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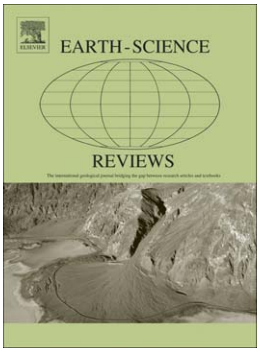

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Tolerable versus actual soil erosion rates in Europe

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Abstract

Erosion is a major threat to soil resources in Europe, and may impair their ability to deliver a range of ecosystem goods and services. This is reflected by the European Commission's Thematic Strategy for Soil Protection, which recommends an indicator-based approach for monitoring soil erosion. Defined baseline and threshold values are essential for the evaluation of soil monitoring data. Therefore, accurate spatial data on both soil loss and soil genesis are required, especially in the light of predicted changes in climate patterns, notably frequency, seasonal distribution and intensity of precipitation. Rates of soil loss are reported that have been measured, modelled or inferred for most types of soil erosion in a variety of landscapes, by studies across the spectrum of the Earth sciences. Natural rates of soil formation can be used as a basis for setting tolerable soil erosion rates, with soil formation consisting of mineral weathering as well as dust deposition. This paper reviews the concept of

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tolerable soil erosion and summarizes current knowledge on rates of soil formation, which are then compared to rates of soil erosion by known erosion types, for assessment of soil erosion monitoring at the European scale.

A modified definition of tolerable soil erosion is proposed as ‘any actual soil erosion rate at which a deterioration or loss of one or more soil functions does not occur’, actual soil erosion being ‘the total amount of soil lost by all recognised erosion types’. Even when including dust deposition in soil formation rates, the upper limit of tolerable soil erosion, as equal to soil formation, is ca. 1.4 t ha^{-1} yr^{-1} while the lower limit is ca. 0.3 t ha^{-1} yr^{-1}, for conditions prevalent in Europe. Scope for spatio-temporal differentiation of tolerable soil erosion rates below this upper limit is suggested by considering (components of) relevant soil functions. Reported rates of actual soil erosion vary much more than those for soil formation. Actual soil erosion rates for tilled, arable land in Europe are, on average, 3 to 40 times greater than the upper limit of tolerable soil erosion, accepting substantial spatio-temporal variation. This paper comprehensively reviews tolerable and actual soil erosion in Europe and highlights the scientific areas where more research is needed for successful implementation of an effective European soil monitoring system.

Key words: erosion tolerance; soil formation; climate change; soil protection; monitoring; dust deposition

1. Introduction

1.1 General

Soil loss occurs mostly through physical pathways but can also occur as a result of biochemical processes, including weathering of mineral particles in soil, which is known as chemical denudation. Removal of particles or even small aggregates from the in situ soil system then takes place in suspension or solution, as bed load or by gaseous export. Organic soil material is lost mainly through decomposition processes, except in the case of peat erosion where organic particles are removed and transported by water or wind. Physical pathways of soil loss predominate and fall within the domain of soil erosion, which is defined as “the wearing away of the land surface by physical forces such as rainfall, flowing water, wind, ice, temperature change, gravity or other natural or anthropogenic agents that abrade, detach and remove soil or geological material from one point on the earth's surface to be deposited elsewhere” (Soil Science Society of America, 2001; Jones et al., 2006, p.24-5). With respect to soil degradation, most concerns about erosion are related to ‘accelerated soil erosion’, where the natural (or ‘normal’, or ‘geological’) rate has been increased significantly by human activity.

The cause and extent of accelerated soil erosion are influenced by a number of factors (Morgan, 2005) and the most significant are:

- soil erodibility or susceptibility to erosive forces, as determined by soil physical, chemical and biological properties (Chepil, 1950: Bryan, 1968; Wischmeier and Mannering, 1969; Aspiras et al., 1971; Wischmeier et al., 1971; Tisdall and Oades, 1982; Rauws and Govers, 1988; Forster, 1989; Chenu, 1993; Oades, 1993; Marinissen, 1994; Edgerton et al., 1995; Le Bissonnais, 1996; Degens, 1997; Ketterings et al., 1997 ; Kiem and Kandeler, 1997; Hallett and Young, 1999; Czarnes et al., 2000; Doerr et al., 2000;

Scullion and Malik, 2000; Boix-Fayos et al., 2001; Ritz and Young, 2004; Allton, 2006; Shakesby and Doerr, 2006)

- erosivity or energy of the eroding agent, e.g. rainfall, overland flow or wind (Wischmeier and Smith, 1958; Skidmore and Woodruff, 1968; Fournier, 1972; Zachar, 1982; Morgan et al., 1986; Knighton, 1998)
- slope characteristics, gradient, length and form (Zingg, 1940; Musgrave, 1947; Kirkby, 1969; Horváth and Erödi, 1962; Chepil et al., 1964; Meyer et al., 1975; D'Souza and Morgan, 1976; Wischmeier and Smith, 1978)
- land cover use and management (Wischmeier and Smith, 1978; Wiersum, 1979; De Ploey, 1981; Dissmeyer and Foster, 1981; Laflen and Colvin, 1981; Foster, 1982; Temple, 1982; Lang and McCaffrey, 1984; Armstrong and Mitchell, 1987; Quinton et al., 1997; Lal, 2001; Gyssels et al., 2005; Zhang et al., 2007)

This paper reviews the dominant causes and rates of soil loss that occur in Europe via the process of detachment (e.g. water, wind, tillage, crop harvesting and land levelling), and subsequent transport and deposition of the detached soil material. Whilst all pathways of soil loss need to be considered and monitored carefully, once detachment of soil particles occurs, the functionality of the remaining soil is impaired to a greater or lesser extent depending on the amount of soil lost. Thus prevention of the detachment phase of the erosion process (Meyer and Wischmeier, 1969) is crucial if the functionality of the soil system is to be safeguarded for future generations.

This review focuses on erosion of mineral soils in Europe, because this is the dominant type of soil loss on the continent (Boardman and Poesen, 2006). Mineral soils are here defined as those that consist predominantly of, and have properties mainly determined by, mineral matter, and usually contain less than 20% organic carbon (SSSA, 2001). Relatively recent research (Holden and Burt, 2002; McHugh et al., 2002; Holden, 2005) has shown that erosion processes also account for substantial losses from organic soils, for example by piping and gullying in peatlands. However, organic soils are far less extensive than mineral soils in Europe (Montanarella et al., 2006) and constitute a different eco-system; thus consideration of their erosion is not included in this paper.

1.2 Scale

Soil erosion research has considered various spatial and temporal scales at which the different erosion processes operate. The experience and knowledge gained from these studies is generated by, and serves, a very wide audience, ranging from developers of sub-process, physically based erosion models, such as EUROSEM (Morgan et al., 1998) and WEPP (Nearing et al., 1989), through to regional planners and policy makers. Ciesiolka and Rose (1998) observe that smaller scale studies tend to focus on ‘on-site’ impacts of soil erosion, whilst larger spatial-scale studies concentrate on the ‘off-site’ impacts.

Table 1

The temporal scale variation in erosion processes is implicit in Table 1, with small spatial scale processes such as raindrop impact occurring in fractions of seconds, and catchment scale processes usually being monitored over much longer time scales (i.e. seasons, years, decades or even geological timescales). Sediment delivery ratios are also time-dependent, ranging from effectively no sediment delivered at the exact moment of detachment to sediment delivery ratios at the catchment scale approaching 100% over geological timescales (van Rompaey et al., 2005).

The comparison of, and connectivity between different spatial and temporal scales is a major challenge in erosion research currently. This complex spatio-temporal process and the lag times involved, make it intrinsically difficult to compare directly a series of plot scale measurements with data generated for the whole catchment. The results of soil loss and sediment delivery obtained at one spatial scale cannot and should not be extrapolated to another (Walling, 1990; de Vente and Poesen, 2005).

Simple ‘scaling up or down’ of erosion rates is not possible (Pierson et al., 1994). According to van Noordwijk et al. (1998), there are no ‘scaling rules’ in erosion research. It appears that the mean value of erosion per unit area will change at different spatial scales, all other factors being equal. At small spatial scales (e.g. individual aggregate), better control of variables, ease of replication and understanding of erosion mechanisms can be gained, but such fragmenting or deconstructing of processes may exclude many of the factors affecting the true rates of erosion (e.g. slope topography) as observed at a larger spatial scale in the field. On small plots, the process of rainsplash detachment (especially) and transport will dominate erosion rates, due to the limited slope lengths over which erosive overland flow can generate. It follows that certain erosion processes such as gully erosion or mass movements cannot be simulated at small spatial scales, but they may dominate at larger scales. As spatial scale increases, overland flow becomes the dominant agent of erosion, but different experimental conditions have shown rates of erosion per unit area to both increase and decrease with increasing slope length (Zingg, 1940; Meyer et al., 1975; Abrahams et al., 1991; Smith and Quinton, 2000). Morgan (2005) states “with such a great range of possible conditions, a single relationship between soil loss and slope length cannot exist”. Also, plot boundary / edge effects on erosion processes and rates are proportionately more significant at smaller spatial scales.

To improve understanding of the effect of spatial scale on erosion processes, the links or connectivity between different scales can be studied by applying experimental methods which encompass a range of spatial scales simultaneously. There has been some work on converting field-scale to catchment-scale erosion data, based on the concept of sediment delivery ratios (Osterkamp and Toy, 1997; Walling, 1983, 1990). Hudson (1993) reports on the ‘nested catchments’ approach in soil erosion research, which was developed from biological research methods, investigating biodiversity and species richness at different scales. Turkelboom and Trebuil (1998) developed a methodology for erosion process analysis at the field, farm and catchment scales, and ways of linking these different scales. Their multiscale approach involves the physical, economic and social aspects affecting erosion. Kirkby (2001) describes the hierarchical MEDRUSH model, which simulates erosion and runoff processes operating at a scale of 1 m^2 in the first instance. These results are then ‘nested’ or ‘embedded’ within representative ‘flow strips’ of up to 100 m wide, oriented up/down the slope. Water and sediment generated at this scale are then ‘routed’ via computed linear transfer functions into the sub-catchment scale (1–10 km^2). Output from this scale then feeds the main catchment-scale channel network, which may be up to

2500 km^2 in area. Kirkby (2001) argues that MEDRUSH demonstrates that ‘coarse and fine scaled models can be linked together consistently with a sound physical basis’.

Until we understand the connections between the different spatial scales, soil erosion research should encompass as wide a range of scales as possible. This has the multiple benefits of linking soil erosion rates generated at varying spatial scales, supplying knowledge which will be of interest to many parties (from physically based erosion modellers through to policy makers) and identifying if there are any rules to be applied when upscaling or downscaling the results of soil erosion research.

This discussion on the effect of scale on erosion is intended for completeness, but the focus of this paper is on the plot-to-field scale, because this is the position in the landscape at which removal of the in situ soil takes place. As a result, it is here that soil functioning will be most adversely affected by soil erosion.

1.3 Consequences, mitigation, costs and monitoring

Soil erosion rates are known to increase significantly following anthropogenic activities such as stripping of natural vegetation, especially clearing of forests for cultivation; other changes in land cover through cultivation or urbanisation and infrastructural development; over-grazing; wildfires or controlled burning; re-sculpturing of the land surface for example terrace construction; inappropriate intensification of land use and management, for example cultivation of steep slopes beyond their inherent ‘capability’ (Klingebiel and Montgomery, 1961) or collapse of terrace structures through poor maintenance (Temple and Rapp, 1972). The consequences of soil erosion for society can be severe, for example annual costs have been estimated to be £205 million in England and Wales alone (see Table 2) and $44 billion in the U.S.A. (Pimentel et al., 1995).

Table 2

As Table 2 demonstrates, the costs associated with soil erosion are often categorised into ‘on-site’, i.e. where the soil loss takes place, and ‘off-site’ impacts, the temporary or permanent destination of the eroded sediment. Over time, attitudes have changed with regard to the most damaging effects of soil erosion. Where crop productivity has been a significant driver of soil erosion, the on-site impacts of erosion are paramount through the. loss of rooting medium, nutrients, seeds, seedlings, agro-chemicals, organic matter, microbial communities, trace elements and water holding capacity. The production function of soil is likely to become even more important, in view of the projected increase in global human population and consequent demands for food. More than 99% of food supplies (calories) for human consumption come from the land, whereas less than 1% comes from oceans and other aquatic ecosystems (FAO, 2003).

However, where food security is not an issue, or any declines in crop yield can be masked by applications of agro-chemicals, the focus has often been on off-site impacts. These include flooding, often due to deposition of eroded sediments restricting the capacity of water channels to carry peak flows, and reductions in water quality, due to turbidity and preferential transport of contaminants on eroded sediment surfaces, which, in turn, have impacts on aquatic biota (Lloyd, 1987; Lloyd et al., 1987; Newcombe and Macdonald, 1991; Cooper, 1993). The value of soil in situ (i.e. not eroded) is once again acknowledged (Vandekerckhove et al., 2004), as the concept of soil resources being able to deliver ecosystem goods and services gains acceptance as advocated in the EU draft Soil Framework Directive (European Commission, 2006a,b).

To evaluate the impact of agricultural and other land use policies in Europe, Gobin et al. (2002, 2004) proposed selecting a set of soil erosion indicators that can be calculated objectively, validated against measurements or observations and evaluated by experts. This advice has been heeded in the design of a European soil monitoring system by the ENVASSO project - Environmental Assessment of Soil for Monitoring – funded under the European Commission's 6th Framework Programme (Morvan et al., 2008). Indicators for soil erosion proposed for implementation at the first tier (Eckelmann et al., 2006), are: i) estimated soil loss by water via rill, inter-rill and sheet erosion, ii) estimated soil loss by wind erosion, and iii) estimated soil loss by tillage erosion. Each of these indicators can be modelled and is accompanied by a measured indicator of soil loss for calibration and validation of modelled estimates. At the present time, there is no reliable model for estimating or predicting gully erosion in the same way as models for rill and inter-rill erosion (Poesen et al. 2006, p528-30). However, it is likely that advances in remote sensing and data processing technology will allow more reliable and accurate estimation of soil loss as a result of gully erosion in future (Jones et al., 2004).

The clear impact of erosion on society and individuals, combined with the political drive for developing a harmonised European system for monitoring erosion as a threat to soil, has identified the need for scientifically sound and robust threshold values against which to appraise the monitoring data. This paper sets out to review tolerable soil erosion, as a concept and in rates, for European conditions, and assesses actual soil erosion rates by discussing all (known) types of erosion.

2 Tolerable soil erosion rates

2.1 Concept

Since soil loss includes the removal of soil material by both physical processes (erosion), and biochemical processes (solute/gaseous export of mineral matter and decomposition of organic matter), the term ‘tolerable soil erosion’ is preferable when referring to soil lost by erosion in the context of soil protection. A number of (near) synonymous terms are used in the literature: ‘soil loss tolerance’, ‘permissible soil loss’, ‘acceptable rates of erosion’, ‘allowable soil loss’, etc. (see Table 3). It is important to note the difference between concept and unit. ‘Tolerable soil erosion’ is a conceptual term, with judgements of affected soil functions etc., that can be quantified in ‘tolerable rates of soil erosion’ with units conventionally in t ha^{-1} yr^{-1}.

Table 3

Reviewing the different definitions for tolerable soil erosion in the literature (Table 3), two themes emerge. The first interpretation is to view tolerable soil erosion as maintaining the dynamic equilibrium of soil quantity (mass/volume) in any location under any circumstances. The second interpretation takes a functional approach by relating soil erosion tolerance to the biomass production function of soil. Roose (1996) highlighted difficulties with both interpretations. The first interpretation ignores soil quality by focusing only on soil quantity. The second approach ignores many soil functions by focusing only on the biomass (particularly crop) production function of soil (see also Table 4). In addition, it creates temporal ambiguity:. 'a long time', 'indefinitely', 'an extended period of time', and '20-25 years'. Interestingly, the Soil Quality Vocabulary of the SSSA (2001) lists both interpretations, without indicating the conditions under which these should apply.

Both interpretations incorporate value judgements of how much soil erosion human societies should tolerate. The first interpretation judges that it is tolerable to ensure that the rate of soil formation exceeds the rate of soil loss by erosion, but that it is not tolerable for the soil erosion rate to exceed the soil formation rate. The value judgement in the functional approach links the soil erosion tolerated to the performance of one particular soil function, for example the crop production function.

At the end of the Second World War much of Europe was in ruins and crop production systems were destroyed or at best seriously malfunctioning in many areas. International aid, through the Marshall Plan in the 'western' world, focused on food supplies, which were scarce and insecure. It was during this period that the concept of tolerable soil erosion was developed most actively, which may explain the focus on the crop production function of soil. The agricultural surpluses of the 1980s lead in the 1990s to a more comprehensive/holistic concept of soil functions (e.g. Blum, 1993; Sombroek and Sims, 1995; Brady and Weil, 2002; De Groot, 2002; Blum, 2005; Nikitin, 2005; and the European Commission, 2006a,b). These are generally based on five primary soil functions (see Table 4).

Table 4

The need to include the regulation function in establishing tolerable rates of soil erosion was realised by Mannering (1981) and Skidmore (1982), who included it in a function of ‘soil loss tolerance’ (modified from Stamey and Smith, 1964), although only as secondary to the production function. Roose (1996) stated that tolerable soil erosion should consider “respect for the environment in terms of water quality, especially runoff sediments”. Despite these appeals, definitions for tolerable soil erosion that were published later only incorporated the crop production function (see Table 3).

The remaining three soil functions (i.e. information, engineering and habitat) do not appear to have been considered in ‘tolerable soil erosion’ definitions in the literature. This can probably be explained by the relatively recent development of the holistic soil function concept, compared to the development of the tolerable soil erosion concept. Sparovek and De Maria (2003) point out that tolerable soil erosion is the most multidisciplinary field of soil erosion research and that only contemplation of this multi-perspective nature may be successful. It appears, therefore, that the time has come to integrate both concepts. Tolerable soil erosion may then be defined as ‘any mean annual cumulative (all erosion types combined) soil erosion rate at which a deterioration or loss of one or more soil functions (Table 4) does not occur’.

Clearly, this definition still leaves the problem of value judgement and scale: at what stage is a soil function considered to have deteriorated, and at what scale is this assessed? Also, it is a rather negative approach, where action is only required when a tolerable rate of soil erosion in a specific location is reached. This approach also assumes that no technological advances may occur over time, such as the invention of ‘super-fertilisers’, which could (albeit unsustainably) mask declines in crop yield due to loss of soil though erosion processes. It may be a more effective policy to provide incentives to land owners and managers to ensure that actual soil erosion rates remain much closer to, or preferably equal to or below, the soil formation rate. This would be an exemplary application of the precautionary principle (i.e. to preferably err on the side of caution), and ensure that soil functions were maintained for the benefit of current and future generations.

Rates of soil formation provide an invaluable benchmark to use as a ‘basis’ for determining tolerable rates of soil erosion, that is soil functions can generally be judged not to deteriorate as long as soil erosion does not exceed ‘natural’ or ‘geological’ (or ‘normal’) erosion rates. At present, this assumption remains largely untested, but applying the precautionary principle appears to be a reasonable starting point. A second assumption is that ‘natural’ soil erosion rates equate to soil formation rates. This implies a meta-stabile situation where all soils are in dynamic equilibrium in terms of quantity (mass/volume). Clearly, young soils or any soil that could accumulate under current conditions, and thereby improve the soil regulation, production, and habitat functions, would not be in dynamic equilibrium. Nevertheless, soil formation rates form the best basis upon which to establish tolerable rates of soil erosion.

2.2 Current evidence for soil formation rates

The natural process of soil accumulation at any location has been described as soil production, soil formation, soil genesis, pedogenesis, or soil renewal (Brady and Weil, 2002). The term ‘soil formation’ is used here for reasons of general acceptance, noting that this includes both dust deposition and parent material weathering.

Ideally, soil formation models (e.g. Hoosbeek and Bryan, 1992; Minasny and McBratney, 2001) would have been developed and validated to such an extent that for any soil type, under any land use, soil management practice, in any region, accurate estimates of soil formation rates could be derived. Better still would be a degree of model development that could also estimate soil formation rates for future climate change scenarios. It is generally acknowledged that ‘natural’ erosion rates have varied significantly throughout geological history as the climate changed (Wilkinson and McElroy, 2007). However, fundamental scientific knowledge on soil formation processes is still insufficient at present to support the use of mechanistic soil formation models for establishing tolerable rates of soil erosion in the context of environmental protection. Therefore, the most useful contribution that science can make to the policy process would be to arrive at a consensus on mean rates of soil formation and soil erosion.

2.2.1 Soil formation rates by weathering

Very few direct measurements of soil formation rates are available. This is due in part to the extremely slow rate of soil formation in relation to the human life span, and consequent difficulties in accurate field measurement. However, from studies using different methodologies over different scales, an overall picture of the range of soil formation rates can be built up (Table 5), although differentiation of these rates by dominant factors remains elusive. Mass balance measurement studies have been performed to investigate soil formation rates. Alexander (1988a) determined soil formation rates for 18 small, non-agricultural, non-carbonate substrate watersheds (located in North America, Europe, Australia (Victoria) and Zimbabwe) with shallow to moderately deep soils, by measuring values of silica inputs and outputs and relating these to soil formation. The range for non-peaty soils was from 0.02 to 1.27 (mean=0.49) t ha^{-1} yr^{-1}. If, and to what extent, these soil formation rates would increase under agricultural land use is not known. Wakatsuki and Rasyidin (1992) used similar geochemical mass balance methodologies on seven elements (Al, Fe, Ca, K, Mg, Na and Si) to calculate soil formation at a global scale as ranging from 0.37 to 1.29 (mean=0.7) t ha^{-1} yr^{-1}. Much greater rates were calculated for well draining, high precipitation watersheds in southwestern Japan, but environmental conditions there are not typical for the rest of the world. Soil formation rates by weathering in limestone-dominated catchments, or those with a mainly igneous lithology, have been estimated at < 0.1 t ha^{-1} yr^{-1} (Alexander, 1985). Soil chronosequence studies can be used as an alternative method for deriving soil formation rates, although most appear to focus on processes that are responsible for specific soil parameters rather than overall soil formation rates. See Huggett (1998) and Yoo and Mudd (2008) for discussions of methodological issues of classic soil chronosequence work.

Table 5

Landscape scale ‘soil formation functions’ (i.e. the relationship between soil formation and soil depth) have been derived from studies in the disciplines of geology and geomorphology. Humphreys and Wilkinson (2007) describe a useful overview of this theme and recommend that the basic idea of soil formation may be used for the determination of tolerable soil erosion rates. Heimsath et al. (1997) used measurements of in situ produced cosmogenic ^{10}Be and ^{26}Al concentrations with measured soil depths to show an inverse relationship between soil formation rates and soil depth in northern California. Soil formation rates ranged from ca. 0.39 t ha^{-1} yr^{-1} for deeper soils (ca. 50 cm) to ca. 0.91 t ha^{-1} yr^{-1} for shallower soil (ca. 5 cm), assuming a bulk density of 1.3 t m^{-3}. Shakesby and Doerr (2006) reviewed evidence in the literature of fire weathering, that is where wildfire ‘weathers’ rocks by spalling (detachment of lensoid-shaped rock flakes) and other fracturing effects, and showed that where fires are relatively frequent this may be an important additional weathering process, although erosion rates are likely to increase concomitantly.

Natural soil erosion rates, assumed to be equivalent to soil formation rates (see section 1) when studied over geological time scales, have been estimated by studying continental erosion and sedimentation. Wilkinson and McElroy (2007) gave an exhaustive analysis of rates of subaerial denudation in the Phanerozoic, a period of 542 million years spanning the Lower Cambrian to the Tertiary Pliocene. They estimate that erosion averaged 5 Gt yr^{-1} during this period.. The global land area fluctuated throughout the Phanerozoic, but using a continental area of 118 million km^{2}, 5 Gt yr^{-1} equates to an average natural erosion rate of 0.4 t ha^{-1}yr^{-1} (over 542 million years. Schaller et al. (2001) measured in situ produced radionuclides (^{10}Be) in the bedload of middle European rivers to infer average soil erosion rates, over the last

10,000-40,000 yr, at 0.26-1.3 t $ha^{-1}yr^{-1}$ (assuming a bulk density of 1.3 t m^{-3}). Mabit et al. (2008) discusses the advantages and limitations of fallout radionuclides for assessing soil erosion. Bennett (1939) reported that soil formation rates in the USA range from 0.3-1.1 t $ha^{-1}yr^{-1}$ (assuming a bulk density of 1.3 t m^{-3}), although he did not specify the methodology used. However, in areas where aeolian deposition occurs, the picture of soil formation is more complex.

2.2.2 Soil formation rates by dust deposition

Simonson (1995) reviewed the significance of air-borne dust to soils and discussed that when dust is deposited onto a soil from a desert source area, it may be regarded as ‘more valuable’ for soil functions in its new location, in a similar way that Sahelian dust boosts biomass production in Amazonian forests (e.g. Swap et al., 1992). Although this is a contentious view, wind erosion of fine particles in the Sahel may contribute to not allowing local vegetation cover development. In the present paper Simonson’s suggestion is accepted as long as the amount deposited is of an order of magnitude that enables the soil to incorporate it (i.e. not being buried by it).

Research into dust transport and deposition has increased substantially over the last decade (Engelstaedter et al., 2006). Satellite imagery and isotopic composition analyses have revealed that the Sahara is the main source of dust deposited in Europe (Middleton and Goudie, 2001), although dust originating from China has also been recorded in the French Alps (Grousset et al., 2003). Remote sensing analysis, employing the Total Ozone Mapping Spectrometer absorbing Aerosol Index (TOMS AI), has identified dust pathways from North Africa to the Mediterranean Basin (Middleton and Goudie, 2001; Israelevich et al., 2002).

North Africa is considered to be the largest source of dust on Earth with estimates of the strength of the Saharan source to be 130 to 760 million t yr^{-1}, compared to 1000 to 3000 million t yr^{-1} globally (Engelstaedter et al., 2006). The greater part of Saharan and peri-Saharan or Sahelian dust is delivered to the North Atlantic, but substantial amounts are estimated to be deposited on the European continent. D'Almeida (1986) used sun-photometer readings taken in the early 1980s to estimate Saharan dust delivery to Europe at 80-120 million t yr^{-1}. Löye-Pilot et al. (1986) extrapolated their field data from Corsica to estimate dust delivery to the western Mediterranean at 3.9 million t yr^{-1}.

Field measurements of dust deposition are summarised in Table 6. As Middleton and Goudie (2001) and Engelstaedter et al. (2006) observed, both the frequency of dust deposition and the mean annual quantity of deposited dust are greater for southern than for northern Europe. For Mediterranean Europe, up to the Pyrenean, Alpine, and Carpathian mountain ranges, dust deposition rates range from 0.05 to 0.39 t ha^{-1} yr^{-1}. North of this mountain divide, dust deposition rates are below 0.01 t ha^{-1} yr^{-1}. For the purpose of setting soil formation rates as thresholds for soil erosion (i.e. tolerable rates), it seems a reasonable generalisation to set dust deposition rates at ca. 0.2 t ha^{-1} yr^{-1} south of the trans-European mountain divide, and to regard dust deposition rates as negligible relative to soil erosion rates north of the divide, accepting potentially substantial but presently unquantifiable local variation to this.

Table 6

The value of 0.2 t ha^{-1} yr^{-1} for southern Europe is of the same order of dust deposition rates found in California, where Reheis and Kihl (1995) measured dust deposition rates to range from 0.04-0.16 t ha^{-1} yr^{-1} in southern Nevada and south-eastern California, and determined an average value of 0.30 t ha^{-1} yr^{-1} in south-western California. Simonson (1995) reviewed the significance of dust deposition to soils and quoted estimates of approximately 3.0 t ha^{-1} yr^{-1} of dust deposition on average for soils between the Rocky Mountains and the Mississippi River. This is a much greater value than those reported for Europe or California, and may be explained by the source area in the semi-arid south west U.S.A. delivering most of its dust eastward.

2.2.3 Overall soil formation rates

For the purpose of deriving overall soil formation rates in the evaluation and monitoring of soil erosion and its impacts, it appears to be reasonable to estimate dust deposition at no more than 0.2 t ha^{-1} yr^{-1} in southern Europe and at 0.0 t ha^{-1} yr^{-1} in northern Europe. By contrast, estimated soil formation rates (by weathering) for current conditions in Europe range on average from ca. 0.3 t ha^{-1} yr^{-1} to ca. 1.2 t ha^{-1} yr^{-1}. Much lower rates (e.g. 0.004 t ha^{-1} yr^{-1} for basaltic parent material in semi-arid Australia – Pillans, 1997) and greater rates (e.g. 5.7 t ha^{-1} yr^{-1} for a very well draining high precipitation watershed in southwestern Japan – Wakatsuki and Rasyidin, 1992) have been reported for environmental conditions generally not found in Europe. Therefore, considering soil formation rates by both weathering and dust deposition, it is estimated that for the majority of soil forming factors in most European situations, soil formation rates probably range from ca. 0.3 – 1.4 t ha^{-1} yr^{-1}. Although the current agreement on these values seems relatively strong, how the variation within the range is spatially distributed across Europe and how this may be affected by climate, land use and land management change in the future remains largely unexplored. It may be expected that dust deposition rates in the Mediterranean will increase in a climate change scenario that brings increasing droughts to the Sahel region, but if this will also mean that more dust will be deposited further northwards in Europe is more uncertain, as is the regional/local scale variation in dust deposition rates. Chemical weathering can be expected to increase where precipitation increases, particularly where the parent material is well draining, although soil erosion rates may concomitantly increase at the same or a greater rate (particularly when the rainfall intensity increases). Soils formed in limestone or granitic lithology are reported to have formation rates towards the smaller part of the range, although the body of evidence is relatively small and more experimental research is urgently needed into soil formation rates for these lithologies, since they cover a substantial area in Europe. Soil formation by sedimentation in water is only significant in the floodplains of large river systems, and is, therefore, omitted from this paper.

2.2.4 Tolerable rates of soil erosion in Europe

Although reported rates of soil formation suggest an upper limit of approximately 1.4 t ha^{-1} yr^{-1} for mineral soils (see also Alexander, 1988b), it would be advisable to apply the ‘precautionary principle’ to any policy response to counteract soil erosion, otherwise soils with particularly slow rates of formation will steadily disappear, even when subjected to low erosion rates. Therefore, future differentiation of soil formation rates for soil–landuse–climate combinations is needed, and quantitative pedogenesis modelling (e.g. Hoosbeek and Bryan, 1992; Minasny and McBratney, 2001) may provide an appropriate methodology.

In some cases, rates of soil erosion greater than those of soil formation have been regarded as tolerable only from the wider perspective of society as a whole, for example because of a perception that certain crops (such as some vines) favour eroded soil profiles. In Switzerland, the threshold tolerated for soil erosion is generally 1 t ha^{-1} yr^{-1}, though this threshold is increased to 2 t ha^{-1} yr^{-1} for some soil types (Schaub and Prasuhn, 1998). In Norway, 2 t ha^{-1} yr^{-1} is adopted as the threshold for tolerable soil loss (A. Arnoldussen, personal communication.). However, the data reviewed here confirm that a precautionary approach to environmental protection should regard soil erosion losses of more than 1 t ha^{-1} yr^{-1} in Europe as unsustainable in the long term (Jones et al., 2004). In the USA, soils have been assigned tolerable rates (so-called ‘T values’) by using a range of methodologies, mainly the USLE model and expert judgement, and differentiated mainly by soil depth and crop productivity. Approaches and assumptions for deriving T values have been revised (e.g. Mannering, 1981; Pierce et al., 1984) and continue to be discussed (Johnson, 1987; Mirtskhulava, 2001; Johnson, 2005; Montgomery, 2007). Another way of expressing tolerable soil erosion is to calculate the ‘life span’ of soil. This is the number of years it will take, at current soil formation/erosion rates, for a soil to reach its finite point (i.e. the minimum soil depth required before it becomes economically unsustainable to maintain the current land use - Stocking and Pain, 1983). For commercial farming the finite point has been defined at which yields fall to 75% below the maximum possible (Morgan, 1987). However, this value is highly dependent on socio-economic conditions and available technology and these factors are notoriously difficult to predict accurately in the future. For other soil functions this approach has not been applied, possibly in part because of some (components of) soil functions do not allow for straightforward economic sustainability assessments (e.g. soil biodiversity).

Setting a limit of 1 t ha^{-1} yr^{-1} is also supported when considering the impact of soil erosion / sediment production rates on water quality. Eroded soil, delivered to water bodies can be a physical and chemical pollutant in terms of water turbidity and as a carrier of contaminants which may have detrimental effects on aquatic ecosystems. Qualitative limits for eroded sediment in water bodies are advocated in policy drivers such as the EU Water Framework Directive, which states that surface waters should be kept in ‘good ecological status’. EU Member States are currently deciding on the level of sediment, which will give such a status, but it is unlikely that absolute standards for biological quality will be set across the whole community, because of ecological variability. It is expected that the specified controls will allow “only a slight departure from the biological community which would be expected in conditions of minimal anthropogenic impact”. Quantitative targets have also been set to control pollution from sediment (e.g. the United States Department of Agriculture uses a target of 1 t ha^{-1} yr^{-1} to maintain water quality).

3. Actual soil erosion rates

Section 3.1 introduces the main types of soil erosion while section 3.2 reviews the erosion rates reported in the literature.

3.1 Soil erosion types

Soil loss by coastal and riparian erosion is not reviewed in this study, because this constitutes the loss of land, which is not directly linked to human activities although it constitutes a ‘permanent’ loss of soil. Furthermore, it is not clear that human influence through land management and land use practices has any significant effect on increasing or decreasing coastal erosion, although a number of studies have shown that attempts to mitigate by erecting engineering structures (e.g. impervious sea walls and breakwaters) can actually aggravate the problem elsewhere along the coastline (McInnes et al., 2000; Lee and Clark, 2004; Lee and Jones, 2004; Bromhead and Ibsen, 2006).

3.1.1 Soil loss by water erosion

Water erosion takes place through rill and/or inter-rill (sheet) erosion, and gullies, as a result of excess surface runoff, notably when flow shear stresses exceed the shear strength of the soil (Kirkby et al., 2000; Jones et al., 2004; Kirkby et al., 2004). This form of erosion is generally estimated to be the most extensive form of erosion occurring in Europe. De Ploey (1989) identified different domains where these processes take place, as a function of soil, slope and land cover characteristics in any location. Sheet and rill erosion will cause surface soil to be removed from the in situ soil mass. Assuming this surface soil has not been disturbed previously (e.g. by inversion tillage or preceding erosion events), it will contain considerable amounts of organic matter and plant nutrients that are crucial to perform effective soil functions (Fullen and Brandsma, 1995). This eroded soil material may not necessarily travel very far and may remain in the same field from where it was eroded. Indeed, the area of deposition may benefit from the accumulation of highly fertile, eroded surface soil, in the same way that river flood plains receive substantial depositions of highly fertile sediment. However, this accumulation of eroded soil may only be temporary, until the next erosion event, especially as the recently deposited sediments often lack aggregation and remain highly erodible.

Where there is little vegetative cover or root network below the surface, and slopes are steep, the eroded soil from these surface processes can move into the stream network and thus cause further detrimental off-site impacts (Cerdan et al., 2006). The transport of eroded material will be enhanced further by erosion features such as gullies which provide a conduit for the eroded surface soil (Blong et al., 1982), as well as being a source of sediments in their own right. Long term field plots are often used for direct measurement of soil loss by rill and inter-rill erosion; as demonstrated by Boix-Fayos (2005). Models of rill erosion have been shown by some researchers to be in disagreement with current experimental evidence (Govers et al., 2007; De Vente et al., 2008), but direct measurements of soil erosion are both scarce and do not fully represent the soil-climatic landscapes that experience rill erosion in Europe.

Gully erosion is common in Mediterranean Europe, in particular, Spain, Italy and Greece (Vandekerckhove et al., 2000). These areas are characterised by long-term gullies (i.e. that cannot be obliterated by ploughing), which have been described as relatively deep, recently formed, eroding channels that form on valley sides and on valley floors where no well-defined channel previously existed (Schumm et al., 1984). Ephemeral gullies (i.e. that can be obliterated by ploughing) commonly occur in the arable loess soil, as seen in the loess belt of Belgium and the sandy soils of the South and West Midlands of England. These gullies develop rapidly, are ploughed in and often reappear the following year. The occurrence of gullies, and variations in the type of gully erosion, are related to particular soil properties, climate and topography of these areas (Nachtergaele and Poesen, 1999; Nachtergaele et al., 2001). It is notoriously difficult to predict where and when gully erosion will occur in the landscape by the extension of an existing gully or a new gully forming, as well as associated rates of sediment production (Poesen et al., 2003).

3.1.2 Soil loss by wind erosion

Wind erosion occurs predominantly on the North European Plain (northern Germany, eastern Netherlands and eastern England) and in parts of Mediterranean Europe (De Ploey, 1989; Evans, 1990, 1996; Chappell, 1999; Chappell and Thomas, 2002; Warren, 2002; Barring et al., 2003; Breshears et al., 2003; Riksen et al., 2003; Jones et al., 2004; Quine et al., 2006). Wind erosion is caused by the simultaneous occurrence of three conditions: high wind velocity; susceptible surface of loose particles; and insufficient surface protection. The transport of soil material (between erosion and sedimentation) can occur in three main modes: saltation, creep and suspension. Factors that exacerbate wind erosion are similar to those for erosion by water: namely soil erodibility, as determined by physical, chemical and biological properties including texture, organic matter content, moisture content, land use and cover, and energy of the force causing the erosion (wind erosivity). Riksen et al. (2003) point out that wind erosion is not as significant or as widespread a problem in Europe as in drier parts of the world, which might explain the relatively limited research on wind erosion to date compared to water erosion studies. The present review concludes that there are few accurate data on the extent and magnitude of the problem, or the costs of the remediation (Owens et al., 2006a,b,c). Goossens et al. (2001) studied the dynamics of Aeolian dust emitted from agriculture in northwest Germany, over a 15 month period. The dust emission was caused by wind erosion combined with tillage activities and the dust emitted consisted of mineral as well as organic particles.

3.1.3 Soil loss by tillage erosion

This erosion type has been recognised for several decades, but the magnitude of soil lost by this process in Europe has only been appreciated and documented during the last 10-15 years (Lindstrom et al., 1992; Govers et al., 1993; Lobb et al., 1995; Govers et al., 1996; Lobb et al., 1999; Van Muysen et al., 1999; Lindstrom et al, 2000; Van Oost et al., 2000a,b; Quine and Zhang, 2004a,b; Van Oost et al., 2005a,b; Owens et al., 2006a,b; Quine et al., 2006; Van Muysen et al., 2006; Van Oost et al., 2006; Van Oost et al., in press). Mech and Free (1942) concluded that soil movement by tillage was far from insignificant and that its intensity was related to slope gradient. Soil translocation by tillage results in soil loss from convex slope positions, such as crests and shoulder slopes, because of an increase in-slope gradient and a consequent increase in soil translocation. Spatial patterns of tillage erosion differ from those of water erosion, because the principal agent is different. Soil loss by tillage can be greatest from landscape positions where water erosion is minimal (i.e. in concavities and near upslope field boundaries), whereas soil deposition by tillage can occur in areas where water erosion is often maximal (i.e. on slope convexities). Measurements on the magnitude of tillage erosion are few, but studies in Europe highlight the importance of the magnitude of tillage erosion relative to water erosion (Govers et al., 1993; Quine et al., 1994; Owens et al., 2006a). Van Oost et al. (2005a) have compared rates of soil erosion by tillage with those by water. By comparing two time periods, they found that there has been a shift from water-dominated to tillage-dominated erosion processes in agricultural areas during the past few decades. This reflects the increase in mechanized agriculture and the authors concluded that where soil is cultivated, tillage erosion may lead to larger losses than overland flow.

3.1.4 Soil loss by crop harvesting

This erosion type refers to soil removed during crop harvesting, for example of root crops, mainly in northern Europe. Soil can be removed from a location or field by adhering to farm machinery (e.g. wheels, tines, ploughs and discs). Much larger amounts of soil can be removed by soil co-extraction with a root crop, particularly . sugar beet, potatoes, carrots and chicory) (Jaggard et al., 1997; Ruysschaert et al., 2005). This mechanism of soil loss is known as ‘soil loss due to crop harvest (SLCH)’ in the scientific literature (Ruysschaert et al., 2004, 2005), and as ‘soil/dirt tare’ in the agricultural industry. SLCH is a particular problem in areas growing early potatoes in northern Europe because harvesting normally takes place when the topsoil is moist or very moist and soil particles readily adhere to the surface of the potatoes. However, preparation of the crop for marketing usually involves cleaning (washing) and removing the soil but returning it to the fields from whence it came is not always advised by the agricultural extension services, because of the possibility of spreading disease**.**

3.1.5 Soil loss by slope engineering

Slope engineering is the mechanical translocation of soil by bulldozers and other earth moving equipment to adapt slope surfaces to mechanised agriculture. Some authors refer to this practice as ‘land levelling’, which implies a reduction of slope gradient, which in turn would actually reduce erosion risk. However, as is seen in the construction of bench terraces for example, whilst the bench of the terrace is levelled, the ‘riser’ or back wall component of the terrace has to compensate for this, and is constructed at an angle which is steeper than the original land slope. This back slope is thus highly susceptible to surface erosion and mass movement. During terrace construction, soil loss can be aggravated as natural vegetation is mechanically removed from the land to enable soil to be cultivated, often in the form of modern specialised orchards, vineyards and olive groves. Often, marginal land with poor quality soils is used, so deep ploughing to about 1 m depth is required to ensure a sufficient depth of rootable soil (Jones et al., 2004). Such soil disturbance can destroy any soil structure, and increase soil erodibility and exacerbate soil losses. This form of erosion is common in many parts of Europe, especially in Italy, where it is widespread in the Apennines and hilly pre-alpine regions. Such techniques are also practised in southern Spain, where intensive horticulture under polythene canopies has spread onto the foothills of Andalusia. The climate there is arid to semi-arid. Thus, when heavy rain falls soil losses are exacerbated by steep slopes, lack of natural vegetation cover and the unstable disturbed soil (Kibblewhite et al., 2007).

3.2 Current evidence for actual soil erosion rates

There have been attempts to map soil erosion rates and risk in a number of EU Member States (De Ploey, 1989; Schaub and Prasuhn, 1998; Sanchez et al., 2001; Ministry of Environment of the Slovak Republic and Slovak Environmental Agency, 2002; Van der Knijff et al., 2002; Hennings, 2003; Øygarden, 2003; Kirkby et al., 2004; Dostal et al., 2004; Boardman and Poesen, 2006; Kertéz and Centeri, 2006), but to establish an accepted overall baseline for erosion in Europe remains a challenging task. Rates of soil erosion have been determined using several approaches: i) plot and field measurements, ii) soil erosion modelling, iii) mass/energy balance modelling, iv) radionuclide measurement, v) suspended sediment load in rivers and streams, vi) chronosequence studies, and vii) geological (sedimentological) studies. Trimble and Crosson (2000a,b) reviewed soil erosion rates in the U.S. and concluded that models should only be used with caution, taking account of all the assumptions and potential inaccuracies of the model chosen. These authors recommended that it would be better if resources were directed more towards measurements of soil erosion.

In this review, the focus is placed on measured soil erosion rates where available, and validated modelled rates for important but relatively unexplored soil erosion types. Publications on mean soil erosion rates refer mostly to water erosion, yet baseline values for other forms of erosion, for example by wind and tillage, are also needed.

3.2.1 Rates of soil loss by water (sheet, rill and gully) erosion

Pimentel et al. (1995) have reviewed erosion rates around the world and suggested an average of 17 t ha^{-1} yr^{-1} for arable soils in Europe. This is a crude approximation since it is based on plot data, which only exist for very small areas where measuring equipment has been installed and monitored. Furthermore, data from plot experiments are known to be a poor basis for regional generalisation (Boardman, 1998). This is because to obtain long-term estimates of soil erosion, plot estimates must be scaled up by integrating over time and surface runoff generated locally may not reach the base of a slope to deliver sediment to a channel (Kirkby et al., 2008). Thus, some soil removed from an experimental plot may be deposited downslope but not lost completely from the regional parcel or catchment. In addition, the location of soil erosion plots across Europe may not be representative, because erosion plots tend to be selected in places where erosion is known to occur and where resources are available to measure it. Yang et al. (2003) applied the RUSLE model on a 0.5° global grid using a 1 km resolution DEM to estimate rates of soil erosion by water, and found an average value of 11.1 t ha^{-1} yr^{-1} for Europe compared to 10.2 t ha^{-1} yr^{-1} globally. In addition Yang et al. (2003) evaluated the human induced proportion of the soil erosion by modelling the difference between current land cover and potential land cover without human activity. Human-induced erosion was estimated to be ca. 60% globally, but ca. 88% for Europe.

The occurrence and rate of water erosion processes are influenced by regional climate, local soil properties, and past and present land use. A number of localised erosion rates are given for various plots around Europe, some containing only one or two forms of erosion, depending on the spatial scale of the plots (Morgan, 2005). Cerdan et al. (2006) extensively reviewed the experimental data for soil loss by sheet and rill erosion in Europe, and compiled a database of 208 plots on 57 experimental sites in 13 countries. The mean erosion rate was 8.8 t ha^{-1} yr^{-1}, although aggregation of the data by land use showed large variations. Geographical comparisons, (i.e. Mediterranean versus the rest of Europe) showed no significant overall difference and no large differences between most land uses, except for bare soil (ca. 32 t ha^{-1} yr^{-1} for the Mediterranean zone and ca. 17 t ha^{-1} yr^{-1} for the rest of Europe).

Poesen et al. (2006) present a comprehensive list of published rates for gully erosion, including both ephemeral and permanent gullies. Ephemeral gully rates derived from studies conducted in the loess belt of Belgium while the majority of permanent gully erosion rate estimates are from the Mediterranean region of Europe. These rates vary from 1.1 to 455 t ha^{-1} yr^{-1} (Poesen et al., 2006). This wide range gives an indication of the complexities of quantifying soil loss by gully erosion owing to the episodic and highly variable nature of soil loss within these eroded channels; variable regional climatic effects; the haphazard nature of gully distribution in the landscape; propensity of vertically variable soil properties to exacerbate gully erosion; the stage at which the gully is in its erosion cycle (active or stable); current or previous topographic position in the landscape; and the historical and present land use influencing the gully (Valentin et al., 2005).

Martinez-Casasnovas et al. (2003) highlighted the complexities of measuring gully erosion rates in a study of one gully system located in north eastern Spain. Using aerial photographs and a detailed digital elevation model (DEM), they estimated the annual average sediment production rate of the gully from 1975 to 1995 to be 846 (± 40) t ha^{-1} yr^{-1}. The net erosion, taking account of some eroded material being deposited, was 576 (± 58) t ha^{-1} yr^{-1}, averaged over the 20-year period. During the study the authors measured and analysed a 1 in 100 year rainfall event when 205 mm fell over the study area in 2h 15 min leading to a net soil loss of 207 (± 21) t ha^{-1} with a sediment production rate of 487 (± 13) t ha^{-1} by ephemeral gully, rill and inter-rill erosion (Martinez-Casasnovas et al., 2003). The authors see this comparison as good evidence that gully erosion accounts for 1.7 times more soil loss than the other forms of erosion in this study area. However, averaging gully erosion on an annual basis probably gives an unrealistic rate, owing to the episodic nature of the gully forming process (Betts and De Rose, 1999)

Few studies have considered erosion from gullies at a regional or catchment scale. However, Nachtergaele and Poesen (1999) considered ephemeral gullies at four sites in Belgium (ranging from 216 to 1095 ha), using sequential aerial photographs from 1952 to 1996. Each site contained 18 to 38 gullies on average and it was estimated that the reasonably long-term (44 yr) average for soil loss was between 3.2 and 8.9 t ha^{-1} yr^{-1}. These figures are considerably different to those given by Martinez-Casasnovas et al. (2003), even though the measurement methods were similar (interpretation of sequential aerial photographs), and reveal the importance of differentiating between type of gully erosion and regional influences (Mediterranean versus western Europe) when assessing gully erosion rates.

Jones et al. (2004) report a number of other soil erosion studies which provide a European overview, but these are based mostly on models or expert judgement (including observation). These approaches more commonly produce assessments of erosion risk rather than estimates of actual soil loss, without reference to baseline and/or threshold values.

3.2.2 Rates of soil loss by wind erosion

Recent work in Eastern England reported mean wind erosion rates of 0.1-2.0 t ha^{-1} yr^{-1} (Chappell and Thomas, 2002), although severe events can move much larger quantities (>10 t ha^{-1} yr^{-1}) of soil. Böhner et al. (2003) estimated average soil loss at 1.6 t ha^{-1} yr^{-1}, and a mean maximum of 15.5 t ha^{-1} yr^{-1} from simulation modelling. Despite research studies in these areas, Chappell and Warren (2003) report that little is known about the true extent and magnitude of wind erosion in Europe.

3.2.3 Rates of soil loss by tillage erosion

Mean gross rates of tillage erosion have been reported to be in the order of 3 t ha^{-1} yr^{-1} for Belgium, the north of France, and the east of England (Govers et al., 1996; Owens et al., 2006a). Boardman and Poesen (2006) reviewed measurement data for tillage erosion rates in Europe and concluded that it often exceeds 10 t ha^{-1} yr^{-1}, particularly on fields with complex topography. Van Oost et al. (2005a) estimated that the average erosion and soil redistribution rate, over the last ca. 35-40 years due to tillage, is ca. 9 t ha^{-1} yr^{-1}. Long-term erosion rates based on soil profile truncation data demonstrated that, over the longer term, erosion has been dominantly by water by overland flow.

Hinz (2004) reported rates of soil loss between 18.6 and 29.5 kg ha^{-1} for harvesting operations, and between 0.8 and 1.4 kg ha^{-1} for normal tillage operations. The latter data are for the production of cereals but they may give a good idea of the order of magnitude for other adjacent crops. Funk and Reuter (2004) investigated emissions for various tillage operations and arrived at values of between 3 and 6 kg ha^{-1}, that is about 3 times greater than those of Hinz (2004).

At Dalicott Farm in Shropshire (UK), ^{137}Cs data and a numerical erosion model were used to estimate erosion on a hillslope (Govers et al., 1993; Quine et al., 1994). The proportions of overall erosion that was caused by water or tillage erosion were estimated to be similar for the last ca. 6 centuries (57% and 43%, respectively), and greater for water erosion over the last 40 years (76% and 24%, respectively), based on ^{137}Cs data.

3.2.4 Rates of soil loss by crop harvesting

Ruysschaert et al. (2004) provided an excellent review of the research on soil loss due to crop harvesting (SLCH) in Europe. They reported mean losses ranging from 1.3 to 19 t ha^{-1}yr^{-1} for a variety of crops. SLCH was greatest for chicory, sugar beet and potatoes. Boardman and Poesen (2006) also reviewed soil loss by crop harvesting, confirming the variation in Europe, according to crop types and climate, concluding that average values of 2 t ha^{-1} yr^{-1} for a potato crop and 9 t ha^{-1} yr^{-1} for a sugar beet crop can be expected. Soil moisture content at harvest is the driving factor.

3.2.5 Rates of soil loss by slope engineering

Recently, P. Bazzoffi (pers.com.) estimated that in Italy the area highly prone to risk of land levelling is about 10% of the area under permanent crops. After levelling, land is in a vulnerable condition and a few storms can easily cause severe soil losses. Bazzoffi et al. (1989) measured 454 t ha^{-1} yr^{-1} of water erosion with the formation of a gully after six rainfall events of medium intensity in central Italy.

In Norway during the late 1970s, extensive land levelling was stimulated by subsidies. This led to a two- to three-fold increase in soil erosion. The increase was especially large when former ravine landscapes used for pasture were levelled and turned into arable land that was ploughed in autumn. The clearly visible erosion and increasing negative offsite effects on water quality, together with overproduction, put an end to the subsidies for land levelling, but not before 13% of the agricultural area had been levelled with the support of these subsidies. The most visible effect was erosion caused by concentrated flow, including severe ‘gullying’ resulting from reduced infiltration, longer slopes and inadequate measures to handle concentrated flow (Jones et al., 2004). Now, land levelling is only allowed in Norway with special permission.

3.2.6 Overall soil erosion rates

Breshears et al. (2003) researched the relative importance of soil erosion by wind and by water in a Mediterranean ecosystem and found wind erosion to exceed water erosion from shrubland and forest sites, but not from a grassland site. Wind-driven transport of soil material from horizontal flux measurements were projected to annual timescales for shrubland (ca. 55 t ha^{-1} yr^{-1}), grassland (ca. 5.5 t ha^{-1} yr^{-1}) and forest (ca. 0.6 t ha^{-1} yr^{-1}). In a similar study, Goossens et al. (2001) found lower values (ca. 9.5 t ha^{-1} yr^{-1}) for arable fields in lower Saxony, Germany.

Owens et al. (2006a) proposed a tentative comparison between the various forms of soil loss, including water erosion processes in England and Wales. The rates quoted suggest that the likely range of annual soil loss rates may be similar for all forms of erosion. There will be temporal and spatial variations in the relative magnitude and extent of the different processes, with arable land being susceptible to all forms of erosion, and uncultivated land only at risk of water and, to some extent (i.e. exposed sandy and peaty soils), wind erosion.

3.2.7 Soil erosion rates for Europe

In the context of soil erosion, the true baseline is the amount of soil that is lost from a defined spatial unit under current environmental conditions. However, to determine a universal baseline it is not practicable to measure the actual loss of soil caused by erosion processes over the whole of Europe. It is more realistic to estimate baseline data for Europe by modelling the factors known to cause erosion, validating estimated baseline soil losses using actual measurements from the few experimental sites that currently exist, and augmenting by measurements from additional ‘benchmark’ sites. This leaves the spatial unit over which any baseline would apply undefined.

For soils under arable land use, several researchers quote soil erosion rates in Europe of between 10 and 20 t ha^{-1}yr^{-1} (Richter, 1983; Lal et al., 1998; Yang et al., 2003), whereas Arden-Clarke and Evans (1993) report that water erosion rates in Britain vary from 1-20 t ha^{-1} yr^{-1} but that the higher rates are rare events and localised. Boardman (1998) challenged the usefulness of an average rate of soil erosion for Europe, concluding that the rates vary too much in time and space to specify precise amounts. This variation is evident in Table 7 which shows ranges of the mean rates of soil lost by the recognised erosion types for agricultural land, and the actual soil erosion rates in tilled, arable agriculture by different combinations of erosion types (ca. 3-40 t ha^{-1}yr^{-1}). Although soil type, slope and climate are important factors, the greater part of the actual soil erosion rates relate to soil cover, soil management, and crop management. These factors can all be influenced by policy measures.

Table 7

4. Summary and conclusions

Figure 1

Tolerable soil erosion is a concept that has been developed over the last 60 years. Its definition has been related to the production function of soil by numerous authors. Inclusion of the regulation function of soil was realised, but not implemented in these definitions. Over the last 15 to 20 years a more holistic concept of soil functions has been developed, which this paper suggests should be applied to defining tolerable soil erosion: ‘any actual soil erosion rate at which a deterioration or loss of one or more soil functions (Table 4) does not occur’, with actual soil erosion meaning ‘the cumulative amount of soil lost by all recognised erosion types’.

Soil formation rates are proposed as a basis for establishing tolerable soil erosion. For Europe, the current state of scientific knowledge indicates that tolerable soil erosion rates range from ca. 0.3 – 1.4 t ha^{-1} yr^{-1} depending on the driving factors of weathering (e.g. parent material, climate, land use) and dust deposition (e.g. geographic position; distance to source). Relevant local components of soil functions that are impacted by soil erosion (e.g. surface water turbidity effects on aquatic wildlife or siltation of reservoirs) can be used to set tolerable soil erosion rates below the upper limit determined by soil formation rates.

Soil erosion research has focused traditionally on erosion by water (rill, gully etc.) and, to a lesser extent, by wind. However, over the last 10 - 15 years, the focus has broadened to include other important types of erosion, namely tillage erosion, crop harvesting and slope engineering or land levelling. Estimates of soil erosion rates for evaluation in a soil monitoring system need to consider all types of erosion, although mitigation should focus on the dominant type in any particular location. For all types of soil erosion, and particularly wind erosion and land levelling, there is a need for more spatially differentiated evidence of current rates.

The range of reported erosion rates for tilled arable soils is many times greater than the range of reported soil formation rates. This can be because soil formation is affected little by human activities, whereas today most soil erosion is anthropogenically induced. It should also be noted that soil erosion only appears to exceed tolerable rates when the soil is under cultivation or affected by other human disturbance. Furthermore, Boardman and Poesen (2006) estimated that arable agriculture accounts for ca. 70% of soil erosion in Europe, while Yang et al. (2003) developed a coarse-scaled global model from which they estimated that ca. 88% of soil erosion in Europe to be human-induced. Figure 1 gives an overview of the concept and rates of tolerable soil erosion and actual soil erosion (i.e. ‘the total amount of soil lost by all recognised erosion types’), and suggests directions for developing more detailed tolerable rates by applying the soil function concept and numerical soil formation modelling. The right side describes the components of soil erosion and the reported variation in their rates (mean and maximum). Tolerable soil erosion rates and approaches for deriving them are described on the left. At present, best estimates for mean rates in Europe are ca. 0.3-1.4 t ha^{-1}yr^{-1} for soil formation and ca. 3-40 t ha^{-1}yr^{-1} for actual soil erosion. These results are comparable with the 10-40 times greater than tolerable global estimate reported by Pimentel (2006). The figure also highlights areas for more research. Apart from the need for more detailed and differentiated values for soil erosion and formation rates (experimentally), it is also needed to identify yet unknown erosion types and further develop concepts such as the soil function system and numerical soil formation models, to implement soil erosion mitigation policies at appropriate spatial scales (differentiated by dominant factors). In addition, soil erosion work and policies should include a wide range of spatial and temporal scales until the connections between scales are better understood. Clearly, the spatial and temporal variation of tolerance-exceeding soil erosion is substantial and is likely to change, or possibly intensify, when climate and land use change. Therefore, the recommendation from Trimble and Crosson (2000a,b) and Brazier (2004), that resources should focus more on monitoring soil erosion by field measurements than on modelling, is supported by this review. Ideally, the approaches to field measurement (e.g. considering scale and spatial heterogeneity) would be developed in conjunction with process-based models.

However, if these measured and estimated ranges for soil formation and erosion are correct, and current conditions and management persist (a ‘business as usual’ scenario), then topsoils of tilled arable land on hill slopes (i.e. not flood plains) in Europe could be ca. 2 to 30 cm thinner in 100 years time (assuming a blanket tolerable rate of 1 t ha^{-1} yr^{-1} and a bulk density of 1.3 t m^{-3}) than today. Where in the range an area will be, depends on physical factors (e.g. climate, drainage, soil texture and structure) and on land management factors (see Table 7). For many topsoils in Europe this would mean a substantial deterioration in their production, regulation, habitat, and information functions (Table 4), if not a cessation of some of them. For areas where slope engineering and/or gully erosion occurs, even more soil could be lost. Thus, the status quo is not compliant with the intergenerational equity argument, i.e. that future generations should have the same rights to natural resources as those enjoyed by the current generation. A substantial effort is required to reduce soil erosion losses closer to tolerable levels, particularly in tilled, arable agriculture. In the future, climate change looks likely to increase rainfall intensity, if not annual totals, thereby increasing soil erosion by water, although there is much uncertainty about the spatio-temporal structure of this change as well as the socio-economic and agronomic changes that may accompany them (e.g. Boardman and Favis-Mortlock, 1993; Phillips et al., 1993; Nearing et al., 2004). Similarly, as a response to climate change, soil formation rates may change and the development of ‘moving tolerable rates’ with climate change scenarios may be required to support the policy sector with sound scientific guidelines.

This review of rates of soil loss by erosion, in the mineral soils of Europe, has clarified the tolerable rate of soil erosion to which modern land use systems should aspire. Furthermore, the evidence of well-founded tolerable rates of soil erosion, evaluated against actual soil erosion rates, is vital for developing policies to ensure that soil receives a level of protection comparable to that accorded to water and air in Europe.

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References

Abrahams, A.D., Parsons, A.J. and Luk, S.H., 1991. The effect of spatial variability in overland flow on the downslope pattern of soil loss on a semi-arid hillslope, southern Arizona. Catena 18, 255-70.

Alexander, E.B., 1985. Rates of soil formation from bedrock or consolidated sediments. Physical Geography 6(1), 25-42.

Alexander, E.B., 1988a. Rates of Soil Formation - Implications for Soil-Loss Tolerance. Soil Science 145(1), 37-45.

Alexander, E.B., 1988b. Strategies for Determining Soil-Loss Tolerance. Environmental Management 12(6), 791-796.

Allton, K.E., 2006. Interactions between soil microbial communities, erodibility and tillage practices, PhD thesis, Cranfield University, Cranfield, 224 pp.

Arden-Clarke, C. and Evans, R., 1993. Soil erosion and conservation in the United Kingdom. In: D. Pimentel (Editor), World soil erosion and conservation. Cambridge University Press, Cambridge, pp. 193-215.

Armstrong, C.L. and Mitchell, J.K., 1987. Transformations of rainfall by plant canopy. Transactions of the Asae 30(3), 688-696.

Aspiras, R.B., Allen, O.N., Harris, R.F. and Chesters, G., 1971. The role of microorganisms in the stabilization of soil aggregates. Soil Biology and Biochemistry 3(4), 347-353.

Barring, L., Jonsson, P., Mattsson, J.O. and Ahman, R., 2003. Wind erosion on arable land in Scania, Sweden and the relation to the wind climate - a review. Catena 52(3-4), 173-190.

Bazzoffi, P., Chisci, G. and Missere, D., 1989. Influenza delle opere di livellamento e scasso sull'erosione del suolo nella collina cesenate. Rivista di Agronomia 3, 231-221.

Bennett, H.H., 1939. Soil Conservation. McGraw-Hill, New York and London, pp 993.

Betts, H.D. and De Rose, R.C., 1999. Digital elevation models as a tool for monitoring and measuring gully erosion. International Journal of Applied Earth Observation and Geoinformation 1, 91–101.

Blong, R.J., Graham, O.P. and Veness, J.A., 1982. The role of sidewall processes in gully development; some N.S.W. examples. Earth Surface Processes and Landforms 7, 381-385.

Blum, W.E.H., 1993. Soil protection concept of the council of Europe and integrated soil research. In: H.A.P. Eijsackers and T. Hamers (Editors), Integrated Soil and Sediment Research: A Basis for Proper Protection. Kluwer Academic Publishers, pp. 37-47.

Blum, W.E.H., 2005. Functions of soil for society and the environment. Reviews in Environmental Science and Biotechnology 4 (3), 75-79.

Boardman, J., 1998. An average soil erosion rate for Europe: Myth or reality? Journal of Soil and Water Conservation 53(1), 46-50.

Boardman, J. and Favismortlock, D.T., 1993. Climate-change and soil-erosion in Britain. Geographical Journal 159, 179-183.

Boardman, J. and Poesen, J., 2006. Soil Erosion in Europe. John Wiley & Sons, UK, 855 pp.

Bohner, J., Schafer, W., Conrad, O., Gross, J. and Ringeler, A., 2003. The WEELS model: methods, results and limitations. Catena 52(3-4), 289-308.

Boix-Fayos, C., Calvo-Cases, A., Imeson, A.C. and Soriano-Soto, M.D., 2001. Influence of soil properties on the aggregation of some Mediterranean soils and the use of aggregate size and stability as land degradation indicators. Catena 44(1), 47-67.

Boix-Fayos, C., Martinez-Mena, M., Calvo-Cases, A., Castillo, V. and Albaladejo, J., 2005. Concise review of interrill erosion studies in SE Spain (Alicante and Murcia): erosion rates and progress of knowledge from the 1980s. Land Degradation & Development 16, 517–528.

Brady, N.C. and Weil, R., 2002. The Nature and Properties of Soils, 10. Prentice-Hall.

Brazier, R., 2004. Quantifying soil erosion by water in the UK: a review of monitoring and modelling approaches. Progress in Physical Geography 28(3), 340-365.

Breshears, D.D., Whicker, J.J., Johansen, M.P. and Pinder, J.E., 2003. Wind and water erosion and transport in semi-arid shrubland, grassland and forest ecosystems: Quantifying dominance of horizontal wind-driven transport. Earth Surface Processes and Landforms 28(11), 1189-1209.

Bromhead, E.N. and Ibsen, M.L., 2006. A review of landsliding and coastal erosion damage to historic fortifications in South East England. Landslides 3(4), 341-347.

Bryan, R.B., 1968. The development, use and efficiency of indices of soil erodibility. Geoderma 2(1), 5-26.

Cerdan O., Poesen J., Govers G., Saby N., Le Bissonnais Y., Gobin A., Vacca A., Quinton J., Auerswald K., Klik A., Kwaad F.J.P.M., Roxo M.J., 2006. Sheet and rill erosion. In: J. Boardman and J. Poesen (Editors), Soil Erosion in Europe. Wiley & Sons, Chichester, pp. 501 - 518.

Chappell, A., 1999. The limitations of using Cs-137 for estimating soil redistribution in semi-arid environments. Geomorphology 29(1-2), 135-152.

Chappell, A. and Thomas, A.D., 2002. Modelling to reconstruct recent wind erosion history of fields in eastern England. In: J.A. Lee and T.M. Zobeck (Editors), ICAR5/GCTE-SEN Joint Conference. USA Publication International Center for Arid and Semiarid Lands Studies, Texas Tech University, Lubbock, Texas, pp. 309.

Chappell, A. and Warren, A., 2003. Spatial scales of Cs-137-derived soil flux by wind in a 25 km(2) arable area of eastern England. Catena 52(3-4), 209-234.

Chenu, C., 1993. Clay polysaccharide or sand polysaccharide associations as models for the interface between microorganisms and soil - water related properties and microstructure. Geoderma 56(1-4), 143-156.

Chepil, W.S., 1950. Properties of soil which influence wind erosion. II. Dry aggregate structure as an index of erodibility. Soil Science 69, 403-414.

Chepil, W.S., Siddoway, F.H. and Armbrust, D.V., 1964. Wind erodibility of knolly terrain. Journal of Soil and Water Conservation 19, 179-181.

Ciesiolka, C.A.A. and Rose, C.W., 1998. The measurement of soil erosion. In: Penning de Vries, F.W.T., Agus, F. and Kerr, J. (eds) Soil Erosion at Multiple Scales: Principles and Methods for Assessing Causes and Impact. CAB International, Wallingford, pp. 287–301.

Cooper, C.M., 1993. Biological effects of agriculturally derived surface-water pollutants on aquatic systems - a review. Journal of Environmental Quality 22(3), 402-408.

Czarnes, S., Hallett, P.D., Bengough, A.G. and Young, I.M., 2000. Root- and microbial-derived mucilages affect soil structure and water transport. European Journal of Soil Science 51(3), 435-443.

Dalmeida, G.A., 1986. A model for saharan dust transport. Journal of Climate and Applied Meteorology 25(7), 903-916.

De Groot, R.S., Wilson, M.A. and Boumans, R.M.J., 2002. A typology for the classification, description and valuation of ecosystem functions, goods and services. Ecological Economics 41(3), 393-408.

De Ploey, J., 1981. The ambivalent effect of some factors of erosion. Mémoirs, Institute de Géologie, L'Université de Louvain 31, 171-181.

De Ploey, J., 1989. A Soil Erosion Map for Western Europe. Catena Verlag.

De Vente, J., Poesen, J., Verstraeten, G., Van Rompaey, A., and Govers, G., 2008. Spatially distributed modelling of soil erosion and sediment yield at regional scales in Spain. Global and Planetary Change 60 (3-4), 393-415.

De Vente, J. and Poesen, J., 2005. Predicting soil erosion and sediment yield at the basin scale: scale issues and semi-quantitative models. Earth Science Reviews 71, 95-125.

Degens, B.P., 1997. Macro-aggregation of soils by biological bonding and binding mechanisms and the factors affecting these: A review. Australian Journal of Soil Research 35(3), 431-459.

Dissmeyer, G.E. and Foster, G.R., 1981. Estimating the cover-management factor-(c) in the universal soil loss equation for forest conditions. Journal of Soil and Water Conservation 36(4), 235-240.

Doerr, S.H., Shakesby, R.A. and Walsh, R.P.D., 2000. Soil water repellency: its causes, characteristics and hydro-geomorphological significance. Earth-Science Reviews 51(1-4), 33-65.

Dostal, T., Krasa, J., Vaska, J. and Vrana, K., 2004. Map of soil erosion risk and sediment transport within the Czech Republic, Department of Drainage and Landscape Engineering, Faculty of Civil Engineering, CTU Prague.

D'Souza, V.P.C. and Morgan, R.P.C., 1976. Laboratory study of effect of slope steepness and curvature on soil erosion. Journal of agricultural engineering research 21(1), 21-31.

Eckelmann, W., Baritz, R., Bialousz, S., Carre, F., Jones, B., Kibblewhite, M., Kozak, J., Le Bas, C., Tóth, G., Várallyay, G., Yli Halla, M. and Zupan, M.., 2006. Common Criteria for Risk Identification according to Soil Threats, European Soil Bureau Research Report No. 20, EUR 22185 EN, Office for Official Publications of the European Communities, Luxembourg, pp. 94.

Edgerton, D.L., Harris, J.A., Birch, P. and Bullock, P., 1995. Linear relationship between aggregate stability and microbial biomass in 3 restored soils. Soil Biology & Biochemistry 27(11), 1499-1501.

Engelstaedter, S., Tegen, I. and Washington, R., 2006. North African dust emissions and transport. Earth-Science Reviews 79(1-2), 73-100.

Environment Agency, 2002. Agriculture and natural resources: benefits, costs and potential solutions, Bristol, U.K.

European Commission, 2006a. Proposal for a Directive of the European Parliament and of the Council establishing a framework for the soil protection of soil and amending Directive 2004/35/EC. COM (2006)232 Brussels, 22/09/2006; 30 pp., last accessed: 04/04/2007: http://ec.europa.eu/environment/soil/.

European Commission, 2006b. Communication from the Commission to the Council, the European Parliament, the European Economic and Social Committee and the Committee of the regions: thematic strategy for soil protection. COM (2006)231; last accessed 04/04/2007: http://ec.europa.eu/environment/soil/.

Evans, R., 1990. Soils at risk of accelerated erosion in England and Wales. Soil Use and Management 6(3), 125-130.

Evans, R., 1996. Soil Erosion and its Impacts in England and Wales, London.

FAO, 2003. Food Balance Sheet. Last modified 08/06/2008: http://faostat.fao.org/site/502/default.aspx.

Forster, S.M., 1989. The role of microorganisms in aggregate formation and soil stabilization: types of aggregation. Arid Soil Research 4, 85-98.

Foster, G.R., 1982. Modelling the erosion process. In: C.T. Haan, H.P. Johnson and D.J. Brakensiek (Editors), Hydrologic modelling of small watersheds, pp. 297-380.

Fournier, F., 1972. Soil conservation nature and environment series, Council of Europe, pp. 320.

Fullen, M.A. and Brandsma, R.T., 1995. Property changes by erosion of loamy sand soils in east Shropshire, UK. Soil Technology 8, 1 -15.

Funk, R. and Reuter, H. I. 2004. Dust production from arable land caused by wind erosion and tillage operations. In: International Symposium on Sand and Dust Storms, Beijing, China, 12-14 September 2004, Abstracts, 64. National Satellite Meteorological Center, Beijing, China.

Gobin, A., Govers, G., Jones, R.J.A., Kirkby, M.J. and Kosmas, C., 2002. Assessment and reporting on soil erosion: Background and workshop report. EEA Technical Report No. 84, 131 pp. Copenhagen.

Gobin A., Jones R., Kirkby M., Campling P., Govers G., Kosmas C. and Gentile R.A., 2004. Pan-European assessment and monitoring of soil erosion by water. Environmental Science and Policy 7, 25-38.

Goossens, D., Gross, J. and Spaan, W., 2001. Aeolian dust dynamics in agricultural land areas in lower Saxony, Germany. Earth Surface Processes and Landforms 26(7), 701-720.

Goudie, A.S. and Middleton, N.J., 2001. Saharan dust storms: nature and consequences. Earth-Science Reviews 56(1-4), 179-204.

Govers, G., Gimenez, R. and Van Oost, K., 2007. Rill erosion: Exploring the relationship between experiments, modelling and field observations. Earth-Science Reviews 84(3-4), 87-102.

Govers, G., Quine, T.A., Desmet, P.J.J. and Walling, D.E., 1996. The relative contribution of soil tillage and overland flow erosion to soil redistribution on agricultural land. Earth Surface Processes and Landforms 21, 929-946.

Govers, G., Quine, T.A. and Walling, D.E., 1993. The effect of water erosion and tillage movement on hill slope profile development: a comparison of field observations and model results. In: S. Wicherek (Editor), Farm Land Erosion: In Temperate Plains Environments and Hills. Elsevier, pp. 285-300.

Grousset, F.E., Ginoux, P., Bory, A. and Biscaye, P.E., 2003. Case study of a Chinese dust plume reaching the French Alps. Geophysical Research Letters 30(6), 10-1.

Gyssels, G., Poesen, J., Bochet, E. and Li, Y., 2005. Impact of plant roots on the resistance of soils to erosion by water: a review. Progress in Physical Geography 29(2), 189-217.

Hallett, P.D. and Young, I.M., 1999. Changes to water repellence of soil aggregates caused by substrate-induced microbial activity. European Journal of Soil Science 50(1), 35-40.

Heimsath, A.M., Dietrich, W.E., Nishiizumi, K. and Finkel, R.C., 1997. The soil production function and landscape equilibrium. Nature 388(6640), 358-361.

Hennings, V., 2003. Erosionsgefährdung ackerbaulich genutzter Böden durch Wasser (Karte im Maßstab 1:2,750,000), Nationalatlas Bundesrepublik Deutschland, Bd. 2: Relief, Boden und Wasser. Inst. für Länderkunde [Hrsg.]. Spektrum Akademischer Verlag, Heidelberg-Berlin.

Hinz, T., 2004. Agricultural PM10 emission from plant production. Proc. PM Emission Inventories Scientific Workshop. Lago Maggiore, Italy, 18. October 2004.

Holden, J., 2005. Controls of soil pipe frequency in upland blanket peat. Journal of Geophysical Research-Earth Surface 110(1), art. no. F01002.

Holden, J. and Burt, T.P., 2002. Infiltration, runoff and sediment production in blanket peat catchments: implications of field rainfall simulation experiments. Hydrological Processes 16(13), 2537-2557.

Hoosbeek, M.R. and Bryant, R.B., 1992. Towards the quantitative modeling of pedogenesis - a review. Geoderma 55(3-4), 183-210.

Horváth, V. and Erödi, B., 1962. Determination of natural slope category limits by functional identity of erosion intensity. International Association of Scientific Hydrology Publication 59, 131-143.

Hudson, N.W., 1993. Field Measurement of Soil Erosion and Runoff. FAO Soils Bulletin No. 68. FAO, Rome, pp. 139.

Huggett, R.J., 1998. Soil chronosequences, soil development, and soil evolution: a critical review. Catena 32, 155-172.

Humphreys, G.S. and Wilkinson, M.T., 2007. The soil production function: A brief history and its rediscovery. Geoderma 139 (1-2), 73-78.

International Society of Soil Science (ISSS), 1996. Terminology for Soil Erosion and Conservation, ITC-Enschede/ISRIC-Wageningen, The Netherlands, pp 313.

Israelevich, P.L., Levin, Z., Joseph, J.H. and Ganor, E., 2002. Desert aerosol transport in the Mediterranean region as inferred from the TOMS aerosol index. Journal of Geophysical Research D: Atmospheres 107(21), art. no. 4572.

Jaggard, K.W., Clark, C.J.A., May, M.J., McCullagh, S. and Draycott, A.P., 1997. Changes in the weight and quality of sugarbeet (Beta vulgaris) roots in storage clamps on farms. Journal of Agricultural Science 129, 287-301.

Johnson, L.C., 1987. Soil Loss Tolerance - Fact or Myth. Journal of Soil and Water Conservation 42(3), 155-160.

Jones, R.J.A., Le Bissonnais Y., Bazzoffi P., Sanchez Diaz J., Düwel O., Loj G., Øygarden L., Prasuhn V., Rydell B., Strauss P., Berenyi Uveges J., Vandekerckhove L. and Yordanov Y., 2004. Nature and Extent of Soil Erosion in Europe. In: L. Van-Camp, B. Bujarrabal, A-R Gentile, R.J.A. Jones, L. Montanarella, C. Olazabal and S-K Selvaradjou. (2004). Reports of the Technical Working Groups established under the Thematic Strategy for Soil Protection. Volume II Erosion. EUR 21319 EN/2, p.145-185. Office for Official Publications of the European Communities, Luxembourg.

Jones, R.J.A., Le Bas, C., Kozak, J., Düwel, O. and King, D., 2006. Identifying Risk Areas for Soil Erosion in Europe. In: W. Eckelmann et al. (Editors), Common Criteria for Risk Identification according to Soil Threats, European Soil Bureau Research Report 20, EUR 22185 EN, 23-33, Office for Official Publications of the European Communities, Luxembourg.

Kertész, A. and Centeri, C. 2006. Hungary. In: J. Boardman and J. Poesen (Editors), Soil Erosion in Europe. Wiley & Sons, Chichester, pp.139-153.

Ketterings, Q.M., Blair, J.M. and Marinissen, J.C.Y., 1997. Effects of earthworms on soil aggregate stability and carbon and nitrogen storage in a legume cover crop agroecosystem. Soil Biology & Biochemistry 29(3-4), 401-408.

Kibblewhite, M., Rubio, J-L., Kosmas, C., Jones, R., Arrouays, D., Huber, S., and Verheijen, F., 2007. Environmental Assessment of Soil for Monitoring Desertification in Europe. Eighth session of the Conference of the Parties (COP 8) to the United Nations Convention to Combat Desertification (UNCCD). Cranfield University, UK, Madrid, Spain, 3 - 14 September 2007, ISBN 1-8713115-97-2, pp. 62.

Kiem, R. and Kandeler, E., 1997. Stabilization of aggregates by the microbial biomass as affected by soil texture and type. Applied Soil Ecology 5(3), 221-230.

Kirkby, M.J., 1969. Erosion by water on hillslopes. In: R.J. Chorley (Editor), Water, earth and man, Methuen, London, pp. 588.

Kirkby, M.J., Jones, R.J.A., Irvine, B., Gobin, A., Govers, G., Cerdan, O., Van Rompaey, A. J. J., Le Bissonnais, Y., Daroussin, J., King, D., Montanarella, L., Grimm, M., Vieillefont, V., Puigdefabregas, J., Boer, M., Kosmas, C., Yassoglou, N., Tsara, M., Mantel, S., Van Lynden, G. J., Huting, J., 2004. Pan-European Soil Erosion Risk Assessment: the PESERA map. Version 1 October 2003. Explanation of Special Publication Ispra 2004 No.73 (S.P.I.04.73), European Soil Bureau Research Report No.16, EUR 21176. Luxembourg: Office for Official Publications of the European Communities; 2004. 18 pp.

Kirkby, M.J., Le Bissonais, Y., Coulthard, T.J., Daroussin, J. and McMahon, M.D., 2000. The development of Land Quality Indicators for Soil Degradation by Water Erosion. Agriculture, Ecosystems and Environment 81, 125 – 136.

Kirkby, M., 2001. From plot to continent: reconciling fine and coarse scale erosion models. In: Stott, D.E., Mohtar, R.H. and Steinhardt, G.C. (eds) Sustaining the Global Farm. Selected papers from the 10th International Soil Conservation Organisation meeting, held 24–29 May 1999 at Purdue University and USDA-ARS National Soil Erosion Research Laboratory.

Kirkby, M.J., Irvine, B.J., Jones, R.J.A., Govers, G., Boer, M., Cerdan, O., Daroussin, J., Gobin, A., Grimm, M., Le Bissonais, Y., Kosmas, C., Mantel, S., Puigdefabregas, J. and Van Lynden, G., 2008. The PESERA coarse scale erosion model for Europe: I – Model rationale and implementation. European Journal of Soil Science 59(6), 1293-1306.

Klingebiel, A.A. and Montgomery, P.H., 1961. Land Capability Classification, Agricultural Handbook 210, U.S. Department of Agriculture, Washington.

Knighton, D., 1998. Fluvial Forms and Processes: A new perspective. John Wiley and Sons, New York.

Kok, K., Clavaux, M.B.W., Heerebout, W.M. and Bronsveld, K., 1995. Land degradation and land cover change detection using low-resolution satellite images and the CORINE database - A case study in Spain. ITC Journal 3, 217-228.

Laflen, J.M. and Colvin, T.S., 1981. Effect of crop residue on soil loss from continuous row cropping. Transactions of the American Society of Agricultural Engineers 24(3), 605-609.

Lal, R., 1998. Soil erosion impact on agronomic productivity and environment quality. Critical Reviews in Plant Sciences 17(4), 319-464.

Lal, R. 2001. Soil degradation by erosion. Land Degradation & Development 12(6), 519 – 539.

Lang, R.D. and McCaffrey, L.A.H., 1984. Ground cover: its effects on soil loss from grazed runoff plots, Gunnedah. Journal of the Soil Conservation Service NSW 40, 56-61.

LeBissonnais, Y., 1996. Aggregate stability and assessment of soil crustability and erodibility .1. Theory and methodology. European Journal of Soil Science 47(4), 425-437.

Lee, E.M. and Clark, A.R., 2004. Investigation and management of soft cliffs. Thomas Telford, London,.

Lee, E.M. and Jones, D.K.C., 2004. Landslide risk assessment. Thomas Telford, London, pp. 454.

Lindstrom, M.J., Nelson, W.W. and Schumacher, T.E., 1992. Quantifying tillage erosion rates due to moldboard plowing. Soil & Tillage Research 24(3), 243-255.

Lindstrom, M.J., Schumacher, J.A. and Schumacher, T.E., 2000. TEP: A Tillage Erosion Prediction model to calculate soil translocation rates from tillage. Journal of Soil and Water Conservation 55(1), 105-108.

Lloyd, D.S., 1987. Turbidity as a Water Quality Standard for Salmonid Habitats in Alaska. North American Journal of Fisheries Management 7(1), 34-45.

Lloyd, D.S., Koenings, J.P. and Laperriere, J.D., 1987. Effects of Turbidity in Fresh Waters of Alaska. North American Journal of Fisheries Management 7(1), 18-33.

Lobb, D.A., Kachanoski, R.G. and Miller, M.H., 1995. Tillage translocation and tillage erosion on shoulder slope landscape positions measured using cs-137 as a tracer. Canadian Journal of Soil Science 75(2), 211-218.

Lobb, D.A., Kachanoski, R.G. and Miller, M.H., 1999. Tillage translocation and tillage erosion in the complex upland landscapes of southwestern Ontario, Canada. Soil & Tillage Research 51(3-4), 189-209.

Löye-Pilot, M.D., Martin, J.M. and Morelli, J., 1986. Influence of saharan dust on the rain acidity and atmospheric input to the mediterranean. Nature 321(6068), 427-428.

Mannering, J.V., 1981. The use of soil loss tolerances as a strategy for soil conservation. In: R.P.C. Morgan (Editor), Soil conservation: problems and prospects. John Wiley & Sons, New York, N.Y., pp. 337-349.

Marinissen, J.C.Y., 1994. Earthworm populations and stability of soil-structure in a silt loam soil of a recently reclaimed polder in the Netherlands. Agriculture Ecosystems & Environment 51(1-2), 75-87.

Martinez-Casasnovas, J.A., Anton-Fernandez, C. and Ramos, M.C., 2003. Sediment production in large gullies of the Mediterranean area (NE Spain) from high-resolution digital elevation models and geographical information systems analysis. Earth Surface Processes and Landforms 28, 443-456.

McHugh, M., Harrod, T. and Morgan, R., 2002. The extent of soil erosion in upland England and Wales. Earth Surface Processes and Landforms, 27(1): 99-107.

McInnes, R.G., Tomalin, D. and Jakeways, J. (Eds.), 2000. Coastal change, climate and stability. European Commission LIFE project LIFE-97 ENV/UK/000510 (1997-2000), Isle of Wight Council, Isle of Wight, U.K.

Mech, S.J. and Free, G.A., 1942. Movement of soil during tillage operations. Agricultural Engineering 23, 379-382.

Meyer, L.D. and Wischmeier, W.H., 1969. Mathematical simulation of the process of soil erosion by water. Transactions of the American Society of Agricultural Engineers 12, 754-762.

Meyer, L.D., Foster, G.R. and Römkens, M.J.M., 1975. Source of soil eroded by water from upland slopes, Present and prospective technology for predicting sediment yields and sources. USDA-ARS Publication, pp. 177-189.

Middleton, N.J. and Goudie, A.S., 2001. Saharan dust: sources and trajectories. Transactions of the Institute of British Geographers 26(2), 165-181.

Minasny, B. and McBratney, A.B., 2001. A rudimentary mechanistic model for soil formation and landscape development II. A two-dimensional model incorporating chemical weathering. Geoderma 103(1-2), 161-179.

Ministry of Environment of the Slovak Republic and Slovak Environmental Agency, 2002. Landscape Atlas of the Slovak Republic. Bratislava and Banska Bystrica, pp. 344.

Mirtskhulava, T.E., 2001. On the maximum soil loss tolerance. Eurasian Soil Science 34(3), 321-325.

Montanarella, L., Jones, R.J.A. and Hiederer, R. (2006). Distribution of peatland in Europe. Mires and Peat 1, 1-10.

Montgomery, D.R., 2007. Soil erosion and agricultural sustainability. Proceedings of the National Academy of Sciences of the United States of America 104(33), 13268-13272.

Morgan, R.P.C., 2005. Soil Erosion and Conservation. Blackwell Publishing, pp. 299.

Morgan, R.P.C., Martin, L. and Noble, C.A., 1986. Soil erosion in the United Kingdom: a case study from mid-Bedfordshire. Silsoe College Occasional Paper 14, Silsoe College, Cranfield University, Silsoe, uk.

Morgan, R.P.C., Brandt, C.J. and Studdard, M.J., 1988. The role of plant covers in controlling soil detachment by rainfall. Silsoe College pamphlet.

Morgan, R.P.C., 1987. Sensitivity of European soils to ultimate physical degradation. In: Barth, H. and L'Hermitage, P. (eds) Scientific basis for soil protection in the European Community. Amsterdam, Elsevier Applied Science, 147-157.

Morvan, X, Saby, N.P.A., Arrouays, D., Le Bas, C., Jones, R.J.A., Verheijen, F.G.A., Bellamy, P.H., Stephens, M., Kibblewhite, M.G., 2008. Soil monitoring in Europe: A review of existing systems and requirements for harmonisation. Science of The Total Environment 391(1), 1-12.

Musgrave, G.W., 1947. The quantitative evaluation of factors in water erosion: a first approximation. Journal of Soil and Water Conservation 2, 133-138.

Nachtergaele, J. and Poesen, J., 1999. Assessment of soil losses by ephemeral gully erosion using high-atitude (stereo) aerial photographs. Earth Surfaces Processes and Landforms 24, 693-706.

Nachtergaele, J., Poesen, J., Steegen, A.,Takken, I., Beuselinck, L., Vandekerckhove, L. and Govers, G., 2001. The value of a physically based model versus an empirical approach in the prediction of ephemeral gully erosion for loess-derived soils. Geomorphology 40, 237–252.

Nearing, M.A., Foster, G.R., Lane, L.J. and Finkner, S.C., 1989. A process based soil erosion model for USDA water erosion prediction project technology. Transactions of the American Society of Agricultural Engineers 32, 1587–1593.

Nearing, M.A., Pruski, F.F. and O'Neal, M.R., 2004. Expected climate change impacts on soil erosion rates: A review. Journal of Soil and Water Conservation 59(1), 43-50.

Newcombe, C.P. and Macdonald, D.D., 1991. Effects of Suspended Sediments on Aquatic Ecosystems. North American Journal of Fisheries Management 11(1), 72-82.

Nikitin, E.D., 2005. Soil Ecology and the Study of Soil Functions. Eurasian Journal of Soil Science 38(9), 922-930.

Oades, J.M., 1993. The role of biology in the formation, stabilization and degradation of soil structure. Geoderma 56(1-4), 377-400.

Osterkamp, W.R. and Toy, T.J. 1997. Geomorphic considerations for erosion prediction. Environmental Geology 29, 152–157.

Owens, P.N., Rickson, R.J. and Clarke, M.A., 2006a. Review of how appropriate current mechanisms and advice on best practice for control and mitigation of erosion is to wind, tillage co-extraction with root vegetables in England and Wales, and consideration of alternative methods and advice, Cranfield University, UK. http://randd.defra.gov.uk/Default.aspx?Menu=Menu&Module=More&Location=None&ProjectID=14183&FromSearch=Y&Publisher=1&SearchText=wind%20erosion&SortString=ProjectCode&SortOrder=Asc&Paging=10#Description;%20[last%20accessed%2014/02/2009

Owens, P.N., Rickson, R.J., Clarke, M.A., Dresser, M., Deeks, L.K., Jones, R.J.A., Woods, G.A., Van Oost, K. and Quine, T.A., 2006b. Review of the existing knowledge base on magnitude, extent, causes and implications of soil loss due to wind, tillage and co-extraction with root vegetables in England and Wales, and recommendations for research priorities, Cranfield University, UK. http://randd.defra.gov.uk/Default.aspx?Menu=Menu&Module=More&Location=None&ProjectID=14183&FromSearch=Y&Publisher=1&SearchText=wind

%20erosion&SortString=ProjectCode&SortOrder=Asc&Paging=10#Description;%20[last%20accessed%2014/02/2009

Owens, P.N., Rickson, R.J., Clarke, M.A., Dresser, M., Deeks, L.K., Jones, R.J.A., Woods, G.A., Van Oost, K. and Quine, T.A., 2006c. Scoping study of soil loss through wind erosion, tillage erosion and soil co-extracted with root vegetables, DEFRA Research Project SP08007. http://randd.defra.gov.uk/Document.aspx?Document=SP08007_6584_FRA.pdf

Oygarden, L., 2003. Rill and gully development during an extreme winter runoff event in Norway. Catena 50(2-4), 217-242.

Patsukevich, Z.V., Gennadiev, A.N. and Gerasimova, M.I., 1997. Soil loss tolerance and self-rehabilitation of soils. Eurasian Soil Science 30(5), 557-563

Phillips, D.L., White, D. and Johnson, B., 1993. Implications of Climate-Change Scenarios for Soil-Erosion Potential in the USA. Land Degradation and Rehabilitation 4(2), 61-72.

Pierce, F.J., Larson, W.E. and Dowdy, R.H., 1984. Soil Loss Tolerance - Maintenance of Long-Term Soil Productivity. Journal of Soil and Water Conservation 39(2), 136-138.

Pierson Jr, F.B., van Vactor, S.S., Blackburn, W.H. and Wood, J.C., 1994. Incorporating small scale variability into predictions of hydrologic response on sagebrush rangelands. In: Blackburn, W.H., Pierson Jr, F.B., Schuman, G.E. and Zartman, R. (eds) Variability in Rangeland Water Erosion Processes. Proceedings of a symposium of the Soil Science Society of America, Minneapolis, Minnesota, November, 1992. SSA Special Publication No. 38. SSSA, Madison, WI.

Pillans, B., 1997. Soil development at snail's pace: Evidence from a 6 Ma soil chronosequence on basalt in north Queensland, Australia. Geoderma 80(1-2), 117-128.

Pimentel, D., 2006. Soil erosion: a food and environmental threat. Environment, Development and Sustainability 8, 119-137.

Pimentel, D. Harvey, C. Resosudarmo, P., Sinclair, K., Kurz, D., McNair, M., Crist, S., Shpritz, L., Fitton, L., Saffouri, R. and Blair, R., 1995. Environmental and economic costs of soil erosion and conservation benefits. Science 267(5201), 1117-1123.

Poesen, J., Nachtergaele, J., Verstraeten, G. and Valentin, C., 2003. Gully erosion and environmental change: importance and research needs. Catena 50(2-4), 91-133.

Poesen, J. Vanwalleghem, T., De Vente, J., Knapen, A., Verstraeten, G. and Martinez-Casasnova, J. A., 2006. Gully erosion in Europe. In: J. Boardman and J. Poesen (Editors), Soil Erosion in Europe. Wiley & Sons, Chichester, pp. 515 - 536.

Quine, T.A., Desmet, P., Govers, G., Vandaele, K. and Walling, D., 1994. A comparison of the roles of tillage and water erosion in landform development and sediment export on agricultural land, near Leuven, Belgium. In: L.J. Olive, R.J. Loughran and J.A. Kesby (Editors), Variability in Stream Erosion and Sediment Transport. IAHS Publication, 77-86.

Quine, T.A., Van Oost, K., Walling, D.E. and Owens, P.N., 2006. Development and application of GIS-based models to estimate national rates of soil erosion by tillage, wind and root crop harvest, University of Exeter, Exeter.

Quine, T.A. and Zhang, Y., 2004a. Re-defining tillage erosion: quantifying intensity-direction relationships for complex terrain. 1. Derivation of an adirectional soil transport coefficient. Soil Use and Management 20, 114-123.

Quinton, J.N., Edwards, G.M. and Morgan, R.P.C., 1997. The influence of vegetation species and plant properties on runoff and soil erosion: results from a rainfall simulation study in south east Spain. Soil Use and Management 13(3), 143-148.

Rauws, G. and Govers, G., 1988. Hydraulic and soil mechanical aspects of rill generation on agricultural soils. Journal of Soil Science 39(1), 111-124.

Reheis, M.C. and Kihl, R., 1995. Dust deposition in southern Nevada and California, 1984-1989 - relations to climate, source area, and source lithology. Journal of Geophysical Research-Atmospheres 100(D5), 8893-8918.

Richter, G., 1983. Aspects and problems of soil erosion hazard in the EEC countries. In: A.G. Prendergast (Editor), Soil Erosion. Commission of the European Communities Report No. EUR 8427 EN, pp. 9-17.

Riksen, M., Brouwer, F. and de Graaff, J., 2003. Soil conservation policy measures to control wind erosion in northwestern Europe. Catena 52(3-4), 309-326.

Ritz, K. and Young, I.M., 2004. Interactions between soil structure and fungi. Mycologist 18(2), 52-59.

Roose, E., 1996. Land Husbandry: components and strategy. FAO soils Bulletin N° 70. FAO, Rome, 380 pp.

Ruysschaert, G., Poesen, J., Verstraeten, G. and Govers, G., 2004. Soil loss due to crop harvesting: significance and determining factors. Progress in Physical Geography 28(4), 467-501.

Ruysschaert, G., Poesen, J., Verstraeten, G. and Govers, G., 2005. Interannual variation of soil losses due to sugar beet harvesting in West Europe. Agriculture Ecosystems & Environment 107(4), 317-329.

Sánchez, J., Recatalá, L., Colomer, J.C. and Añó, C., 2001. Assessment of soil erosion at national level: a comparative analysis for Spain using several existing maps. In: Y. Villacampa, C.A. Brevia and U. J.L. (Editors), Ecosystems and Sustainable Development III. Advances in Ecological Sciences 10. WITT Press, Southampton, pp. 249-258.

Schaller, M., von Blanckenburg, N., Hovius, N. and Kubik, P.W., 2001. Large-scale erosion rates from insitu-produced cosmogenic nuclides in European river sediments. Earth and Planetary Science Letters 188, 441-458.

Schaub, D. and Prasuhn, V., 1998. A map on soil erosion on arable land as a planning tool for sustainable land use in Switzerland. In: H.P. Blume et al. (Editors), Towards Sustainable Land Use, Volumes I & II - Furthering Cooperation between People and Institutions. Advances in Geoecology, pp. 161-168.

Schumm, S.A., Harvey, M.D. and Watson, C.C., 1984. Incised Channels: Morphology, dynamics and control. Water Resources Publications, Littleton, Colorado, U.S.A., pp. 200.

Scullion, J. and Malik, A., 2000. Earthworm activity affecting organic matter, aggregation and microbial activity in soils restored after opencast mining for coal. Soil Biology & Biochemistry 32(1), 119-126.

Shakesby, R.A. and Doerr, S.H., 2006. Wildfire as a hydrological and geomorphological agent. Earth-Science Reviews 74(3-4), 269-307.

Simonson, R.W., 1995. Airborne Dust and Its Significance to Soils. Geoderma 65(1-2), 1-43.

Skidmore, E.L., 1982. Soil loss tolerance (Chapter 8), Determinants of soil loss tolerance. ASA, SSSA, 677 South Segoe Road, Madison, pp. 87-93.

Skidmore, E.L. and Woodruff, N.P., 1968. Wind erosion forces in the United States and their use in predicting soil loss. USDA Agricultural Research Service Handbook, 346.

Smith, R and Quinton, J.N., 2000. Dynamics and scale in simulating erosion by water. In: Schmidt, J. (ed.) Soil Erosion: Application of Physically Based Models. Springer, Berlin, pp. 282–294.

Soil Science Society of America, 2001. Glossary of Soil Science Terms. Soil Science Society of America, Inc., Madison WI, pp. 140.

Sombroek, W. G. (Ed.) and D. Sims (Ed.) (1995). "Planning for sustainable use of land resources: towards a new approach." FAO Land and Water Bulletin 1024-6703 (2), ISBN 92-5-103724-8, FAO, Rome, Italy.

Sparovek, G. and De Maria, I.C., 2003. Multiperspective analysis of erosion tolerance. Scientia Agricola 60, 409-416.

Stamey, W.L. and Smith, R.M., 1964. A conservation definition of erosion tolerance. Soil Science 97, 183-186.

Stocking, J.M. and Pain, A., 1983. Soil life and the minimum soil depth for productive yields: developing a new concept. Discussion paper 150, School of development studies, University of East Anglia.

Swap, R., Garstang, M., Greco, S., Talbot, R. and Kallberg, P., 1992. Saharan dust in the Amazon basin. Tellus Series B-Chemical and Physical Meteorology 44(2), 133-149.

Temple, D.M., 1982. Flow retardance of submerged grass channel linings. Transactions of the Asae 25(5), 1300-1303.

Temple, P.H. and Rapp, A., 1972. Landslides in the Mgeta area, western Uluguru Mountains, Tanzania. Geografiska Annaler 54-A, 157-193.

Tisdall, J.M. and Oades, J.M., 1982. Organic-matter and water-stable aggregates in soils. Journal of Soil Science 33(2), 141-163.

Trimble, S.W. and Crosson, P., 2000a. Land use - US soil erosion rates - Myth and reality. Science 289(5477), 248-250.

Trimble, S.W. and Crosson, P., 2000b. Measurements and models of soil loss rates - Response. Science 290(5495), 1301-1301.

Turkelboom, F. and Trebuil, G., 1998. A multi-scale approach for on-farm erosion research: application to Northern Thailand Highlands. In: Penning de Vries, F.W.T., Agus, F. and Kerr, J. (eds) Soil Erosion at Multiple Scales: Principles and Methods for Assessing Causes and Impact. CAB International, Wallingford, pp. 51–72.

Valentin, C., Poesen, J. and Yong, L., 2005. Gully erosion: Impacts, factors and control. Catena 63, 132 - 153.

Van der Knijff, J.M., Jones, R.J.A. and Montanarella, L., 2002. Soil erosion risk assessment in Italy. In: J.L. Rubio, R.P.C. Morgan, S. Asins and V. Andreu (Editors), the third International Congress Man and Soil at the Third Millennium. Geoforma Ediciones, Logrono, pp. 1903-1913.

Van Muysen, W., Govers, G., Bergkamp, G., Roxo, M. and Poesen, J., 1999. Measurement and modelling of the effects of initial soil conditions and slope gradient on soil translocation by tillage. Soil & Tillage Research 51(3-4), 303-316.

Van Muysen, W., Van Oost, K. and Govers, G., 2006. Soil translocation resulting from multiple passes of tillage under normal field operating conditions. Soil and Tillage Research 87(2), 218-230.

Van Noordwijk, M., van Roode, M., McCaillie, E.L. and Lusiana, B., 1998. Erosion and sedimentation as multi-scale, fractal processes: implications for models, experiment and the real world. In: Penning de Vries, F.W.T., Agus, F. and Kerr, J. (eds) Soil Erosion at Multiple Scales: Principles and Methods for Assessing Causes and Impact. CAB International, Wallingford, pp. 223–253.

Van Oost, K., Govers, G. and Desmet, P., 2000a. Evaluating the effects of changes in landscape structure on soil erosion by water and tillage. Landscape Ecology 15, 577-589.

Van Oost, K., Govers, G., Van Muysen, W. and Quine, T.A., 2000b. Modeling translocation and dispersion of soil constituents by tillage on sloping land. Soil Science Society of America Journal 64(5), 1733-1739.

Van Oost, K., Govers, G., Quine, T. A., Heckrath, G., Olesen, J. E., De Gryze, S. and Merckx, R., 2005a. Landscape-scale modeling of carbon cycling under the impact of soil redistribution: The role of tillage erosion. Global Biogeochemical Cycles 19(4), 1733-1739.

Van Oost, K., Van Muysen, W., Govers, G., Deckers, J. and Quine, T.A., 2005b. From water to tillage erosion dominated landform evolution. Geomorphology 72(1-4), 193-203.

Van Oost, K., Govers, G., de Alba, S. and Quine, T.A., 2006. Tillage erosion: a review of controlling factors and implications for soil quality. Progress in Physical Geography 30(4), 443-466.

Van Oost, K., Govers, G., de Alba, S. and Quine, T.A., 2006. Tillage erosion: a review of controlling factors and implications for soil quality. Progress in Physical Geography 30(4), 443-466.

Van Rompaey, A.J.J., Bazzoffi, P., Jones, R.J.A. and Montanarella, L., 2005. Modeling sediment yields in Italian catchments. Geomorphology 65, 157–169.

Vandekerckhove, L., Arnoldussen, A., Bazzoffi, P., Böken, H., Castillo, V., Crescimano, G., Düwel, O., Esteve, J.F., Imeson, A., Jarman, R., Jones, , .J.A., Kobza, J., Lane, M., Le Bissonnais, Y., Loj, G., Owens, P.N., Øygarden, L., Mollenhauer, K., Prasuln, V., Redfern, P., Sánchez Díaz, J., Strauss, P. and Üveges Berényi, J., 2004. Monitoring Soil Erosion in Europe. In: L. Van-Camp et al. (Editors), Reports of the Technical Working Groups established under the Thematic Strategy for Soil Protection, volume II Erosion, EUR 21319 EN/2, Office for Official Publications of the European Communities, Luxembourg, pp. 295-309.

Vandekerckhove, L., Poesen, J., Oostwoud Wijdenes, D., Gyssels, G., Beuselinck, L. and de Luna, E., 2000. Characteristics and controlling factors of bank gullies in two semi-arid Mediterranean environments. Geomorphology 33, 37-58.

Wakatsuki, T. and Rasyidin, A., 1992. Rates of weathering and soil formation. Geoderma 52(3-4), 251-263.

Walling, D.E. 1983., The sediment delivery problem. Journal of Hydrology 65(1-3), 209-237.

Walling, D.E., 1990. Linking the field to the river: sediment delivery from agricultural land. In: J. Boardman, I.D.L. Foster and J.A. Dearing, Editors, Soil Erosion on Agricultural Land, Wiley, Chichester, West Sussex, England (1990), pp. 129–152.

Warren, A., (Ed.) 2002. Wind erosion on agricultural land in Europe. EUR 20370 EN, Office for the Official Publications of the European Communities, Luxembourg, pp 76..

Wiersum, L.K., 1979. Comparison of the behavior of some root systems under restricted aeration. Netherlands Journal of Agricultural Science 27(1), 92-98.

Wilkinson, B.H. and McElroy, B.J., 2007. The impact of humans on continental erosion and sedimentation. Geological Society of America Bulletin 119(1-2), 140-156.

Wischmeier, W.H. and Smith, D.D. 1958. Rainfall energy and its relationship to soil loss. Transactions of the American Geophysical Union 39, 285-291.

Wischmeier, W.H., Johnson, C. and Cross, V., 1971. A soil erodibility nomograph for farmland and construction sites. Journal of Soil and Water Conservation 26(5), 189-193.

Wischmeier, W.H. and Mannering, J.V., 1969. Relation of soil properties to its erodibility. Soil Science Society America Proceedings 33(1), 131-137.

Wischmeier, W.H. and Smith, D.D., 1978. Predicting rainfall erosion losses –a guide for conservation planning. In: Agricultural Handbook 537, U.S. Department of. Agriculture, Washington.

Yang, D., Kanae, S., Oki, T., Koike, T. and Musiake, K., 2003. Global potential soil erosion with reference to land use and climate changes. Hydrological Processes 17, 2913-2928.

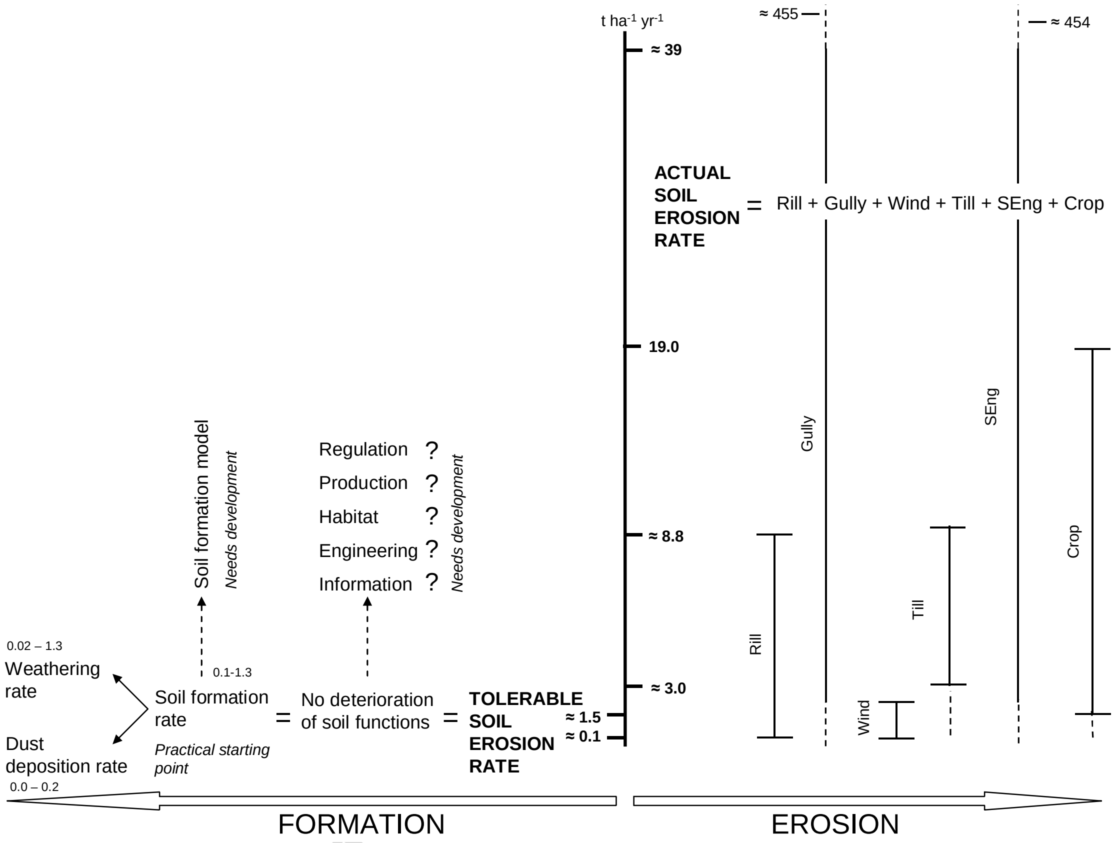
Yoo, K. and Mudd, S.M., 2008. Discrepancy between mineral residence time and soil age: Implications for the interpretation of chemical weathering rates. Geology 36(1), 35-38.

Zachar, D., 1982. Soil erosion (Development in Soil Science 10). Elsevier, Amsterdam, pp. 84-86..

Zhang, G.S., Chan, K.Y., Oates, A., Heenan, D.P. and Huang, G.B., 2007. Relationship between soil structure and runoff/soil loss after 24 years of conservation tillage. Soil & Tillage Research 92(1-2), 122-128.

Zingg, A.W., 1940. Degree and length of land slope as it affects soil loss in runoff. Agricultural Engineering 21, 59-64.

Figure 1



Tolerable vs. actual soil erosion, concept and rates. See the text for a detailed explanation. All numbers are in t $ha^{-1}yr^{-1}$. Please see relevant sections of this paper for more detailed information and references. Rill=rill and sheet erosion; Gully=gully erosion; Wind=wind erosion; Till=tillage erosion; SEng=erosion by slope engineering; Crop=erosion by crop harvesting.

Table 1

Range of spatial scales of soil erosion research (Rickson, 2006; after Wickenkamp et al., 2000).

Erosion research technique	Area	Dimension descriptors (Wickenkamp et al., 2000)		Dominant processes operating	Selected References
Splash cup	mm^2	Nanoscale	Subtope	Rain splash dominant; overland flow/deposition limited. No gullies, stream bank erosion or mass movements.	Ellison (1944); Kinnell (1974); Morgan et al. (1988); Salles, C. and Poesen, J. (2000)
Laboratory tray	cm^2	Nanoscale	Subtope	Rain splash dominant?; overland flow/deposition limited. No gullies, stream bank erosion or mass movements.	Idowu (1996)
Runoff rig	m^2	Microscale	Tope	Rain splash and overland flow; some deposition possible. No gullies, stream bank erosion or mass movements.	Kamalu (1993); Govers (1989)
Field plot	m^2	Microscale	Tope	Rain splash and overland flow; some deposition. Some gullying and mass movements possible; no stream bank erosion.	Wischmeier and Smith (1978); Ciesiolka and Rose (1998); Pierson et al. (1994)
Field	ha	Mesoscale	Chore	Rain splash, overland flow and deposition. Gullying and mass movements possible. No stream bank erosion.	Evans and Boardman (1994); Walling and Quine (1991)
Sub-catchment	ha – km^2	Mesoscale	Chore	Rain splash, overland flow and deposition. Gullying possible. Some stream bank erosion.	Hudson (1981); Rapp et al. (1972)
Catchment/landscape	km^2	Macroscale	Region	Rain splash, overland flow and deposition. Some gullying and mass movement possible. Stream bank erosion.	Dickinson and Collins (1998)

Table 2

Estimated annual costs of soil erosion to UK economy in £million (2000 prices)

	£ million	% contribution from agriculture
Soil organic matter loss, leading to increased emissions of carbon dioxide	74	95%
On-farm costs (additional fertilisers, etc.)	8	100%
Accidents/stream channels (i.e. off-site costs mainly related to clean-up operations)	8.2	95%
Effects of flooding	115	14%
TOTAL ANNUAL COST (£ million)	205	

Source: Environment Agency (2002).

Table 3

Interpretations and definitions for 'tolerable soil erosion'

Tolerable soil erosion - definition	Reference
The maximum volume of erosion-removed topsoil that provides high, or economically feasible, fertility for a long time	Patsukevich et al., 1997.
Soil loss balanced by soil formation through weathering of rocks	in Roose (1996)
Erosion that does not lead to any appreciable reduction in soil productivity	in Roose (1996)
The maximum rate of soil erosion that permits an optimum level of crop productivity to be sustained economically and indefinitely	ISSS (1996)
The average annual soil loss a given soil type may experience and still maintain its productivity over an extended period of time (permissible soil loss)	Kok et al. (1995)
The maximum permissible rate of erosion at which soil fertility can be maintained over 20-25 years	Morgan (2005)
(i) The maximum average annual soil loss that will allow continuous cropping and maintain soil productivity without requiring additional management inputs. (ii) The maximum soil erosion loss that is offset by the theoretical maximum rate of soil development which will maintain an equilibrium between soil losses and gains	SSSA (2001)
Rate of soil erosion is not larger than the rate of soil production (acceptable rates of soil erosion)	Boardman and Poesen (2006)

Table 4

Harmonised primary soil functions scheme.

Primary soil functions	Components
Habitat	Refugium function; nursery function; medicinal resources; gene pool; seed bank
Information	Cultural information (archaeological and palaeontological); science and education; spiritual and historic; recreation; aesthetic information
Production	Food; fodder; fibre; raw materials; renewable energy
Engineering	Technical, industrial and socio-economic structures
Regulation	Gas regulation; climate regulation; disturbance resistance; disturbance resilience; water supply; water filtering; pH buffering; biotransformation of organic carbon; soil retention; soil formation; nutrient regulation; biological control; waste and pollution control

Table 5

Reported soil formation rates by weathering (large scale); na=not available.

Methodology	Spatial scale	Temporal scale	Lower limit	Upper limit	Reference
Mass balance (Si)	Non-carbonate; non-arable; North America, Europe, Australia (Victoria), Zimbabwe	na	0.02	1.27	Alexander (1988a)
Mass balance (Al, Fe, Ca, K, Mg, Na, Si)	Global		0.37	1.29	Wakatsuki and Rasyidin (1992)
In situ cosmogenic ^{10}Be and ^{26}Al	Northern California	na	0.39	0.91	Heimsath et al. (1997)
In situ cosmogenic ^{10}Be	Middle European rivers	10-40 Kyr	0.26	1.3	(Schaller et al. (2001)
Continental scale erosion/sedimentation	Global	542 Myr	0.4	1.4	Wilkinson and McElroy (2007)
Na	USA	na	0.3	1.1	Bennett (1939)

Table 6

Soil formation rates by dust deposition (adapted from Goudie and Middleton, 2001)

Location	Dust deposition ($t\ ha^{-1}\ yr^{-1}$)
Aegean Sea	0.112 - 0.365
Southern Sardinia	0.06 – 0.13
Swiss Alps	0.004
French Alps	0.002
NE Spain	0.051
Corsica	0.12
Corsica	0.125
Central France	0.01
Crete	0.1 – 1.0
Crete	0.195
Pyrenees	0.30 – 0.39

Table 7

Actual soil erosion rates in Europe (tolerable rate < 1.0 t ha^{-1} yr^{-1}). For references, please see relevant sections in this paper.

Erosion type	Mean rates (t ha^{-1} yr^{-1})	Maximum rates (t ha^{-1} yr^{-1})	comment	Main factors
Rill, sheet erosion	0.1 - 8.8	23.4		Land use, soil cover, slope
Gullies	na	455		Climate, land use
Wind erosion	0.1 - 2.0	15		Soil type, soil cover, climate
Tillage erosion	3.0 - 9.0	na		Soil management
Slope engineering	na	454		Soil management
Crop harvesting	1.3 – 19.0	na	For a variety of crops	Crop type (Table 6); soil moisture content at time of harvesting
Cumulative mean soil erosion rates in tilled agriculture	3.0 - 10.0 3.2 - 19.8 4.5 – 38.8	na	Tillage only Water + wind + tillage Water + wind + tillage + crop harvesting	

na = not available