has been much published over the past twenty years on the selection and use of possible indices and indicators of soil health and soil quality in relation to the evaluation of current policies (see Brouwer & Crabtree, 1999; Paoletti, 1999; Pankhurst *et al*., 1997; Rickson, *et al*., Undated; Natural England, 2015; Kibblewhite *et al*., 2016; Schroder *et al*., 2016). For example, soil carbon is now commonly used as an indicator of soil health owing to its functional relationships with soil aggregate stability and soil erosion, soil fertility, and its relevance to the global carbon cycle with respect to atmospheric carbon fluxes and climate change (Smith *et al*., 2015). Others have considered the use of soil physical indicators such as bulk density with largely inconclusive results (e.g. Corstanje *et al*., 2016) and even operational measures such as the trafficability (i.e. accessibility) of land for cultivation (Bouma & Wösten, 2016).

~~Whilst~~ Although the deployment of such indicators enable the formulation of management practices and remedial actions, they have largely been concerned with the resilience of soils to productive farming policies and practices. As Rickson *et al*. (2012) point out, there is the question of whether individual quantitative indicators can ever be related to ecosystem services, let alone being the basis for a conceptual model. What is needed is an overarching framework, which considers soil condition (health) at its core and values the soil at least as much as the livelihood and wealth derived from it.

With the possibility of greater land use intensification and efficiency measures, it seems almost inevitable that separate frameworks for the environment and farming will be proposed despite the call for a holistic approach to soil conservation (Usher, 2005; Parliamentary Office of Science and Technology, 2012a). A more holistic approach to soil conservation would be to consider all land uses and soils together. Only when this is established, can appropriate indicators be identified and applied. Graves *et al*. (2015) suggested the development of the Landis based 'Soilscape' (Cranfield University, 2017a) as an integrator of soil textural and habitat types and Kibblewhite *et al*. (2014) used this to assess the spatial risk of soil erosion in England and Wales. However, currently, the application of Soilscapes for policy making is limited as it is not real-time based and cannot differentiate between land use histories on a land utilisation basis (i.e. individual fields or parcels of land).

A Soil Condition-Based Framework

Importantly, any re-thinking should be independent and based on principles and evidence, and not governed by commercial self-interest and/or political malaise. An ecological-based approach is most compelling as soils are part of the plant-soil land-based ecosystems (Jenny, 1980). Such an approach is timely given the renewed interest in the nature and functioning of soil ecosystems and services (Bardgett *et al*., 2005; Bardgett & Wardle, 2010; Wall, 2012), and sits well scientifically with the concept of healthy (functioning) soils from a land use perspective (Pankhurst *et al*., 1997; Doran & Zeiss, 2000; Wall, 2012) and that of ecosystem services.

Any new framework will need to encompass and recognise both agricultural and non-agricultural soils as part of the wider UK resource capital as a whole and not treat them as separate entities in policy formulation, as is the case at present. It would also need to recognise self-maintaining plant-soil systems as those which have the full cyclic elements of primary production, decomposition and cycling, along with their attendant biological components (Bardget *et al*., 2005; Wall, 2012). Such situations are manifest in the form of mature largely undisturbed semi-natural woodlands, grasslands, and other habitat types found across the UK (JNCC, 2010). In these states soils are likely to maintain their fully functioning ecosystems in terms of nutrient cycling, biodiversity etc. Although ~~Whilst~~ ~~some~~some of these habitats are used commercially for food and fibre, such as rangeland grazing or rotational harvesting of timber, these are likely to be undertaken with little intervention and disruption of the ecosystem processes that support their ability to function and provide services.

Land use practices such as the repeated cultivation of soil, application of quantities of ~~artificial~~ manufactured fertilisers and manures, and establishment of mono-cultures are known to lead to the alteration and degradation of soil ecosystems, and ultimately their dysfunction and reduced service provision (Graves et al, 2015; Smith *et al*., 2015; Environmental Audit Committee, 2016a). Hence, there are concerns about erosion, compaction, degradation of soil structure and loss of soil carbon in the UK and the concentration on policies which remediate and minimise their occurrence (Posthumus *et al*., 2011; Verspecht *et al*., 2011; DEFRA, 2016; Kibblewhite *et al*., 2016).

It is evident from the above that there is sufficient understanding of soil ecosystem composition and functioning to be able to ascribe diagnostic traits to indicate soil health of habitats and land use types. A fully functioning ecosystem will be associated with definable traits ~~which~~ that would signify healthy soils and differentiate them from dysfunctional ones. It is notable that the metaphor 'health' has been used to confer a condition in relation to soils and UK policy (DEFRA, 2009), but not synonymously with ecosystem function.

An Existing Ecosystem Condition Framework

An existing framework and methodology known as Common Standards Monitoring (CSM) provides a standard and consistent approach for the evaluation of the conservation status and dynamics for the long term maintenance and risks to the integrity (extent) and supporting functional attributes (structure and natural processes, regeneration potential, distinctiveness) of important biological resources (JNCC, 2004a). The CSM was initially developed for the UK's semi-natural resources such as ~~Upland~~ upland ~~Grasslands~~grasslands, ~~Woodland~~woodland and~~,~~ ~~Mires~~mires,~~,~~ ~~but~~ later being extended to earth science geological capital. CSM defines ~~the~~ ecosystem~~'s~~ attributes and the conservation objectives, ~~and~~ sets targets for judging the current condition according to defined terminology (Favourable/Unfavourable/Destroyed) ~~and~~ relative to a past condition, thereby providing an indication of trend (Maintained, Recovering/Declining). Importantly, CSM provides a methodology for practical application at a scale ranging from individual features to larger catchment, regional and national scales. The outcomes can be presented as tabulated data ~~and/~~or as maps that can be interrogated and used in strategic policy making, management planning and reporting the effectiveness of their implementation (Williams, 2006).

A soil resource-based CSM would also be based on habitat or ~~/~~land use feature~~-based~~. Here, the conservation objectives would be to maintain the integrity of the entirety of the ~~feature's~~ soil resource of that feature and the favourable status of the ecosystem's structural, biological and regenerative functional attributes. The soil conservation status and trends can be inferred from well-established associations with habitat type and land use practice. For example, the favourable condition of structural integrity and functioning of soils is known to be maintained in the long-term absence of or infrequent disturbance (e.g. cultivation) of the soil, ~~along~~ together with the maintenance of the functional biological processes with persistent vegetation cover and diversity, and root longevity (Bardgett *et al*., 2005; Bardgett & Wardle, 2010; Stockdale & Watson, 2012; Natural England, 2015). Conversely, soil structure and biological capacity are degraded by disturbance and certain cropping practices, resulting in unfavourable soil conditions and which, in agricultural practice, are often a focus of intervention or changes in practices to mitigate and maintain production levels (Watts *et al*., 2001; Roger-Estade *et al*., 2004; Hamza and Anderson, 2005; Bilotta *et al.*, 2007; Batey, 2009; Alvarez *et al.*, 2012; Ball *et al.*, 2012; Munkholm *et al.*, 2013; Osman, 2013; Abdollahi *et al.*, 2014; Cui *et al.*, 2014; Abdollahi *et al.*, 2015). Soil ecosystem recovery is possible, but dependent on time and the reinstatement of favourable habitats and land use, and (if needed) intervention practices. Hence, soil condition can be inferred from the ~~feature's~~ habitat type and land use history of the feature, which can be collected by remote sensing, historical records and numerous local and national data bases, site inspection, interview and census.

The following is work in progress, however, it is set out here for illustration of the mechanics of a soil-based CSM approach to soil resource conservation.~~-~~ Conservation objectives, for each ~~of the feature's~~ attributes of the feature would be set to maintain the integrity of the physical extent of the resource~~,~~ and the functioning of its structural composition and natural processes, potential for regeneration~~,~~ and, where appropriate, the local distinctiveness of particular soil types, associations and geomorphological features ~~etc~~among others. (Table 1). Threshold limits would be set to enable the evaluation of the conservation status and established trends. For the extent and distinctiveness of soil resources, these can be directly determined, but for the other attributes it is anticipated that they can be determined indirectly from the extensive and long established scientific evidence-base supporting associated habitats and land use histories.

Soils used for agricultural cropping could be a separate land use or ~~/~~ecosystem category. ~~Whilst~~Although~~,~~ such soils are equitable to disturbed land, if the intent is for cropped land to be a functional land use (rather than a transitionary state, such as ~~like~~ glacial moraines) it would be expected to have structural and process~~/~~ or functional attributes that equate~~d~~ with a 'favourable' and sustainable soil condition. It could be argued that cropped soils having 'Good' structural characteristics (according to visual evaluation methods (e.g. Ball & Munkholm, 2015)) could qualify as being in a functional and hence favourable condition, whereas, 'Poor' soils would be intrinsically unfavourable. Such outcomes would take into account soil-type potential and agricultural practice. In this respect, there is seemingly a strong association between risk of soil erosion and run-off, and cropping and land-use practices (Table 2). The likelihood of cultivated soils being in a favourable condition could be assessed from land use practice and histories, and verified using visual evaluation methods. Hence, cropped land need not necessarily rank lower than other land, but would be a function of climate, land use practice, soil type and landscapes (Environmental Agency, 2007).

Table 3 provides an illustration of the CSM approach for some habitat features and their hypothetical land use histories. Here, outcomes range from where the soil resources were being maintained in a ‘F~~f~~avourable condition’~~,~~ to those in U~~u~~nfavourable recovering’ and ‘~~declining~~ Declining’ conditions, and those in ‘F~~f~~avourable recovered’ or ‘U~~u~~nfavourable no change’ states (where intervention measures might have been applied). Where particular types of soils are of uncommon occurrence or perform special (environmental) functions, the integrity of these assets would be part of the reporting process.

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Discussion

As CSM is process-based, the principles are capable of being extended to general situations, other than the legally designated nature conservation and geological sites in the UK for which it was originally developed. For example, it has been successfully applied in a general context of assessing ecosystem health in the rehabilitation of highly disturbed land after mining (Humphries, 2014; Humphries, 2016). Hence, it is argued here that a CSM-like approach based on habitats~~/~~ or land use and integrity~~/~~ or functional attributes would provide for a generally applicable and possibly a more ‘global’ approach for the long-term monitoring and management of UK and EU soil resources.

The development of a soil-based CSM approach could enable an inventory of the local and national extent and condition of soil capital in the UK and elsewhere. It could contribute significantly to the formulation and monitoring of protection and strategic policies, and the evaluation of their effectiveness. For example, future policies might encourage and even support the conversion of arable usage to woodland or grassland land uses that are ~~un~~non-intensive, ~~grassland land uses~~ where particular types of soil or climate change make intervention practices less effective and there is a high risk of degradation ~~and/~~or loss of soil resource (Environment Agency, 2007).

Prioritisation and Division of Responsibility

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As introduced earlier, the importance of soils and the risks to the future of the Earth's finite land resources ~~has~~ have long been argued by influential members of the land user and scientific communities. This has been acted upon by some government bodies and intergovernmental organisations, such as the EC, UK and FAO, but with arguably mixed results. Despite the continuing accumulation of evidence, there is seemingly little sign of a widespread change in the attitudes of policy makers. Seemingly, too many of those using land see soils as inert entities, rather than as living and supporting ecosystems that should be conserved and maintained accordingly. As a consequence, it has been argued that current policy making in the UK and the EU continues to be open to short-term exploitative production foci as opposed to long-term conservation of the soil resource and sustainability of its living functions (Environmental Audit Committee, 2016a). The reasons for the seeming lack of prioritising soil conservation are likely to be a complex mix of political and commercial influences, and institutional inertia as illustrated by the failure of EU Member States~~'~~ ~~failure~~ to agree a Directive for the protection of soil and land (European Commission, 2016a). Irrespective of the lack of progress towards a EU wide Directive, with the prospect of the UK leaving the EU, there is an opportunity for the UK Government to rethink its approach to soils and land use, and agricultural policy in general (Humphries, 2017).

The division of responsibilities for natural resources, environment and land use policy is known to contribute to the inhibition of strategic and integrated policy making (Parliamentary Office of Science and Technology, 2012b). This raises the question which bodies or organisations in the UK and the EU should be charged with the development of soils and land use policy? Should ~~it~~ such a policy be led by ~~one~~ an organization concerned with the production of food and fibre, rather than one focused on the environment or the conservation of resources, or should it be a new body? Delegation of responsibility will probably not be enough and satisfactory progress may only be made by legislation for the conservation of soil resources; as has been the case for biodiversity and water resources in the EU and its transposition into the UK Statutes. The science-based lead for the conservation of biodiversity natural and earth science natural capital, and the deployment of the CSM in the UK is by DEFRA's conservation advisory body, the Joint Nature Conservation Committee (JNCC). The ~~JNCC's~~ current advisory remit of JNCC for influencing government policy could be widened to include the conservation of soil resources.

A Risk Based or Conservation Approach

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The proposed EC Directive on soil protection (European Commission, 2006) was founded on a risk-based approach and measures necessary to protect soils from wind erosion, reduction in soil organic matter, compaction, salinization, landslides and contamination in order that they have the capacity to carry out their environmental, economic, social and cultural functions (i.e. soil ecosystem provisions and services). This risk-based approach was adopted by DEFRA (2009) in their strategy for safeguarding soils in the UK. The approach has been criticised for being inadequate for future soil conservation policy making by some of those making submissions to the Environmental Audit Committee (Environmental Audit Committee, 2016a).

The risk approach taken by the EC is strikingly different from that of conservation for biological resources enacted by their Habitats Directive (European Commission, 1992), and appears to reflect the focus and expertise of the stakeholder communities involved. For example, there was no legal requirement of the Member States to have in place policies and actions that aim to maintain their soil and land resource in a favourable biological condition, other than when there are threats to the provisioning and service capacities (which may be provided for in less than favourable circumstances). Importantly, there were no requirements for conservation objectives and targets to be set, nor the monitoring and reporting of the conservation status and long-~~–~~term dynamics, as for the Habitats Directive. Notably, it was to be left to the Member States to decide and administer, thereby frustrating the development and enactment of an EU wide soil conservation policy.

The Habitat Directive imposed a legal requirement on Member States to monitor and report on the long-term maintenance of a favourable conservation status of nationally and internationally important biodiversity capital in the European Union. In response, the UK Government's nature conservation advisor, the JNCC, devised a condition-based framework and methodology which were subsequently adopted by DEFRA (Rowell, 1993; JNCC, 2004a; JNCC, 2017). The approach has been accepted by the European Commission as being compliant with the Directive's monitoring and reporting requirements at the pan-European and international levels (Williams, 2006). The CSM framework and methodology has also been applied by the JNCC and adopted by DEFRA for evaluating the conservation status of the UK's important earth science (geology and geomorphology) natural capital (JNCC, 2004b). Importantly, the JNCC has used CSM to inform DEFRA's policy making and monitoring of policy outcomes, as well as for the management of the natural capital and for strategic planning and budgeting purposes. Here, it is argued that the CSM approach ~~is~~ potentially ~~of~~ has wider geographical and natural resource application than just the UK and biodiversity, and could inform future policy making for a more 'global' approach to the conservation of soil resources.

A Resource Rather than a Classification Approach

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The UK's vegetation is a complex of different community compositions, mixes, gradients and mosaics represented by the National Vegetation Classification (Rodwell, 2006), similarly the multitude of types of European vegetation communities approach (Mucina *et al.*, 2016). The CSM ~~approach~~ avoided the difficulty posed by the complexity and stricture of classification by being broad habitat and land use based. The soil-based CSM could use the same approach. Hence, the inherent complexity of the types of soils and their classifications (Kubiëna, 1953; Soil Survey Staff, 1960; Clayden and Hollis, 1984; Buol *et al.*, 2011), that has seemingly frustrated the formulation of the European Commission's proposed Soil Directive, should also not be a conceptual bar to the deployment and the development of the CSM approach for national and pan-national applications. Where there are distinct traits or particular conservation or ecosystem service qualities of particular soil resources, these could be accommodated by the CSM methodology, as has been the case for the UK's nature conservation assets.

Adoption of the CSM approach is likely to be criticised on account of over simplification of the complexity of soils and their condition. Here, there are basically two contrasting mind-sets of how this is approached by scientists and practitioners alike. There are those who are concerned in the detail and accounting of the contributing individual factors, ~~whilst~~ whereas others approach it as their aggregate, as processes~~/~~ or functions (see Sherwood & Uphoff, 2000; Kibblewhite *et al.*, 2008). Inherently, CSM is an aggregated approach as ~~is~~ are the visual soil evaluation methodologies of soil condition (see Ball & Munkholm, 2015).

The same arguments of over simplification of the complexity and variation between soil types and characteristics could be levelled in the application of visual approaches such as VESS, VSA, SOIL.pak (see Ball & Munkholm, 2015; Ball *et al.*, 2017) in the evaluation and subsequent management of soils. Despite this, the visual approach has been widely adopted across different climates and soils. Visual assessment is accepted as valid and repeatable methods for both practitioners and academic researchers. The approach has been adopted by the UK's Environment Agency in its 'Think Soils Manual' guidance to farmers for reducing soil loss and degradation (Environment Agency, 2007). In essence the CSM approach could be considered as an extension of the visual evaluation family.

Also, the functional ecosystem focus of the CSM approach for evaluating the condition status of nature conservation assets is in concert with the more recently developed and widely proffered ecosystem services soil health approach, ~~that~~ which seems to be replacing the risk-based approach. Both CSM and ecosystem services are founded on similar ecosystem function attributes of cover (extent), soil structure, biological composition, nutrient cycling, and recuperation, all being necessary for 'healthy' ecosystems (JNCC, 2004a; Kibblewhite *et al.*, 2008). Notably, both CSM and ecosystem service approaches are not based on reductionist hierarchical classifications of composition.

Cost Effective Implementation and Infrastructure

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Conventionally, the approach to monitoring soils has been the systematic collection and analysis of samples of selected biological ~~and/~~or chemical components that are used as indicators to assess condition (health). The sampling approach has been extensively researched and reported for over 30 years (e.g.Pankhurst *et al.*, 1997; Paoletti, 1999; Doran & Zeiss, 2000; Kibblewhite *et al.*, 2008). However, its deployment, other than at the local scale ~~and/~~or very limited sampling, requires considerable investment of time, and technical and financial resources, as would the wholesale deployment of qualitative visual techniques (see Ball & Munkholm, 2015). This sampling limitation and the restriction to a few easily measured parameters makes meaningful interpretation difficult (Smith 2004; Saby *et al.*, 2008; Brazier *et al.*, 2014; O'Sullivan *et al.*, 2017). Even if based on existing data sets (e.g. Agri-Food Biosciences Institute, Undated; European Commission, 2016b; Countryside

Survey, 2017; Cranfield University, 2017b) the parameters available may not be appropriate for the development and evaluation of policies or a changing environment (Keay *et al*., 2013), besides issues concerning the standardisation of methodologies (O'Sullivan *et al*., 2017). In practice, the conventional sample-based approach is more suited for research and verification purposes.

The ecosystem services approach is also said to be based on the selection and periodic analysis of independent sample-based indicators (Kibblewhite *et al*., 2008), making it uncertain how an evaluation might be arrived at. CMS relies on existing research knowledge and established associations of soil condition with habitat and land use practices, similar to those proposed by Sherwood & Uphoff (2000). Hence, potentially, the CMS approach does not have the same financial and practical limitations as sampling~~/~~ or analytical methodologies. Past and current habitat and land use data would be collected on a real-time (annual) basis from existing historical records, remote sensing and annual returns. Consequently, development costs and time will be ~~low~~ small and short. There would be no need for the time consuming and costly repetition of the extensive collection and analysis of samples; other than initially and for updating, in the setting and refining of conservation targets, limits and ranges associated soil-habitat or ~~/~~land use during the development of a soil-CSM. Of course, an ongoing and structured sampling programme for quality control purposes and verification of conservation status would be needed, as would be the case for any approach.

In the UK and the EU an administrative infrastructure already exists for the national reporting of land use through the implementation of the CAP payment system, which could be re-engineered. In the UK this is currently provided by DEFRA's Rural Payments Agency or the Member Country's own agencies. Alternatively, implementation and administration might become an expansion of the remit of DEFRA's JNCC and the UK's Country conservation agencies.

Finally, in preparing the above argument for the adoption of the CSM approach to soil conservation policy, it has rightly been pointed out (during the peer review process) that there will be technical, socio-economic and political concerns that will need to be answered in more detail before it is likely to become widely accepted. For example, the reliance on habitat~~/~~ or land use practice as the basis of assessment of soil condition as opposed to the traditional approaches based on soil types and their origin. Also, there is the concern that the CSM approach is inherently biased against the agricultural use of soils and would place the priority of soil conservation above food production with the concomitant result of lower levels of production and the implications this has for commerce and living standards. These and other considerations are clearly relevant and will need to be debated and addressed as the methodology is trialled and developed. These matters are beyond the scope of this paper; given that its purpose was simply to introduce the CSM methodology for developing soil resource conversation policies.

Conclusions

For some time there has been a growing case for a change in attitude and policy making ~~in~~ with respect ~~of~~ to the UK's and the EU's soil resources. A change from a production to a sustainable soil resource and function perspective could drive this. It ~~could~~ might be better facilitated by approaching soils as a matter of resource conservation rather than risk management.

CSM is an existing and proven conservation-based framework and methodology which is currently used in natural resource management and for policy making. The same approach could be deployed to set the conservation objectives and report on the status of soil resources consistently at individual local features as well as at national scales. A soil-based CSM is complementary to and supportive of the emerging policy drivers of ecosystem services and natural capital.

A CSM for soil resources ~~could~~ would be quick and cheap to develop, and has the potential to be simple to administer and cost effective, and capable of being implemented through existing UK and EU governmental administrative 'infrastructure'.

~~Whilst~~ Although acknowledging there are likely to be concerns about the adoption of a conservation approach to soil resources and policies governing their use, there are compelling reasons why a soil-CSM should be explored and debated further.

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Table 1 Draft generic CSM guidance on conservation objectives for soil resources

FEATURE ATTRIBUTE	CONSERVATION OBJECTIVE	TARGET / THRESHOLD	METHODS OF ASSESMENT	COMMENT
Extent	Maintain topsoil cover and topsoil resource	No significant physical loss to erosion, change in use or development	Direct by remote sensing / survey / historical records / census	Change would be determined by a pre-set date
Structural composition	Maintain potential climatic and textural soil structure development	No significant degradation	Indirect by association with current and historic habitat and land use practice determined by remote sensing / survey / historical records / census	Inferred status determined from literature and updated from verified experience. Scope for intervention measures to be included.
Natural Processes	Maintain carbon transformations, nutrient cycling and biological population regulation	No significant degradation		
Regeneration Potential	Maintain persistent complete plant cover, diversity of plant species and continuity of living root system	No significant degradation	Indirect by association with current and historic habitat and land use practice determined by remote sensing / survey / historical records / census	Inferred status determined from literature and updated from verified experience. Scope for intervention measures to be included.
Indicators of Local Distinctiveness	Maintain soils of local and uncommon distribution and soils having important environmental function	No significant physical loss to erosion, change in use or development	Direct by remote sensing / survey / historical records / census	Does not contribute to the evaluation of conservation state and trend per se, but alerts to provision of other ecosystem services and/or conservation of diversity or scientific interest of resource

Table 2 Land use practices and soil types in the UK at risk of soil erosion and run-off (derived from Think Soils Manual, Environment Agency, 2007)

Land Use	Soil Type	Conditions	Risk of Erosion	Risk of Run-off
All cereals	Light	Fine smooth seed beds	When dry and exposed	When capped
All cereals	All	Harvested when soils wet		When surface compaction / wheelings
Winter cereals	All	Seed bed formed when soils wet		When surface compaction or shallow underlying compaction
Spring root crops/vegetables – potatoes, carrots, onions, sugar beet	Light, Peaty	Soils wet at crop establishment and cultivations	When repeated trafficking of headlands & tramlines	When surface compaction / tramlines / wheelings / beds orientated up/down slopes
Autumn harvested crops	All	Soils wet at harvesting		When surface compaction / wheelings
Grassland livestock & vehicular movements	All	Outside over-wintering of stock / stock feeding / access places when soils wet		When surface compaction / poaching
Grassland livestock & vehicular movements	All	Timing of stock and vehicular access when soils wet		When surface compaction / poaching
Forage crop harvesting		Timing of vehicular access when soils wet / low ground cover in early spring		When surface compaction / wheelings
Vehicular movements – e.g. spreading manure	All	Timing when soils wet		When surface compaction / wheelings

Table 3 Illustrative hypothetical CSM outcomes of conservation state and trend associated with some feature-based habitats and land use histories

Feature's Soil Integrity & Functional Attributes	**Mature Deciduous Woodland**	**Upland Hay-Meadow**	**Lowland Permanent Pasture – Un-intensive / controlled grazing**	**Short-term Leys – Intensive Grazing**	**Continuous Arable**	**Rotational Cropping Without Breaks in Cover**	**Conversion of Arable to Deciduous Woodland**
Land Use History	Woodland established in mid-C19th	Known not to have been cultivated since 1940s	Pasture for 15 years and without restorative intervention, prior continuous arable.	Ryegrass Leys resown 2/3 year cycle.	Including cereals, root crops and maize	Change from continuous arable to rotation of mixed fodder/cereal crops (excluding root crops/maize)	10 year old plantation
Extent	FM – no net loss	FM – no net loss	UNC* – no net further loss	UD*# – annual loss by erosion	UD – annual loss by erosion	UNC* – no further net loss by erosion	UNC* – no further net loss
Structural Composition	FM – no anticipated degradation	FM – no anticipated degradation	UR* – re-establishing	UD# – tillage and/or compaction	UD – tillage and/or compaction	UD# – tillage and/or compaction	UR – re-establishing
Natural Processes	FM – no anticipated degradation	FM – no anticipated degradation	FR – re-established	UD# – tillage and/or compaction	UD – tillage and/or compaction	UD# – tillage and/or compaction	UR – re-establishing
Potential for Regeneration	FM – components present	FM - components present	FR - components re-established	UNC – some components present	UD – absence of components	UNC – some components present	UR – some components present
Overall Conservation Status & Trend	FM	FM	UR*	UD#	UD	UD#	UR

Key FM = favourable maintained FR = favourable recovered UNC = unfavourable no change UD = unfavourable declining UR = unfavourable recovering PD = partially destroyed D = destroyed.

Notes: with intervention measures * = potentially FR # = potentially UNC