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Status, methods, drivers and effects on ecosystem services

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RECARE
Preventing and Remediating degradation of soils in Europe through Land Care

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2 SOIL EROSION BY WATER

Jacob Keizer, Hakan Djuma and Volker Prasuhn

2.1 *Description of soil erosion by water*

Soil erosion in general can be defined as a three-phase process that consists of: (i) the detachment of individual soil particles from the soil mass; (ii) their subsequent transport by an erosive agent; and, ultimately, (iii) their deposition when the erosive agent lacks sufficient energy for further transport (Morgan, 2005). In the case of soil erosion by water, both rainsplash and water running over the soil surface detach and then move the detached particles, but rainsplash is the most important detaching agent whereas running water is the principal transporting agent. The transport of soil particles resulting from the direct impact of falling raindrops is designated as rainsplash erosion, while the transport of soil particles by running water is commonly divided into interrill and rill erosion. Interrill erosion then refers to water running as a shallow sheet ("overland flow") and removing a relative uniform thickness of soil, whereas rill erosion refers to water running as concentrated flow and removing soil by "digging out" channels of increasing deepness and/or width. In turn, rill erosion is generally divided into rill and gully erosion depending on channel dimensions. A cross-sectional area of at least 1 ft^2 (Poesen, 2003) is a widely recognized criterion to distinguish gullies from rills. Poesen (2003) calculated that at larger scales around 80% of detachment/soil loss comes from gullies.

Not only water running over the soil surface as described above but also water moving laterally through the soil matrix in downslope direction ("interflow") can detach and transport soil particles, including as concentrated flow in macro-pores or subsurface pipes (Morgan, 2005). These subsurface erosion processes mainly occur in peatlands (Holden, 2005) as well as in areas where man-made subsurface drainage systems have been installed (Russel *et al.*, 2001).

Soil erosion appears to have been recognized by mankind since the early civilizations of China and the Mediterranean Basin (Morgan, 2005). Nonetheless, scientific research into soil erosion did not gain impetus till the 1920s and 1930s, with Hugh Hammond Bennett leading the soil conservation movement in the USA. In Western Europe, by contrast, the importance of soil erosion only started to be duly recognized from the 1970s onwards.

2.2 *State of the soil erosion by water*

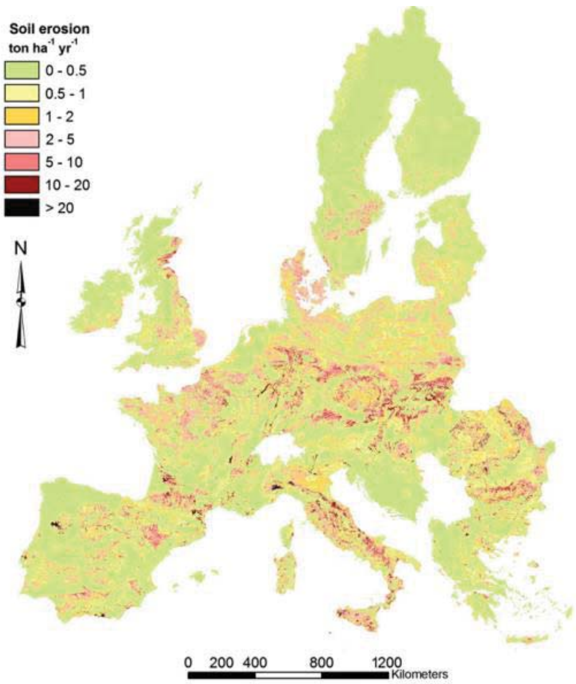

Figure 2.1: Maps of the risk of soil erosion by water across Europe based on erosion plot data (Cerdan et al., 2010).

The work done by Boardman & Poesen (2006) is so far the most comprehensive research into the extent, seriousness and impacts of soil erosion in Europe. It involved erosion experts from 33 European countries who collaborated to compile and analyse existing data and information at the national scale and/or from typical case studies, with a strong emphasis on field observations and measurements. These erosion data, however, are not always directly comparable, as there is a lack of harmonization among the different European countries on which methods, approaches and models to use over which spatial and temporal scales. Furthermore, European countries differ markedly in the amount of erosion data they have. Because of this lack of harmonised measurement data, soil erosion risk has frequently been used as a surrogate indicator in national as well as European-wide risk assessments. Risk assessment involves the identification of the risk and the quantification of the exposure to that risk (Jones *et al.*, 2004; Grimm *et al.*, 2002).

The risk of erosion by water has been assessed at the European scale using various models and expert-based approaches. The most recent attempts are those of Kirkby *et al.* (2004): applying the PESERA model, Cerdan *et al.* (2010): based on erosion plot data, Vanmaercke *et al.* (2012): based on sediment yield data, the OECD (2013), and Bosco *et al.* (2014): applying the eRUSLE model, whereas Grimm *et al.* (2002) and Jones *et al.* (2004) provided a list of earlier approaches and the corresponding maps. The former are described in more detail underneath, starting with those based on measurement data and with data collected over the smallest spatial extent. In addition, the latest OECD is briefly presented.

Table 2.1: Overwiev of recent estimates of the risk of soil erosion by water in European countries.

source	Cerdan *et al.*, 2010	Kirkby *et al.*, 2004	Bosco *et al.*, 2014	Panagos *et al.*, 2014	OECD 2013
indicator	Country-wise mean soil loss risk	Country-wise mean soil loss risk	Country-wise mean soil loss risk	Country-wise mean soil loss risk	area with mean soil loss risk > 11 ton ha^{-1} yr^{-1}
unit	ton ha^{-1} yr^{-1}	ton ha^{-1} yr^{-1}	ton ha^{-1} yr^{-1}	ton ha^{-1} yr^{-1}	%
estimates based on	erosion plots	PESERA	eRUSLE	EIONET	OECD
Austria	1.6	0.5	4.8	0.7	3
Belgium	1.4	1.1	2.3	3.7	9
Bulgaria	1.9	0.6	2.2	1.9	
Czech Republic	2.6	1.3			4
Denmark	2.6	2.3			
Finland	0.2				0
France	1.5	1.6			4
Germany	1.9	0.9	2.7	1.4	
Greece	0.8	5.8			20
Hungary	1.0	0.4			25
Ireland	0.5	0.2			
Italy	1.0	3.1	7.4	6.6	30
Latvia	1.3	0.1			
Lithuania	1.0	0.3			
Luxembourg	1.3	0.5			25
Netherlands	0.4	0.1	0.6	0.3	0
Norway	0.2				3
Poland	1.5	0.7	1.6	1.5	29
Portugal	1.2	4.6			
Romania	1.8	0.4			
Slovakia	3.2	1.3	2.3	1.0	55
Slovenia	1.2	0.9			38
Spain	1.0	2.4			28
Turkey	0.3				39
United Kingdom	0.9	0.3			17

(i) Erosion plot data (Figure 2.1): Cerdan *et al.* (2010) compiled data from 81 experimental sites in 19 countries, amounting to a total of 2,741 plot-years, and calculated mean inter-rill and rill erosion rates for the area in Europe covered by the CORINE database. The authors used correction factors for topography and soil properties to extrapolate the plot data to the European scale, producing a map with a 100 m resolution. The estimated inter-rill plus rill erosion rates were, on average, 1.2 ton ha-1 yr-1 for the whole CORINE-covered area and 3.6 ton ha-1 yr-1 for the arable lands within that area. These estimates were much lower than earlier estimates, as these earlier figures involved erroneous extrapolation of local plot measurements. Erosion rates were comparatively high (2–10 ton ha-1 yr-1) in the hilly loess areas of Western and Central Europe, and revealed marked spatial variation in the Mediterranean Zone, being high in many areas in Italy (Apennine slopes and Sicily) as well as in some areas in Spain (southern part of the Guadalquivir basin and the area around Zaragoza). Erosion rates also varied strongly for Europe as a whole, as 70% of the total erosion originated from 15 % of the territory. At the country level, the highest mean erosion rates were predicted for Slovakia, Denmark, Czech Republic and Italy (Table 2.1). Erosion rates further differed markedly between land covers. The highest rates

(ii) were estimated for vineyards (17.4 ton ha-1 yr-1), arable lands (3.6 ton ha-1 yr-1) and orchards (3 ton ha-1 yr-1), respectively, whereas all other land uses revealed mean values well below 1 ton ha-1 yr-1.

(ii) Sediment yield data: Vanmaercke *et al.* (2012) compiled annual sediment yield data for 1,794 catchments in Europe, which corresponded to at least 29,203 catchment-years of observations. They compared these data with annual erosion rates (n = 777) from runoff plots located at 187 study sites that were relatively well spread across Europe as well as with the above-mentioned map produced by Cerdan *et al.* (2010) and the PESERA map produced by Kirkby *et al.* (2004). The authors found that the sediment yield data and the runoff plot data indicated significantly higher soil loss rates than the two maps, even though sediment yields do not take into account that large proportions of eroded sediment may be deposited before reaching the catchment outlet. To the authors, this clearly demonstrated the importance of erosion processes other than inter-rill and rill erosion for catchment-scale sediment yields, in particular gully erosion, channel erosion, mass movements, and glacial erosion. These findings were later confirmed by De Vente *et al.* (2013). Thus, soil erosion by water is only one possible source of the sediments that leave a catchment outlet, as a result caution is needed when comparing soil erosion rates and sediment yields (Verheijen *et al.*, 2009).

(iii) PESERA model predictions (Figure 2.2): the Pan-European Soil Erosion Risk Assessment (PESERA) model is a process-based and spatially distributed model that was developed to estimate the risk of soil erosion by water across Europe (Kirkby *et al.*, 2004). The PESERA results were also selected by the OECD as basis for its agri-environmental indicator of soil erosion (IRENA fact sheet No. 23; EEA, 2006). According to PESERA, about 105 million ha or 17% of the total land area of Europe (excluding Russia) is subject to some degree of soil erosion risk. Furthermore, Europe can be divided in three zones w here erosion risk is significant: (i) a southern zone characterised by a severe risk of erosion by water; (ii) a northern loess zone with a moderate risk; and (iii) an eastern zone where the two prior zones overlap. Within all three zones, however, hot spots of soil erosion risk do occur. At the country level, Greece, Italy, Portugal, Italy and Spain stand out with the highest mean annual rates of soil erosion risk (Table 2.1). Spain is the country with the largest area subject to a high erosion risk, comprising southern and western Spain and covering 44% of the country's territory. Portugal ranks second, with one-third of its territory country revealing a high erosion risk. In Central and Eastern Europe, soil erosion risk is most widespread in Bulgaria and Slovakia, affecting some 40% of the territory of both countries.

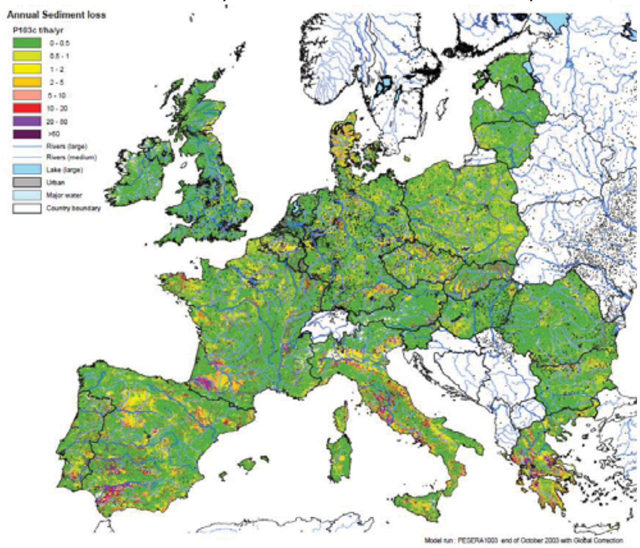

Figure 2.2: Maps of the risk of soil erosion by water across Europe based on PESERA model predictions (Kirkby et al., 2004).

(iv) eRUSLE model predictions (Figure 2.3): Bosco *et al.* (2014) presented a new, extended version of the Revised Universal Soil Loss Equation (RUSLE). The authors validated their eRUSLE predictions through comparison with national datasets as well as based on expert judgement. The eRUSLE results indicated that 130 million ha in the EU-27 countries are at risk of being affected by soil erosion by water and that this risk is moderate to high for about 14 % of the European territory. Almost 20% was subjected to soil loss in excess of 10 ton ha^{-1} yr^{-1} (EEA, 2012). Soil erosion rates exceeding 11 ton ha^{-1} yr^{-1}, defined as moderate to severe erosion by the OECD, were foreseen to affect just over 7% (= 115,410 km^2) of the cultivated lands (arable and permanent cropland) in the EU-24 (excluding Greece, Cyprus and Malta) (Jones *et al.*, 2012). The average rate of soil erosion by water across the EU-27 (excluding CY, GR and MT) was estimated at 2.76 ton ha^{-1} yr^{-1}; rates were higher in the EU-15 (3.1 ton ha^{-1} yr^{-1}) than in the EU-12 (1.7 ton ha^{-1} yr^{-1}), probably as the EU-15 includes the Mediterranean area where overall erosion rates were higher.

(v) OECD assessment: the OECD assessed soil erosion risk through questionnaires to the experts of the individual countries, using a standard table linking erosion risk to erosion rates (OECD, 2013). The OECD's table classifies soil erosion risk into five categories ranging from tolerable (< 6 ton ha^{-1} yr^{-1}) to severe erosion (> 33 ton ha^{-1} yr^{-1}). However, not all countries employed the class limits proposed by the OECD and, in particular, various countries use lower upper thresholds for the class of tolerable soil erosion. During the period 1990-2010, nine of the 20 European OECD member countries had more than 20% of their agricultural lands exposed to a moderate to severe erosion risk. These nine countries were Slovak Republic, Turkey, Slovenia, Italy, Poland, Spain, Luxembourg, Hungary and Greece (Table 2.1).

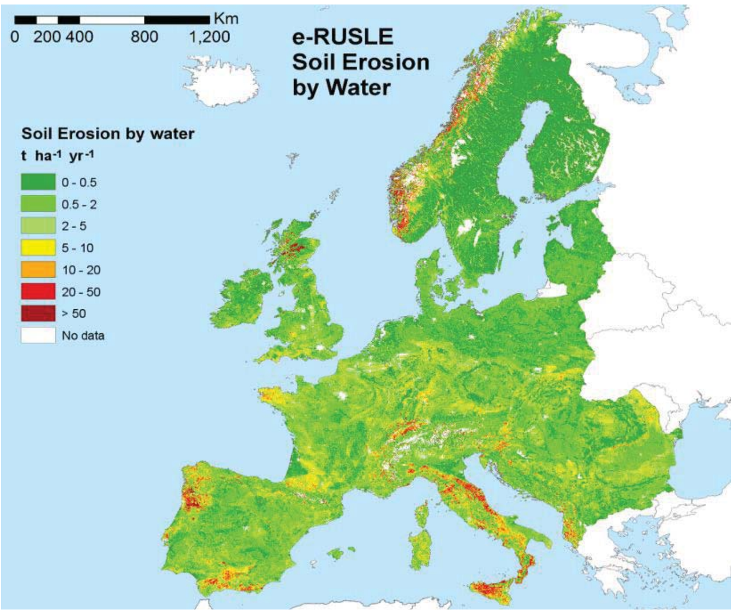

Figure 2.3: Maps of the risk of soil erosion by water across Europe based on eRUSLE model predictions (Bosco et al., 2014).

The results of the various erosion risk models and approaches that have been applied at the European-scale differ quite considerably. This relates to differences in modelling approaches, differences in model input data and their quality as well as to differences in the models' spatial and temporal resolutions. Model input data lacking sufficient quality and/or spatial resolution can result in substantial errors and uncertainties in model predictions. Also the geographical extents of the model-based assessments differ, depending on the countries that are being considered as European.

The fundamental importance of model input data is well illustrated by the following three examples.

Example 1: Panagos *et al.* (2014a) presented new values for the soil erodibility factor K based on a pan-European harmonised soil data set and taking into account stoniness. The countries with the lowest mean value for the K-factor (< 0.025 ton ha^{-1} h^{-1} ha^{-1} MJ^{-1} mm^{-1}) ranged from Portugal, Ireland, Denmark, Greece, Netherlands, United Kingdom, Estonia to Finland, whereas the countries with the highest mean value for the K-factor (> 0.032 ton ha^{-1} h^{-1} ha^{-1} MJ^{-1} mm^{-1}) included Belgium, Luxembourg, Czech Republic, Hungary and Slovakia.

Example 2: Panagos *et al.* (2014b) compared soil losses predicted by PESERA with the plot data compiled by Cerdan *et al.* (2010) as well as with the national data from eight countries collected through EIONET-SOIL. Overall, the PESERA figures were not only lower than the mean soil losses of the EIONET data set but also than those of the Cerdan *et al.* (2010) data set, except in the case of Italy.

Example 3: Hessel *et al.* (2014) applied the MESALES model (Modèle d'Evaluation Spatiale de l'ALéa Erosion des Sols) to three geographical areas using two distinct soil data bases. This resulted in noticeable differences in soil erosion risk, in spite of the fact that the risk estimates were on a semi-quantitative scale ranging from very low to very high.

Model assessment is often constrained by a lack of measurement data with the necessary spatial resolution, so that it is often impossible to determine which of the models is performing best. Nonetheless, it is widely recognised that a model such as RUSLE tends to overestimate soil losses. Furthermore, model-based estimates can be expected to overestimate soil erosion risk, since soil conservation measures are not taken into account. This is first and foremost due to the absence of EU-wide data on the application of practices such as sequential cropping, reduced tillage and strip tillage. By contrast, plot-based studies and models such as PESERA and eRUSLE may underestimate soil erosion rates, since they assess inter-rill and rill erosion but not (ephemeral) gully erosion. The existing measurements of gully erosion rates mainly concern the Mediterranean region of Europe. They revealed a huge variation, with figures ranging from 1 to 455 ton ha^{-1}

yr^{-1}, depending on rainfall and site conditions. Ephemeral gullies at four sites in Belgium were estimated to produce medium-term soil losses between 3.2 and 8.9 ton ha^{-1} yr^{-1} (Verheijen *et al.*, 2009).

The thresholds above which soil erosion should be regarded as a serious problem continue to be controversial, including because soil formation processes and rates seem to differ substantially across Europe (Bosco *et al.*, 2014). Nonetheless, direct measurements of soil formation rates are very scarce. Soil formation rates (by weathering) in Europe under current conditions are estimated to vary in between 0.3 and 1.2 ton ha^{-1} yr^{-1} (Verheijen *et al.*, 2009). At such slow rates of soil formation, soil losses exceeding 1 ton ha^{-1} yr^{-1} can be considered irreversible and unsustainable within a time span of 50-100 years (Jones *et al.*, 2004; Verheijen *et al.*, 2009). Soil losses ranging from 5 to 20 ton ha^{-1} yr^{-1} can have serious impacts, both at the site where the soil is lost and off-site in downstream flood zones and aquatic habitats. Soil losses of 20 to 40 ton ha^{-1} yr^{-1} by individual storms with a return interval of two or three years are measured regularly in Europe, whereas extreme rainfall events have been found to produce soil losses exceeding 100 ton ha^{-1} yr^{-1}. Such large soil losses can have catastrophic on-site effects as well as serious off-site consequences (Grimm *et al.*, 2002).

An alternative approach to modelling soil erosion by water is to represent the role of running water in an explicit manner, predicting the generation of runoff as well as its detachment and/or transport capacity. The blueprint for this approach was presented as early as 1969, by Meyer & Wischmeier (1969), but it was not implemented until more than a decade later, in the semi-empirical model of Morgan, Morgan and Finney (MMF); Morgan *et al.* (1984) as well as in the bulk of the physically-based models (e.g. CREAMS by Knisel, (1980), WEPP by Nearing *et al.* (1989), EUROSEM by Morgan *et al.* (1998) and PESERA by Kirkby *et al.* (2004). Due to their large data-demands, such models are difficult to apply at EU-scale.

2.3 Drivers/pressures

The factors controlling soil erosion are commonly divided into:
(i) erosivity of the erosive agent or its capacity to detach and transport soil particles;
(ii) erodibility of the soil or the inverse of the soil's resistance against the detachment and transport of its particles;
(iii) plant and litter cover; and
(iv) slope of the terrain (Morgan, 2005).

In the case of soil erosion by water, erosivity typically focuses on the detaching power of raindrops, ignoring that of running water, whereas erodibility usually refers not just to the soil's resistance to rainsplash and running water but also to the likelihood that water will actually be running over the soil surface. This conceptual framework is intimately linked to the Universal Soil Loss Equation (USLE) by Wischmeier & Smith (1978) which estimates annual soil losses from plots, fields and hillslopes as the product of the four above-mentioned factors. USLE does, in fact, include a fifth multiplication factor but specifically to predict how effective land management practices such as contouring and bench terracing are to reduce soil losses.

Climate drivers: Climate and, in particular, rainfall is the primary driver of soil erosion by water. Rainfall is not only the main agent of detachment of soil particles but also the principal source of water running over the soil surface (Morgan, 2005). In cold climate regions, however, also freezing-thawing cycles can play a key role in detachment, while snow melt can be an important additional source of runoff. The erosivity of rainfall is typically related to the kinetic energy of the raindrops striking the soil surface and, as such, calculated as a function of the intensity and duration of a rainfall event as well as of the mass, diameter and velocity of the raindrops. The measurement of these raindrop characteristics has long posed considerable challenges, at least till the development of disdrometers. Therefore, the kinetic energy of rainfall is typically estimated based on its relationship with rainfall intensity, often using relationships that are adjusted to local climate conditions. Nonetheless, these local relationships can reveal marked variations between and within individual rain storms, especially depending on their origins in terms of synoptic weather conditions (e.g. convectional vs frontal rain) and on wind speeds. The rainfall-runoff response of soils is typically explained as a function of the two main runoff generating processes. Infiltration-excess overland flow occurs when rainfall intensity exceeds a soil's so-called infiltration capacity or, in other words, the rate at which a soil can take in water that has accumulated at its surface. By contrast, saturation overland flow occurs when a soil's water storage capacity has been exceeded, typically due to prolonged antecedent rainfall.

Climate also affects soil erosion by water indirectly, through its impacts on soil properties, soil cover by (whether of (semi-natural) vegetation or of croplands and sown pastures) as well as interactions between these impacts. For instance soil properties strongly determine a soil's infiltration and storage capacities and, thus, its hydrological response. This includes properties that tend to be time-invariant such as soil texture, soil depth and the presence of impermeable layers as well as properties that vary markedly in time such as the presence of a surface crust, soil aggregate stability, soil water repellency or groundwater level. In the case of soil hydrological properties, the indirect role of climate is well-illustrated by the importance of dry spells in the formation of a structural surface crust or in the appearance and severity soil water repellency. In the case of soil properties determining erodibility, the indirect role of climate can be exemplified by the marked increase that freezing and thawing can produce (Coote *et al.*, 1988). In the case of plant cover, the indirect role of climate is perhaps most obvious in semi-arid and arid regions, where the protective cover provided by plants against rainsplash tends to decrease with increasing aridity.

Human drivers: Arguably, human activities have become the most important driver of soil erosion by water in modern times and places, especially those witnessing strong increases in population and/or rapid advances in slope- and landscape-engineering capabilities. The concept of a new geological age – the Anthropocene – has become a topic of serious debate (Zalasiewicz *et al.*, 2011), including based on the observed and modelled impacts of humans on sediment flux (Syvitski and Kettner, 2011). The paramount importance of human activities in soil erosion by water is also evidenced by the commonly-made distinction between "natural" (or "geological") erosion rates and human-induced, "accelerated" erosion rates (Verheijen *et al.*, 2009). In turn, the concept of accelerated erosion is closely linked to that of tolerable soil erosion, as (changes in) land management are implied in avoiding to exceed "any actual soil erosion rate at which a deterioration or loss of one or more soil functions does not occur" (Verheijen *et al.*, 2009).

Human activities can accelerate soil erosion by water in a wide variety of ways but always in an indirect manner, by provoking changes in especially the first three of the erosion-controlling factors listed at the beginning of the present section. Some examples will follow to illustrate this for each of these three factors. Rainfall erosivity is expected to increase under likely climate change scenarios, especially in Mediterranean climate regions as autumn rainfall events become more intense. Erosivity of surface runoff can be enhanced by ploughing, leading to concentration of overland flow in furrows and to reduction of micro-topographic variations and, thereby, of the resistance to flow. Overland flow generation can be enhanced by compaction of the topsoil in the wheel tracks of heavy machinery, provoking a reduction in infiltration capacity. Soil erodibility can be enhanced by ploughing, both directly by destroying soil aggregates and indirectly by reducing soil organic matter content and, thereby, the formation of new aggregates. Soil cover will typically be less in croplands than in the original vegetation, leading to an overall reduction in the protection of the soil surface against rainsplash as well as in the resistance to overland flow.

In contrast, human activities can also reduce accelerated and even natural rates of soil erosion by water, through so-called soil conservation techniques (Morgan, 2005). Soil conservation techniques can be divided in three groups: agronomic, vegetative, structural and management. In a nutshell, agronomic measures target plant cover, soil management measures aim at soil erodibility and infiltration capacity, and mechanical measures are directed towards terrain shape and drainage network, often involving engineering solutions. Typical examples of these three measures are mulching with organic residues, contour-tillage and terracing, respectively. Bench terraces have existed for over 2000 years and from ancient civilizations across the globe.

Socio-economic-politics drivers: land use and land management, which are influenced by the socio-economy and policy, can have an important impact on soil erosion by water (Schwilch *et al.*, 2012). Nonetheless, more detailed assessments of the impacts of specific socio-economic factors and of past and present agricultural, forestry and soil conservation legislation and plans seem to be lacking. Such assessments also have a requisite that is typically lacking: adequate erosion monitoring schemes. For example, the common-grounds in the mountains of Portugal were afforested on a large scale by the "Estado Novo" ("New State") following the 1930s, among others on the official grounds of preventing the silting-up of rivers and the truncation of soil profiles (Estevão, 1983). The effectiveness of this afforestation plan in terms of erosion reduction, however, cannot be easily quantified, as no erosion measurements were carried out before and after afforestation and/or to compare afforested and non-afforested lands.

With respect to political drivers, one notable example is the Norwegian political decision to subsidise farmers who levelled their fields in the 1970's. Land levelling causes very high erosion in Norway (Lundekvam *et al.*, 2003) resulting in new guidelines and regulations prohibiting land levelling without specific permission.

2.4 Key indicators of soil erosion by water

The European Environmental Agency (EEA, 2000) used the driving force–pressure–state–impact-response (DPSIR) framework (which also underpins the approach by RECARE) to identify a list of agri-environmental indicators of soil erosion by water that were considered relevant to pan-European policy making. This section will focus on the two indicators of the state of soil erosion and the combined indicator of state and impact, i.e. area affected by soil erosion (in km^2), extent of area affected by soil erosion (in %), and magnitude of soil erosion or sediment delivery (in tons), respectively.

Gobin *et al.* (2004) critically reviewed the EEA indicators in terms of policy relevance and utility, analytical soundness (including data availability) and measurability, and based on this analysis, provided recommendations. In relation to the two state indications, the authors recommended the implementation of a combined measurement-modelling-expert approach, considering that: (i) erosion measurements by themselves are unsuitable for European-wide assessments but indispensable to validate model-based predictions of the risk of erosion across Europe under present environmental and land-cover/use conditions; (ii) expert knowledge is required for verification of regional-scale assessments of actual erosion risk. In relation to the combined state-impact indicator, Gobin *et al.* (2004) stressed that measurements of sediment yield at catchment outlets or of sediment deposition in lakes/reservoirs provide at best an indirect validation of catchment-scale model predictions. The main reasons according to Gobin *et al.* (2004) are that the origin(s) of the sediments are mostly uncertain (e.g. due to riverbank or channel erosion) and that sediment yield/deposition data typically lack the required accuracy. In addition, sediment delivery ratios (i.e. the proportions of the sediments eroded from the land surface that discharges into a river) are estimated to vary widely, from less than 5 to 90% (Walling 1983).

Measurement-based indicators of soil erosion by water can be divided into two broad classes, those referring to the actual transport process of soil particles and therefore expressed as ton ha^{-1} yr^{-1} (or equivalent unit); and those related to changes in land surface or soil characteristics resulting from soil erosion (Morgan, 2005). The transport process indicators encompass the amount of particles transported by rainsplash (splash erosion) or running water, as sheet flow (inter-rill erosion) and/or concentrated flow (rill and gully erosion). The changes in land surface or soil characteristics resulting from soil erosion indicators include differences in contents of radioactive tracers, differences in ground levels, altered soil profiles (e.g. truncated profiles without A-horizon), and the presence/extent of so-called erosion features such as pedestals, rills, gullies and recently deposited sediments. They reflect cumulative erosion processes and, thus, require a well-defined time basis to be converted into the same measurement units as the former indicators. For example in the case of rills, this can be achieved by measuring their extent and dimensions at regular intervals, in combination with measurements of the bulk density of the removed soil.

2.5 Methods to assess the status of soil erosion by water

Although models are indispensable for assessing the status of soil erosion by water for larger areas and for larger time frames (both past and future), this section will be limited to the methods used for measuring erosion as such. Measurements are a prerequisite for the validation of model predictions.

The transport of soil particles by rainsplash (splash erosion) can be measured in the field by splash boards as well as by funnels and cups of various designs, which are typically less than 15-20 cm in diameter (Morgan, 2005). Rainsplash can be measured under natural rainfall conditions as well as artificial rainfall conditions, for which a wide range of portable rainfall simulators can be employed (e.g. Iserloh *et al.*, 2013).

The transport of soil particles by sheet flow (inter-rill erosion) can be measured in the field by using plots that are sufficiently small to avoid that the overland flow occurs as concentrated flow. Arguably, inter-rill erosion has mainly been studied by means of field rainfall simulators, applying rainfall with typically high intensities but low kinetic energies to bounded plots of small dimensions rarely exceeding 1 m^2. Nonetheless, these so-called micro-plots have also been employed to measure inter-rill erosion under natural rainfall conditions, including for assessing the representativeness of the results obtained under simulated rainfall conditions.

The transport of soil particles by combined sheet and concentrated overland flow (inter-rill + rill erosion) can be measured in the field by using appropriately sized plots, typically more than 10 m long (Morgan, 2005). A widely-used approach is the so-called “Wischmeier” plot. It is standard 22 m long and 1.8 m wide, bounded by sheets of, for example, metal that stick out 150-200 mm above the soil surface, has a collecting through or gutter at the bottom end where the runoff, with its sediments, is channelled into one or more collecting tanks, depending on runoff volumes. Nonetheless, bounded plots of other designs and especially smaller dimensions have also been frequently used. An alternative design consists of unbounded plots, avoiding edge effects and possible sediment exhaustion but introducing uncertainty about the contributing area. Unbounded plots such as Gerlach troughs and sediment fences have especially been used for measuring runoff and/or sediment losses at larger spatial scales such as agricultural fields, permanent crop or tree plantations or entire hillslopes, i.e. including by gully erosion.

The transport of soil particles beyond the hillslope scale can be measured at the outlets of catchments with a hydrometric station, where typically water level is recorded continuously and sediment yield is estimated by multiplying the streamflow's suspended sediment concentration by discharge. Water level recordings are converted to discharge estimates based on the stage-discharge curve at the catchment outlet, which, in turn, is derived from discharge measurements that should ideally cover the full range of water levels. Hydraulic structures such as weirs and flumes can greatly reduce the need for repeating discharge measurements, especially if the channel section at the outlet is subject to marked changes. The suspended sediment concentration can be determined through runoff samples collected throughout runoff events, either manually or by means of one or more automatic samplers, or through turbidity recordings. While turbidity sensors have the advantage of providing continuous estimates of suspended sediment concentration, the quality of these estimates does depend critically on the relationship between the two parameters.

The status of cumulative soil erosion can be described through a survey of (selected) erosion features, mapping either their presence/absence or their extent and dimensions. These features can include pedestals (evidencing rainsplash erosion), soil profile characteristics (e.g. truncated profiles without A-horizon or profiles with buried A-horizons), rills, gullies and sediment depositions. A simple method of estimating the cumulative volume of soil removed by rill or gully erosion on a slope is to determine the cross-sectional area of the rills/gullies along a series of transects of 20-100 m long across the slope (Morgan, 2005). The bulk density of the removed soil is then needed to estimate the sediment losses by weight. A similar approach can be used to estimate the volume and weight of sediments recently deposited on hillslopes or at footslopes, measuring their length, width, depth, and bulk density. More precise estimates of the volume of removed soil/deposited sediments can be obtained by classical topographic methods, also depending on the dimensions involved. This is of particular relevance in the context of repeated surveys. Terrestrial and aerial photogrammetry, terrestrial and airborne 3-D laser scanning as well as satellite imagery can equally be useful for (repeated) mapping of erosion features, as long as the precision of the resulting digital terrain models (DTM) match the dimensions of the features and the changes therein. A dense cover of high-stature vegetation can, in this respect, be a constraining factor.

Changes in ground level can be estimated not only from sequential DTMs, as mentioned above, but also through erosion pins as well as by means of an erosion bridge (Morgan, 2005). Typically, erosion pins are installed in large numbers, and the distance between the pin's head and a washer (originally placed at the soil surface) measured at regular intervals. An erosion bridge is a device that allows the repeated measurement of the distance to the soil surface from a fixed height at fixed points along a fixed transect. Sediment pins have been used to measure the sedimentation rate in the irrigated fields particularly in spate irrigation systems where farmers divert flood water that contains soils and nutrients to adjacent irrigable fields (Tesfai and Sterk, 2002). Also changes in the level of sediments in ponds, reservoirs and lakes can be used to estimate sedimentation rates at the catchment scale. Besides sedimentation rates, the efficiency to trap these sediments must be estimated to arrive at sediment yields. Trap efficiency is particulary difficult to measure with sufficient accuracy to avoid large uncertainties in the resulting sediment yields (Morgan, 2005).

Differences in the concentrations of radioactive isotope tracers in soil profiles can provide not only qualitative information on the patterns of soil erosion/deposition in a landscape over time depending on the decay rate of the isotope, but also estimates of soil erosion rates when combined with conversion models such as the proportional approach or the mass balance model (Morgan, 2005). The most commonly radioactive isotope tracer in erosion studied has been ceasium-137. Among innovative tracers, magnetic iron oxides attached to soil particles deserve special mention as they can be measured easily, cheaply, and directly in the field (Guzman *et al.*, 2013).

2.6 Effects of soil erosion by water on other soil threats

Soil erosion by water can have an important impact on other soil threats especially for decline in soil organic matter (SOM), flooding risk, and decline in soil biodiversity.

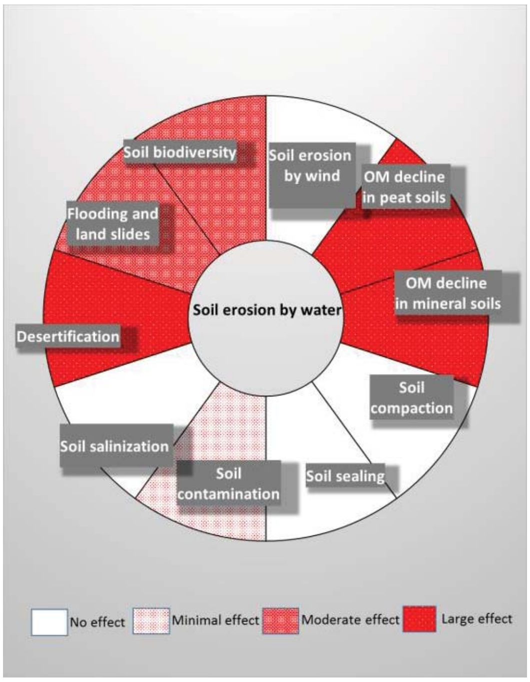

Figure 2.4: Effects of soil erosion by water on other soil threats. Red is negative

Soil erosion by water can reduce the SOM stocks of soils directly as well as indirectly. The direct effect involves the transport, by running water, of organic compounds in dissolved and especially particulate form, aggregated to mineral soil particles. This effect can be especially relevant in recently burnt areas to the extent that an ash layer is present and, arguably, that it resulted from the burning of the litter layer as opposed to the above-ground standing biomass. In the case of mineral soils, the reduction in SOM stock is further aggravated when inter-rill erosion is the predominant process, as organic matter contents tend to decrease with soil depth. The indirect effect of soil erosion results in denudation of the upper soil layer, exposing of SOM at greater soil depths to conditions propitious to its decomposition.

Soil erosion by water can also enhance flooding risk directly as well as indirectly. The same overland flow that causes soil erosion will typically constitute an important component of the hydrological response of catchments during flooding events. This is especially true for flash floods associated to infiltration-excess overland flow and less so for regional-scale floods associated to saturation overland flow due to a larger baseflow component. Flooding risk can further be increased by the silting-up of the channel network resulting from the deposition of soil eroded during prior erosion events. The sediment load carried by the water also increases the volume of the flood, and that results in larger damage off-site effects of erosion.

Soil erosion by water is often regarded as one of the most intense and widespread desertification processes (e.g. Rubio & Bochet, 1998; Vanmaercke, *et al.*, 2011). This has lead to the use of various desertification indicators that are related to soil erosion.

Eroded sediment can contain contaminants (agricultural or other) that cause contamination downstream where the sediment is deposited. Furthermore, sediment itself is also considered contamination by some. Situation becomes critical if highly contaminated sites are eroded, such as mine-spills.

Soil erosion by water can result in direct losses of soil biodiversity through the removal of soil flora, fauna and micro-organisms in the running water. This has been demonstrated for nematodes as well as seeds. Soil erosion by water can also lead to losses in soil biodiversity in an indirect manner, by changing the environmental conditions of the soil habitat, for example in terms of SOM as mentioned above.

2.7 Effects of soil erosion by water on soil functions

Soil erosion by water can affect the soil function of food and other biomass production both directly and indirectly. Possible direct effects are the removal of seeds by runoff and damage to above- and below ground plant organs. Possible indirect effects can be related to plant growth itself such as reduced rooting space for support, reduced available soil water and reduced soil nutrient pool, or to land management operations such as the removal of recently applied agrichemicals and additional efforts required to fill-up rills or circumvent gullies.

Soil erosion by water can have negative consequences for a soil's capacity for storage, filtering, buffering and transformation. In the case of storage and buffering, these consequences would seem to depend fundamentally on the net reduction of soil depth or, in other words, the difference between soil loss and soil accretion relative to the total soil stock. In the case of filtering and transformation, however, the impacts of soil erosion would seem to depend first and foremost on how important the soil layer that is being eroded is for the respective filtering or transformation process.

Soil erosion by water can be expected to have important implications for the soil function of biological habitat and, possibly, also that of gene pool if the removal of an organism by runoff is significant in terms of its existing population. The habitat effect would seem to depend strongly on the degree to which the organism depends on the topsoil for its habitat.

Soil erosion by water and especially gully erosion can affect the soil function of physical heritage by the resulting changes in the aspect of the landscape. It can also affect the function of cultural heritage through the removal and re-deposition of archeological artifacts as well as through the burial of archeological artifacts under sediments eroded upslope or upstream.

Soil erosion by water can have major consequences for a soil's function as a platform for man-made structures, either through the removal of the soil underneath these structures or through the deposition of eroded sediment again or on these structures.

Soil erosion by water can play an important role in the provision of raw materials. This is well-illustrated by sands accumulated in river beds which are exploited for civil construction purposes.

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