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Understanding soil erosion impacts in temperate agroecosystems: bridging the gap between geomorphology and soil ecology

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Abstract

Soil is a key asset of natural capital, providing a myriad of goods and ecosystem services that sustain life through regulating, supporting and provisioning roles, delivered by chemical, physical and biological processes. One of the greatest threats to soil is accelerated erosion, which raises a natural process to unsustainable levels, and has downstream consequences (e.g. economic, environmental and social). Global intensification of agroecosystems is a major cause of soil erosion which, in light of predicted population growth and increased demand for food security, will continue or increase. Elevated erosion and transport is common in agroecosystems and presents a multidisciplinary problem with direct physical impacts (e.g. soil loss), other less tangible impacts (e.g. loss of ecosystem productivity), and indirect downstream effects that necessitate an integrated approach to effectively address the problem. Climate is also likely to increase susceptibility of soil to erosion. Beyond physical response, the consequences of erosion on soil biota have hitherto been ignored, yet biota play a fundamental role in ecosystem service provision. To our knowledge few studies have addressed the gap between erosion and consequent impacts on soil biota. Transport and redistribution of soil biota by erosion is poorly understood, as is the concomitant impact on biodiversity and ability of soil to deliver the necessary range of ecosystem services to maintain function. To investigate impacts of erosion on soil biota a two-fold research approach is suggested. Physical processes involved in redistribution should be characterised and rates of transport and redistribution quantified. Similarly, cumulative and long-term impacts of biota erosion should be considered. Understanding these fundamental aspects will provide a basis upon which mitigation strategies can be considered.

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1 Introduction

Soil sustains humanity and, with an increasing global population, demands on soil are increasing (UN, 2004). The Millennium Ecosystem Assessment (MA) estimated that global food production increased by around 168 % since 1963, and that 80 % of increased agricultural outputs resulted from intensification of agro ecosystems (MA, 2005).

Soil productivity is threatened by transport and redistribution of biota by erosion and downstream sediment transport (Powlson et al., 2011). Land use changes enhance mobility of biota (Chapin et al., 2000) over multiple scales, and egress of organisms from ecosystems may decrease system functionality (Diaz et al., 2006; Brussaard et al., 2007). Soil erosion redistributes organic and inorganic materials across the landscape (Dungait et al., 2013); driven by climate, topography, land management and wider anthropogenic impacts (Collins and Owens, 2006; Helming et al., 2006; Powlson et al., 2011). Lal et al. (2007) estimated that 45 % of arable land is degraded to some extent, a component of degradation is moving into more erosion-vulnerable areas as good agricultural land is eroded (Pimentel, 2006). With growing intensification of food production systems the security of soil as a provisioning resource and the supporting services it provides is uncertain (Banwart, 2011). Moreover, future climate changes may increase uncertainty and vulnerability of production systems, and may lead to increased incidence of extreme weather events (IPCC, 2007). There has been a tendency in the literature to overlook if and how soil can tolerate greater demands for increased agricultural yields creating a risk that potential vulnerabilities are ignored and thus mismanaged (Banwart, 2011).

Pankhurst (1997) distinguishes between soil quality and soil health, the former a measure of the capacity of soil to meet plant and ecosystem requirements, and the latter more broadly describing the condition of the soil ecosystem and its functionality. Through their discrete roles in the soil food web, biota promote soil quality by maintaining soil health, and in turn maintain productivity, for example in the form of agricultural outputs (cf. Pimentel and Kounang, 1998; Hunt and Wall, 2002; Lavelle et al., 2006; Coleman, 2008). In agro ecosystems soil quality is essential for ensuring food productivity (Moebius-Clune et al., 2011) and can be characterised by chemical, physical and biological parameters (Chen et al., 2010). Soil productivity is threatened by the loss of important ecosystem components. The loss may be by transport and redistribution of biota as erosion and downstream sediment transport, a natural process characteristically accelerated by human activities. Where soil biota, such as nematode populations, are threatened so too are the functions they provide in terms of supporting and regulating services (Pimentel, 2006). There is limited research quantifying transport of soil biota by erosive processes and to our knowledge very few studies that have characterised mechanisms responsible for nematode movement. It is therefore essential to improve our understanding of the impacts of erosion to soil biota to further our knowledge of potential threats to soil.

One of the biggest challenges for soil ecology is the integration of complex processes (Lavelle, 2002). Integrating different scientific disciplines into agricultural research (including the social sciences) has led to a better understanding of impacts of agricultural practices and has improved good practice (Lichtfouse et al., 2010). We propose integrating geomorphology and soil ecology with agricultural science to assess effects of soil erosion on biota.

Here we relate soil physical processes to biological impacts of erosion in temperate agroecosystems. We focus on nematodes as a model group (Gupta and Yeates, 1997; Bongers and Ferris, 1999) for assessing water-induced soil erosion effects from plot to catchment scale.

2 Soil erosion and sediment budgets

Soil erosion involves a series of processes (the detachment and transport of soil particles and associated biota) from slopes into channel networks (natural and man-made channels that drain water from land), for which a mass balance, or sediment budget,

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can be calculated at various scales from plot to catchment (Walling, 1983; Walling and Collins, 2008). By quantifying sources (e.g. sheet, rill and gully erosion), sinks (e.g. footslopes) and outputs (catchment sediment efflux), an understanding of sediment delivery dynamics can be achieved (Ferguson, 1981; Walling et al., 2001). While gross upland erosion rates may be substantial, net efflux of sediment from the catchment system may only be a fraction of that mobilised, with the majority remaining in storage (Phillips, 1991; Trimble and Crosson, 2000).

2.1 Soil erosion by water

Arable agro ecosystems are particularly vulnerable to water erosion, compared with natural grasslands or forest systems, because soil is highly perturbed during conventional cultivation, and may be left bare following planting until crop-cover is established (Davidson and Harrison, 1995).

Rainsplash and slaking initiate the erosion process, occurring frequently but contributing minimally to erosion rates, relative to other processes. Energy from raindrops impact the soil surface and compact and break-down aggregates dispersing soil particles. With increased wetting, slaking and related mechanisms release clays and fine silt particles, resulting in surface sealing (Warrington et al., 2009), decreasing infiltration and water retention rates. Overland flow occurs either when precipitation exceeds infiltration capacity (infiltration excess overland flow), or when soil pore spaces are full as a result of groundwater flow or interflow (saturation excess overland flow) (Nash et al., 2002). The onset of overland flow depends on antecedent soil water, hydraulic conductivity of soil, and precipitation rate and volume (Emmett, 1978). Initiation of overland flow is typically associated with ponding and sheet flow, developing into complex patterns, influenced by micro-topography and local obstacles. Hydrological flow is typically laminar and selective entrainment, transport and downslope deposition of fine soil particles occurs. Rills can form as a result of convergence of overland flow, or occasionally by sapping mechanisms, into concentrated micro-channels (dimensions 2–250 mm wide and deep). Confined flow causes scour of rill beds, often accompanied by collapse of steep side-walls that generates sediment of non-specific particle size for transport. Typically rill networks develop by headcut retreat upslope, resulting in stepped-longitudinal profiles (Bryan, 1987). These channels form an efficient transport network for runoff and sediment. Though less widespread or frequently activated than inter-rill erosion, rills erode soil, and biota, at greater rates (Brunton and Bryan, 2000). Gullies can develop through multiple mechanisms (Bull and Kirby, 1997). They are less common but where locally formed, erode soil at greater rates. Gully dimensions are highly variable, but are typically 0.5–30 m deep and again particle size of eroded sediment is non-selective (Bull and Kirkby, 1997). Soil can also be lost in agro ecosystems through mass movements, in the form of rotational and transitional landslides. Such features are usually localised and tend to be associated with the steepest ground ($> 20^{\circ}$) with relatively little vegetation and with significant variations in water table height (cf. Ruhe, 1975; Beguera, 2006; Turkelboom et al., 2008). Although localised, landslides can result in massive and non-selective soil losses downslope. The susceptibility of soil to erosion, its erodibility, is determined by physical, chemical and biological properties, which include soil texture, amount and nature of organic matter present, water content, land cover and the energy of the eroding force (Mamedov et al., 2000; Verheijen et al., 2009).

The processes involved in water erosion have specific magnitude-frequency relations which determine erosion rates. DEFRA (2006) estimated a typical range of combined water-erosion rates for England and Wales of 0.1–15 $t\,ha^{-1}\,yr^{-1}$. These erosion rates have a direct effect on the redistribution of soil biota, which until now have remained largely under-investigated, the relationship between magnitude and frequency of different erosive processes are presented as Fig. 1.

2.2 Soil erosion by wind

Wind erosion is caused by the combined effect of high wind velocity, loose surface-soil particles and insufficient soil-surface protection (Verheijen et al., 2009). Through surface creep, saltation and suspension, soil particles can be eroded from vulnerable

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soils, and in the England and Wales rates have been estimated at 0.1–2.0 $t\,ha^{-1}\,yr^{-1}$ (DEFRA, 2006). Erosion of soil and organic particles by wind can occur when wind speeds exceed 3 $m\,s^{-1}$ (Carroll and Viglierchio, 1981). In Europe, where wind erosion poses less risk than most other regions, impacts can be localised but extreme and an estimated area of 3 million ha of the north-western European lowlands are at high potential risk, as well as parts of the Mediterranean (Riksen et al., 2003; Verheijen et al., 2009). Therefore there is potential for soil biota to be transported by the same erosive mechanisms. This may exacerbate soil susceptibility to wind erosion, since it has been demonstrated that biota are important in aggregating soil (cf. Tisdall et al., 2012).

2.3 Soil erosion due to tillage

Tillage is recognised as one of the major agents of soil erosion (Powlson et al., 2011) and rates can exceed those of natural processes, particularly on slopes (Govers et al., 1994; DEFRA, 2006). As land is cultivated, ploughing displaces (translocates) soil, breaking aggregates. Compaction, as a result of loading from agricultural machinery, alters soil properties promoting increased surface runoff and susceptibility to water and wind-borne soil loss (Addiscott and Thomas, 2000, Bailey et al., 2013). Greatest soil loss from tillage occurs on slope crests and shoulders where slopes are steepest, and deposition occurs in concave depressions (Van Oost et al., 2006). Increased depth and frequency of tillage, ploughing downslope and ploughing on steeper slopes increase the severity of erosion, and accumulations of translocated soil will be most susceptible to subsequent erosion by wind and water (Van Oost et al., 2006; Dlugoß et al., 2012). Translocation distance of soil as a direct result of tillage has been recorded to be as much as 10 m (Van Muysen et al., 2006) and estimates of gross tillage erosion in Europe are quantified at 3.3 $t\,ha^{-1}\,yr^{-1}$ (Van Oost et al., 2009). Soil can also be lost during crop harvesting operations. During harvests soil can be eroded when adhering to farm machinery, and can be 'co-extracted' with crops, particularly root crops, and

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rates of co-extraction in England and Wales have been estimated at 0.1- 5 t ha^{-1} yr^{-1} (Ruysschaert et al., 2004; DEFRA, 2006; Verheijen et al., 2009).

Mechanised agriculture increases erosion above background levels, decreasing on-farm productivity and increasing problems associated with diffuse-pollution to downstream channel networks (Verstraeten and Poesen, 1999). An under-recognised dimension of this is the simultaneous loss of soil biota, which are also physically redistributed and subject to comparable downstream delivery processes (Pimentel and Kounang, 1998).

2.4 Scales of erosion and the sediment budget

Figure 1 highlights the link between different mechanisms of erosion and their contribution to erosion rates. They have specific magnitude-frequency relationships that determine runoff rates, which in turn determine sediment and sediment-associated biota fluxes. For example, rain-splash erosion is low magnitude, involving distances of mm^2 to cm^2 (Rickson, 2006), but occurs at high frequency across hillslopes, in contrast, rills occur less frequently but the magnitude of sediment and associated biota transport is greater. The inefficient delivery of sediment and erosion-associated biota to the channel network (e.g. $< 10\,\%$ of eroded sediment) due to the dominance of features like overland flow and rill networks (low magnitude) create a process of in-field deposition and intermediate storage (e.g. 50 % of total erosion) (Ferguson, 1981). Transport and redistribution of sediment and biota to depositional zones like foot slopes, field margins and buffer strips may occur. Therefore within-field redistribution may have significant consequences for environments of net erosion and deposition in terms of biota transport and soil-ecosystem productivity. Moreover the active movement of biota in response to changes in their environment, for example movement upwards in the soil profile due to increased soil water by rainfall may increase the susceptibility of organisms to transport by erosive processes (cf. Roots, 1956).

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3 Soil erosion and climate change

Global agricultural systems have a strong record of resilience, but the uncertainty and regional variability presented by climate change may challenge the strength of this resilience (Burton and Lim, 2005). Despite technological improvements in agriculture, including mechanisation and improved irrigation and the trend to conservation agriculture, weather and climate are still important factors that influence production (Parry et al., 1999). In order for agroecosystems to adapt to potential climate changes, namely increased variability of temperature and precipitation, resilience must be built into production systems (Smith and Olesen, 2010). A higher frequency of extreme rainfall events (high-intensity precipitation), as a result of increased temperature and pressure gradients (Favis-Mortlock and Guerra, 1999), is likely to exacerbate erosion-related impacts to agroecosystems, and thus understanding these impacts to soil biota is essential in building resilience.

4 Impacts of erosion to soil biota

Literature on the impacts of erosion to soil biota in arable agro ecosystems is scarce. Of the few studies exploring erosion-biota relationships (e.g. Cadet and Albergel, 1999; Planchon et al., 2000; Cadet et al., 2002; Villenave et al., 2003; Chabrier and Quénéhervé, 2008; Chabrier et al., 2009) most are from the tropics. In temperate settings, dispersal of plant-parasitic nematodes (PPNs) has been reported through irrigation channels indirectly linked to catchment water and sediment transport (Faulkner and Bolander 1966, 1970a, b). The presence of nematodes in drinking water has also been reported (Tombes et al., 1979; Mott et al., 1981; Mott and Harrison, 1983). However the consequences of soil erosion to biota in terms of ecosystem service provision have hitherto been overlooked.

4.1 Importance of soil biota in the context of erosion

The importance of soil to ecosystem services is established (MA, 2005). Biota are fundamental to the delivery of both goods and underpinning services through roles they serve in soil food webs (Barrios, 2007; Ritz et al., 2009). Key roles include decomposition; nutrient cycling and storage; carbon sequestration; maintaining environmental quality; regulating parasites, and creating and maintaining soil aggregation, critical processes for the sustainable management of agro ecosystems (Hindell et al., 1997; Neilson et al., 2000; Ferris et al., 2001; Lavelle, 2002; Lavelle et al., 2006; Barrios, 2007).

Biodiversity is central to ecosystem functioning, and soil subject to stress may display decreased diversity compared with unstressed systems (Brussaard et al., 2007). Ecosystem perturbation, for example by erosion, is unlikely to be restricted to a single trophic level but would cascade through the food web at a range of scales (Pimentel and Kounang, 1998; Raffaelli et al., 2002), impacting biodiversity and potentially leading to a loss of ecosystem function.

Erosion estimates in Europe range from 3.3–17 $t\,ha^{-1}\,yr^{-1}$ (Pimentel, 1995; Van Oost et al., 2009), and the range of tolerable erosion rates (rates less than or equal to soil formation rates) have been estimated from 0.3–1.4 $t\,ha^{-1}\,yr^{-1}$ (Verheijen et al., 2009), therefore estimated erosion rates in Europe are at least an order of magnitude greater than rates of soil formation. Soil erosion decreases productivity through loss of water, soil organic matter, nutrients, soil depth and decreased abundance and diversity of soil biota (Atlavinyte, 1964, 1965; Pimentel and Kounang, 1998). In order to follow the assertion by Verheijen et al. (2009) that the definition of "tolerable soil erosion" should include loss or impact to soil function, we must first investigate how erosion impacts biota and the ecosystem services they deliver.

Ecological processes contributing to ecosystem services vary across time and space, from micrometres for microbial processes to hundreds of meters and beyond for soils and landscapes (Lavelle et al., 2004). Similarly erosive processes (Fig. 1) vary spatially and temporally (Verstraeten et al., 2002); thus so will the impacts of erosion

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on ecosystems and the services they provide. Decreased soil depth, water storage and infiltration capacity, and concentrations of nutrients and soil organic matter are key impacts of erosion that damage the soil ecosystem (Pimentel and Kounang, 1998), impacts of erosion on biota requires further investigation to be added to this list.

Recent estimates suggest that a quarter of all faunal species live exclusively in soil and soil litter (Decaëns et al., 2006), including micro-biota (e.g. bacteria, fungi, protozoa and nematodes), meso-biota (e.g. collembola, mites and enchytraeids) and macro-biota (e.g. ants, beetles and earthworms) (Giller et al., 1997).

Lavelle et al. (2006) identified five scales of soil function, microbial bio-films (micro-biota), the micro-food web (micro- and meso-biota), functional domains of ecosystem engineers (macro-biota), mosaics of functional domains where groups of macro-biota that affect the soil differently are nested together (macro-biota), and landscapes (all biotas). Beare et al. (1995) described five spheres at which biodiversity influences soil structure and function, the detritusphere- decaying plant and animal matter, the drilosphere- zone of earthworm influence, the porosphere-solid particles and air and water filled voids, the aggregatusphere-organic and mineral particle aggregates, and the rhizosphere-zone of primary plant root influence. Combining these definitions with the erosion mechanisms described here clearly identifies erosion as a potentially significant dispersal mechanism that operates over scales ranging from individual soil aggregates and micro-habitats to catchment and indeed the landscape scale.

Soil micro-aggregates comprised of mineral and organic matter are $< 250\,\mu m$ diameter (Tisdall and Oades, 1982). Micro-biota within biofilms and the micro-food web (Lavelle et al., 2006) are involved in ecosystem processes associated with all five spheres of biodiversity. Micro-biota are likely to colonise and inhabit niches within micro-aggregates from the time of aggregate-formation until it is disturbed by mechanical breakdown, e.g. associated with tillage or water erosion (Hattori and Hattori, 1993).

Aggregates are greater than $250\,\mu m$ (Tisdall and Oades, 1982), and micro- and meso-biota may dwell within individual aggregates or in the water-filled voids between them, the porosphere. Rain splash erosion may have considerable impact on the structure of surface soils where aggregates are disrupted, broken apart and primary particles displaced. Soil structural changes alter the habitat of biota by clogging pores, as well as physical displacement of biota in the direction of transport. Fine soil particles are selectively entrained by overland flow erosion, and this may also be a transport mechanism for biota similar in size and mass to fine soil particles (Cadet and Albergel, 1999).

Macro-biota occupy the pedosphere, the functional domain of ecosystem engineers (Lavelle et al., 2006). The increased mobility and size of macro-biota give them advantage over meso- and micro-biota in responding to pressures like tillage treatments or physical redistribution through erosive soil loss because they are able to move away from such perturbations. However, under very wet soil conditions macro-biota may move closer to the surface towards preferable environmental conditions (cf. Roots, 1956).

Rain splash and slaking may only be significant erosive processes for relatively small surface-dwelling (or near-surface) organisms, but the drilosphere may be impacted as worm burrows become clogged and blocked by fine particles. Moving up through the energy and effective-erosion-depth continuum (rills, gullies and mass movements) typically means less frequent but higher magnitude events that result in soil being lost to greater depths. This has the consequence for wholesale transport of the contained biota, with micro- and meso-biota transported in association with the soil, and macro-biota escaping the destruction of the habitat. The impact of erosion at high magnitude, e.g. as a consequence of rill and gully erosion, results in greater stresses translating into loss of habitable space (Pimentel and Kounang, 1998), and indeed loss of habitat, e.g. the detritusphere. The nested relationship of erosion processes, spheres of soil function and soil biota are presented as Fig. 2.

4.2 Transport and passive dispersal of soil biota

Dispersal of biota plays a key role in the evolution of populations and species (Gibbs et al., 2010), and drives the spatial and temporal distribution of genotypes (Ronce, 2007).

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Dispersal can be by movement of organisms (active) or through transport by natural and anthropogenic processes (passive) (Eijsackers, 2011). A body of literature has been dedicated to understanding active dispersal and its role in the evolution of population dynamics, community structure and spatial heterogeneity (Bowler and Benton, 2011) that influence patterns of ecosystem services (Ettema and Wardle, 2002).

Wind dispersal has been investigated as a dispersal mechanism for plant parasitic nematodes (PPNs) (cf. Carroll and Viglierchio, 1981; Andrade and Asmus, 1997; de Rooij-van der Goes et al., 1997; Wharton, 2004; Nkem et al., 2006; de la Peña et al., 2011). Dry conditions are necessary and typical transport distances of 5km have been reported, as well as extremes of 40 km (Carroll and Viglierchio, 1981). Nematode ability to enter a state of anhydrobiosis (a dry-state where metabolism is reduced to undetectable levels to survive desiccation) has been reported as an adaptation strategy to enable passive dispersal (Carroll and Viglierchio, 1981; Nkem et al., 2006). However, whilst wind transport may be capitalised upon as a dispersal strategy in some habitats, in others, it can cause mortality. In a study of wind blow outs in sand dunes de Rooij-van der Goes et al. (1997) reported that scour forces caused by saltation of sand particles killed nematodes, and when set into suspension fine particles effectively sieved nematodes out of sand.

Water is a mechanism for passive dispersal of soil biota (cf. Freckman and Baldwin, 1990; Dighton et al., 1997; Terhivuo, 1988; Terhivuo and Saura, 2006; Eijsackers, 2011). Moreover, the role of soil loss through erosion, tillage, harvesting and other agricultural activities is also potentially important in passive dispersal. Boag (1985) demonstrated that virus vector nematodes are passively dispersed at small scales (e.g. plot and field) when soil adheres to farm vehicles and machinery. Rainfall has been identified as a passive dispersal mechanism of PPNs from plot to catchment scale (Freckman and Baldwin, 1990, Hugo and Malan, 2010). Terhivuo (1988) reported that the wide spatial distribution of earthworm (*Lumbricidae*) species in Finland can in part be attributed to passive dispersal during flooding due to their presence close to the soil surface. Terhivuo and Saura (2006) reported that the earthworm (*Eiseniella tetraedra*) capitalises on passive dispersal, where eggs are washed into flowing water and over large distances in streams and rivers.

4.3 Understanding biological consequences of erosion

It is necessary to consider how soil ecosystems and the services they provided are impacted by redistribution of biota by erosion. Debate remains in the literature regarding the role and importance of the contribution of individual species to ecosystem function (Barrios, 2007), but net loss of biota and physical restructuring of habitats as a consequence of erosion may be damaging. The ecological redundancy hypothesis (Walker, 1992) argues that losing one species would not necessarily lead to loss of ecosystem services, provided key functions were fulfilled by remaining biota. However where there is increasing dependence on species that compensate for losses, resilience of the ecosystem is compromised (Naeem, 1998). Transport of biota by erosion is unlikely to be selective to particular species and therefore multiple species loss may occur, including loss of key drivers of ecosystem services. Even with redundancy in the function provided by individual species, effects of cumulative erosion over larger timescales (e.g. years to decades) will impact compensating species as well as the physical habitat. In this context the rivet-popping hypothesis (Ehrlich and Ehrlich, 1981) seems more fitting, where the exponential loss of species causes loss of system productivity, leading to eventual system collapse. In the context of agro ecosystems, collapse is likely to be offset by increased use of chemical fertilisers to maintain yields.

Therefore the impacts of erosion to soil biota are twofold with (a) physical erosion, transport and redistribution of soil biota, and (b) modification of habitats through erosion-induced restructuring. Erosion can passively disperse soil biota, impacting ecosystem services that underpin productive soils (Pimentel, 1995; Pimentel and Kounang 1998). Over the time scale of years to decades the spatial patterns of erosion and their cumulative consequences to ecosystem structure and function may emerge, and these patterns in turn impact ecosystem services. Furthermore the process of erosion, characterised by the jerky conveyor belt (Ferguson, 1981), changes the physical (e.g. soil texture, structure and hydrology) and chemical (e.g. nutrients and soil organic matter) environment, and degrades top and upslope erosional environments enriching downslope depositional environments, modifying the quality of habitat for soil organisms.

5 Soil nematodes for investigating biota redistribution by erosion

Unravelling complexities of the soil food web using a multi-trophic approach is challenging (Ferris et al., 2001; Koch et al., 2011; Santorufo et al., 2012). The diversity and distribution of biota in soils, coupled with the heterogeneity of the soil matrix (Nielsen et al., 2010), require that appropriate biota be selected when seeking to answer questions about ecosystem impacts (Barrios, 2007).

Soil nematodes are aquatic and range from 40 µm to 5 mm long. In soil, they live on thin water films (1–5 µm) surrounding soil particles and move through soil pores 25 to 100 µm diameter (Neher, 2010). Nematodes serve important roles in the soil food web, are ubiquitous, and all of the > 20 000 described species (Hugot et al., 2001) can be sampled and extracted with relative ease (Bongers and Ferris, 1999; Ritz and Trudgill, 1999). Their high diversity, abundance and trophic heterogeneity can provide insight into the condition of a food web and can aid understanding of ecosystem health and quality (Bongers, 1990). Morphological characteristics of nematodes can be translated into trophic groups (Yeates et al., 1993) and the relative abundances of these groups in an assemblage can infer the state of an ecosystem (Neher et al., 1999). Nematodes are central to decomposition and nutrient cycling and are good indicators of changes in biological and physico-chemical properties of soils due to their sensitivity to disturbance (Neher, 2001; Landesman et al., 2011). Moreover they are widely recognised as a useful group for measuring environmental impact and change (e.g. Gupta and Yeates, 1997; Porazinska et al. 1999; Wall et al., 2002; Yeates, 2003; Griffiths et al., 2012). Therefore trophic groupings in nematode assemblages could be used to test whether erosion selectively transports biota based on size or mass.

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Griffiths et al. (2002) demonstrated that nematode communities respond to changes in the soil environment and that a number of factors are involved, including previous land use, soil structure, soil water regime and species factors. Griffiths et al. (2003) further demonstrated that soil properties have an effect alongside biological factors, like microbial (food source) and faunal (predators) communities. Nematodes can be used as a model to understand more complex soil ecosystem components (Ritz and Trudgill, 1999) and therefore can be used to investigate the impact of erosion on soil biota.

5.1 Nematodes reported in rainfall runoff in tropical agro ecosystems

There are limited data of erosion effects on nematode transport, particularly in temperate agroecosystems and studies identified here are from tropical regions. Cadet and Albergel (1999) reported water-borne nematode transport in the Sudano-Sahelian region of Senegal during the wet season, likening nematode size and mass to that of fine soil particles. This highlighted the potential for redistributed PPNs to impact on crop productivity. Planchon et al. (2000) quantified the relative importance of rain-splash and surface wash on nematode redistribution on erosion plots in Senegal. This demonstrated that rain splash was responsible for detachment as an important precursor to entrainment by surface runoff which was the key downslope transport mechanism.

Field-plot rainfall-simulations showed that nematode entrainment occurred at discharges 25 % of that necessary for loss of soil particles, that drought-adaptation strategies influenced abundance, and more beneficial than PPNs were found in runoff assemblages (Cadet et al., 2002). Furthermore an important paradox was suggested, PPN dissemination into uninfested areas may be damaging, but the introduction of other nematodes may be beneficial. Villenave et al. (2003) observed selective transport of bacterivorous and omnivorous nematodes in the same experiment. This selectivity was explained by the combined effect of erosion of the surface layers of the soil and presence of these trophic groups close to the soil surface.

Chabrier and Quénéhervé (2008) divided a steep fallow field into plots to test the importance of leaching in translocation of *Radopholus similis* a plant-parasitic nematode, within a soil profile. They reported that runoff water is a dominant dispersal mechanism at field scale and advocated the use of drainage ditches to hydrologically isolate crops and thus control dispersal. The factors identified as controlling loss rates included rainfall intensity, soil water content and transport-path lengths, e.g. from erosion source to field boundaries or the channel network.

Whilst using drainage ditches to isolate crops from potential PPN infestation has been proposed, their use may also create effective pathways for nematodes to reach other environments (Chabrier and Quénéhervé, 2008). Waliullah (1984) reported higher PPN abundances in irrigation canals within the Upper Ganges during the rainy season, suggesting that field runoff is an important transport mechanism. Similar results were found in Kashmiri canals (Waliullah, 1989) with higher PPN abundances associated with areas of highest runoff and upland soil erosion. In contrast, comparatively lower PPN abundances were reported in irrigation canals of southern Italy (Roccuzzo and Cianco, 1991), most likely due to the concrete lining of canals that decreased bank erosion and soil-derived PPNs, and the use of flow-regulating dams that allowed PPNs to settle out of the water column (Roccuzzo and Cianco, 1991).

5.2 Relevant nematode studies from temperate regions

Evidence from studies in the tropics illustrates the potential role of dispersal of PPN and other nematodes by runoff and associated sediment transport. These examples are from tropical regions that have common climatic characteristics involving long dry periods and intense wet seasons with corresponding hydrological regimes. Nematode communities in these regions are adapted to the climate, raising the question of transferability from the tropical studies to temperate agro ecosystems.

Early studies into the source of nematodes in irrigation canals in temperate agroecosystems reported rainfall runoff as a key transport mechanism (Faulker and Bolander, 1966, 1970a, b). Landesman et al. (2011) investigated nematode community response to varying rainfall patterns and showed a positive relationship between increased precipitation and nematode abundance. Certain trophic groups were reported

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to benefit from resource increases by primary production, and nematodes were found to be highly sensitive to dry conditions. Soils able to retain water will therefore favour nematode survival for longer under drought stress. Conversely, under very wet conditions when soil becomes saturated, nematodes are likely to be transported up through the soil profile, and thus more vulnerable to transport by erosion.

Nematode densities in drinking water from treatment plants have been positively correlated with precipitation; streamflow and suspended sediment concentrations, with nematodes mobilised from channel-beds and from hillslopes (Tombes et al., 1979; Mott et al., 1981). Similar work undertaken by Mott and Harrison (1983) reported that a substantial proportion of nematodes recorded in streams were from soil habitats, noting a strong correlation between nematode densities and periods of high precipitation, and from snow-melt water.

6 Progress for understanding soil erosion and redistribution of biota

Multiple in transport of nematodes from soil and channel banks, mechanisms are involved erosive including rain splash, dispersal in surface flow and deposition in receiving environments. The vertical distribution of nematodes within soil profiles, in terms of both absolute numbers and relative abundances, is an important factor controlling their likelihood of being transported (cf. Chabrier et al., 2009) and may depend on soil water status. By extension this also means that different trophic groups are likely to be more or less susceptible to different sets of erosion processes.

Redistribution of soil biota by erosion may be detrimental to soil ecosystem services through loss of important components of the food web, or decreased resilience of the system, alongside a suite of soil chemical and physical impacts. Agriculture is a major source of soil erosion, and in the US alone the annual cost has been estimated at $37.6 Billion (Uri, 2000), but this figure does not account for the loss of ecosystem services resulting from impacts to soil biota. There exists a need and opportunity to value changes in ecosystem services resulting from soil erosion.

Further work is necessary to understand biological impacts of erosion by land use activities (e.g. agriculture) in temperate regions. The twofold issue of erosion to soil biota requires investigation of mechanisms responsible for transport and redistribution and the significance of this to the ecosystem. Moreover the consequences to soil biota of physical and chemical restructuring of soils by erosion must be explored. Only once such effects have been identified and understood can mitigation strategies be considered.

We advocate nematodes as a useful model organism to develop such an understanding. In the context of climate change, with a prediction of increased precipitation in northern Europe, and increased risk of extreme weather events (IPCC, 2007), rainfall-induced transport should be investigated to determine the scale and extent of nematode redistribution and moreover to investigate implications for affected soils.

Erosion risks adversely impacting soil and climate change may exacerbate these risks. As the world population grows, and demands for increased production from agro ecosystems continue, how soil is managed will determine its ability to sustain humanity. Soil biota are central to sustainable, productive agro ecosystems through the important processes they mediate, yet the relationship between erosion and biota is under-investigated. This knowledge-gap creates uncertainty about the long term sustainability of the multiple ecosystem services soil provides, and therefore further work is required.

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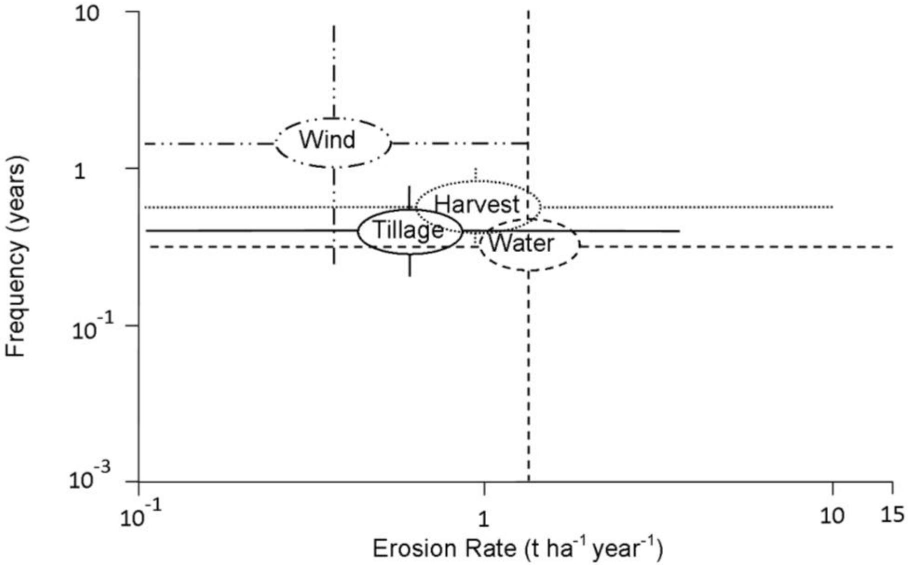

Fig. 1. Frequency and magnitude of erosive processes affecting arable agroecosystem soils.

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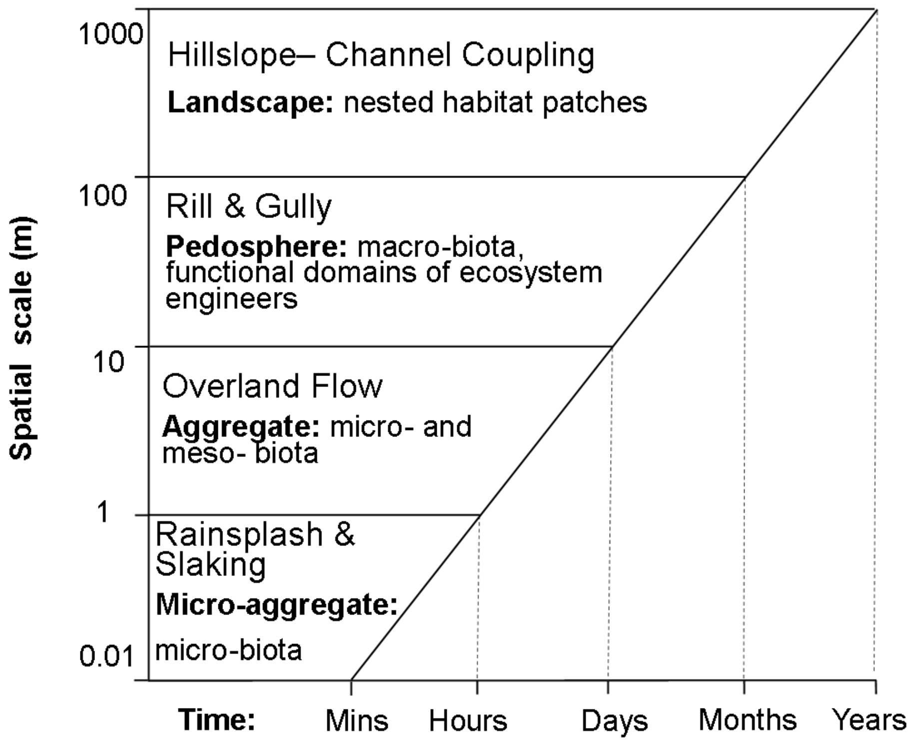

Fig. 2. Schematic diagram of the nested spatial and temporal scales of water-induced erosion processes, and the associated spheres of soil ecosystem function.

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