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Review

A Functional Land Management conceptual framework under soil drainage and land use scenarios

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ABSTRACT

Agricultural soils offer multiple soil functions, which contribute to a range of ecosystem services, and the demand for the primary production function is expected to increase with a growing world population. Other key functions on agricultural land have been identified as water purification, carbon sequestration, habitat biodiversity and nutrient cycling, which all need to be considered for sustainable intensification. All soils perform all functions simultaneously, but the variation in the capacity of soils to supply these functions is reviewed in terms of defined land use types (arable, bio-energy, broadleaf forest, coniferous forest, managed grassland, other grassland and Natura 2000) and extended to include the influence of soil drainage characteristics (well, moderately/imperfect, poor and peat). This latter consideration is particularly important in the European Atlantic pedo-climatic zone; the spatial scale of this review. This review develops a conceptual framework on the multi-functional capacity of soils, termed Functional Land Management, to facilitate the effective design and assessment of agri-environmental policies. A final functional soil matrix is presented as an approach to show the consequential changes to the capacity of the five soil functions associated with land use change on soils with contrasting drainage characteristics. Where policy prioritises the enhancement of particular functions, the matrix indicates the potential trade-offs for individual functions or the overall impact on the multi-functional capacity of soil. The conceptual framework is also applied by land use area in a case study, using the Republic of Ireland as an example, to show how the principle of multi-functional land use planning can be readily implemented.

1. Introduction

Sustainable intensification is necessary to respond to the need for global food security so that increased food production can be achieved in an environmentally sustainable manner (Lal, 2009; Benton et al., 2011; Schulte et al., 2014). The Food and Agriculture Organisation (FAO) of the United Nations suggests that primary food production will need to increase by as much as 60% by 2050 to meet the needs of a rapidly increasing global population (Alexandratos and Bruinsma, 2012; McBratney et al., 2014). The soil resource occupies a central role in primary production, which is reliant on effective, tailored planning and land management policies.

In tandem with achieving primary production goals (which includes fibre and fuel provision as well as food production), dual purpose agri-environmental policies must ensure that soils are managed to achieve environmental targets in a socially acceptable and economically viable manner. Within the European Union (EU), for example, environmental targets are set under current European legislation on water quality (Schulte et al., 2010), greenhouse gas emissions (Schulte and Lanigan, 2011) and ecological protection (Schulte et al., 2014). By definition, sustainable intensification requires that any emphasis placed on increasing primary productivity is matched with an equal emphasis on sustainability to enable the delivery of food and ecosystem services into the future (Garnett and Godfray, 2012). Policies by the European Commission such as the Green Infrastructure Strategy and "greening measures" in the reformed Common Agriculture Policy (CAP; European Commission 2014) demonstrate that strengthening of ecosystem services is integral to achieving sustainable intensification. It is anticipated that the Functional Land Management (FLM) framework will support the more effective use of landscape specific agri-environmental policies, such as those considered within the document "Transforming our world: the

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2030 agenda for sustainable development" by the United Nations (2015), rather than the application of uniform measures (such as those in the Nitrates Directive) across diverse farming regions (Bouma et al., 2012; Pacini et al., 2015). The importance of tailored land management policies will become even more prominent in the EU due to the crop diversification measure under the CAP (Mahy et al., 2015).

A spectrum of soil based ecosystems services, which are referred to as soil functions, have been identified in a number of studies and comprise of; supporting, provisioning, regulating and cultural services (Blum et al., 2004; Bouma and Droogers, 2007; Haygarth and Ritz, 2009; Breure et al., 2012; Pulleman et al., 2012; Dominati et al., 2014; Horrocks et al., 2014). Schulte et al. (2014) identified the following five soil functions as key, specifically, for agricultural land use;

1) Production of food, fibre and (bio)fuel, which is a provisioning service. This is the primary function in the agricultural sector.
2) Water purification, which is a regulating service. This comprises the soil's capacity for storage, filtration and transformation.
3) Carbon sequestration.
4) Soil as a habitat for biodiversity and a gene pool.
5) Recycling of (external) nutrients/agro-chemicals, which is a supporting service.

The key to achieving all of these multiple demands on the soil through agri-environmental policy is to understand the intrinsic nature of the soil in order to get the most out of the system sustainably. However, soils vary in their relative capacity to deliver multiple soil functions, owing to the heterogeneous nature of soil, the type of land use and management practices (Haygarth and Ritz, 2009; McBratney et al., 2014; Schulte et al., 2014). Understanding how individual soil properties relate to the functional capacity of a soil has been the basis of much research (Sauer et al., 2011; Breure et al., 2012; Pulleman et al., 2012; Dominati et al., 2014) but less so on a comprehensive view of the interrelationships between soil properties and the full suite of soil functions. To optimise the delivery of individual soil functions, or to maximise the suite of soil functions, requires an understanding of the expected delivery of soil functions as influenced by key soil properties under different land management regimes, as well as the effect that incentivising one function has on the delivery of the other soil functions. Therefore, an integrated framework is required to consolidate the intricacies of such complex multi-faceted information in a simplified, transparent and coherent manner to support the design of a tailored land use policy.

1.1. Objectives

In a previous study, Schulte et al. (2014) related soils' functions to land use but emphasised the need to extend this to include soil type. Therefore, in this paper, the aim was to extend the framework of FLM theory to consider dominant soil type constraints, with two objectives. The first objective was to review the literature to derive concise conceptual models for the interrelationships between land use, soil type and the five aforementioned soil functions of agricultural landscapes. The second objective was to apply these models to build on the initial FLM matrix, which was introduced by Schulte et al. (2014), in order to complete the conceptual matrix of the suites of soil functions. This could subsequently be used to show changes in soil functional capacity in response to policies involving the alteration of land use.

At large spatial scales, these interrelationships are likely to be dependent on, and thus vary between, agro-ecological zones. The scope of this paper is, therefore, limited to the Atlantic climatic zone of Europe, with potential similarities and hence relevance to other temperate maritime climates. Under this climatic regime, excess soil moisture is considered the dominant biophysical constraint to achieving sustainable intensification because of: (a) reductions in herbage yields and growing season, (b) restricted pasture utilisation and trafficability owing to the potential threat of soil compaction, (c) reduced nutrient uptake by plants and (d) nutrient loss to water bodies (Shalloo et al., 2004; Schulte et al., 2005, 2006, 2012; Creamer et al., 2010; Humphreys et al., 2011; O'Sullivan et al., 2015). Based on this, drainage class is used here to encompass key soil properties that represent the soil natural capital, which performs soil functions that contribute to ecosystem services (Calzolari et al., 2016). With respect to climatic change, future strategic planning will need to consider temporal and spatial changes in the soil water balance, which determines land use options and management practices (Rivington et al., 2013).

2. Soil functions and soil management

2.1. Primary productivity

As conceptualised in Fig. 1, primary production is possible under a range of soil moisture conditions and deficits but it varies considerably with the type of land use. Herbage growth in grass pasture is greatest at soil moisture conditions around field capacity, which occur more frequently on moderately drained soils than on poorly drained heavy clays (which may carry water surpluses in excess of field capacity for prolonged periods) (Shalloo et al., 2004; Fitzgerald et al., 2008) or well drained soils that may be prone to (moderate) drought conditions (Laidlaw, 2009; Kroes and Supit, 2011). However, trafficability and herbage utilisation are higher on well drained soils (O'Loughlin et al., 2012; Schulte et al., 2012; Gregory and Nortcliff, 2013).

Most arable crops grow best on well drained soils that have a moisture status beyond field capacity for longer periods throughout the year. However, drought conditions impair plant physiology and limit the transportation of nutrients to the plant roots (Batey, 1988; Briggs and Courtney, 1989; Gardner et al., 1999).

At the other extreme, challenges posed for sensitive plant species on waterlogged soils include poor root development, impaired nutrient uptake (Rechcigl, 1982; Laidlaw, 2009), reduced rates of photosynthesis (Schulte et al., 2012), the production of toxic substances, such as ethylene and methane (Batey, 1988; Briggs and Courtney, 1989; Gardner et al., 1999; Schulte et al., 2012), nitrogen loss by denitrification (Rechcigl, 1982) and increased susceptibility to disease (Rechcigl, 1982; Briggs and Courtney, 1989). Compacted soils are more prone to impeded drainage and saturation, which in turn increases their vulnerability to degradation of the soil structure by livestock and farm machinery (Batey, 1988; Ellis and Mellar, 1995; Schulte et al., 2012; Kuncoro et al., 2014). Alternatively, tolerant species of deciduous and coniferous trees may be selected, in addition to bioenergy crops such as willow (Stolarski et al., 2011), in order to increase primary production on soils prone to saturation.

In summary, moderately drained soils offer the highest capacity for primary production (Ellis and Mellar, 1995). Primary productivity on excessively drained and poorly drained soils tends to be reduced with the exception of some tolerant species of deciduous/coniferous trees and bioenergy crops.

2.2. Water purification

The enrichment of water bodies by residual inorganic plant nutrients, such as nitrogen (N) and phosphorus (P) (Mason, 1998), is a global water quality issue. In general nitrate (NO_3^-) is considered to pose a greater risk to groundwater and transitional water bodies and P to freshwater (Schulte et al., 2006).

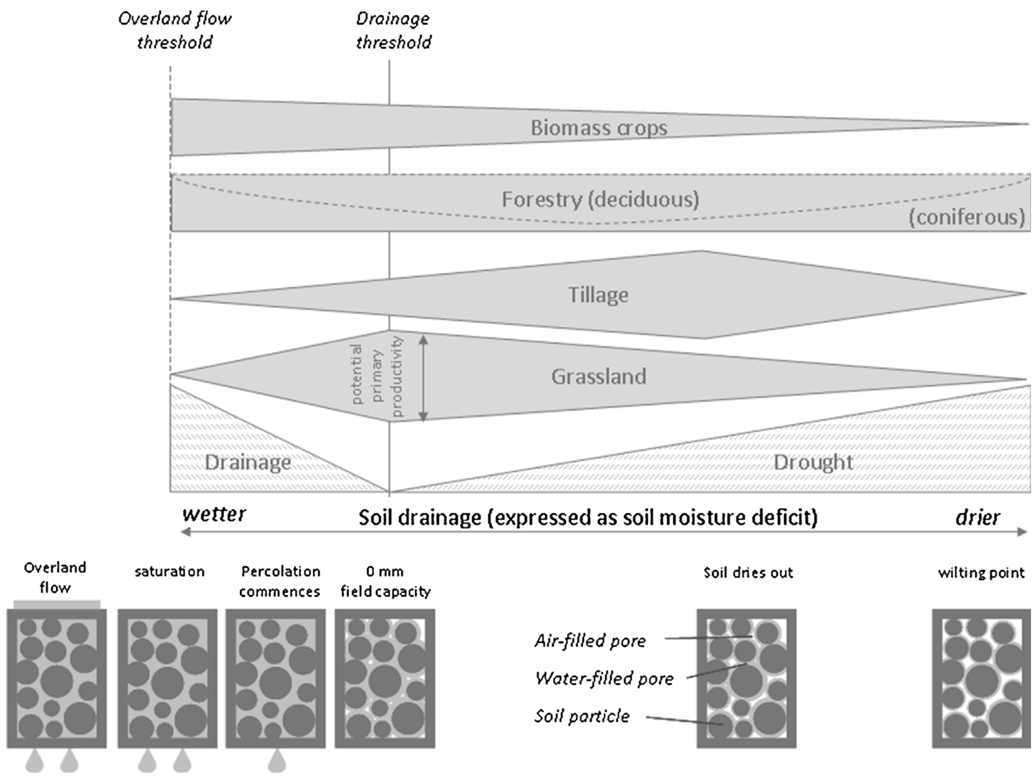

Fig. 1. Conceptual diagram of drivers of function: primary productivity, in relation to land management and soil drainage status.

Denitrification to N_2 gas and soil P sorption are, respectively, two proxy indicators of water purification potential and are considered here.

2.2.1. Denitrification, soil drainage and land use

Denitrification constitutes the conversion of oxidised forms of nitrogen, primarily NO_3^-, to gaseous forms under poorly oxygenated conditions by facultative anaerobic heterotrophic bacteria (Smith and Tiedje, 1979). Fig. 2a presents the denitrification process in soils in response to the main landscape (distal) and soil property (proximal) regulators that control this process. Soil moisture status is the major distal regulator of the process in soils, but denitrification is also influenced by proximal indicators such as the concentration of NO_3^- and soil organic carbon (SOC) (Loehr, 1977; Delwiche, 1981; Luo et al., 1997; Galloway et al., 2003; Ullah and Faulkner, 2006; Oehler et al., 2007; Saggar et al., 2013). Saturated soils with poor aeration have the highest denitrification potential (DP) (Delwiche, 1981; Luo et al., 1997). Under optimal soil moisture conditions, denitrification is controlled largely by the supply of SOC (Luo et al., 1997; Ruser et al., 2006), which provides a substrate for the growth of denitrifying bacteria (Delwiche, 1981).

In step one of the denitrification process (Fig. 2a), NO_3^- is reduced to form nitrite (NO_2^-), nitric oxide (NO) and nitrous oxide (N_2O) by a series of individual reductase enzymes, which are inhibited by oxygen and dependent on adequate substrate (carbon) availability (Paul, 2015). During complete denitrification (step two), these undesirable intermediate products undergo chemical reduction to form dinitrogen gas (N_2), which is inert and is released to the atmosphere. Saturated conditions in soils are required for complete denitrification (high N_2 to N_2O ratio) and provide the greatest capacity for water purification (Maag and Vinther, 1996). As shown in Fig. 2b, incomplete denitrification in moderately drained soils can result in the release of the greenhouse gas N_2O. This can lead to pollution swapping whereby water quality is protected from residual N at the expense of contributing to climate change (Haygarth and Jarvis, 2002). It is acknowledged that denitrification leads to the loss of nitrogen, which is an important macronutrient for plant growth, from agricultural systems (Gregory and Nortcliff, 2013). The denitrification potential (DP) is low in drier well aerated soils, which favour nitrification over denitrification.

In addition to soil drainage, DP depends on the N surplus. Despite a seasonal leaching risk from low soil cover on arable land (Beaudoin et al., 2005), the N surplus tends to be higher in grasslands, in particular intensive systems, compared to arable land and forestry plantations, as a result of livestock excreta (Wrage et al., 2004; Menneer et al., 2005). Higher rates of denitrification will occur from organic manures rather than mineral fertilisers due to the higher microbial biomass content associated with organic sources, which provide an easily mineralisable organic carbon content (Mogge et al., 1999).

Therefore, poorly drained grassland soils provide a greater DP overall. Alternatively, the installation of deciduous riparian buffers, which prosper in the wetter near-stream soil conditions (Heilman and Norby, 1998; Ullah and Faulkner, 2006; Hayakawa et al., 2012; Bonnett et al., 2013; Boz et al., 2013), provides a mechanism to enhance denitrification due to high biomass and SOC from the riparian trees.

2.2.2. P-sorption, soil drainage and land use

Soil P exists largely as inorganic forms that complex with calcium (Ca), iron (Fe) or aluminium (Al) (Yang et al., 2012) and as organic forms, as shown in Fig. 3. Organic P is formed by the incorporation of water-soluble P from the soil solution into living biomass (Massey et al., 2013). Owing to their high cation exchange capacity (CEC) and specific surface area, soils with high clay content have the greatest P-sorption capacity (McGechan and Lewis, 2002; Heredia and Cirelli, 2007; Qiong et al., 2008; Brady and Weil, 2010). The bioavailability of P is further reduced in acidic clay soils owing to the formation of Fe/Al/Mn hydrous oxide

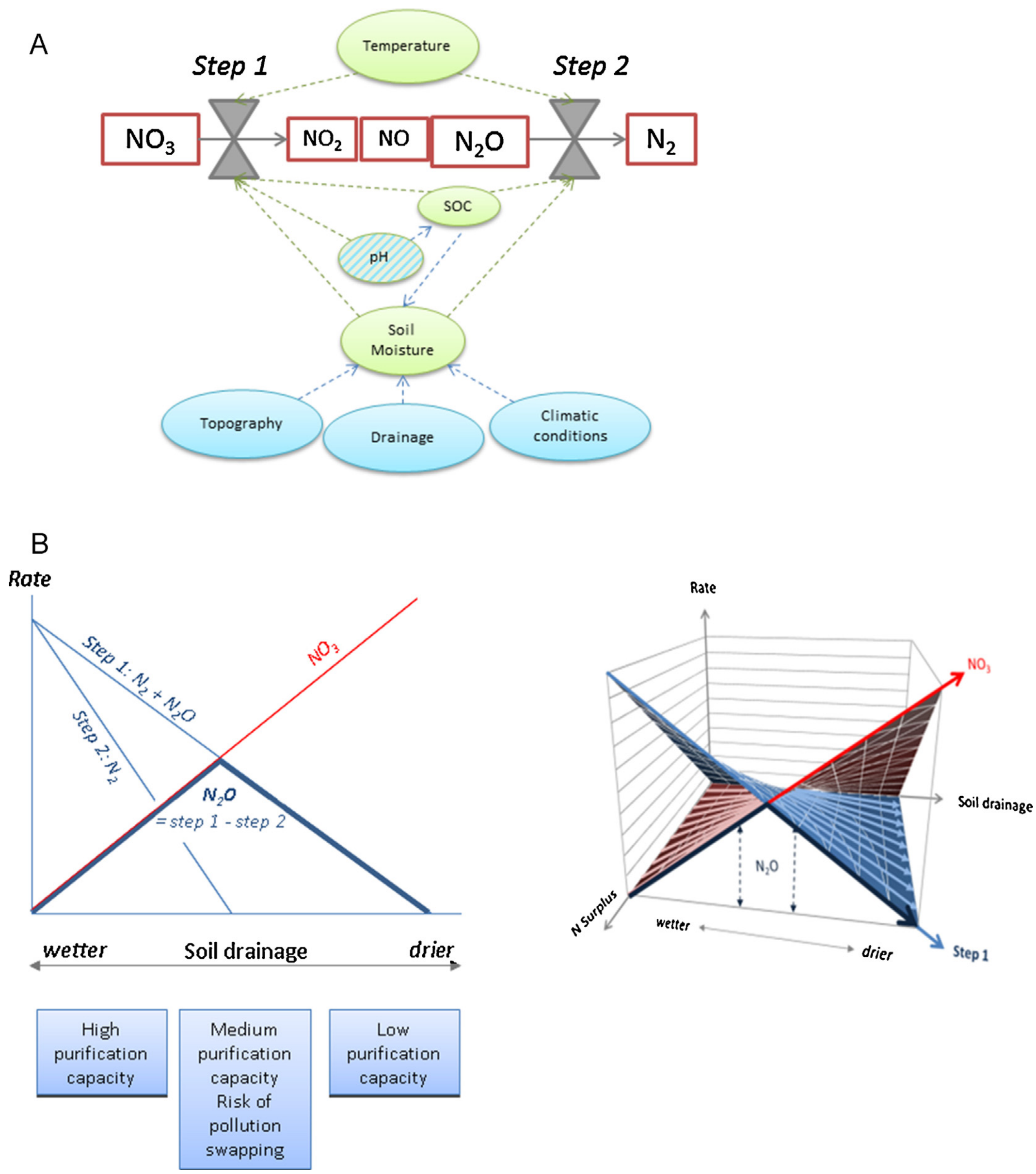

Fig. 2. (a) Key parameters affecting the denitrification process in soils. Boxes explain the process steps; valves represent the rate of a step; blue circles represent distal (landscape controlling) indicators and green circles represent proximal (soil property) indicators. (b) Conceptual diagram of drivers of function: water purification (represented by proxy indicator: denitrification) in relation to drainage status (left hand side) and in relation to a combination of drainage status and N surplus (right hand side).

complexes, while alkaline conditions immobilise P by the precipitation of calcium phosphates (Sharpley, 1995; Daly et al., 2001; McGechan and Lewis, 2002; Jordan et al., 2005; Ellison and Brett, 2006; Brady and Weil, 2010; Yang et al., 2012).

Although P-sorption increases with the clay content, P loss by surface runoff is exacerbated on soils with low permeability (Schulte et al., 2006; Melland et al., 2012), which are commonly utilised in grassland production systems (Peukert et al., 2014). Waterlogged conditions also encourage the dissolution and re-mobilisation of the adsorbed P into soil water by bringing the pH towards neutral conditions and lowering the soil redox potential conditions (Sharpley, 1995; Scalenghe et al., 2012; Schoumans et al., 2014). Organic soils, such as peat, have a low capacity to strongly fix and retain P due to their low absorptive properties (low clay, Fe/Al and Ca content) and poor drainage status (Daly et al., 2001; Heredia and Cirelli, 2007; Brady and Weil, 2010).

Forested land, including riparian areas, has the highest capacity for P adsorption in soil, as the more acidic nature of coniferous forests or anoxic conditions within deciduous riparian buffers favour the complexation of P in more recalcitrant forms (Ellison and Brett, 2006; McDowell and Stewart, 2006; Qiong et al., 2008; Yang et al., 2012). In contrast, intensive grassland dairy systems receive a high P input from excreta from livestock and fertilisers (McDowell and Stewart, 2006), while arable sites receive high P fertiliser inputs. In these situations, soil drainage has a pivotal role in determining the pathway of P.

2.3. Carbon sequestration

Fig. 4 presents the conceptual model for the soil carbon (C) sequestration function. This refers to the rate of a “permanent” increase in SOC and comprises the transfer and retention of atmospheric carbon dioxide (CO_2) into the SOC pool (Lal, 2006). The rate of soil C sequestration is highly variable and is influenced by a range of factors including climate, soil type, land use and management practices (Ostle et al., 2009).

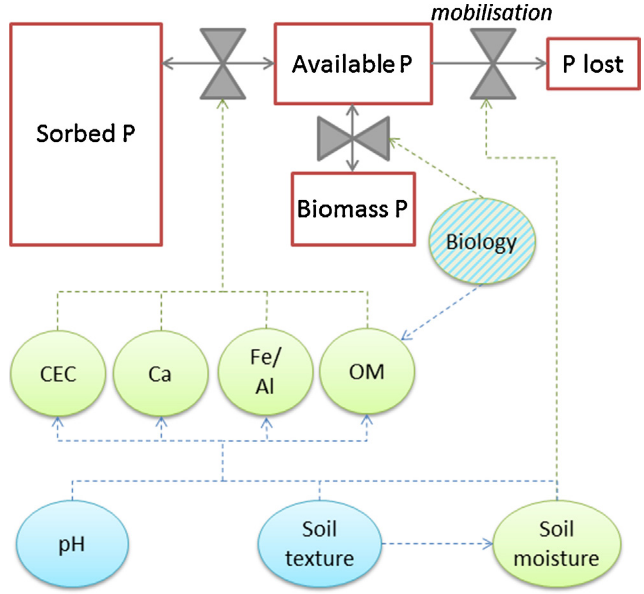

Fig. 3. Key parameters affecting the P-sorption process in soils. Boxes explain the process steps; valves symbols control the release and movement of phosphorus from/between the different forms in which it is present in the soil; green circles represent proximal indicators and blue circles represent distal indicators.

Undisturbed peat soils and the maintenance and establishment of forestry ecosystems (natural or plantation), in particular hardwood species, provide the largest C sequestration potential in the long term (Wigley and Schimel, 2000; Kirk, 2004; Xiong et al., 2014). Conversely, degradation of peat soils by drainage or cultivation can result in a rapid depletion of soil C pools (Freibauer et al., 2004).

The initial rate of C sequestration in grassland and arable systems can be similar and is dependent upon plant biomass cover and clay content/aggregates present in the soil. Potential sequestration requires the allocation of SOC from short term pools, such as those found in large macro-aggregates, to micro-aggregates in silt and clay fractions, which have longer turnover times (Briggs and Courtney, 1989; Wigley and Schimel, 2000; Freibauer et al., 2004; Marland et al., 2004; Brady and Weil, 2010; Heywood and Turpin, 2013). SOC in conventional tillage systems declines as a result of regular disruption of soil aggregates, through inversion practices, resulting in increased microbial decomposition of organic matter (Briggs and Courtney, 1989; Lal, 2004).

The retention of SOC in wetter soils with a higher clay content (Heywood and Turpin, 2013; Kong et al., 2009) increases due to stable soil aggregates and clay-humus complexes which provide an increased surface area for C-sorption (Krull et al., 2001).

While it is acknowledged that C retention will enhance other functions such as primary production and water purification, natural ecosystems (on peat soils) or forestry plantations on poorly drained, fine textured soils are expected to offer the best opportunity for higher C sequestration with grassland providing a moderate capacity.

2.4. Habitat for biodiversity

Soil biological communities are dynamic and vary according to the local conditions and, when determining the status of soil biodiversity, proxy indicators are often employed (Bispo et al., 2009). In this review, earthworms have been used as a proxy indicator of the status of soil biodiversity within the soil because they are considered the keystone engineers of soil biological communities. They serve as good bio-indicators for soil quality and habitat functioning (Breure, 2004; Jänsch et al., 2013). The use earthworms as indicators of biodiversity are supported by studies by Bispo et al. (2009) and Crotty et al. (2015).

Land use type has a large influence on soil biodiversity, which is represented by the abundance, biomass and species richness of earthworms. As shown on the simplified conceptual diagram (Fig. 5), earthworm biodiversity is greater under improved grassland than arable land, forestry and semi-improved grassland systems (Van Eekeren et al., 2008; Schmidt et al., 2011). Earthworm biodiversity is lower in arable production systems than improved grassland due to injury/death as a result of

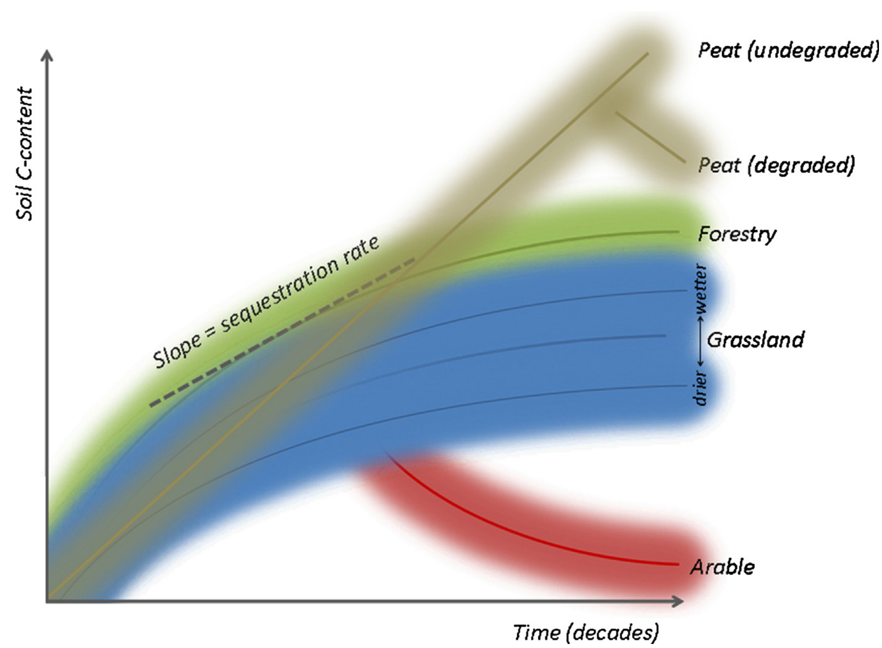

Fig. 4. Conceptual diagram of drivers of function: carbon sequestration, in relation to land management and drainage status.

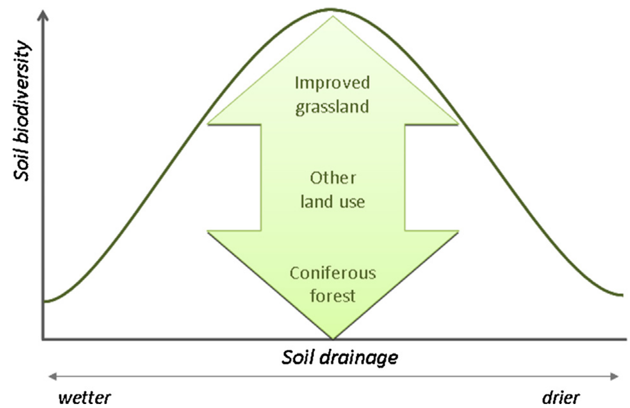

Fig. 5. Conceptual diagram of drivers of function: habitat for biodiversity, in relation to land management and drainage status.

cultivation, soil erosion, compaction, pesticide use, increased predation and lower availability of food (organic matter) (Bardgett, 2005; Brussaard et al., 2007; Van Eekeren et al., 2008; Turbé et al., 2010). Furthermore, forestry and semi-improved grassland systems are the least favourable land use types with respect to earthworm biodiversity due to a higher organic matter content (including peat) and associated lower pH in soils (Schmidt et al., 2011).

In order to enhance and sustain earthworm biodiversity, improved grassland on moderately drained soil provides the most favourable option. Although, different species of earthworms are suited to varying levels of soil moisture (Rombke et al., 2005), they lose weight in excessively dry soils and enter a diapause phase whereas poorly drained soils tend to be too cold (Turbé et al., 2010).

2.5. Nutrient cycling

Soil has the ability to absorb and detoxify organic wastes (Tan, 2009) by biogeochemical and physical processes, which are influenced by its drainage characteristics (Gardiner, 1986). The application of external sources of pig slurry is used as a proxy indicator in this study in order to determine the nutrient cycling capacity of soil under different types of land use, as shown in Fig. 6. Similar to other EU countries, the Irish pig industry, for example, has intensified by more than fivefold since the mid-1970s (Fealy and Schroder, 2008), which has increased the volume of animal waste generated. Limited land availability at intensive pig rearing units means that most of the slurry produced is exported equating to 5674 t P per year requiring soil recycling (Schulte et al., 2014).

Short rotational forests (SRF) provide the best opportunity to maximise nutrient cycling, especially nitrogen, due to the need for the regular replenishment of soil fertility following whole tree harvesting within short cycles and to optimise production (Heilman and Norby, 1998). However, site access to SRFs in remote areas may be difficult and expensive (see Fig. 6). In conventional forest systems, certain deciduous species, such as poplar, can tolerate substantial amounts (up to 100 tonnes ha^{-1}) of pig slurry whereas coniferous forests are not considered suitable for the long-term disposal of livestock waste (Gasser et al., 1980). Intensive arable systems have a higher capacity for nutrient cycling than intensive grassland systems, especially grasslands with high P-Index soils. This is due to the large input of nutrients required to replace those rapidly accumulated by arable crops (Sullivan et al., 1999) and those lost due to biomass export at harvest time (Briggs and Courtney, 1989). However, areas with the greatest number of pig farms generally have amongst the lowest amount of tillage and transport of slurry from such areas beyond 50–75 km can become prohibitively expensive (Fealy and Schroder, 2008).

Soils on intensively managed grasslands tend to have a low capacity to sustainably accept imported nutrients because of contributions of excreta from grazing livestock and applications of on-farm slurry arising from animal housing during the winter months. As shown on Fig. 6, nutrient management planning is complicated on intensive grassland by the mandatory requirement for paperwork for the import of externally produced slurry.

Moderately drained soils enhance the nutrient cycling capacity (Gardiner, 1986) because the rate of decomposition and mineralisation is greater than in waterlogged or drought prone soils (Gasser et al., 1980). Owing to their moderate permeability, these soils also minimise the migration of nutrients by overland flow to surface water bodies and by leaching to groundwater in comparison to lower and higher

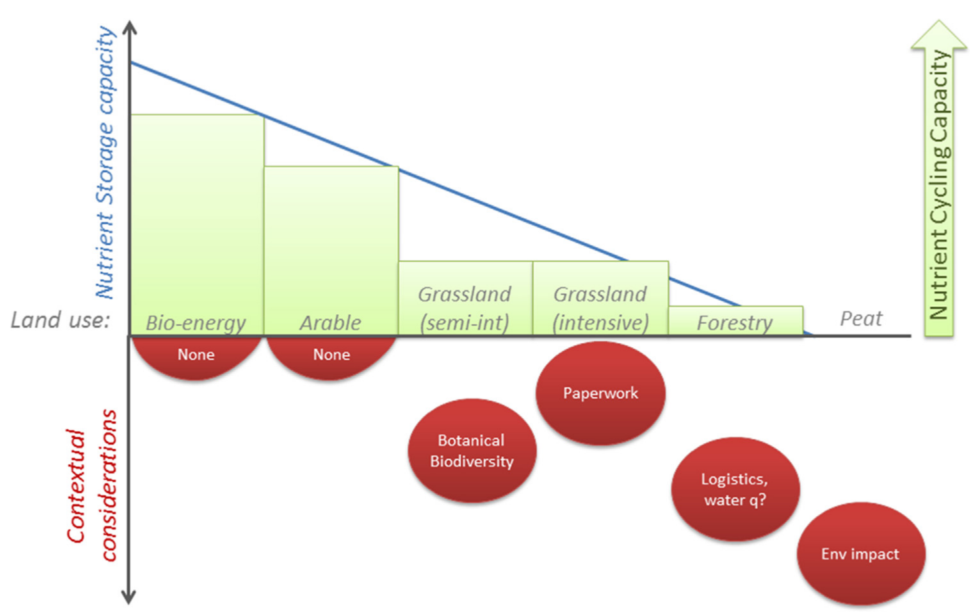

Fig. 6. Conceptual diagram of drivers of function: external nutrient cycling, in relation to land management. The text in the red oval shapes concern a number of key contextural considerations.

permeability soils, respectively (Loehr, 1977; Gasser et al., 1980; Gardiner, 1986). Peat soils are not considered suitable for the application of slurry due to their low winter rain acceptance potential (Gardiner, 1986) and the potential for nutrient migration by overland flow to surface water.

In summary, notwithstanding contextual considerations such as logistics (transportation and waste permits), the nutrient cycling capacity of soil is highest in SRF systems and intensive arable farming on moderately drained soil.

3. Discussion

3.1. Soil functional links

This review has extended the original Functional Land Management (FLM) conceptual framework to include soil drainage class. Moreover, the review, in conjunction with the conceptual schema, has presented how soil drainage characteristics in combination with land use are the dominant drivers that govern the magnitude of soil functions in Atlantic pedo-climatic conditions. This shared dependence on drainage and land use means that soil functions are interlinked and this needs to be incorporated into tailored soil management planning. The use of different soil characteristics will apply in contrasting pedo-climatic zones as the dominant drivers of soil functional capacity (Dominati et al., 2014; Calzolari et al., 2016), such as in the delineation of intermediate Less Favoured Areas by the European Commission (Eliasson et al., 2010). This is also the subject of further research in the EU Horizon 2020 project LANDMARK: Land Management: Assessment, Research, Knowledge base.

The individual soil function narratives are integrated and visualised in the matrix shown in Fig. 7. Here, the five soil functions are colour coded in line with the original diagrams by Schulte et al. (2014). The size of each of the boxes that represent the various soil functions, illustrates (conceptually) their relative proportions within each combination of land use and drainage category. These 'suites' of soil functions, and their response to drainage and land use, are produced by combining the relationships shown in Figs. 1, 2a, 2b, 3–6.

This matrix illustrates two important principles: (1) (almost) all soil and land use combinations perform all functions and (2) at the same time, the capacity to supply each individual function may differ significantly between soil and land use combination. For example, the capacity to supply biodiversity (residing within the soil) is greatest on moderately drained soils under improved grassland; the capacity to supply primary productivity is highest on well drained arable soils; while the capacity to sequester carbon is utmost in poorly drained coniferous forestry. Similarly, the capacity to supply water purification (by denitrification) is greatest under poorly drained broadleaf forests, while the potential for recycling of external nutrients is highest under moderately drained SRF systems.

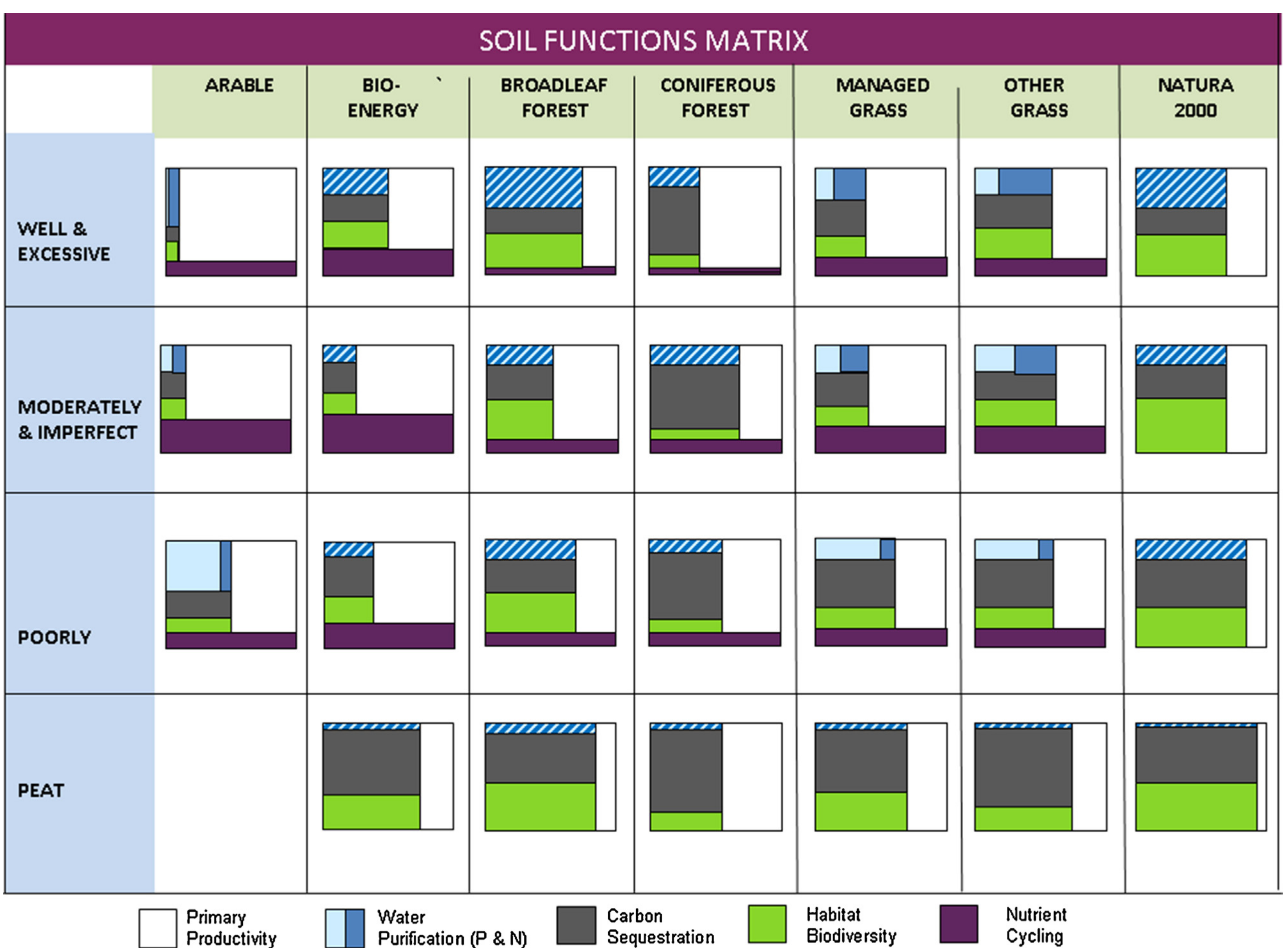

Fig. 7. Multi-functional soil management matrix showing the suite of soil functions under the different combinations of land use type (*x* axis) and soil drainage (*y* axis). For the water purification category, dark blue refers to P and light blue refers to N. Dark and light blue diagonal stripes denote where the predominance of P and N purification is less clear. The proportional differences between boxes can be referred to as the Delta (area), as defined by the conceptual models for each of the functions.

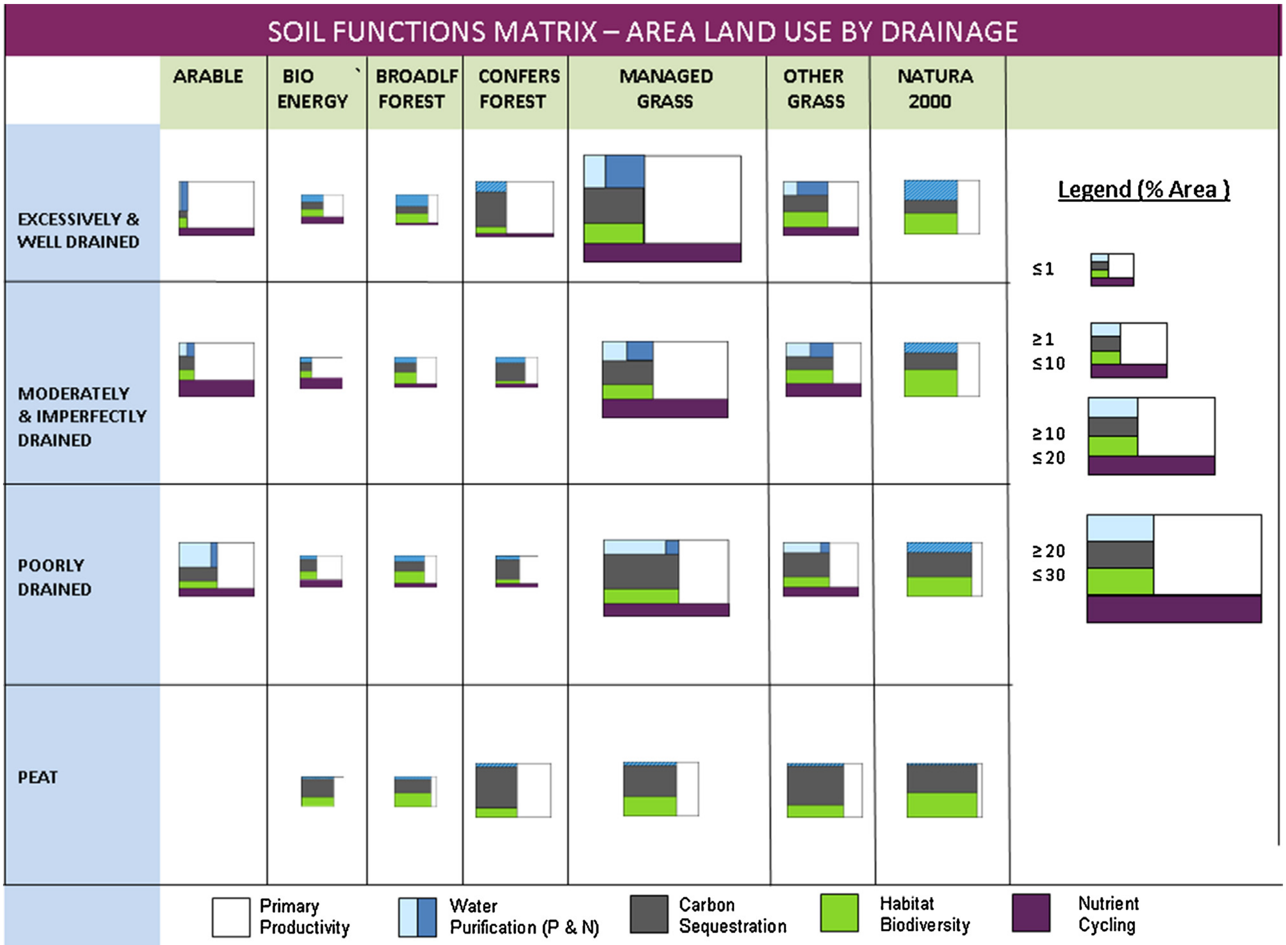

Fig. 8. Spatial extent of land area covered in Ireland by components of the matrix on land management by drainage class (legend as for Fig. 8).

3.2. A case-study: national supply of soil functions in Ireland

In Fig. 8, the suites of soil functions as displayed in the matrix (Fig. 7) are combined with the spatial extent of each land use by soil drainage combination for Ireland. This spatially explicit matrix, which is country-specific, is a powerful visual tool to assess both the supply of soil functions in one country and, when coupled with the review findings here, enables the identification of potential pathways towards optimising this supply through FLM. The concept of FLM can be applied at different spatial scales, ranging from local to continental level (Schulte et al., 2015)

3.3. Potential applications

Figs. 7 and 8 synthesise complex physical, biological and chemical reactions into a format that is useful at a policy level. These frameworks lend themselves to more effective policy making as they can incorporate the complex interaction between land use and biophysical constraints/utilities, therein providing the scope to support the sustainable intensification of agriculture. A recent example of this was provided by O'Sullivan et al. (2015), who assessed the impact of the installation of drainage systems on two soil functions, namely primary productivity and carbon sequestration at the regional scale. Drainage installation aims to upgrade a poorly drained soil into the moderately/imperfectly drained category. Fig. 1 shows that this can be expected to increase the potential for primary productivity, while Fig. 4 suggests that this may be at the expense of the carbon-sequestration potential. In addition, a change in drainage class may be associated with an increase in P-sorption capacity, a decrease in denitrification capacity (Fig. 2b) and a potential increase in earthworm abundance as an indicator of soil biodiversity (Fig. 5). Importantly, FLM is able to provide a simplified framework to support land use policies that are based on understanding the role of inherent soil properties in defining many of these soil functions (Schulte et al., 2014) and managing the expectations and consequences of land use decisions. As a development of this qualitative framework, an empirical understanding of these potential changes will help to guide assessments further as data become available (e.g. www.teagasc.ie/soil/SQUARE) and which have a catchment or sub-catchment soils' scale to augment the regional example provided by O'Sullivan et al. (2015).

4. Limitations

The soil function indicators selected in this paper were based on a comprehensive literature review and are representative of the dominant drivers of soil behaviour in Atlantic pedo-climatic zone conditions. However, these may not be relevant to all pedo-climatic zones and must be adjusted for the specific pedo-climatic scenario under consideration.

5. Conclusions

The functional capacity of a soil refers to its ability to provide a range of functions that underpin both agricultural production and environmental protection, which are integral in agri-environmental policies. This ability depends primarily on land use and soil characteristics. In the Atlantic pedo-climatic zone of Europe, soil drainage characteristics represent the dominant classifier.

The key to sustainable intensification is that the demand for soil functions, as defined by agri-environmental policies, is matched by the supply at a multitude of spatial scales. This can be achieved either through optimisation of soil management at the local level, or optimisation of land use and land management at larger spatial scales. It is reasonable to assume that farmers and foresters who are generally responsible for the supply of soil functions will manage land on the basis of the soil resource. Whilst acknowledging this, FLM can provide a framework for more targeted management. Equally, for policy makers, the provision of a landscape overview, can allow for more nuanced policy making that facilitates better objective setting or assessment of policies. Both approaches require quantification, or at least a ranking, of the capacity of contrasting soils and regions to perform each of the five soil functions. In both cases, there is greater cognisance of the trade-offs and impacts on other soil functions. The concept can be adapted and applied to other pedo-climatic conditions, with axes of the matrix relevant to the zone of interest.

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