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Towards constraining the magnitude of global agricultural sediment and soil organic carbon fluxes

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ABSTRACT: Reliable quantitative data on the extent and rates of soil erosion are needed to understand the global significance of soil-erosion induced carbon exchange and to underpin the development of science-based mitigation strategies, but large uncertainties remain. Existing estimates of agricultural soil and soil organic carbon (SOC) erosion are very divergent and span two orders of magnitude. The main objective of this study was to test the assumptions underlying existing assessments and to reduce the uncertainty associated with global estimates of agricultural soil and SOC erosion. We parameterized a simplified erosion model driven by coarse global databases using an empirical database that covers the conterminous USA. The good agreement between our model results and empirical estimates indicate that the approach presented here captures the essence of agricultural erosion at the scales of continents and that it may be used to predict the significance of erosion for the global carbon cycle and its impact on soil functions. We obtained a global soil erosion rate of 10.5 Mg ha^{-1} y^{-1} for cropland and 1.7 Mg ha^{-1} y^{-1} for pastures. This corresponds to SOC erosion rates of 193 kg C ha^{-1} y^{-1} for cropland and 40.4 kg C ha^{-1} y^{-1} for eroding pastures and results in a global flux of 20.5 ($\pm$10.3) Pg y^{-1} of soil and 403.5 ($\pm$201.8) Tg C y^{-1}. Although it is difficult to accurately assess the uncertainty associated with our estimates of global agricultural erosion, mainly due to the lack of model testing in (sub-)tropical regions, our estimates are significantly lower than former assessments based on the extrapolation of plot experiments or global application of erosion models. Our approach has the potential to quantify the rate and spatial signature of the erosion-induced disturbance at continental and global scales: by linking our model with a global soil profile database, we estimated soil profile modifications induced by agriculture. This showed that erosion-induced changes in topsoil SOC content are significant at a global scale (an average SOC loss of 22% in 50 years) and agricultural soils should therefore be considered as dynamic systems that can change rapidly. Copyright © 2012 John Wiley & Sons, Ltd.

KEYWORDS: global soil erosion; soil carbon fluxes; global carbon cycle

Introduction

Under natural conditions, soil systems are considered to be sustainable as the loss of soil material by erosion is approximately balanced by the production of soil as a result of weathering (Montgomery, 2007; Verheijen *et al.*, 2009). A shift from natural to agricultural land use removes the protective natural vegetation and this typically increases soil erosion by one to two orders of magnitude (Montgomery, 2007), thereby accelerating physical erosion well over equilibrium levels. Research on the implications of accelerated erosion has traditionally focused on on-site nutrient and crop productivity losses (Bakker *et al.*, 2004) and erosion effects on water quality (Lal, 2007a). More recently, the erosion-induced changes in bio-geochemical cycles (Kutsch *et al.*, 2010) have received considerable attention, in particular resulting C flows between soils and the atmosphere.

The substantial human impact on soil resources is raising global sustainability issues (Pimentel, 2000; Lal, 2007a; Kummerer *et al.*, 2010) and thus reliable quantitative data on the extent and rates of soil erosion are needed to understand how agricultural ecosystems can maintain their delivery of critical services such as food production (Den Biggelaar *et al.*, 2004; Pimentel 2006) and influence global biogeochemical cycles (Lal, 2007b; Quinton *et al*. 2010). For example, several global assessments of the influence of erosion processes on the carbon cycle suggest a net uptake or sink of carbon but results are very divergent and range between 0.1 and 1.0 Pg C y^{-1}. Although it is clear that there is insufficient understanding of the interactions between erosion and carbon cycling at the process level (Liu *et al.*, 2003), this wide range also reflects uncertainties associated with estimates of the magnitude of agricultural erosion (Boardman, 2006; Quinton *et al.*, 2010).

The quantification of agricultural soil erosion and resulting C fluxes at a global scale is a difficult task because the temporal and spatial variability of soil erosion is extremely high. Soil erosion processes occur at very fine scales in heterogeneous patches on the landscape and their response to environmental conditions varies between locations. When moving to larger spatial scales, the quantification of landscape heterogeneity is therefore crucial but large information gaps exist. These gaps are due to the fact that spatially comprehensive methods, such as remote sensing, are currently lacking for examining and synthesizing erosion processes over the Earth's surface (Stallard, 1998). Often, global estimates are therefore based on the extrapolation of results derived from soil erosion experiments at the plot scale. Using these methods, existing estimates of agricultural soil erosion range between 24 and 120 Pg y^{-1} and 0.5–3.7 Pg C y^{-1}, respectively (Table I). Several factors, however, may introduce substantial uncertainty in these estimates. First, it has already been shown that the experimental data used in these estimates are derived from only a few observations which may be strongly biased towards steep slopes, bare soil conditions, extreme rainfall events and hence high rates of erosion (Cerdan *et al.*, 2010). Second, although global model applications may provide an indication of areas at risk, they have to rely on coarse-scale input data which are not compatible with the scales at which erosion models were parameterized. The fact that previous assessments based on global models remain unverified, mainly due to the lack of model-testing data at larger spatial scales (i.e. regions or continents) further adds to the uncertainty.

Implicit in the assumption that erosion plot data can be extrapolated to global scales is that the experimental conditions are representative for global agricultural land. Similarly, global estimates based on the application of hillslope scale erosion models assume that small-scale processes do not require re-parameterization when used at the global scale. The main objective of this study is therefore to test these assumptions underlying current assessments and to reduce the uncertainty associated with global estimates of accelerated soil and organic carbon erosion. To do this, we present a hybrid approach: we parameterized a simplified erosion model driven by coarse global databases but that was first calibrated and validated with an empirical database that covers large spatial and temporal scales and that is consistent with observations at the hillslope scale. First, we used the NRI database (USDA, 2000) to identify the key controls on accelerated erosion at the continental scale. This spatial database presents estimates of soil erosion across the conterminous USA representing a wide range of climatic, soil, topographic, cropping and management conditions. In a second step, we developed and validated a mechanistic model that describes quantitatively the relationships between erosion and the key controlling factors. We used the results of the NRI to parameterize our model and compared the results with the original NRI estimates. Third, we evaluated our estimates, which are based on coarse but global-scale datasets, against independent estimates derived from high resolution and high precision datasets for Europe, South Africa and Australia. As such, we explicitly address scaling issues and fine-scale processes operating at hillslopes. Finally, we applied our model to estimate global fluxes of sediment and organic carbon on agricultural land and we discuss the global significance of accelerated carbon erosion as well as uncertainties in relation to previous estimates.

Material and Methods

NRI

The NRI is a statistical survey of land use and natural resource conditions and trends on US non-Federal lands (Nusser and

Table I. Comparison of global erosion estimates based on different methods

Source	Type of erosion	Method	Estimated soil or sediment [Pg y^{-1}]	Estimated C eroded [Pg y^{-1}]
Oldeman *et al.*, 1991	Total erosion	Glassod survey, Model extrapolation	50.0	-
Lvovich *et al.*, 1991 (recalculated by Lal 2003)	Total erosion	Extrapolated suspended sediment	393.0	-
Milliman and Syvitski, 1992	Large river fluxes	Plot river sediment data, model extrapolation	16.0–24.0	-
Myers, 1993	Total erosion	Extrapolation of plot data	75.0	-
Pimentel *et al.*, 1995	Erosion on agricultural land	Extrapolation of plot data	73.5	-
Stallard, 1998	Total erosion	Model extrapolation of different sediment yield scenarios	23.7–64.9	0.45–1.23
Hooke, 2000	Construction and agricultural practices	Erosion data per capita, extrapolation using population figures	120.0	-
Yang *et al.*, 2003	Total erosion and human increase of soil erosion	Modified RUSLE based simulations	132.7 Soil erosion increase by humans 60%	-
Walling and Webb, 1983	Total erosion	Extrapolation of river sediment estimates	201.0	-
Lal, 2003	Total erosion	Extrapolation of river sediment estimates	201.0	4.0–6.0
Berhe *et al.* 2007	Erosion on agricultural land	Extrapolation of plot data, taken from Pimentel 1995	75.0	1.1–3.7
Ito, 2007	Water erosion	Modified RUSLE modelling approach	172.2	1.6
Van Oost *et al.*, 2007	Pasture and Cropland erosion	Erosion plot data, modified RUSLE for local extrapolation	33.4 ± 4.4	0.5 ± 0.06
Wilkinson and McElroy, 2007	Erosion on agricultural land	Plot erosion data, modeled extrapolation	28.1–39.9	-

Goebel, 1997; USDA, 2000). Data are available for several years since 1982 but here we used the results of the survey conducted from July 1997 to October 1998 as this covers the same period as our land use datasets (see further). This extensive database, with 417357 entries for US agricultural land, is to our knowledge the most detailed database describing land use, crop type and factors controlling soil erosion for crop- and pastureland that covers a large region (i.e. the conterminous USA). For each observation point, water erosion (sheet, interrill and rill erosion) was estimated using the widely used USLE (Wischmeier and Smith, 1978). The USLE is an empirical equation that predicts the annual average, long-term water erosion as the product of a rainfall erosivity factor (*R*), a soil erodibility factor (*K*), a slope length factor (*L*), a slope steepness factor (*S*), a crop cover factor (*C*) and a management factor (*P*). In addition to soil erosion estimates, the database also provides estimates for each USLE factor separately based on field observations and information on land cover and use. In our analysis, we spatially aggregated the NRI estimates to two levels: counties and HUC4 watersheds (Seaber *et al.*, 1987), resulting in 2755 and 211 observations points with a mean surface area of *c.* 2900 and 43 000 km^2, respectively. We performed a stepwise linear multiple regression analysis (Forthofer *et al.*, 2006) on the NRI database to identify the key factors controlling soil erosion variability at the continental scale.

Estimating erosion factors with global databases

In our analysis, high precision and fine resolution NRI data were compared with estimates derived from coarse, but global scale, databases. Here, we describe the global databases that are used. Land use data was derived from the gridded HYDE 3.1 dataset (Klein-Goldewijk *et al.*, 2010) which represents the fraction of agricultural land (both cropland and agriculturally used grassland, i.e. sum of pastures and rangelands) with a resolution of 5 by 5 arc-minutes for the year 2000. While a direct comparison between the NRI and HYDE databases can be performed for croplands, this is not the case for grasslands. Erosion related estimates in the NRI database only cover pastures and not rangelands; this prohibited a meaningful spatial correlation with HYDE estimates. As we will discuss below, this limited our analysis and estimates for global pastures. Global estimates for the *R* factor were calculated at a resolution of 5 by 5 arc-minutes from the annual precipitation (Hijmans *et al.*, 2005) using the empirical relationship between annual rainfall and rainfall erosivity presented by Renard and Freimund (1994). We used the GTOPO elevation model (USGS, 1996; Gesch *et al.*, 1999) to derive slope estimates at a 5 arc-minute resolution using the methods described in Van Oost *et al.* (2007). The USLE slope steepness factor was then calculated using the continuous function of Nearing (1997). We used the global crop type abundance map of Leff *et al.* (2004) at a resolution of 5 by 5 arc-minutes to estimate *C* factors. The NRI point estimates were used to calculate mean *C* factor values for all major US crop types and a weighted mean *C* factor was then calculated for each grid using crop type abundance. Our analysis showed that soil erodibility (*K*), slope length (*L*) and management (*P*) did not contribute much to explaining NRI-based variability in soil erosion at the continental scale (see below), and therefore, we did not include it in our analysis. For each erosion factor derived from global databases, we proposed correction functions by minimizing the difference between NRI- and global database-derived USLE factors for both cropland and grasslands.

Model implementation

After identifying the key controls on soil erosion at the continental scale and scaling global databases, we calculated erosion rates for cropland on a 5 by 5 arc-minutes grid using the following formula:

$$E_{wc} = S_r * R_r * C_r * E_{NRIref} \tag{1}$$

where E_{WC} is the estimated average soil loss per year on cultivated cropland by water erosion (Mg ha^{-1} y^{-1}), S_r is the local *S* factor divided by the NRI derived mean for the US, R_r is the local *R* factor divided by the NRI derived mean for the US, C_r is the local *C* factor divided by the NRI derived mean for the US and E_{NRIref} is the average annual erosion rate for US cultivated cropland for 1997 normalized to a slope of 9% (Mg ha^{-1} y^{-1}).

This approach ensured that the mean erosion estimate for the US derived from global databases equals the NRI estimate. Using the HYDE cropland area (L_c, km^2), E_{WC} was then converted into an erosion flux (F_{WC}, in Mg y^{-1}). The same approach was implemented for erosion on agricultural grassland (E_{WAG} and F_{WAG}). For grasslands, C_r was not derived from global databases due to the lack of data but was reduced by 42%, relative to croplands, conform to the findings of Yang *et al.* (2003) for grassland *C* factors and assuming an equal distribution of sparse and dense grassland on eroding pastures.

We also considered the effect of rock fragments on erosion processes in stony soils (Poesen *et al.*, 1994). In our global application, we took into account the protective effect of rock fragments against splash erosion and reduced erosion by 80% for stony soils, which is consistent with field observations (Poesen *et al.*, 1994). Based on the average gravel content reported for WISE 1.1 (Batjes, 2006) map units dominated by litho- and regosols, stony soils were identified as having gravel content above 12%. Using the WISE 1.1 (Batjes, 2006) soil type distribution on a 5 by 5 arc-minutes grid and the WISE 3.1 (Batjes, 2008) spatial soil profile database with information on gravel content from more than 10 000 soil profiles, a global map reflecting the protection by stones against splash erosion has been created. This analysis was performed by only considering profiles that were identified as agricultural soils. In our estimates for pasture and rangeland erosion, only environments where vegetation is sparse and which are subjected to accelerated erosion when grazed were considered. This approach is consistent with observations that erosion rates on grasslands in humid, temperate regions are very low (Cerdan *et al.*, 2006). We did this by only taking into account areas where the potential vegetation is grassland or steppe as described in the ISLSCP Potential Vegetation Types database (Ramankutty and Foley, 1999). We then applied our model to the global scale at a 5 by 5 arc-minutes resolution.

Using the soil carbon content (%), the amount of cropland and pasture/rangeland soil erosion was then converted into carbon erosion. The estimated average percentage of SOC in the toplayer (upper 20 cm) of cropland and pasture/rangeland was taken from the WISE 3.1 database (Batjes, 2008), and only those profiles that were identified as agricultural soils were considered.

Uncertainty assessment

In order to constrain the uncertainty associated with our estimate, we adopted the following approach. First, we evaluated our estimates by comparing them with large-scale estimates of water erosion reported in the literature for cropland outside the US. These observations are therefore outside the data cloud where our model has been calibrated. Secondly, we conducted an analysis to identify the potential error of key parameters and total model error using the NRI dataset as a validation set following the methodology described in Van Rompaey and Govers (2002). The error of the model output associated with the input data was assessed by Monte Carlo analysis. The model was run 100 times for each HUC4 watershed with stochastically generated input layers, where for each input parameter (i.e. *R*, *S* and *C*) a probabilistic distribution was assessed. The error on each parameter was considered to have a normal distribution with a zero mean and a standard deviation of 1.5 times the observed standard deviation for each HUC4 watershed. The latter was extracted from the NRI database. The relative root mean square error (RRMSE) of the model output was then calculated by comparing the 100 model estimates for total erosion with the NRI-derived estimates. In addition, we performed an analysis to assess the model error. We derived three simplified models from the full model (see Equation (1)) where one factor was kept constant. For each simplified model, the spatially distributed input was replaced by a single constant value. The latter was considered to be the mean value of the eliminated factor. The results of this analysis were then interpreted in terms of required model complexity. This analysis was performed using cropland water erosion fluxes. However, we use the resulting uncertainty estimate also on estimates for tillage erosion and pasture water erosion, as the lack of similar data for these types of erosion prohibits a comparable separate assessment.

Results

Identification of key factors controlling erosion in the US

We found that slope steepness and rainfall/runoff erosivity are the most important factors controlling erosion rates on cropland at the continental scale (Table II). About 75% of the water erosion variability can be explained by rainfall and slope alone, while other factors such as slope length, support practice, soil erodibility and crop type contribute little and slope length and management factor are not significant. In contrast to croplands, rainfall erosivity plays a minor role for pastures while up to 64% of the water erosion variability is explained by slope steepness alone. Rainfall erosivity and slope steepness, and to a lesser extent crop type, are thus the primary controls on soil erosion by water at the continental scale. If the variability of these key factors can be captured by global databases, they can be used to provide estimates of soil erosion variability at larger spatial scales.

Characterizing *R*, *S* and *C* with global databases

After aggregation at the HUC4 watershed level, we compared low resolution and low precision rainfall erosivity, slope steepness and crop factor estimates derived from global databases with high resolution and precision estimates from the NRI database.

Rainfall erosivity factor: For both land use types good spatial correlations between NRI and global database estimates ($R^2=0.80$, $P<0.01$) were observed (Figure 1). The rainfall erosivity was underestimated by 2% and 52% for cropland and pastures, respectively, and we corrected for this systematic bias.

Slope steepness factor: The comparison for cropland indicated a substantial overestimation of cropland slopes in watersheds which are situated in mountainous regions. This can be explained by our global slope estimates being based on a study that focused on lowland agriculture (Van Oost *et al.*, 2009). This also indicates that croplands in mountainous areas are preferentially situated on relatively flat terrain; and this is not captured by global databases. Therefore, we reanalyzed the data and explored the predictive power of several topographical indices to identify these areas. We found an optimal fit between NRI and global estimates when the range of elevation was included as follows:

$$Slope_C = \begin{cases} \left((3.21 + Slope - 0.08ER) * 1.20 + (-0.002EL + 0.26)\right)^{1.08} * 0.95 & if\ \ EL > 600\ \ and\ \ ER > 100 \\ \left((3.21 + Slope - 0.08ER) * 1.20\right)^{1.08} * 0.95 & if\ \ EL < 600\ \ and\ \ ER > 100 \\ \left((Slope * 0.82)^{1.43} * 0.53 * 1.20 + (-0.002EL + 0.26)\right)^{1.08} * 0.95 & if\ \ EL > 600\ \ and\ \ ER < 100 \\ \left((Slope * 0.82)^{1.43} * 0.53 * 1.20\right)^{1.08} * 0.95 & if\ \ EL < 600\ \ and\ \ ER < 100 \end{cases} \tag{2}$$

where $Slope_C$ is the corrected NRI adjusted slope (%), *ER* (m) is the range of elevation and *EL* (m) is the mean elevation in a 3 by 3 window (based on GTOPO data (USGS, 1996)). The minimum slope was set to 0%.

This approach resulted in good predictions of cropland slopes at an aggregation level of HUC4 watersheds ($R^2=0.62$, $P<0.01$, Figure 1). A similar trend for pastures towards overestimation was observed. Because of the previously described limited entries for agricultural grasslands in the NRI database, correction for these could therefore only improve systematical errors while a spatial evaluation was not possible.

Crop factor: Using the NRI data, we calculated mean *C* factor values for all major US crop types. We derived spatial estimates based on this information using global crop type abundance maps and compared them with NRI estimates. The NRI database showed that US cropland is dominated by maize, wheat and soy; together these crops represent 70% of the total cropland area. The relative contribution of different crop types for the US is well represented by the global database (Table III). The most important crop types are characterized by a narrow range of *C* factors with mean values that range between 0.22 and 0.24. However, specific crop types have a much higher mean value (e.g. potatoes and sugar beets). Although the global database gave a good prediction of mean *C* factors at the continental scale, the spatial correlation between NRI and global database-derived *C* factors at the HUC4 watershed level was significant, but relatively weak ($R^2=0.20$, $P<0.001$, Figure 1).

We were unable to relate a global soil dataset to NRI estimates for the *K* factor (data not shown). We also did not include those factors that were not contributing significantly

Table II. Step-wise linear regressions between NRI soil erosion estimates (*A*) and USLE factors aggregated at the HUC4 watershed level. *R* = rainfall erosivity factor, *S* = Slope percent, *K* = Soil erodibility factor, *L* = Slope length, *P* = support practice factor). Significance level p expressed as: 1% = **; not significant ($P > 0.1$) = ^{ns}

		US cultivated cropland (N = 211)		US pastures (N = 208)		US dry pastures (N = 120)	
Factor	Model	Model R^2	Partial correlation	Model R^2	Partial correlation	Model R^2	Partial correlation
R	*A* = *R*	0.28	0.53**	0.75	0.87**	0.64	0.80**
R	*A* = *RS*	0.75	0.81**	0.8	0.89**	0.8	0.89**
S			0.81**		0.43**		0.67**
R	*A* = *RSK*	0.77	0.82**	0.86	0.93**	0.84	0.90**
S			0.81**		0.63**		0.74**
K			0.24**		0.55**		0.42**
R	*A* = *RSKC*	0.78	0.83**	0.87	0.93**	0.86	0.90**
S			0.82**		0.63**		0.75**
K			0.28**		0.55**		0.41**
C			0.25**		0.19**		0.37**
R	*A* = *RSKCP*	0.78	0.81**	0.87	0.93**	0.86	0.89**
S			0.79**		0.61**		0.71**
K			0.29**		0.53**		0.41**
C			0.25**		0.18**		0.33**
P			−0.11 ^{ns}		0.09 ^{ns}		0.11 ^{ns}
R	*A* = *RSKCPL*	0.78	0.78**	0.87	0.93**	0.86	0.89**
S			0.79**		0.61**		0.71**
K			0.28**		0.53**		0.41**
C			0.25**		0.18**		0.33**
P			−0.11 ^{ns}		0.09 ^{ns}		0.10 ^{ns}
L			−0.02 ^{ns}		−0.02 ^{ns}		−0.04 ^{ns}

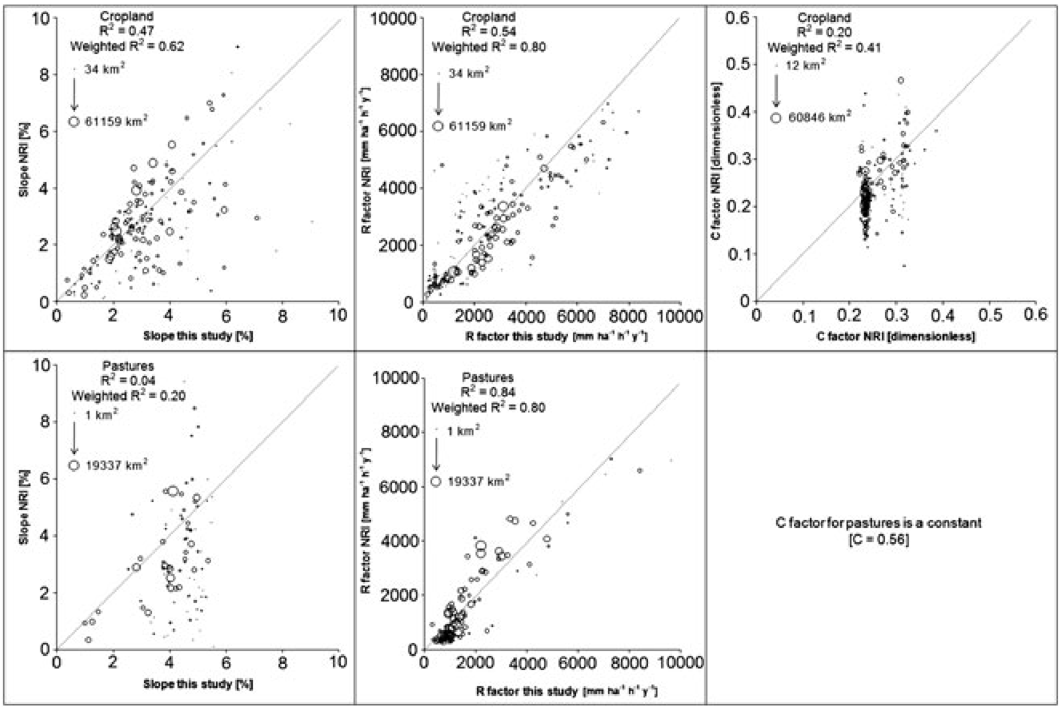


Figure 1. Scatter plots of erosion controlling factors derived from NRI and global databases. Data is aggregated at the HUC4 watershed level.

to the variation in water erosion at the continental scale, i.e. slope length (*L*) and management (*P*). For these three factors, we did not provide spatial estimates but used the average value for the US instead.

Comparison with NRI soil erosion estimates

We compared our estimates, aggregated at the HUC4 watershed level, with NRI derived estimates for cropland and

Table III. *C* factor estimates and differences in abundance for major crop types

Region	USA	USA	USA	World
Source	USDA (2000); Nusser and Goebel (1997)	USDA (2000); Nusser and Goebel (1997)	Leff *et al.* (2004)	Leff *et al.* (2004)
Crop Type	C Factor (± 1 StDev)	Fraction (%)	Fraction (%)	Fraction (%)
Wheat	0.22 (± 0.07)	22.0	22.8	22
Maize	0.22 (± 0.08)	26.4	28.3	13
Rice	0.22 (± 0.08)	1.2	1.3	11
Barley	0.22 (± 0.08)	1.8	2.6	9
Soy	0.24 (± 0.08)	21.3	24.1	5
Potatoes	0.32 (± 0.09)	0.4	0.5	3
Cotton	0.46 (± 0.10)	4.7	6.2	3
Sorghum	0.29 (± 0.08)	3.4	4	3
Sunflower	0.23 (± 0.06)	0.8	1.2	2
Sugar beet	0.31 (± 0.07)	0.4	0.6	1
Other	0.25 (± 0.03)	17.8	8.5	28
Area (10^3 km^2)		**1284**	**1471**	**17920**

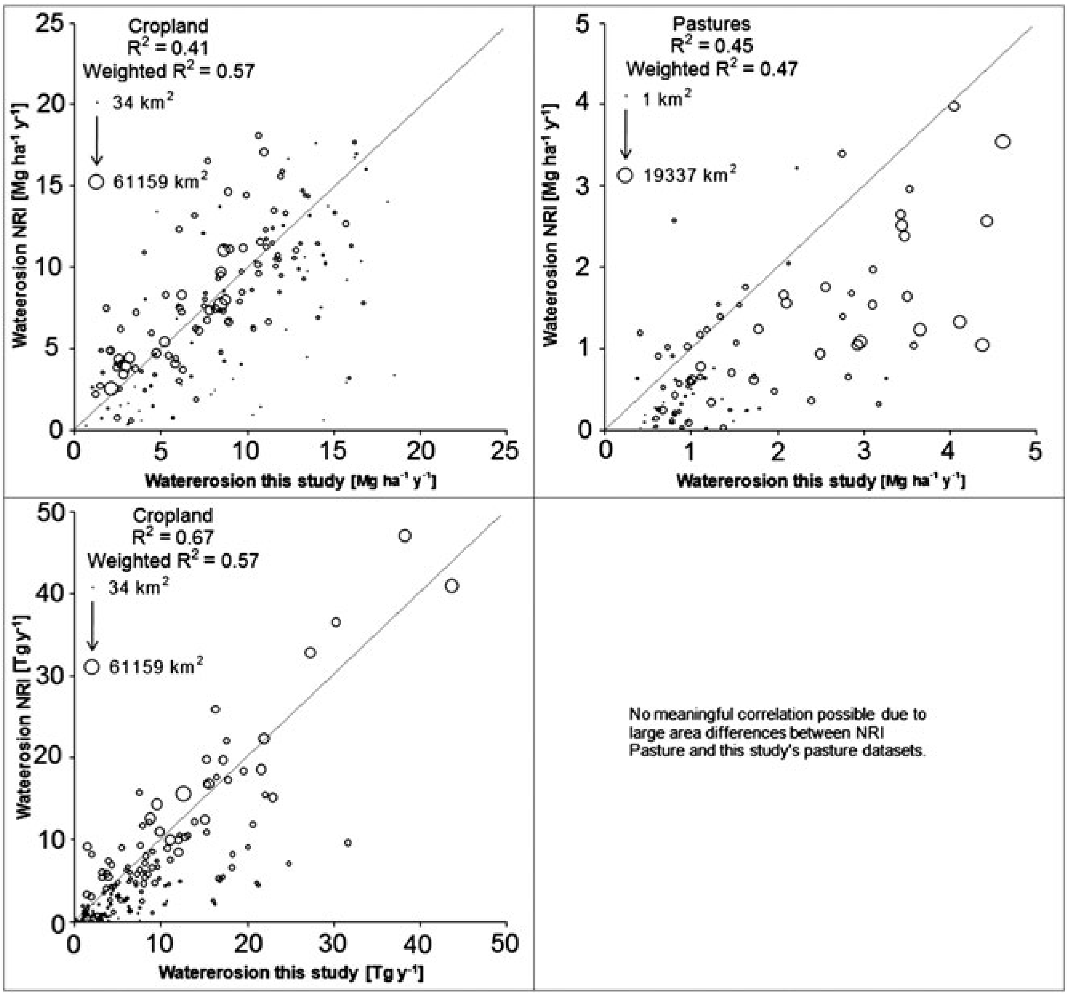


Figure 2. Scatter plots of NRI- and model-derived water erosion estimates for US cropland and pastures.

pastureland erosion. For croplands, we obtained a reasonably good fit (Figure 2), although our estimates for larger watersheds were consistently higher than NRI estimates for erosion rates above 10 Mg ha^{-1} y^{-1} (representing *c.* 20% of US cropland). This results from the fact that our approach focused on a correct prediction of single factors and did not optimize the output of the multiplicative model, i.e. soil erosion rates. Rather than (re)adjusting individual factors, we choose to adjust the erosion estimates for cropland as follows:

$$E_{WCf} = 4.7 * \ln(E_{WC}) - 1.79 \quad (3)$$

where E_{WCf} is our final estimate of erosion (Mg ha^{-1} y^{-1}).

This resulted in an overall model efficiency (R^2) of 0.57 for both water erosion rate and sediment flux (Figure 2). No systematic bias in our predictions for pastureland water erosion was observed and a comparable model performance for erosion rate was achieved. However, due to the previously described limitations of land use datasets, soil water erosion fluxes on pastures were poorly predicted (model efficiency (R^2) <0.1).

Uncertainty analysis

Consistent with the identification of key controls on soil erosion, this analysis showed that the highest parameter uncertainties were associated with the *R* and *S* factors with an RRMSE of the parameter estimates of 52% and 116%, respectively; these are also the parameters with the highest spatial variability. In contrast, the input error for the *C* factor is much lower (RRMSE of 30%). However, the analysis clearly indicated that, when all factors are considered, the effect of uncertainty associated with the model input on model predictions is relatively small. When applying the simplified models where one of the three factors were kept constant, we observed that the full complexity model was the best model with a total error of 33%. Model structures that do not incorporate slope and rainfall erosivity factors resulted in much higher errors (RRMSE between 43 and 35%). This analysis shows that the model used here is relatively robust because the error on the total erosion flux on US cropland is relatively low (RRMSE of 33%).

In a next step we applied our model, which was calibrated with data from the USA, to Europe and compared the results with independent estimates (Cerdan *et al.*, 2010). The latter study provides estimates for water erosion on different types of land use in Europe with a very high spatial resolution (100 m). This study is also based on the spatial extrapolation of plot data, but differs from our approach as (i) the spatial databases used have a much higher precision and resolution than our low precision and coarse global databases, (ii) the effect of soil erodibility is explicitly modeled and (iii) the empirical basis is derived from local, and therefore more representative, plot experiments in Europe. Two key observations can be made (Figure 3). First, although there is substantial scatter, there is a significant spatial correlation for both erosion rates ($R^2 = 0.41$, $P < 0.01$) and fluxes ($R^2 = 0.44$, P 0.01). This indicates that our approach, which is based on coarse input data and simplified process representation, is able to capture the spatial variability of soil erosion. Secondly, although there is a good spatial fit, the absolute values differ substantially. The Cerdan *et al.* (2010) estimates are, on average, 58% lower than our estimates (i.e. 9.1 Mg ha^{-1} y^{-1} vs. 3.9 Mg ha^{-1} y^{-1}). This difference is mainly related to the reference soil erosion parameter (i.e. E_{NRIref}, see Equation (1)), which is 67% lower in the Cerdan *et al.* study. This clearly shows that, in contrast to the spatial patterns, the uncertainty associated with erosion rates is high (RRMSE = 69%).

For other regions, e.g. South-east Asia, South America or Africa, spatial explicit data concerning soil erosion is still scarce. Based on a RUSLE approach, but using high resolution input data, the Australian National Land and Water Resources Audit (NLWRA, 2001) and Lu *et al.* (2003) estimated the average annual soil loss due to sheet and rill erosion for cropland in Australia at 3.4 Mg ha^{-1} y^{-1} and 3.6 Mg ha^{-1} y^{-1}, respectively. For pastures, the NLWRA reported an erosion rate of 5.6 Mg ha^{-1} y^{-1}. These independent estimates at the scale of continents, are in the same order of magnitude as our estimates, particularly for croplands. Our estimates for Australian cropland and pastures are 4.7 Mg ha^{-1} y^{-1} and 0.9 Mg ha^{-1} y^{-1}, respectively. For South Africa, Compton *et al.* (2010) estimated the rate of sheet water erosion at 6.9 Mg ha^{-1} y^{-1} for cropland and 1.1 Mg ha^{-1} y^{-1} for pastures. Their method was based on a quantitative assessment of land degradation in combination with detailed land cover maps. Again, this is similar to our estimates of 8.4 Mg ha^{-1} y^{-1} for cropland and 1.6 Mg ha^{-1} y^{-1} for pastures.

In summary, the results of the Monte Carlo analysis and comparison with independent estimates for Europe, Australia and Africa suggest that the model is able to describe accurately spatial patterns of soil erosion at the continental scale. The error associated with the estimation of the total agricultural sediment flux is much higher. Although a fully analytical error analysis is at present not possible due to data gaps, our analysis suggests that the error is in the order of *c.* 30% to 70%.

Global application

When we apply our model to global agricultural land, we obtain an accelerated sediment flux (water and tillage erosion) of 20.5 (±10.3) Pg y^{-1} for the year 2000, with a relative contribution of water erosion of *c.* 85%. This estimate corresponds to a SOC erosion flux of 0.4 (±0.2) Pg C y^{-1}. Even though total cropland area is around half of the total agricultural grassland area considered, 15.9 (±8.0) Pg y^{-1}, representing 78% of total agricultural erosion, are eroded on croplands. The average global erosion rate for cropland and pastures is estimated at 10.5 Mg ha^{-1} y^{-1} and 1.7 Mg ha^{-1} y^{-1}, respectively, which results in a global average of 4.2 Mg ha^{-1} y^{-1} for total agricultural area. High rates of erosion occur in tropical regions where steep slopes and high rainfall amounts coincide, whereas erosion for most other regions varies between 6 and 10 Mg ha^{-1} y^{-1}. The predicted erosion patterns and rates are presented in Figure 4(a) and (b). Most of the erosion occurs in Asia (43% of total agricultural sediment flux), with high rates especially in

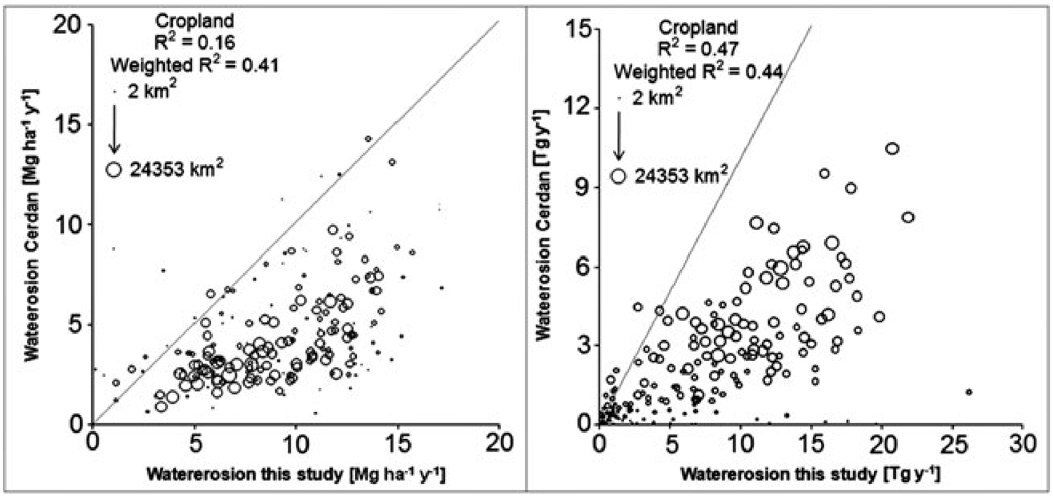


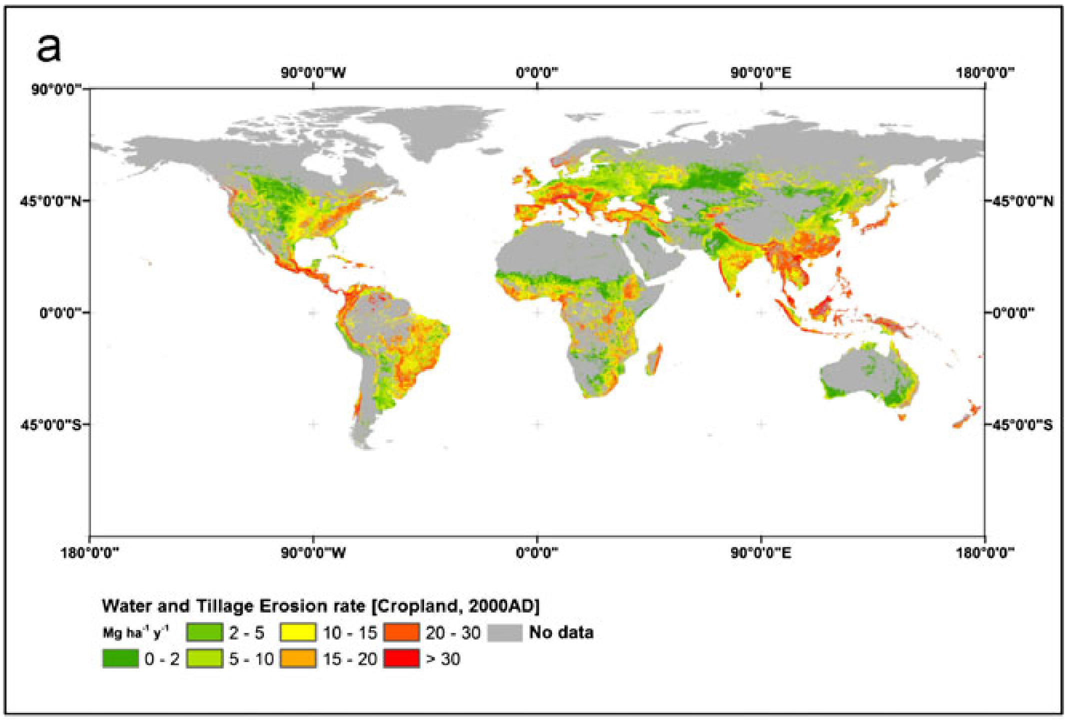
Figure 3. Scatter plots of water erosion estimates derived from the Cerdan *et al.* (2010) study and modelled estimates for European cropland. Data is aggregated on a regular grid, using the mean HUC4 area estimate for the conterminous USA (i.e. 193 km × 193 km).

South-east Asia (Figure 4(a) and Table IV). SOC erosion shows a similar spatial pattern as carbon erosion rates are strongly related to soil erosion rates ($R^2 = 0.95$). This implies that variability in SOC content is less important than variability in erosion rates.

Discussion

Comparison with other global estimates

We obtain a global flux of 13.1 (±6.6) Pg y^{-1} by water erosion on croplands. Montgomery (2007) presented, to our knowledge, the most comprehensive study on soil erosion rates under conventional and conservation agriculture as well as soil production rates. The extrapolation of plot-data to global agricultural lands is the most commonly used method to estimate global fluxes (Pimentel *et al.*, 1995; Hooke, 2000). Here, we take the mean value of the plot studies presented in Montgomery (2007) and compare them with our estimate. Our estimate for cropland erosion is approximately five times lower than the mean value of erosion rates derived from plot experiments (i.e. 10.5 Mg ha^{-1} y^{-1} versus 54.0 Mg ha^{-1} y^{-1}, assuming a soil bulk density of 1.45 g cm^{-3}. A comparison of the cumulative distributions (Figure 5) shows that our approach results in an erosion rate below 20 Mg ha^{-1} y^{-1} for 80% of the cropland area, while this is less than 50% for the experimental studies.



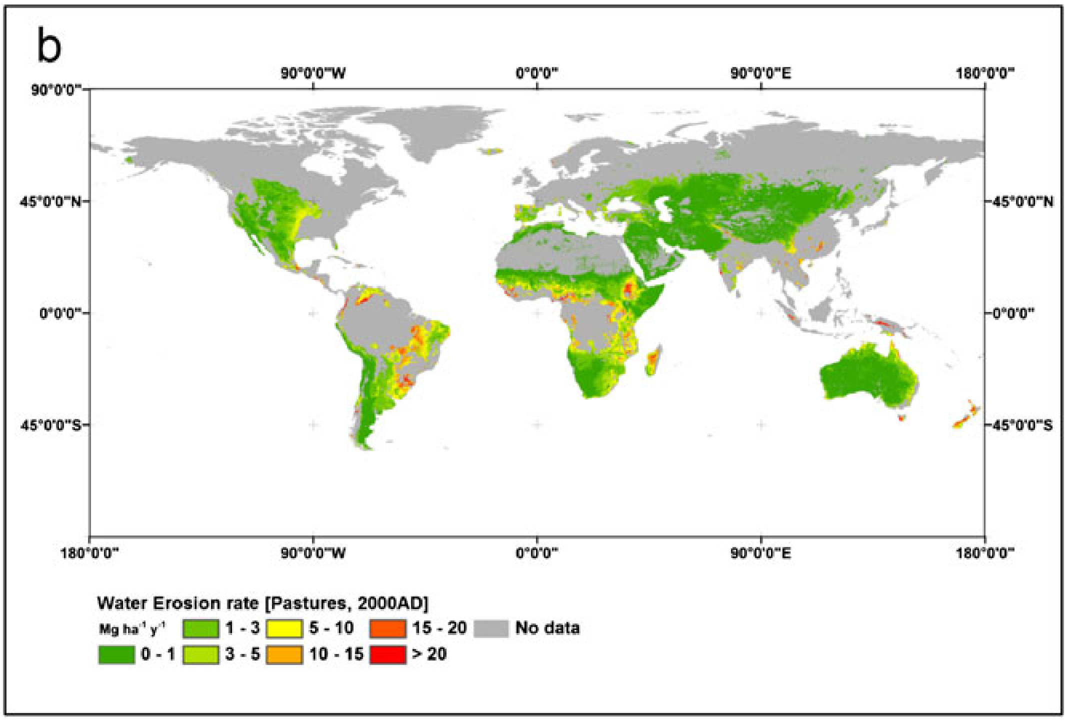


Figure 4. (a) Estimated water and tillage erosion rates for global cropland for the year 2000. (b) Estimated water erosion rates for global pastures for the year 2000. Only pastures in dry environments are considered to erode. This figure is available in colour online at wileyonlinelibrary.com/journal/espl

Table IV. Estimates for globally eroded sediment and linked SOC fluxes for cropland and pastures for the year 2000 (uncertainty 30–70%). Only pastures in dry environments are considered to erode

	AREA 2000 (10^6 km^2)			Sediment flux (Pg y^{-1})			SOC Flux (Tg C y^{-1})		
	C	P	P	C		P	C		p
Region	Total	Total	Dry	Water	Tillage	Water	Water	Tillage	Water
Central America	0.4	1.0	0.6	0.5	0.2	0.1	9.7	3.6	2.6
Central Asia	0.4	2.8	2.8	0.1	0.1	0.1	1.2	1.0	2.7
China	1.6	4.0	3.0	1.3	0.4	0.2	23.7	7.0	5.4
Europe	2.5	1.4	0.5	1.9	0.6	0.1	39.5	12.0	2.4
India	2.0	0.2	0.1	1.9	0.3	0.0	30.4	3.9	0.5
Middle East	0.6	2.2	2.1	0.4	0.4	0.1	6.8	7.4	1.3
North Africa	0.3	0.7	0.7	0.2	0.1	0.0	2.8	1.1	0.7
North America	2.3	2.5	2.0	1.6	0.2	0.3	31.5	4.5	6.1
North East Asia	0.5	1.7	1.5	0.1	0.1	0.1	3.5	2.5	1.3
Oceania	0.5	4.2	4.1	0.3	0.0	0.5	4.3	0.6	10.5
South America	1.2	4.6	2.9	1.4	0.1	1.4	26.7	2.5	36.2
South East Asia	0.9	0.2	0.0	1.5	0.2	0.1	31.2	3.0	3.0
Sub Saharan Africa	2.0	8.1	7.0	1.8	0.2	1.7	29.9	2.5	38.0
Total	**15.1**	**33.4**	**27.4**	**13.1**	**2.8**	**4.6**	**241.1**	**51.7**	**110.7**

Comparison with global estimates (Table I) available in the literature and which are based on extrapolation of erosion plot data (Myers, 1993 [50 Pg y^{-1}]; Pimentel *et al*., 1995 [73.5 Pg y^{-1}]; Wilkinson and McElroy, 2007 [28.1 Pg y^{-1}]), shows that these estimates are two to five times higher than our estimate. Similar, other approaches to derive erosion fluxes based on river sediment fluxes and yield enhancement factors (Stallard, 1998 [23.7–64.9 Pg y^{-1}]), extrapolation of erosion rates multiplied per capita (Hooke, 2000 [120 Pg y^{-1}]), or the application of erosion models on a global scale (Van Oost *et al*., 2007 [28.3 Pg y^{-1}]; Yang *et al*., 2003 [79.6 Pg y^{-1}]; Ito, 2007 [172.2 Pg y^{-1}]) all result in much higher fluxes than our approach of calibration with high-res empirical data and application of a simplified model on a global scale. This indicates that empirical studies are biased towards areas that are characterized by high erosion rates and are probably not fully representative for global cropland. Extrapolating without taking into account the spatial variability of controlling factors or applying erosion models with relatively coarse input data will therefore overestimate total erosion and consequently lead to erroneous conclusions when applied to large areas. Although this suggests that existing estimates are likely to be overestimations, rates of accelerated erosion are well above typical rates of soil formation (which is estimated on average at 0.53 Mg ha^{-1} y^{-1}, assuming a soil bulk density of 1.45 g cm^{-3}; Montgomery, 2007).

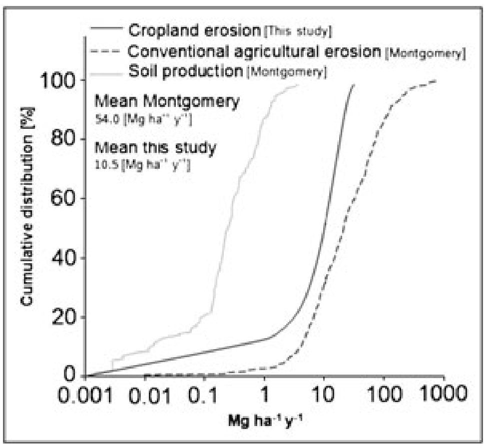


Figure 5. Frequency distributions of soil erosion and soil production derived from point observations (Montgomery, 2007) and our estimates of cropland water erosion for the year 2000.

Syvitski *et al*. (2005) present an assessment of the human impact on global river sediment discharge. They estimate the global human-induced river sediment flux at 2.3 ±0.6 Pg y^{-1}. Here, we discuss this river flux in relation to our estimates of accelerated erosion, which reflects mobilization of sediment in uplands. We do this by taking into account the retention and delivery of sediments to the fluvial system. Although controlled by a series of factors including basin area, climate and topography, between 10 to 30% of the material mobilized by accelerated erosion are typically delivered to the fluvial systems (Walling and Webb, 1996; De Vente *et al*., 2007). When this ratio is applied to our estimates, we obtain a human-induced river flux of 1.8–5.4 Pg y^{-1}. This is similar to the flux estimated by Syvitski *et al*. (2005) and suggests that our global estimates are in the right order of magnitude.

Linking eroded soil and SOC to river sediment and POC

We compared our estimates of agricultural sediment mobilization with estimates of total suspended sediment (TSS) and particulate organic carbon (POC) fluxes to the oceans, provided by NASA-ISLSCP II river discharge data set (Hall *et al*., 2005).

Table V. Comparison mean values for global river TSS, POC and DOC (Hall *et al*., 2005) to our estimates of water induced soil and SOC erosion for the year 2000

Area [10^6 km^2]: 133.2 N = 6070	Fluxes [Pg y^{-1}]
DOC	0.20
POC	0.15
TSS	16.7
Eroded Soil	17.6
Eroded SOC	0.35
POC in TSS [%]	0.9
SOC in eroded soil [%]	1.9

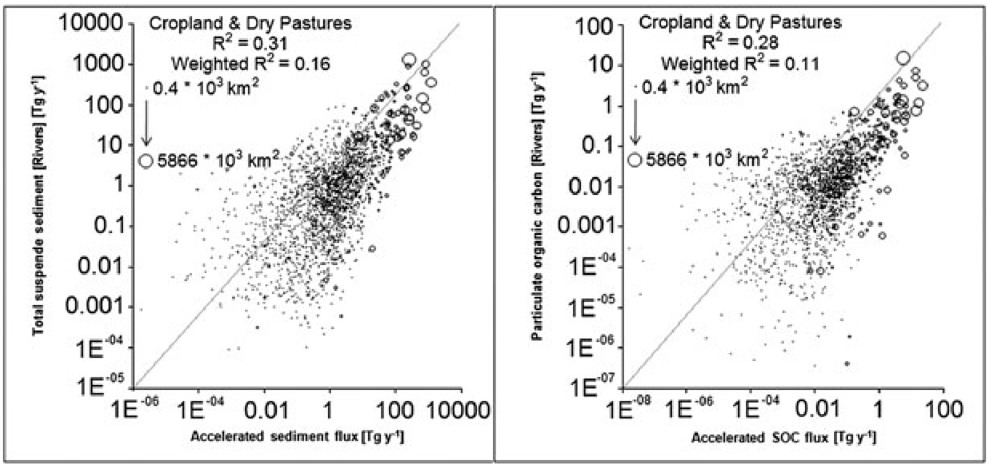


Figure 6. Left panel: Global river sediment vs. accelerated, agricultural soil erosion fluxes for the year 2000. Right panel: Global particulate organic carbon in river systems vs. SOC fluxes induced by accelerated erosion for the year 2000.

The total sediment flux to the global ocean is estimated at 16.7 Pg y^{-1} (Hall *et al.*, 2005) and this integrates both natural and human-induced sediment fluxes. This is very similar to our estimate of accelerated water erosion, i.e. 17.6 (± 8.8) Pg y^{-1} from which only about 10–30% are delivered to fluvial systems (see above). The POC flux to the global ocean is estimated at 153.9 Tg y^{-1}, while our estimates of accelerated SOC erosion are in the order of 348.4 (±174.2) Tg y^{-1} (Table V).

When specific sediment yields and accelerated erosion for all global basins are compared, no significant relation is observed (data not shown). Low correlations ($R^2 = 0.16$, $P < 0.001$) are found for sediment fluxes (Tg y^{-1}) as basin area is accounted for. This indicates that, on a global scale, other factors than agricultural erosion control the export of sediment to the ocean. This can be explained by the fact that natural erosion fluxes are mainly controlled by slope parameters (McElroy and Wilkinson, 2005; Wilkinson and McElroy, 2007) while most of the agricultural regions are situated in lowlands. Similar observations can be made for particulate carbon export (Figure 6).

We illustrate graphically the relative contribution of agricultural erosion for all basins by considering the ratio between agricultural and natural (assuming that TSS estimates are representative for undisturbed conditions) sediment flux (Figure 7). This shows that in areas of intensive agriculture, typically located in lowlands, the relative contribution of accelerated erosion is high and often dominates river sediment fluxes (e.g. Central Europe). In contrast, for basins with mountainous uplands (e.g. South America or Southern Asia) natural processes are more important and the footprint of human activities is smaller which confirms the assumptions made in the previous section.

As discussed above, recent studies indicate that human activities have a relatively small effect on the total flux to the ocean (Syvitski *et al.*, 2005; Walling, 2009). This implies that vast amounts of sediment and SOC eroded on agricultural lands do not reach the fluvial system but are buried in terrestrial depositional zones, such as wetlands, reservoirs, floodplains and colluvial soils (McCarty and Ritchie, 2002; Smith *et al.*

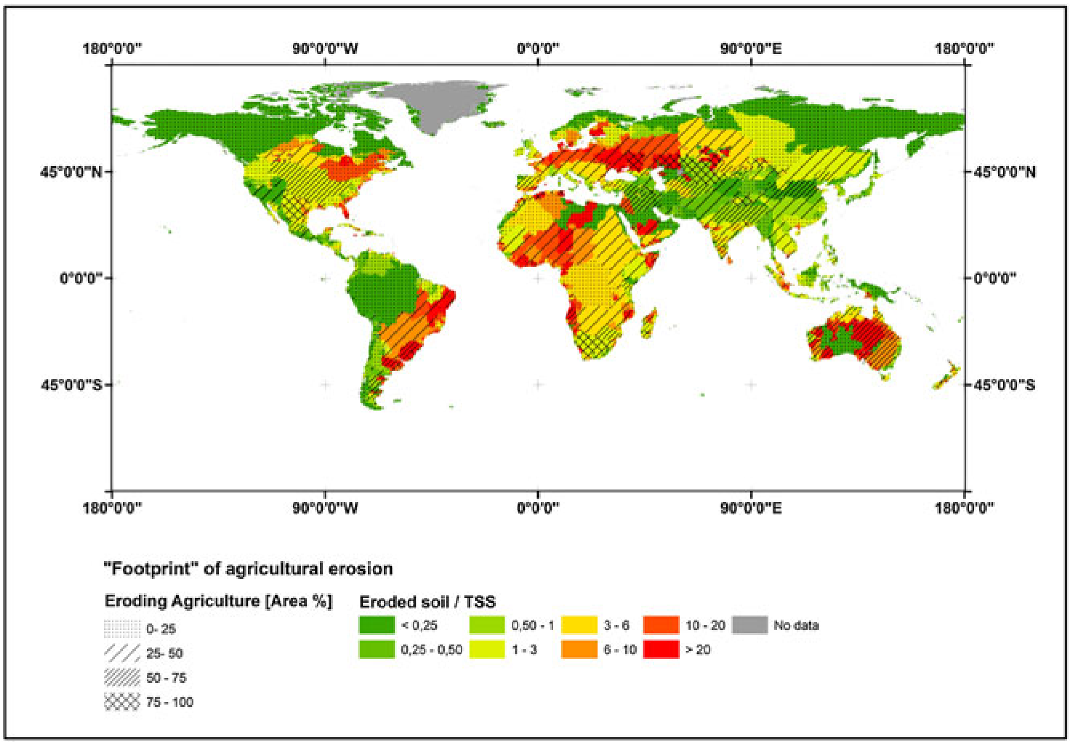


Figure 7. Ratios of eroded soil/TSS for major basins for the year 2000. This figure is available in colour online at wileyonlinelibrary.com/journal/espl

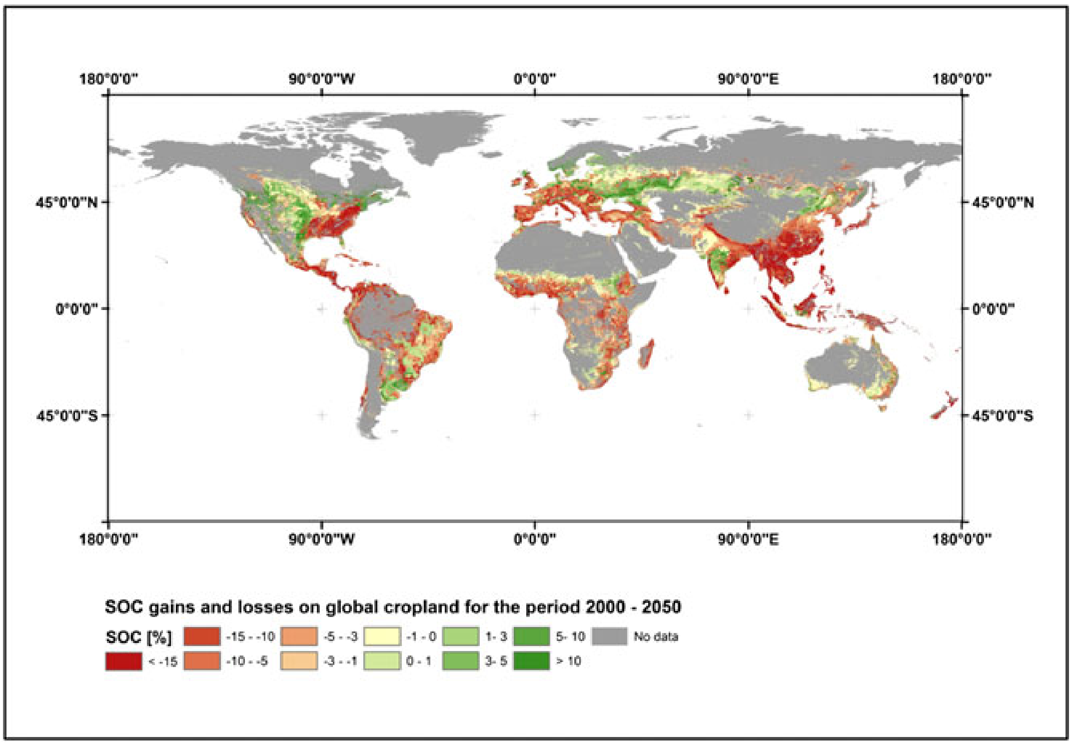


Figure 8. Simulated changes in topsoil SOC contents due to soil erosion for the period 2000–2050. Gains are related due to the inclusion of carbon rich subsoil. This figure is available in colour online at wileyonlinelibrary.com/journal/espl

2005) or are oxidized during the transport or burial process (Lal 2003; Van Hemelryck *et al.*, 2010).

It is therefore important to identify where, and for how long, the sediment and carbon mobilized by accelerated erosion is retained, but at present little information is available (Renwick, 2005; Osterkamp *et al.* 2011). It remains uncertain under which conditions soil erosion can act as a sink or source for carbon (Dlugoß *et al.*, 2011; Dymond, 2010). Recent progress on clarifying what factors eventually control the strength of the erosional forcing (Berhe, 2012; Hilton *et al.*, 2011; Worrall *et al.*, 2011) will help to answer the question of the missing terrestrial C sink.

Soil erosion as a soil modifier

In the previous paragraphs, we addressed the magnitude of the erosional disturbance. Here, we discuss how this lateral transport of sediment and organic carbon is relevant for the cycling of carbon between the terrestrial ecosystem and the atmosphere. On eroded sites, former subsoil mixes with remaining topsoil resulting in SOC depleted topsoils when compared with non-eroded profiles and complete to partial replacement of eroded SOC via crop production may result in net uptake of atmospheric CO_2 (Stallard, 1998). On depositional sites, former topsoil is buried under continuously deposited material, which may inhibit decomposition of OC, and deep profiles enriched in SOC develop. The erosion-induced alterations of the net CO_2 fluxes between the soil and the atmosphere thus critically depend on the rate of dilution and replacement at sites of erosion (Quine and Van Oost, 2007; Van Oost *et al.*, 2007). The approach described above allows us to quantify this dilution process on a global scale by linking our model with the spatially explicit WISE 3.1 soil profile database. Assuming constant erosion rates, we estimate SOC profile modifications induced by agricultural erosion for the period 2000–2050. The SOC profiles change in response to erosion and are updated each year. The amount of SOC eroded is proportional to the erosion rate (derived from our model) and the SOC inventory of the topsoil layer (0–20 cm, derived from WISE 3.1). As a result of erosion, a fraction of the subsoil layer is incorporated into the top soil layer. The results indicate that this dilution effect is important for the vast majority of cropland (Figure 8). For example, the average dilution of the topsoil in the EU-25 states is 22% for the whole period or a loss of 20.59 ($\pm$ 10.3) g C m^{-2} y^{-1} (assuming a soil bulk density of 1.45 g cm^{-3}). In relation to other C fluxes this value is comparably low, as the C balance is dominated by gains and losses due to net primary production (NPP), harvest and respiration (Ciais *et al.*, 2010). However, it is in the same order of magnitude as other disturbances of these dynamics and offsets carbon gains from improved management, e.g. fertilizer C input (+ 26 g C m^{-2} y^{-1}) (Table VI).

On a global scale, the highest dilution rates are simulated in the (sub)tropics, where a combination of steep slopes and high rainfall results in high erosion rates, such as Brazil, South-east Asia or Ethiopia. This is consistent with the observations of Ziegler and Nogareda (2009) and Valentin *et al.* (2008). However, topsoil enrichment is also possible: due to erosion, the organic sublayer of specific soils (e.g. Histosols) is incorporated

Table VI. Cropland C fluxes in the EU-25 compiled by Ciais *et al.* (2010) in relation to our estimate of C losses due to erosion-induced dilution

Flux	C gains (+) and losses (−) g m^{-2} y^{-1}
Net primary production	(+) 490–846
Heterotropic respiration	(−) 299
Harvest	(−) 239–275
Fertilizers	(+) 26
River outflux	(−) 9.6
Net biome production	(−) 8.3($\pm$ 13)–13($\pm$ 33)
Trade net	(+) 6.8
Fires	(−) 3.3($\pm$ 1)
Erosion induced dilution (This study)	(−) 22.6($\pm$10)

Table VII. Representativeness of US cropland and grasslands

Factor	World average cropland (±1 StDev)	US average cropland (±1 StDev)	World average dry pastures (±1 StDev)	US average dry pastures (±1 StDev)
C	0.26 (± 0.02)	0.25 (± 0.03)	-	-
R	2970 (± 3521)	2688 (± 1452)	2147 (± 4158)	1568 (± 1155)
S	0.48 (± 0.51)	0.31 (± 0.29)	0.44 (± 0.23)	0.47 (± 0.18)
K	0.92 (± 0.24)	0.97 (± 0.15)	0.82 (± 0.32)	0.88 (± 0.29)

into the topsoil layer. The cropland area experiencing gains is small (22%) compared with cropland experiencing SOC losses (78%) (Figure 8). It is important to note that the data presented here should be interpreted in terms of a disturbance. It is clear that dynamic processes such as the replacement of eroded carbon by new photosynthate, on-site changes of microbial decomposition or during transportation and erosion-induced changes in soil properties are essential to fully understand SOC changes (Jacinthe and Lal, 2001; Jacinthe *et al*., 2001; Billings *et al*., 2010). However, this analysis shows that erosion-induced changes in topsoil carbon content are significant at a global scale and that agricultural soils should be considered as dynamic systems that can change rapidly. For example, the substantial dilution has implications for soil carbon monitoring studies that usually do not consider erosion and identify all (topsoil) SOC losses as a source for atmospheric CO_2 (Goidts *et al*., 2009).

Caveats

The approach taken here aims to simplify and parameterize the key factors that control accelerated erosion and SOC erosion based on quantities that can be inferred at global scales. As is the case with any global-scale model, it is subject to drawbacks and it does not capture the subtleties that drive small-scale variation. Although our analysis shows that a simplified model and coarse, but global, input data allow for a good prediction of spatial patterns in both the USA and Europe, the uncertainty associated with the magnitude of global accelerated erosion is without any doubt high.

First, US agricultural land may not be fully representative for global agricultural land (Table VII). US slope factors are on average 35% lower than global values because US cropland is biased towards lowland agriculture in the Mississippi valley. In addition, R factors on eroding dry pastures in the USA are about 27% less than the world average, as the US rangelands are typically situated in the very dry continental mountain ranges which might not be fully representative for the (full) range of grassland in other parts of the world (e.g. Australia, Brazil, Central Asia).

Although distributions of the major crop types (e.g. wheat, maize) are similar, the more erodible crop types (e.g. potatoes and sugar beets) are more abundant on global cropland (Table III). In particular, the combination of steep slopes and high rainfall erosivity is rare in the USA, while our global extrapolation shows that these conditions make up most of the global accelerated erosion (e.g. more than 26% of the global flux occurs in South-east Asia and India, Table IV).

At present, accurate continental- or regional-scale assessments of agricultural soil erosion are lacking in these regions and we are therefore unable to verify our model predictions. In addition, the management of agricultural land in the USA, which is representative for high-input agriculture, might be different elsewhere (e.g. in terms of landscape structure, cover crops, conservation techniques, etc.).

Second, our approach is based on an empirical model. Although the limitations and drawbacks of the USLE were discussed recently (Parsons *et al*., 2004), it is based on more than 10 000 plot-years of soil erosion data and should be considered as one of the largest data-collecting efforts in environmental sciences.

Third, other processes, which may dominate accelerated erosion at different spatial scales or under specific conditions are not included in our analysis such as (ephemeral) gully erosion, wind erosion and soil losses associated with harvest (Poesen *et al*., 1996). To our knowledge, neither the required empirical data nor models are currently available to present a meaningful assessment of these processes at larger spatial scales.

In summary, the approach we used is admittedly not very complex – the lack of (spatial) data on how erosion varies with soil erodibility or management (e.g. terracing), as well as information on what controls agricultural erosion in mountainous (sub-)tropical or low-input agricultural systems, precludes a more detailed formulation. However, the good spatial predictions of agricultural erosion suggest that our coarse model is able to capture variability at the continental scale. US conditions are diverse and can be considered to be representative for a large fraction of global agricultural land. It is not possible to assess accurately the uncertainty associated with our estimates of global agricultural erosion due to the lack of model testing data in specific regions. However, comparison with estimates for Europe, South Africa and Australia and the results from our uncertainty assessment indicate that the uncertainty will be in the order of 30–70%.

Conclusions

We parameterized a simplified global-scale model with the NRI data for the USA. The good agreement between our approach and empirical estimates for larger regions suggests that the approach may capture the essence of accelerated erosion. As such, the model may be used to predict the significance of accelerated erosion for the global carbon cycle, soil evolution or sustainability of agricultural practices at continental or global scales. The lack of data on how erosion varies with soil erodibility or management, as well as information on what controls agricultural erosion in mountainous (sub-)tropical or low-input agricultural systems, precludes a more detailed formulation and validation. However, to date our global assessment is the only attempt we know of that is (i) consistent with empirical observations at relevant spatial scales and (ii) is validated at the continental scale. Nevertheless, at present it is not possible to assess accurately the uncertainty associated with our estimates of global agricultural erosion due to the lack of model testing data in regions which differ from US conditions.

We estimate soil/SOC erosion rate on a global scale at 10.5 Mg ha^{-1} y^{-1}/193 kg C ha^{-1} y^{-1} and 1.7 Mg ha^{-1} y^{-1}/40.4 kg C ha^{-1} y^{-1} for croplands and eroding pastures, respectively. This corresponds to a total flux of 20.5 (±10.3) Pg y^{-1} of soil and 403.5 (±201.8) Tg y^{-1} SOC due to agricultural erosion for the year 2000. Although the uncertainty is high, our numbers are significantly lower than former assessments based on extrapolation of local data and show that only a small area of cropland and pastures is subjected to high erosion rates. It is likely that empirical studies are biased towards intensively eroding areas and may therefore overestimate global erosion substantially. Current assessments of the global significance of agricultural erosion for the global carbon cycles are typically based on largely unverified assumptions about the intensity and extent of erosion. Our analysis shows that simplified global-scale approaches may well capture the spatial variability of accelerated erosion and that existing estimates are likely to overestimate the erosion of SOC induced by human activities. Nevertheless, there is no doubt that even these lower estimates of agricultural erosion largely exceed rates of soil formation – especially in (sub)tropical regions – and this should be considered as a serious threat to the soil resource.

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References

Bakker MM, Govers G, Rounsevell MDA. 2004. The crop-productivity-erosion relationship: an analysis based on experimental work. *Catena* **57**(1): 55–76.

Batjes NH. 2006. ISRIC-WISE derived soil properties on a 5 by 5 arc-minutes global grid. Report 2006/02. ISRIC – World Soil Information. Wageningen.

Batjes NH. 2008. ISRIC-WISE Harmonized Global Soil Profile Dataset (Ver. 3.1). Report 2008/02. ISRIC-World Soil Information. Wageningen.

Berhe A. 2012. Decomposition of organic substrates at eroding vs. depositional landform positions. *Plant and Soil* **350**: 261–280.

Berhe A, Harte J, Harden J, Torn M. 2007. The significance of the erosion-induced terrestrial carbon sink. *Bioscience* **57**: 337–346.

Billings SA, Buddemeier RW, Richter DD, Van Oost K, Bohling G. 2010. A simple method for estimating the influence of eroding soil profiles on atmospheric CO2. *Global Biogeochemical Cycles* **24**: GB2001. DOI: 10.1029/2009GB003560. 14 pp

Boardman J. 2006. Soil erosion science: Reflections on the limitations of current approaches. *Catena* **68**(2–3): 73–86.

Cerdan O, Govers G, Le Bissonnais Y, Van Oost K, Poesen J, Saby N, Gobin A, Vacca A, Quinton J, Auerswald K, Klik A, Kwaad FJPM, Raclot D, Ionita I, Reijman J, Rousseva S, Muxart T, Roxo MJ, Dostal T. 2010. Rates and spatial variations of soil erosion in Europe: a study based on erosion plot data. *Geomorphology* **122**(1–2): 167–177.

Cerdan O, Poesen J, Govers G, Saby N, Le Bissonnais A, Gobin A, Vacca A, Quinton J, Auerswald K, Klik A, Kwaad FPM, Roxo MJ. 2006. Sheet and rill erosion. In *Soil Erosion in Europe*, Boardman J, Poesen J (eds). Wiley: New York; 501–513.

Ciais P, Wattenbach M, Vuichard N, Smith P, Piao SL, Don A, Luyssaert S, Janssens IA, Bondeau A, Dechow R, Leip A, Smith PC, Beer C, Van der Werf GR, Gervois S, Van Oost K, Tomelleri E, Freibauer A, Schulze ED and CarboEurope Synthesis Team. 2010. The European carbon balance. Part 2: Croplands. *Global Change Biology* **16**: 1409–1428.

Compton JS, Herbert CT, Hoffman MT, Schneider RR, Stuut JB. 2010. A tenfold increase in the Orange River mean Holocene mud flux: implications for soil erosion in South Africa. *The Holocene* **20**(1): 115–122.

De Vente J, Poesen J, Arabkhedri M, Verstraeten G. 2007. The sediment delivery problem revisited. *Progress in Physical Geography* **31**(2): 155–178.

Den Biggelaar C, Lal R, Wiebke K, Breneman V. 2004. The global impact of soil erosion on productivity. *Advances in Agronomy* **81**: 1–48.

Dlugoß V, Fiener P, Van Oost K, Schneider K. 2011. Model based analysis of lateral and vertical soil carbon fluxes induced by soil redistribution processes in a small agricultural catchment. *Earth Surface Processes and Landforms* **37**(2): 193–208.

Dymond JR. 2010. Soil erosion in New Zealand is a net sink of CO(2). *Earth Surface Processes and Landforms* **35**(15): 1763–1772.

Forthofer RN, Lee ES, Hernandez M. 2006. *Biostatistics, Second edition: A Guide to Design, Analysis and Discovery*. Elsevier Academic Press: Burlington.

Gesch DB, Verdin KL, Greenlee SK. 1999. New land surface digital elevation model covers the earth. *Eos, Transactions, American Geophysical Union* **80**(6): 69–70.

Goidts E, Van Wesemael B, Van Oost K. 2009. Driving forces of soil organic carbon evolution at the landscape and regional scale using data from a stratified soil monitoring. *Global Change Biology* **15**(12): 2981–3000.

Hall FG, Collatz G, Los S, Brown de Colstoun E, Landis D. 2005. ISLSCP Initiative II, NASA, DVD/CD-ROM. NASA. Available online at: http://islscp2.sesda.com/ISLSCP2_1/html_pages/groups/carbon/river_carbon_flux_hdeg.html.

Hijmans RJ, Cameron SE, Parra JL, Jones PG, Jarvis A. 2005. Very high resolution interpolated climate surfaces for global land areas. *International Journal of Climatology* **25**(15): 1965–1978.

Hilton RG, Meunier P, Hovius N, Bellingham PJ, Galy A. 2011. Landslide impact on organic carbon cycling in a temperate montane forest. *Earth Surface Processes and Landforms* **36**(12): 1670–1679.

Hooke RL. 2000. On the history of humans as geomorphic agents. *Geology* **28**(9): 843–846.

Ito A. 2007. Simulated impacts of climate and land-cover change on soil erosion and implication for the carbon cycle, 1901 to 2100. *Geophysical Research Letters* **34**(9). L09403. DOI: 10.1029/2007GL029342.

Jacinthe PA, Lal R. 2001. A mass balance approach to asssess carbon dioxide evolution during erosional events. *Land Degradation & Development* **12**(4): 329–339.

Jacinthe PA, Lal R, Kimble JM. 2001. Assessing water erosion impacts on soil carbon pools and fluxes. In *Assessment Methods for Soil Carbon*, Lal R, Kimble JM, Follett RF, Stewart BA (eds). Lewis Publishers: Boca Raton, FL; 427–450.

Klein-Goldewijk K, Beusen A, Van Drecht G, De Vos M. 2010. The HYDE 3.1 spatially explicit database of human induced land use change over the past 12,000 years. *Global Ecology and Biogeography* **20**(1): 73–86.

Kummerer K, Held M, Pimentel D. 2010. Sustainable use of soils and time. *Journal of Soil and Water Conservation* **65**(2): 141–149.

Kutsch WL, Bahn M, Heinemeyer A. 2010. *Soil Carbon Dynamics - An Integrated Methodology*. Cambridge University Press: New York.

Lal R. 2003. Soil erosion and the global carbon budget. *Environment International* **29**(4): 437–450.

Lal R. 2007a. World soils and global issues. *Soil & Tillage Research* **97**(1): 1–4.

Lal R. 2007b. Anthropogenic influences on world soils and implications to global food security. *Advances in Agronomy* **93**: 69–93.

Leff B, Ramankutty N, Foley JA. 2004. Geographic distribution of major crops across the world. *Global Biogeochemical Cycles* **18**(1): GB1009. DOI: 10.1029/2003GB002108

Liu SG, Bliss N, Sundquist E, Huntington TG. 2003. Modeling carbon dynamics in vegetation and soil under the impact of soil erosion and deposition. *Global Biogeochemical Cycles* **17**(2): 1074.

Lu H, Prosser IP, Moran CJ, Gallant JC, Priestley G, Stevenson JG. 2003. Predicting sheetwash and rill erosion over the Australian continent. *Australian Journal of Soil Research* **41**(6): 1037–1062.

Lvovich MI, Karasik GY, Bratseva NL, Medvedeva GP, Maleshko AV. 1991. *Contemporary Intensity of the World Land Intercontinental Erosion*. USSR Academy of Sciences: Moscow.

McCarty GW, Ritchie JC. 2002. The Impact of soil movement on carbon sequestration in agricultural ecosystems. *Environmental Pollution* **116**: 423–430.

McElroy B, Wilkinson B. 2005. Climatic control of continental physiography. *Journal of Geology* **113**(1): 47–58.

Milliman JD, Syvitski JPM. 1992. Geomorphic tectonic control of sediment discharge to the ocean – the importance of small mountainous rivers. *Journal of Geology* **100**(5): 525–544.

Montgomery DR. 2007. Soil erosion and agricultural sustainability. *Proceedings of the National Academy of Sciences of the United States of America* **104**(33): 13268–13272.

Myers N. 1993. *Gaia: An Atlas of Planet Management*. Anchor Press: Garden City.

National Land and Water Resources Audit (NLWRA). 2001. Australian Agriculture Assessment 2001. Available online at http://www.anra.gov.au/topics/agriculture/pubs/national/agriculture_soil_erosion.html.

Nearing MA. 1997. A single, continuous function for slope steepness influence on soil loss. *Soil Science Society of America Journal* **61** (3): 917–919.

Nusser SM, Goebel JJ. 1997. The National Resources Inventory: a long-term multi-resource monitoring programme. *Environmental and Ecological Statistics* **4**(3): 181–204.

Oldeman LR, Hakkeling RTA, Sombroek WG. 1991. World Map of the Status of Human-Induced Soil Degradation: An Explanatory Note. ISRIC. Wageningen and United Nations Programme, Nairobi.

Osterkamp WR, Hupp CR, Stoffel M. 2011. The interactions between vegetation and erosion: new directions for research at the interface of ecology and geomorphology. *Earth Surface Processes and Landforms* **37**(1): 23–36.

Parsons AJ, Wainwright J, Powell DM, Kaduk J, Brazier RE. 2004. A conceptual model for determining soil erosion by water. *Earth Surface Processes and Landforms* **29**(10): 1293–1302.

Pimentel D. 2000. Soil erosion and the threat to food security and the environment. *Ecosystem Health* **6**(4): 221–226.

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, Blair R. 1995. Environmental and economic costs of soil erosion and conservation benefits. *Science* **267**(5201): 1117–1123.

Poesen JW, Torri D, Bunte K. 1994. Effects of rock fragments on soil erosion by water at different spatial scales – a review. *Catena* **23**(1–2): 141–166.

Poesen J, Vandaele K, Van Wesemael B. 1996. Contribution of gully erosion to sediment production on cultivated lands and rangelands. In *Erosion and Sediment Yield: Global and Regional Perspectives*. Proceedings of the Exeter Symposium, July 1996. IAHS Publ. no. 236.

Quine TA, Van Oost K. 2007. Quantifying carbon sequestration as a result of soil erosion and deposition: retrospective assessment using caesium-137 and carbon inventories. *Global Change Biology* **13**(12): 2610–2625.

Quinton JN, Govers G, Van Oost K, Bardgett RD. 2010. The impact of agricultural soil erosion on biogeochemical cycling. *Nature Geoscience* **3**(5): 311–314.

Ramankutty N, Foley JA. 1999. Estimating historical changes in global land cover: croplands from 1700 to 1992. *Global Biogeochemical Cycles* **13**(4): 997–1027.

Renard KG, Freimund JR. 1994. Using monthly precipitation data to estimate the R-factor in the revised USLE. *Journal of Hydrology* **157**(1–4): 287–306.

Renwick WH. 2005. Contemporary meanings in physical geography: From what to why? *Geographical Review* **95**(4): 609–611.

Seaber PR, Kapinos FP, Knapp GL. 1987. Hydrologic Unit Maps. US Geological Survey Water-Supply Paper 2294.

Smith SV, Sleezer RO, Renwick WH, Buddemeier RW. 2005. Fates of eroded soil organic carbon: Mississippi basin case study. *Ecological Applications* **15**(6): 1929–1940.

Stallard RF. 1998. Terrestrial sedimentation and the carbon cycle: coupling weathering and erosion to carbon burial. *Global Biogeochemical Cycles* **12**(2): 231–257.

Syvitski JPM, Vorosmarty CJ, Kettner AJ, Green P. 2005. Impact of humans on the flux of terrestrial sediment to the global coastal ocean. *Science* **308**(5720): 376–380.

US Department of Agriculture. 2000. Summary Report: 1997 National Resources Inventory (revised December 2000). Natural Resources Conservation Service, Washington, DC, and Statistical Laboratory, Iowa State University, Ames, Iowa.

U.S. Geological Survey. 1996. GTOPO30 Global 30 Arc-Second Elevation Data Set. EROS Data Center (EDC) Distributed Active Archive Center (DAAC), Sioux Falls. Available online at http://edcdaac.usgs.gov/gtopo30/gtopo30.html.

Valentin C, Agus F, Alamban R, Boosaner A, Bricquet JP, Chaplot V, De Guzman T, De Rouw A, Janeau JL, Orange D, Phachomphonh K, Do Duy Phai, Podwojewski P, Ribolzi O, Silvera N, Subagyono K, Thiébaux JP, Tran Duc Toan, Vadari T. 2008. Runoff and sediment losses from 27 upland catchments in southeast Asia: impact of rapid land use changes and conservation practices. *Agriculture, Ecosystems and Environment* **128**: 225–238.

Van Hemelryck H, Govers G, Van Oost K, Merckx R. 2010. Evaluating the impact of soil redistribution on the in situ mineralization of soil organic carbon. *Earth Surface Processes and Landforms* **36**(4): 427–438.

Van Oost K, Cerdan O, Quine TA. 2009. Accelerated sediment fluxes by water and tillage on European agricultural land. *Earth Surface Processes and Landforms* **34**(12): 1625–1634.

Van Oost K, Quine TA, Govers G, De Gryze S, Six J, Harden JW, Ritchie JC, McCarty GW, Heckrath G, Kosmas C, Giraldez JV, Da Silva JRM, Merckx R. 2007. The impact of agricultural soil erosion on the global carbon cycle. *Science* **318**(5850): 626–629.

Van Rompaey AJJ, Govers G. 2002. Data quality and model complexity for regional scale soil erosion prediction. *International Journal of Geographical Information Science* **16**(7): 663–680.

Verheijen FGA, Jones RJA, Rickson RJ, Smith CJ. 2009. Tolerable versus actual soil erosion rates in Europe. *Earth-Science Reviews* **94**(1–4): 23–38.

Walling DE. 2009. The impact of global change on erosion and sediment transport by rivers: current progress and future challenges. Scientific Paper for the UNESCO-IHP International Sediment Initiative (ISI). UNESCO, France.

Walling DE, Webb BW. 1983. Patterns of sediment yield. In *Background to Palaeohydrology*, Gregory KJ (ed). Wiley: Chichester: 69–100.

Walling DE, Webb BW. 1996. Erosion and sediment yield: a global overview. In *Erosion and Sediment Yield: Global and Regional Perspectives*, Proceeding of the Exeter Symposium, July 1996, IAHS publication No 236.

Wilkinson BH, McElroy BJ. 2007. The impact of humans on continental erosion and sedimentation. *Geological Society of America Bulletin* **119**(1–2): 140–156.

Wischmeier WH, Smith DD. 1978. Predicting rainfall erosion losses - a guide to conservation planning. United States Department of Agriculture, Agricultural Handbook 537.

Worrall F, Rowson JG, Evans MG, Pawson R, Daniels S, Bonn A. 2011. Carbon fluxes from eroding peatlands - the carbon benefit of revegetation following wildfire. *Earth Surface Processes and Landforms* **36**(11), 1487–1498.

Yang DW, Kanae S, Oki T, Koike T, Musiake K. 2003. Global potential soil erosion with reference to land use and climate changes. *Hydrological Processes* **17**(14): 2913–2928.

Ziegler A, Nogareda JS. 2009. Environmental management systems and technological environmental innovations: exploring the causal relationship. *Research Policy* **38**(5): 885–893.

