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Comparison of long-term field-measured and RUSLE-based modelled soil loss in Switzerland

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Abstract

Long-term field measurements to asses model-based soil erosion predictions by water are rare. We have compared field measurements based on erosion assessment surveys from a 10-year monitoring process with spatial-explicit model predictions with the Revised Universal Soil Loss Equation (RUSLE). Robust input data were available for both the mapped and the modelled parameters for 203 arable fields covering an area of 258 ha in the Swiss Midlands. The 1,639 mapped erosion forms were digitized and converted to raster format with a 2 m resolution. A digital terrain model using 2 m resolution and a multiple flow direction algorithm for the calculation of the topographic factors and the support practice factor was available for modelling with the RUSLE. The other input data for the RUSLE were determined for each field. The comparison of mapped and modelled soil loss values revealed a substantially higher estimation of soil loss values from modelling by a factor of 8, with a mean mapped soil loss of 0.77 t/ha/yr vs. modelled soil loss of 6.20 t/ha/yr. However, high mapped soil losses of > 4 t/ha/yr were reproduced quite reliably by the model, while the model predicted drastically higher erosion values for mapped losses of < 4 t/ha/yr. Our study shows the value of long-term field data based on erosion assessment surveys for model evaluation. RUSLE-type model results should be compared with erosion assessment surveys at the field to landscape scale in order to improve the calibration of the model. Further factors related to land management like headlands, traffic lanes and potato furrows need to be included before they may be used for policy advice.

Keywords: soil erosion, field assessment, calibration, RUSLE-based modelling, long-term monitoring, Cambisols

Introduction

Soil erosion models are nowadays used worldwide for estimation of soil erosion by water. Numerous different models are currently available (Jetten et al., 2003; Merritt et al., 2003; Aksoy and Kavvas, 2005; Pandey et al., 2016). The Universal Soil Loss Equation (USLE; Wischmeier and Smith, 1978) and its various derivates such as the Revised Universal Soil Loss Equation (RUSLE; Renard et al., 1997) are still the most common empirical models (Alewell et al., 2019). Borrelli et al. (2018) estimated that more than 90% of soil erosion assessments around the world are derived from USLE-based models. Since USLE estimates long-term mean soil loss and thus the risk of erosion, this model is also very popular for policy advice and measure planning, where it is used as a decision-making instrument for agricultural regulations and guidelines (e.g. Prasuhn et al. (2013) in Switzerland, and Swerts et al. (2019) in Belgium).

The USLE was originally based on an extensive dataset of about 10,000 plot years of erosion measurements under natural rainfall and under standard plot conditions with 9 % slope steepness and 22.1 m slope length (Wischmeier and Smith, 1978). Boix-Fayos et al. (2006) presented a review of the advantages and limitations of the use of test plots to measure soil erosion and determine the parameter values of the USLE. For homogeneous test plots, they found an inadequate representation of natural conditions in landscapes, which are characterized by a higher heterogeneity. Boix-Fayos et al. (2006) conclude that an extrapolation of test plot data leads in most cases to an overestimation of erosion at hillslope and catchment scales. Poesen et al. (1996), Boardman (2006) and Evans (2017) have confirmed this overestimation, which can be two to 10 times higher than measurements from farmers' fields. Nevertheless, mean erosion rates for different countries are derived from test plot data (Cerdan et al., 2006; Auerswald et al., 2009; Guo et al., 2015).

Since Panagos et al. (2015) published a soil erosion map of Europe using a derivate of the RUSLE, called RUSLE 2015, which can be used as a basis for political and economic decisions, a vehement controversy has arisen in the scientific community about the use of RUSLE (Evans and Boardman, 2016a, b; Fiener and Auerswald, 2016; Panagos et al., 2016a, b; Fiener et al., 2020). One question is the level of quality and detail of the input data for the modelling needed to achieve a suitable result. Another is the extent to which an erosion model developed on a test plot scale can represent reality for catchment areas, landscapes or entire nations (Gobin et al., 2004; Batista et al., 2019; Boardman and Evans, 2019; Parsons, 2019). Evans (2017) has stressed that the erosion risk map for Europe by Panagos et al. (2015) does not accurately reflect erosion rates and risk in Britain. Fiener et al. (2020) has also demonstrated, using catchment examples in Bavaria, the Czech Republic and Austria, that there are substantial differences in modelled mean soil loss between regionally adapted USLE models and the European soil erosion risk map by Panagos et al. (2015). Furthermore, the study has been frequently criticized in the above-mentioned literature for applying the RUSLE without any calibration or adjustment and it simplifies the calculations of some factors (C- & P-factor). Empirical USLE-type models are often used, usually with the best input data available to the authors, but mostly without any evaluation, calibration or validation.

Today, individual erosion processes are well understood and can be reproduced relatively accurately with models (Nearing et al., 2017). For process-based models, parameterization is also comparatively simple, and verification or validation can be achieved with experiments (e.g. Aksoy et al. 2020). For complex situations on the scale of catchment areas or regions, both parameterization and validation are much more difficult. Accurate erosion risk modelling presents a number of challenges, including parameterization, validation and resolution of the input data (Gobin et al., 2004; Baggaley and Potts, 2017). On one hand, there is an urgent need for sound and appropriate soil loss data to validate erosion models, and on the other hand, the acquisition of real soil loss rates is a very complex issue. Recently, several authors (Evans et al., 2017; Alewell et al., 2019; Batista et al., 2019; Parsons, 2019) have evaluated various soil erosion assessment methods (plot studies, monitoring and measuring, modelling, use of radionuclides) in order to assess their suitability, validity and scientific robustness as well as their benefits and shortcomings in terms of the reliability of the estimated soil loss rates. They have all concluded that every method has its weaknesses and uncertainties.

Many attempts to evaluate or validate the RUSLE and its predictions exist, but validation of spatial soil loss predictions is generally difficult (Gobin et al., 2004). Therefore, these models are rarely tested in the field. Soil erosion often strongly depends on randomly occurring major events (Prasuhn, 2011; Evans, 2017). Long-term studies are required, because they make it possible to minimize the bias resulting from low-frequency high magnitude effects. Evans and Boardman (2016a) stated: "*RUSLE assessments have not, as far as we know, been compared with field-based assessments*". To the best of our knowledge, only one new study with long-term measured field data, from Steinhoff-Knopp and Burkhard (2018) in Germany, is currently available. They found a significant overestimation of the soil loss by modelling and concluded that modelled erosion did not reflect real conditions very well. Evans and Boardman (2016a) also concluded: "*In Britain the two ways of assessing erosion do not relate well to each other, field-based assessment does not validate (ratify) model assessment*".

In Switzerland, there is a longstanding expertise in soil erosion research on arable land, which allows us to learn from field experiences. Long-term measurements with test plots (Schaub and Prasuhn, 1993), field measurements with sediment traps (Rüttimann et al., 1995), various field mappings (Ledermann et al., 2010; Prasuhn, 2011; Prasuhn, 2020) and several types of modelling (Mosimann and Rüttimann, 2006; Ogermann et al., 2006) have been performed.

Leser et al. (2002) previously concluded, based on 25 years of soil erosion measurements, that only long-term measurements under real field conditions provide a realistic assessment of regional erosion risk.

The first simple USLE-based erosion risk map for the whole of Switzerland was produced by Schaub and Prasuhn (1998). This map has been continuously developed and improved several times (Prasuhn et al., 2007; Prasuhn et al., 2013; Bircher et al., 2019b). However, in a first attempt carried out by Ogermann et al., (2006), to compare the mapping of erosion damage in the study area and the calculation of soil erosion with three different models, higher model erosion rates were determined in the computation than in the determination by mapping.

Based on this experience and the conclusions in the literature that the RUSLE overestimates soil loss rates, we have used a long-term study on the monitoring of soil erosion in farmers' fields (Prasuhn, 2011; 2020) for this paper, in order to evaluate the reliability of RUSLE-based modelling of soil erosion. Accordingly, the aim of this study is an analysis and comparison of mapped soil loss with RUSLE-modelled soil loss. Therefore, we compared high-resolution digitized mapped soil loss data gathered over 10 years for 203 fields in Switzerland with results of erosion modeling using an extensive amount of input data adapted to Swiss conditions. In this comparison, we want to show the accuracy of fit between mapping and modelling. The results of this study are intended to be used in the future to calibrate and adapt modelled erosion rates regionally, in order to utilize the soil erosion risk map for policy advice and decision support.

Methods

Study site

The study site is located about 20 km north-west of Bern, in the Cantone of Bern, where five long-term assessed subareas (Frienisberg (FRI), Suberg (SUB), Lobsigen (LOB), Seedorf

(SEE), Schwanden (SCH)) of soil erosion mapping provide a representative reflection of the agricultural used area in the Swiss plateau between the northern Prealps and the Jurassic Alps. The area is situated between the altitudes of 475 and 720 m a.s.l. The region is characterized by a moderate climate, with an annual average temperature of approximately 8.5°C and annual precipitation of 1,048 mm. Most soils are well drained Cambisols and Luvisols on ground moraines and tertiary molasses; they are mostly sandy loams. Farm size is relatively small, averaging 16.7 ha; the average field size is also small at 1.3 ha. Crop rotations are versatile and usually include temporary grassland of about 22 % in the summer half-year and 37 % in the winter half-year. The five selected study sites consist of 203 fields with crop rotation and about 258 ha, or 645,242 pixels at a resolution of 2 x 2 m.

For LS-factor calculation field blocks were formed consisting of several fields on a slope (see chapter (R)USLE modelled soil loss). This region serves as the comparison area of the field mapping and the RUSLE model (Figure 1). A detailed description of the area has been provided by Prasuhn and Grünig (2001) and Prasuhn (2011).

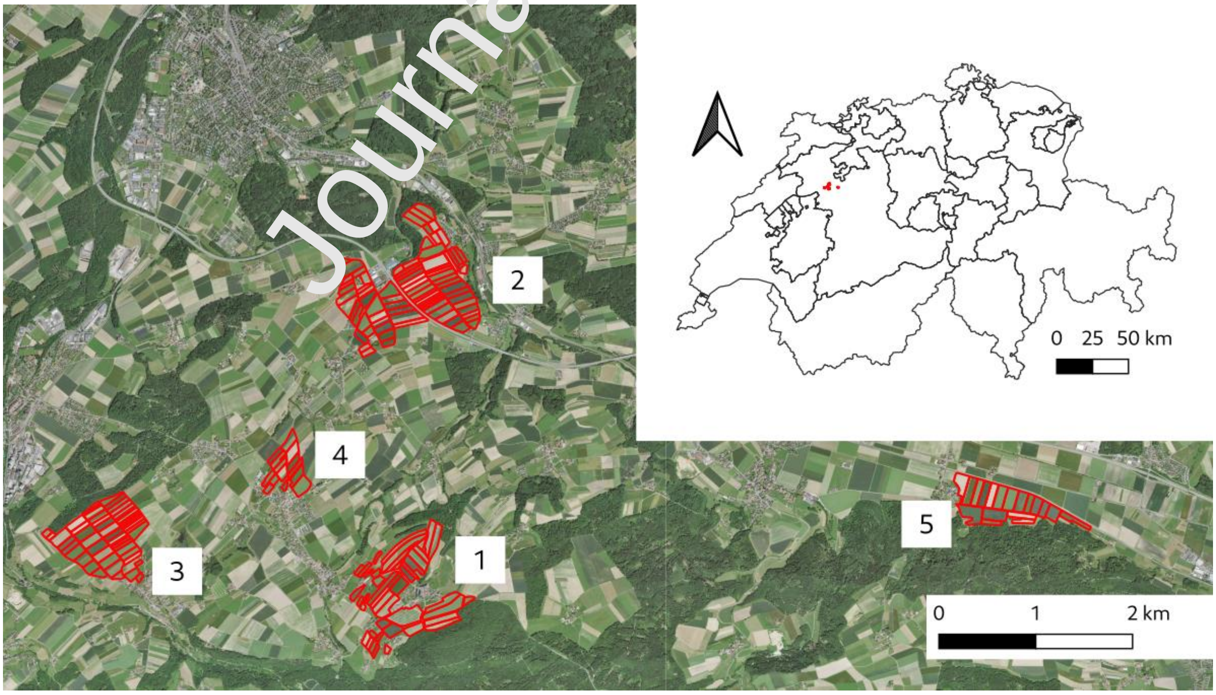

Figure 1: Study site in the Swiss plateau: 203 fields under monitoring from 1997-2007 (red); 1= Frienisberg (FRI), 2= Suberg (SUB), 3= Lobsigen (LOB), 4= Seedorf (SEE), 5= Schwanden (SCH). Insert shows the study site (red) and the boundaries of the Cantons of Switzerland. Background © Swisstopo.

Field mapped soil loss

From autumn 1997 to autumn 2007, event-related erosion damage mapping was carried out during 90 field visits (Prasuhn 2011; 2012). A farmer living in the study area, also a member of the cantonal soil protection department, contacted the surveyor immediately after each precipitation event with visible erosion features. As soon as possible after every precipitation event, after 14 days at the latest, all fields were surveyed (= 18,270 visited fields in 10 years). If several events occurred within a few days of each other, often only the cumulative erosion could be mapped. In 78 out of 90 field observations, soil erosion was mapped on at least one field, in 12 field visits there was no visible erosion damage anywhere although there have been heavy rain events before. Of the total of 18,270 fields visited in 10 years, erosion was mapped on 873 fields or 5% of all visited fields. 89 of the 90 mappings were performed by the same experienced mapper, so that no calibration between different mappers and over time was necessary. This mapper also carried out an accuracy analysis of the mapping method based on repeated independent mappings and statistical analysis (Rüttimann and Prasuhn, 1990). Linear erosion features (rills, ephemeral gullies), sheet-to-linear erosion and sheet erosion were recorded. With linear forms of erosion, the channel lengths and their cross-sections (depth and width) at appropriate intervals alongside the channel were measured following a uniform guideline according to Rohr et al. (1990) for Switzerland and Botschek et al. (2020) for Germany. This method has been used in various other studies (Evans, 2017; Steinhoff-Knopp and Burkhard, 2018; Saggau et al., 2019). The uncertainty of the mapping of linear erosion features amounted to plus/minus 15 % for the experienced mapper and for a careful application (Rüttimann and Prasuhn, 1990). Steinhoff-Knopp and Burkhard (2018) carried out a comparison of multiple measurements from different observers and data derived using structure-from-motion methods and found an error rate of about 15 % for the actual loss rates determined in Lower Saxony, Germany. Soil losses by sheet erosion were estimated visually in a semi-quantitative way, according to Ledermann et al. (2010). Considering data obtained from long-term measurements in the study area with sediment traps (40 sediment traps in 30 fields; measurements over 3 years; Mosimann et al. 1990; Rüttimann et al., 1995), three intensity values ('light' corresponds to 0.5 t/ha; 'moderate' corresponds to 1.0 t/ha; 'severe sheet erosion including small rills < 2 cm depth' corresponds to 1.7 t/ha) were formed. Maximum soil loss rates measured with sediment traps almost never exceeded 2.0 t/ha per event without showing any linear erosion features. Visual indicators observed in the field, such as soil sealing, runoff tracks, small sediment deposits, etc. were combined to determine the level of sheet erosion intensity. Steinhoff-Knopp and Burkhard (2018) have used the same method to estimate sheet erosion. However, we are aware of uncertainties concerning the values for sheet erosion.

The weight of the eroded soil was determined by multiplying the volume of the eroded soil by the bulk density of the topsoil. In the literature, these values range from 0.95 to 1.50 Mg/m^3 (see Prasuhn, 2011). In the present study, a topsoil bulk density of 1.20 Mg/m^3 was assumed for large rills (> 10 cm depth) as well as for rills in tractor lanes and furrows. However, most rills were only a few centimeters deep and erosion occurred immediately after seed bed preparation or sowing, when the topsoil was loosely packed. Therefore, a low bulk topsoil density of 1.00 Mg/m^3 was used for shallow rills (see Ledermann et al., 2008).

Each of the 1,639 erosion forms was plotted as accurately as possible on a field sketch, and the measured soil loss rates recorded in a database. The field sketches were then digitized and quantitatively transferred to a geographic information system (GIS). In order to achieve comparability with the modelled data, the mapped data were converted to a 2 m grid based on the digital elevation model (DEM) of SwissALTI3D (Swisstopo 2015). Rill erosion features were buffered with 8 m on both sides of the linear erosion form in order to take into account the inaccuracy of the field mapping. The soil loss rates of the linear erosion features were distributed weighted with a Multiple Flow Direction (MFD) algorithm in SAGA-GIS (Freeman, 1991) in flow direction. This means that as the length of a rill increases, its soil loss increases. The total amount of soil loss of a digitized erosion feature always corresponds exactly to the amount of mapped soil loss in the database. This procedure was used in order to achieve the best possible representation of the mapped erosion forms, illustrating the spatial pattern of soil loss on a slope. To combine the spatially explicit mapped soil erosion features to high-resolution maps of soil loss all 1,639 digitized erosion forms from the 10 years were finally superimposed onto a map, summed up and divided by 10. The results from the map of the field showed the long-term average soil erosion rate in a 2 m grid (for details see Prasuhn, 2020). The calculation of the soil loss rates for a single field was based on the sum of the soil loss rates of all mapped erosion forms on this field over 10 years. Related to the area of the field and the 10 years of investigation, this results in the mean soil loss in t/ha/yr. This value corresponds to the mean value of the mapped soil loss rates of all pixels of the respective field. Accordingly, the statistical evaluation of the mapped soil loss rates always included all fields and years with and without visible erosion.

(R)USLE modelled soil loss

The RUSLE modelled soil loss is based on the USLE estimation (Wischmeier and Smith, 1978), and consists of six factors, where L is the slope length factor [no unit], S is the slope steepness factor [no unit], K is the soil erodibility factor [t*ha*h/ha/MJ/mm], R is the rainfall and run-off erosivity factor [MJ*mm/ha/h/yr], C is the cover and management factor [no unit], and P is the support practice factor [no unit]. Multiplication of these factors provides the average long-term soil erosion risk in tonnes per hectare and year [t/ha/yr] (Wischmeier and Smith, 1978; Renard et al., 1997).

$$A_{act} = L * S * K * R * C * P$$

A_{act} represents the modelled actual annual soil loss rate [t/ha/yr]. The soil loss rates were modelled as raster GIS layers for the 258 ha arable land of the study area at a resolution of 2 m.

The topographical factor LS was calculated using a 2 m resolution DEM from 2015. The DEM was produced with Light Detection and Ranging (LIDAR) technology with vertical accuracy, at ± 0.5 m (Swisstopo 2015). To calculate the LS-factor, Bircher et al. (2019a) tested 11 different multiple flow algorithms with different convergence settings for the study area. The variation in the LS-factor values was only small. For the present study, we decided to use the Multiple Triangular Flow Direction Algorithms (MTFD) by Seibert and McGlynn (2007) with a convergence value of 1.1. The method of Renard et al. (1997) was used for the S-factor calculation. The L-factor was calculated using the method of Desmet and Govers (1996), which replaced the slope length with the upslope contributing area. The L-factor approach was combined with the multiple flow direction algorithms (MTFD) (for details see Bircher et al., 2019a). LS was calculated as a differentiating LS for each 2 m pixel, which means that the soil loss of the upper increment was subtracted and all different increments along a slope were added. The topographical factor LS was calculated at field block level based on the 2 m DEM and are used as independent flow units. Field blocks divide areas surrounded by artificial or natural borders such as streets, forests, and villages, preventing water flow. A field block can contain several cultivation plots, feature different types of use (arable land, permanent grassland, vineyards, or different field crops), and be cultivated by different farmers. More details of the field block map of Switzerland are available in Bircher et al. (2019b) and Prasuhn et al. (2013).

Detailed soil maps with information on grain size distribution on a scale of 1:25,000, and in some cases 1:10,000, were available. For the calculation of the soil erodibility factor K, for each of the 203 fields, grain size, skeletal content and humus content were additionally determined by an experienced soil expert using a feeling finger test in the field. Laboratory analyses were performed on 21 selected fields (texture, humus content), and the K-factor was determined based on the obtained texture distribution and organic matter content using the formula by Schwertmann et al. (1990). Values for permeability class and soil structure class were estimated. (for details see Prasuhn and Grünig, 2001). A K-factor value was determined for each of the 203 fields. A differentiation within the fields was not possible. However, the fields are relatively small with an average of 1.3 ha and homogeneous with regard to soil properties.

The rainfall erosivity factor R was calculated by Schmidt et al. (2016), using datasets from federal and cantonal sources with a resolution of 1 ha grid cells for Switzerland. For the calculation, 86 rain stations distributed throughout Switzerland with 10-minute rainfall amount values over 20 years were used and interpolated with covariates (DEM, altitudes of snow etc.).

Based on interviews with all farmers and observations during the field visits, the crop rotation and tillage methods (no-till and strip-till; mulch tillage that leaves > 30% of crop residues on the soil surface; reduced tillage which leaves < 30% of the soil surface covered with crop residues; mouldboard or disk plough with soil inversion) were determined for each field for the years 1997 to 2006 in order to calculate the cover and management factor C. The C-factors were determined using a C-factor calculation tool (Mosimann and Rüttimann, 2006), adapted to Swiss conditions. Region-specific dates for growing stages for all crops (sowing, soil cover phases, harvest), area-specific seasonal distribution of rainfall erosivity, various intermediate uses (winter fallow, stubble fallow, freezing or wintering cover crops, etc.) and various correction factors for carry-over effects were taken into account (for details see Prasuhn and Grünig, 2001; Prasuhn, 2022).

In a first step, the support practice factor P was determined in the field on the basis of observations of the tillage direction. If the tillage direction of a field was in the direction of the slope, a P-factor value of 1.0 was used for the whole field or all pixels in this field. In a second step, the effect of cross-slope cultivation was determined as a function of slope gradient and critical slope length for all other fields (Auerswald, 1992; DIN 19708, 2017; Steinhoff-Knopp and Burkhard, 2018). If tillage and cultivation was in a cross-slope direction (along the contour), the P-factor is only effective below a critical slope length (SL). The critical SL was calculated based on the field blocks and the DEM using the following formula:

$$SL_{crit} = 170 * e^{-0.13 * Slope (\%)}$$

For all pixels of a field block exceeding the critical slope length, the P-factor value of 1.00 was used. For all pixels below the critical slope length, the P-factor value was calculated from the DEM as a function of the slope gradient based on the classification according to DIN 19708 (2017) and the formula of Schäuble (2005):

$$P = 0.4 + 0.02 * Slope (\%)$$

Comparison of mapped and modelled soil loss

The comparison was made at different spatial scales:

(a) Pixel: soil loss for 2 m pixels mapped and modelled was compared (n = 645,242). It should be noted that for modelled soil loss, only the LS- and P-factor for 2 m pixels was available. The R-factor was determined at the hectare grid, and the K- and C-factors were determined per field and disaggregated to the 2 m grid.

(b) Fields: for each of the 203 fields, the mean value for the mapped and modelled soil loss and for each factor (LS, R, K, C, P) was used based on the 2 m grid. The range of the number of pixels of the 203 fields varied from 393 to 11,957.

(c) Subareas: for the five subareas, the mean value for the mapped and modelled soil loss based on the 2 m pixels was taken. The range of the number of pixels of the five subareas varied from 37,808 to 241,586.

Results

Compilation of the RUSLE factors

The mean LS-factor based on the 645,242 pixels was 2.23 (Table 1). The subarea FRI was the steepest and has the highest LS value with a mean of 5.10, while the subarea LOB has the lowest with 1.30. As shown in Figure 3, the spatial variability of the LS-factor was also highest in the subarea FRI. In addition to some very steep fields, there were some slope depressions with high LS-factors. For the 203 fields, the LS-value varies between 0.40 and 16.80 (Figure 2).

The average K-factor in the study area was 0.033 t*ha*h/ha/MJ/mm (Table 1). It was lowest in FRI and highest in LOB. The range for the 203 fields included values from 0.017 to 0.042 t*ha*h/ha/MJ/mm (Figure 2).

The average R-factor in the study area was 985 MJ*mm/ha/h/yr and varied between 972 and 1002 MJ*mm/ha/h/yr in the five subareas and between 952 and 1029 MJ*mm/ha/h/yr in the 203 fields (Table 1, Figure 2).

The average C-factor in the study area was 0.099 in the five subareas. The range was large for the 203 fields, with values from 0.036 to 0.247 (Table 1, Figure 2).

The average P-factor in the study area was 0.89. For the 203 fields, the value varied between 0.46 and 1.00 (Table 1, Figure 2).

Table 1: Mean values for the RUSLE-factors of the five subareas and the total area of the 203 fields. FRI = Frienisberg, SUB = Suberg, LOB = Lobsigen, SEE = Seedorf, SCH = Schwanden

Mean	**LS-factor [-]**	**K-factor [t*ha*h/ha/MJ/mm]**	**R-factor [MJ*mm/ha/h/yr]**	**C-factor [-]**	**P-factor [-]**
FRI (n = 138,467)	5.1	0.026	1002	0.108	0.93
SUB (n = 241,586)	1.33	0.033	984	0.088	0.82
LOB (n = 136,229)	1.3	0.037	972	0.103	0.92
SEE (n = 37,808)	2.46	0.037	978	0.073	0.87

SCH (n = 91,152)	1.58	0.033	988	0.118	0.97
Mean five subareas (n = 645,242)	2.23	0.033	985	0.099	0.89

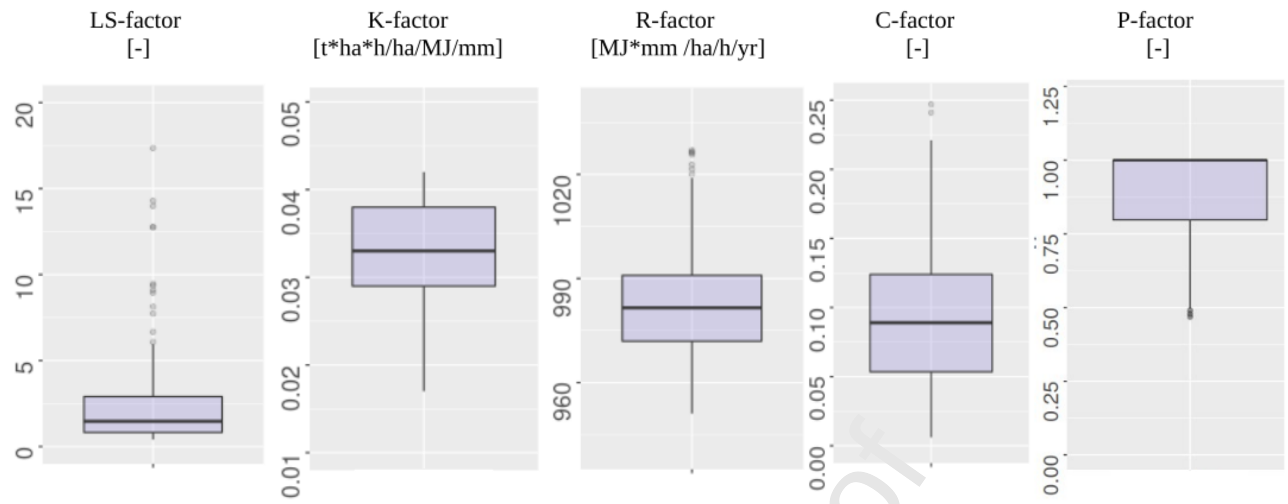

Figure 2: Boxplots for the different factors of the RUSLE (LS, K, R, C, P) as mean values for 203 fields. Boxes indicate median and 25% and 75% quantiles, while whiskers indicate 5% and 95% quantiles (n=203).

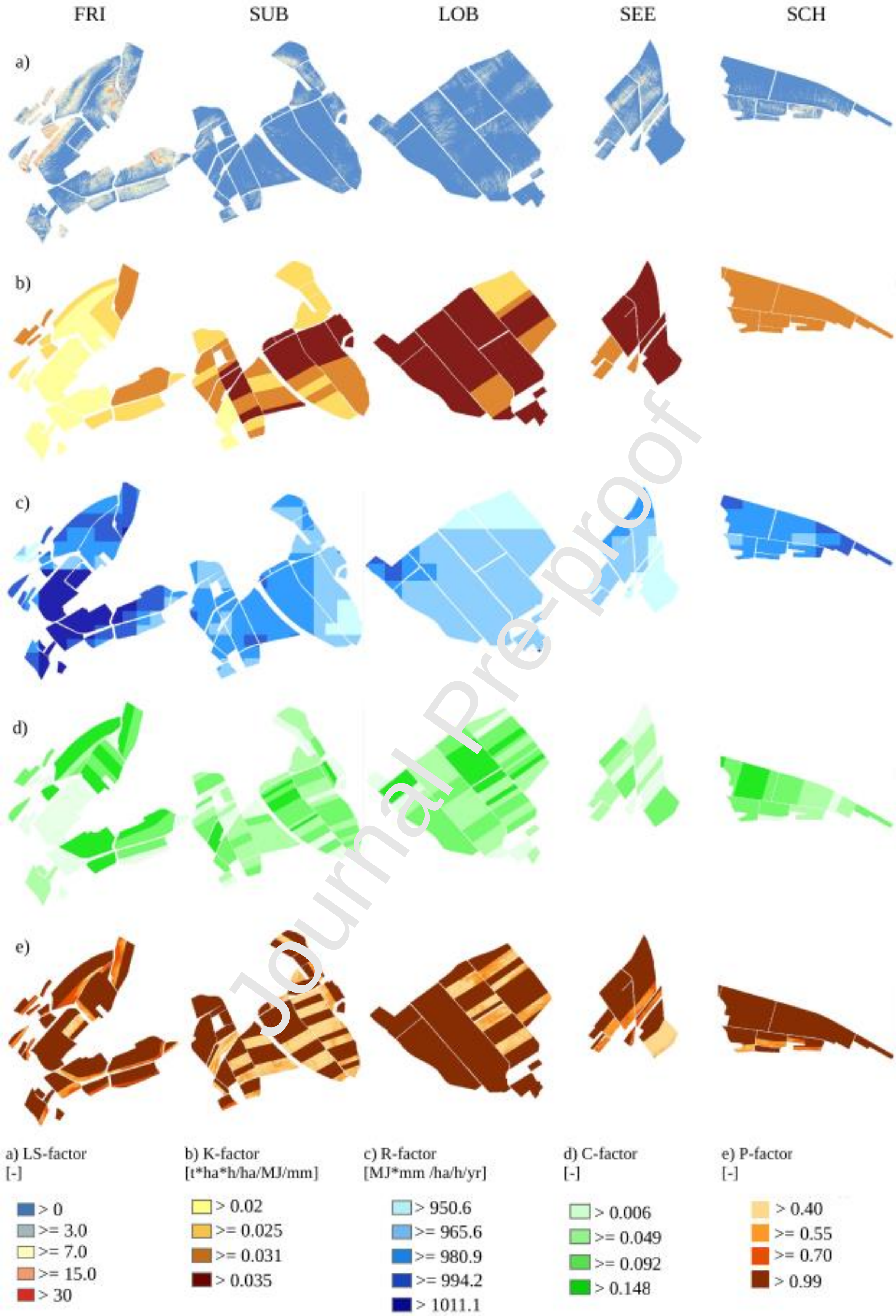

Figure 3: Factor maps of RUSLE, based on a 2 m grid (n= 645,242 pixel): a) topographic factor (LS); b) soil erodibility factor (K); c) rainfall erosivity factor (R); d) cover management factor (C); and e) support practice factor (P), for 203 fields from left to right (FRI = Frienisberg, SUB = Suberg, LOB = Lobsigen, SEE = Seedorf, SCH = Schwanden)

Comparison of mapped and modelled soil loss

For the whole area, based on the analysis of the 645,242 pixel, a mean mapped soil loss of 0.77 t/ha/yr and a modelled actual soil loss of 6.20 t/ha/yr was obtained. Thus, the mean mapped soil loss was only 12% of the modelled loss, meaning modelling estimates higher soil loss by a factor of 8 (Table 2). The subarea FRI had by far the highest mean mapped soil loss (2.00 t/ha/yr), as well as the highest modelled actual (15.30 t/ha/yr) soil loss, meaning the modelled values were higher by a factor of 7.7 compared to mapped soil loss. The other four subareas – SUB, LOB, SEE, and SCH – had significantly lower mapped soil loss, which amounted to about a quarter of the mapped soil loss in FRI. In these four subareas, the modelled soil loss was also significantly higher than the mapped values, ranging from factor 5.9 (SUB) and factor 14 (SEE).

Even though the mean mapped soil loss of the 10 years in the whole area was low at 0.77 t/ha/yr, the maximum mapped annual soil loss in a single field was 96 t/yr or 58 t/ha/yr. Only a few erosion events on a few fields substantially contribute to the total extent of soil loss in the study area. Rill erosion and sheet erosion accounted for 75% and 25% of total soil loss, respectively (Prasuhn, 2011). The mapped soil loss showed a large spatial variability between different areas, between different fields, and within fields (Figure 4). High soil erosion was mainly caused by linear erosion in slope depressions or at the field edges (headlands, tractor lanes), and thus occurred only in certain parts in a field. The modelled soil loss did not represent precisely this small-scale pattern of soil erosion. However, only the LS and P factor could be modelled at a 2 m resolution, while for the other factors R, K and C only averages of each of the 203 fields could be used, although even these factors can vary within a field.

Table 2: Mean mapped and modelled actual soil loss and derived factors for the five subareas, and mean values for the whole area. FRI = Frienisberg, SUB = Suberg, LOB = Lobsigen, SEE = Seedorf, SCH = Schwanden

	Mapped soil loss [t/ha/yr]	**Modelled actual soil loss [t/ha/yr]**	**Factor modelled act/mapped**
FRI (n = 138,467)	2.00	15.30	7.7

SUB (n = 241,586)	0.46	2.70	5.9
LOB (n = 136,229)	0.34	4.10	12.1
SEE (n = 37,808)	0.40	5.60	14.0
SCH (n = 91,152)	0.56	4.90	8.8
Mean total area [n = 645,242 pixel**]**	0.77	6.20	8.0

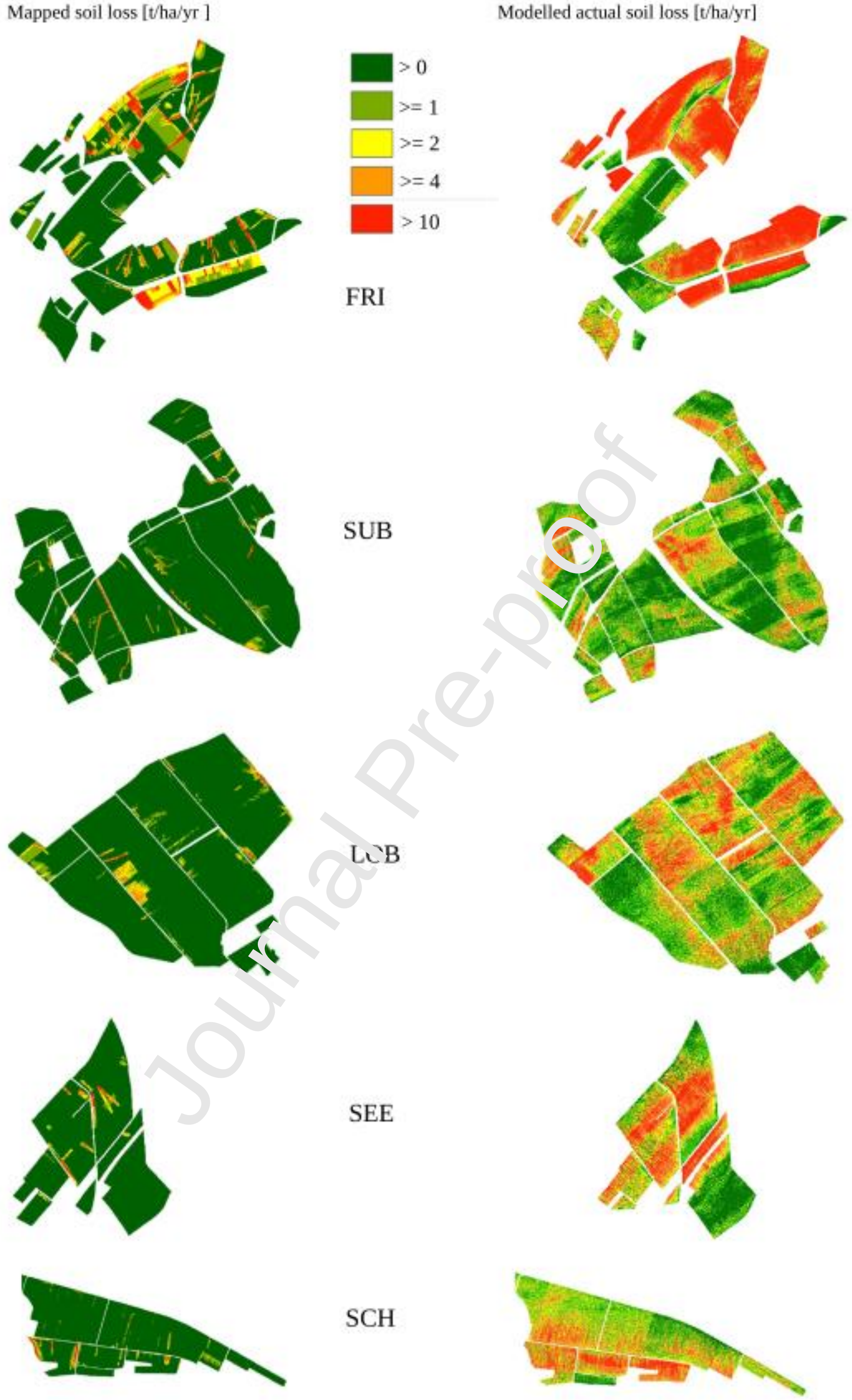

Figure 4: Mapped soil loss over 10 years (left), modelled actual soil loss (right) for the five subareas (FRI = Frienisberg, SUB = Suberg, LOB = Lobsigen, SEE = Seedorf, SCH = Schwanden).

An attempt to correlate mapped and modelled soil loss on a pixel by pixel basis (n = 645,242) did not show any significant correlation (data not shown). This is not surprising, since only the LS- and P-factor could be calculated on the basis of the 2 m pixels, while the other factors were collected at field level (K- and C-factor) or in the hectare grid (R-factor). The soil loss values were classified, based on the guideline values of soil erosion in Swiss legislation. According to the Swiss Ordinance on Soil Protection (Schweizer Bundesrat, 1998), soil erosion is tolerable if it does not exceed a mean of 2 t/ha/yr (for soil depth < 70 cm) or 4 t/ha/yr (for soil depth > 70 cm). Of the mapped soil loss, 90% of the pixels were in class 1 (0-1 t/ha/yr) (Table 3). We have used the values 2 and 4 t/ha/yr of the legal requirements as class boundaries in Table 3 and created some additional classes above and below these legal tolerance values to better show the spatial patterns of soil erosion. From class 1 to 6, the area of the mapped pixels decreased continuously, and the area for class 6 (> 16 t/ha yr) was only 2.1 ha or 0.8% of the total area. Only 3.4% of the pixels had a mapped soil loss of > 4 t/ha/yr. The modelled actual soil loss showed a completely different pattern. The size of area and amount of pixels of the classes 1 to 6 decreased. 33.7% of the pixels and area had a modelled soil loss of > 4 t/ha/yr and 66.3% of the pixels had a modelled soil loss of < 4 t/ha/yr. In contrast to the mapped area, the modelled area contained in class 1 was only 25.1% (64.9 ha), and in class 6 9.1% (23.5 ha) (Table 3).

Table 3: Area proportions and mapped and modelled soil loss for six erosion classes based on the guideline values of soil erosion in Swiss legislation.

	Class 1 0-1 t/ha/yr	**Class 2 1-2 t/ha/yr**	**Class 3 2-4 t/ha/yr**	**Class 4 4-8 t/ha/yr**	**Class 5 8-16 t/ha/yr**	**Class 6 > 16 t/ha/yr**	**Total**
Mapped Soil loss							
Number of pixels [n]	579,253	25,999	18,312	10,236	6,145	5,297	645,242
Area [ha]	231.7	10.4	7.3	4.1	2.5	2.1	258.1
Percent of area mapped	***89.8***	***4.0***	***2.8***	***1.6***	***1.0***	***0.8***	***100***
Mapped soil loss [t/yr]	39.3	14.8	20.5	22.9	27.7	75.3	200.5

Percent of total soil loss mapped	***19.6***	***7.4***	***10.2***	***11.4***	***13.8***	***37.6***	***100.0***
Modelled Soil loss							
Number of pixels [n]	162,352	137,117	128,636	94,669	63,599	58,869	645,242
Area [ha]	64.9	54.8	51.5	37.8	25.4	23.5	258.1
Percent of area modelled	***25.1***	***21.3***	***19.9***	***14.7***	***9.9***	***9.1***	***100***
Modelled soil loss [t/yr]	1121.1	152.9	130.6	74.9	53.2	61.9	1594.8
Percent of total soil loss modelled	***70.3***	***9.6***	***8.2***	***4.7***	***3.3***	***3.9***	***100***
Factor: modelled / mapped soil loss	**28.5**	**10.3**	**6.4**	**3.3**	**1.9**	**0.8**	**7.9**

Despite the fact that almost 90% of the area affected by soil loss was in class 1, the mapped soil loss in these areas only represented 19.6% of the total loss, while 70.3% of the total modelled soil loss belonged to this class (Table 3). On areas with low soil loss of < 4 t/ha/yr (classes 1 - 3) the mapped soil loss was 37.2% of the total mapped soil loss. In contrast, 88.1% of the total modelled soil loss was calculated for these areas. The mismatch between mapped and modelled soil loss was, on average, factor 18.8 for areas with low mapped soil loss. The high mapped soil loss of > 4 t/ha/yr totalled 62.8% of the total mapped soil loss. In the same areas, the modelled soil loss was 11.9% of the total modelled soil loss and was thus of a similar magnitude to the mapped soil loss (mismatch factor 1.5). In total, the higher estimation of soil loss due to modelling was highest in areas with low mapped soil loss (classes 1 - 3) and decreased with high mapped soil loss (classes 4 - 6). In class 6 (> 16 t/ha/yr), the modelled soil loss rate was even slightly lower than the mapped loss rate (Table 3).

As an example, for Frienisberg (FRI), the spatial pattern in Figure 5 shows that in areas where high mapped soil loss (> 4 t/ha/yr) occurred, high soil loss rates were modelled. On the other hand, high soil loss was modelled in many areas where only low soil loss (< 4 t/ha/yr) was mapped.

Figure 5: Mapped soil loss in the soil loss categories >4 t/ha/yr (top left) in the Frienisberg region and modelled soil loss for the corresponding areas (top right). Mapped soil loss in the soil loss categories <4 t/ha/year (bottom left) and modelled soil loss for the corresponding areas (bottom right).

Evaluation of the 203 fields

The evaluation of the 203 fields with regard to mapped and modelled soil loss showed that the mean values of soil losses were significantly above the median values in all cases (Table 4). This demonstrates the large dispersion of the soil loss values and that they were not normally distributed, resp. were left-skewed distributed (Figure 6). The modelled actual soil loss was higher by a factor of 8 than the mapped soil loss (mean values of 203 fields).

With the mapped soil loss, there were some fields with no observed erosion in 10 years and accordingly a mean soil loss of 0.00 t/ha/yr was assumed. In the model calculations with the USLE / RUSLE, some soil loss is always calculated; the modelled minimum value was 0.32 t/ha/yr.

Table 4: Statistical values for the analysis of the 203 fields regarding mapped and modelled actual soil loss.

	Mean	Median	Maximum	Minimum
Mapped soil loss [t/ha/yr] (n = 203)	0.62	0.27	11.18	0.00
Modelled actual soil loss [t/ha/yr] (n = 203)	5.61	3.[illegible]	67.8	0.32

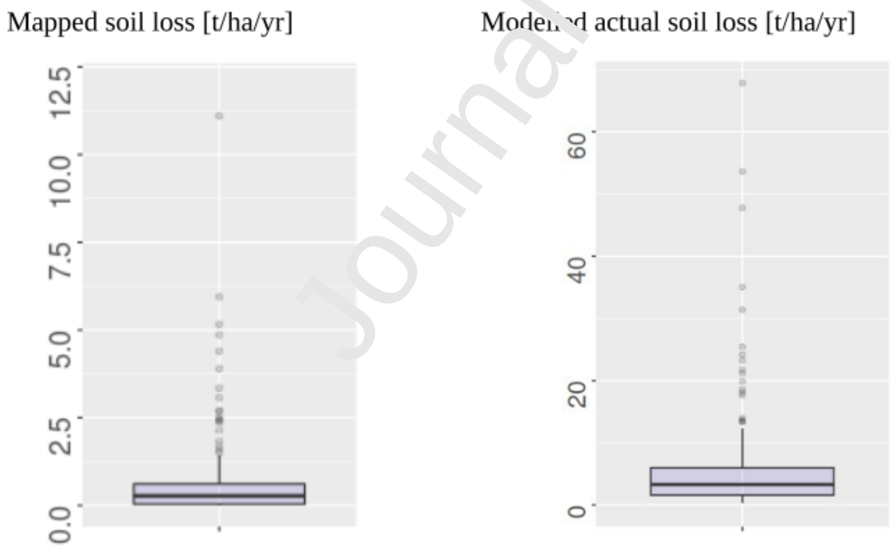

Figure 6: Boxplot of mapped actual soil loss (left) and modelled actual soil loss (right) for the 203 fields. Boxes indicate median and 25% and 75% quantiles, while whiskers 5% and 95% quantiles (n = 203). Note the different scales for the y-axis.

The comparison between mapped and modelled actual soil loss gave a weak relationship (r2 = 0.19) for the area related soil loss values in t/ha/yr when considering the 203 fields (Figure 7). Five out of 203 fields with high mapped mean soil loss > 4 t/ha/yr have relatively high modelled soil loss as well. In contrast, however, there are also fields with no or very low mapped soil loss that show very high modelled soil loss.

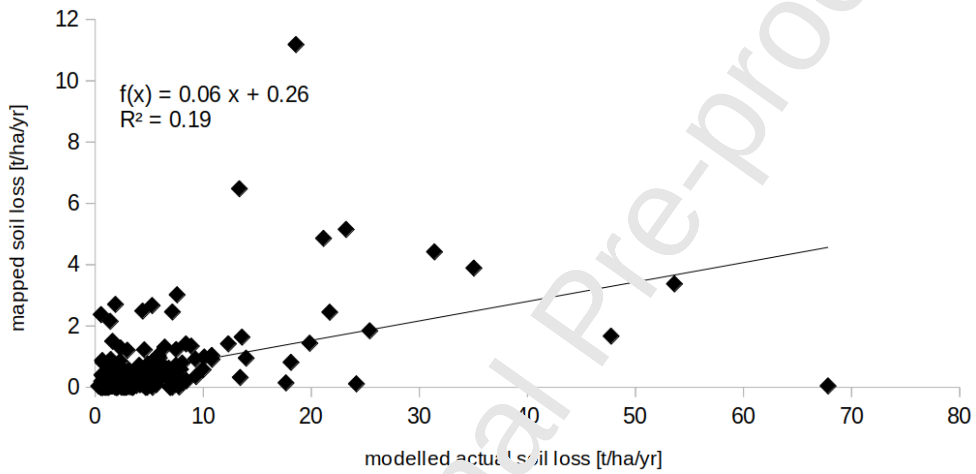

Figure 7: Comparison of mapped and modelled actual soil loss, n = 203 fields.

Discussion

The existing data sets provide the basis to compare spatially distributed mapped and spatially distributed modelled erosion. The 10-year mapping data are of high precision and quality. In 90 field surveys, area-wide mapping was carried out by the same experienced mapper. 1,639 erosion forms were analysed in detail and published (Prasuhn, 2011; 2012; 2020). The field sketches were digitized at the same spatial resolution, using the 2 m grid of the digital elevation model, as the modelling. By using multiple flow algorithms for the digitization of linear erosion features, the spatial pattern of the soil loss on the slopes was implemented in the best way possible (Prasuhn, 2020).

Modelling was also carried out using high-quality and high-resolution input data. Particular attention was paid to the two especially sensitive factors of RUSLE. The C-factor of the USLE is the most sensitive model parameter, followed by the LS-factor (Borrelli et al., 2018; Covelli et al., 2020). The C-factor was calculated for each field over the 10 years on the basis of field mapping and interviews with farmers, using a tool adapted to Swiss conditions and established in Switzerland (Mosimann and Rüttimann, 2006). Region-specific growth stages of all crops, region-specific erosivity values, four different tillage methods, various cover crops, carry-over effects such as temporary ley grass and other correction factors were all taken into account. The calculated mean C-factor of 0.099 in the study area is rather low compared to international studies, due to the high proportion of temporary ley grass in the crop rotation, the use of conservation tillage practices and the cultivation of cover crops (Prasuhn, 2022). Prasuhn (2022) showed that the mean C-factors calculated with the same method over five different periods between 1987 and 2014 in the study area decreased in a similar order of magnitude as the mean mapped soil loss during these periods. The increase in conservation tillage practices was identified as the most important mitigation measure for both modelled C-factors and mapped soil loss.

For the LS-factor, various multiple flow algorithms for this area were compared and analysed in a separate study (Bircher et al., 2019a). With the accurate 2 m DEM and the selected MTFD1.1 from Seibert and Glynn (2007), the LS- and P-factor was determined in the best way possible. Furthermore, for the R-factor, K-factor and P-factor field specific values were available.

Despite the unique data base described above, some critical points should be noted and taken into account when interpreting the results. The mapping period under consideration only lasted 10 years.

Therefore, there are some fields where erosion has not been observed so far, but will occur in the future. In other fields, the average soil loss can also change over time. During the field mapping, it was considered that slight sheet erosion could have been overlooked sometimes. However, this is of little significance for the total amount of soil loss, because there were few major events that determined the total amount of soil loss (Prasuhn, 2011; Evans, 2017; Fiener et al., 2019). The mapping itself is subject to some uncertainties, but an uncertainty analysis of rill erosion is difficult, especially for complex linear erosion forms. According to various studies and our own long-term experience, an error of plus/minus 10-30% can be expected, depending on the complexity of the erosion form and the experience of the mapper (Rüttimann and Prasuhn, 1990; Herweg, 1996; van Oost et al., 2005; Casalí et al., 2006; Ledermann et al., 2010). Since all mapping was performed by the same experienced mapper, it is realistic to expect an error of at most plus/minus 20%. Nevertheless, we are particularly aware of uncertainties concerning the rates for sheet erosion (Ledermann et al., 2010). The conversion from mapped erosion volume (m^3) to mass (tonnes of soil) is another source of uncertainty. The bulk density was assumed to be 1.0 for shallow channels and 1.2 Mg m^{-3} for deeper channels (Prasuhn, 2011). The top soil bulk density increases rapidly over time (Franzluebbers et al., 1995), but we could not take this into account as it would require extensive field measurements. Finally, there are inaccuracies in the spatial representation, which is unlikely to affect the amount of soil loss, but may affect the spatial comparability. Since no Differential Global Positioning System (dGPS) was used for mapping, the positioning accuracy of the individual erosion forms is not exact. This fact was taken into account through the buffering of the linear forms during the digitization, but it explains, nevertheless, why a pixel-wise comparison of mapped and modelled soil loss in the 2 m raster did not match.

Modelling with the RUSLE is also not perfect. "Model predictions are intrinsically more prone to errors than measurements" (Wainwright and Mulligan, 2013). According to our findings, the choice of LS-factor calculation does not have a great impact on the amount of soil erosion for our area with the selected DEM (Bircher et al., 2019a). However, there are some studies that have identified a decreasing soil loss with decreasing DEM resolution. Due to the high-resolution DEM used, the slope becomes more important and the S-factor is higher than with a coarser DEM (Bircher et al., 2019a). Replacing the one-dimensional LS-factor calculation with a two-dimensional LS-factor calculation can lead to an overestimation of the influence of the topography because convergent and divergent flows are better represented in real landscapes. The maximum slope length is another critical parameter. By calculating the LS-factors at field block level, an upper limit is given, since field blocks in Switzerland are relatively small, with an average size of 5.22 ha. When calculating the flow paths using multiple flow algorithms, it is assumed that water and sediment from upslope areas control the soil erosion in downslope pixels. However, it is also assumed that there is a continuous, unimpeded flow of water within the slope or field block. The hydrological connectivity is controlled exclusively by topography; the influence of different vegetation cover is not considered. The L-factor thus represents a theoretically maximum contributing area (Qin et al., 2018). However, if land use varies on a slope, downslope erosion can be reduced by slowing down the runoff. In the study area and in Switzerland in general, agriculture is small-scale with small fields (mean field size = 1.3 ha), so that often several fields with different land use coexist on a slope. Qin et al. (2018) conclude: "rational and reliable soil erosion assessment can only be acquired if the coupled effects of upslope topography and vegetation cover on downslope soil erosion are fully considered in the models". This is not adequately addressed by the C- and P-factor since these factors are independent of the slope. In their study on the Lvergou watershed (China), the new calculation of the LS-factor resulted in a 41% lower average annual soil loss than with the conventional calculation.

However, Borrelli et al. (2018) considered the mapping of soil cover conditions and their spatio-temporal change to be a relevant factor. They developed an enhanced C-factor based on a spatially more accurate and and high temporal resolution assessment of crop dynamics in the medium-size Upper Enziwigger River Catchment in Switzerland. They reported an approximately seven times higher soil loss using traditional C-factor modelling than that predicted by their novel approach. Thus, this may solve the overestimation of factor 8 we reported in this study.

The USLE, resp. RUSLE models, represents long-term average soil loss through sheet and rill erosion. Gully erosion is not included, but this does not occur in the study area. However, the use of multiple flow algorithms allows a more accurate representation of concentrated runoff and the resulting rill and ephemeral gully erosion than one-dimensional approaches (Winchell et al., 2008; Prasuhn et al., 2013). In contrast, mapping has shown that erosion often has specific operational causes such as plough furrows, compacted field headlands, tractor lanes compaction (especially tramlines), and potato furrows (Prasuhn, 2011; Evans, 2017; Steinhoff-Knopp and Burkhard, 2018; Saggau et al., 2019), or is caused by extraneous water inflow from other areas. These conditions cannot be modelled using the RUSLE approach. Tractor lanes compaction and potato furrows are not explicitly spatially considered in the C-factor when modelling with RUSLE. Plough furrows and compacted headlands have not been taken into account in the C-factor calculation so far. In the headlands, the direction of tillage also changes and thus the direction of the tractor lanes. Consequently, the P-factor for the headlands would have to be calculated separately. According to our mapping, they are important in our study area. In future, they should be additionally recorded in the C-factor or a separate C- and P-factor should be developed for the headlands. However, this would require systematic measurements in the headlands in comparison to the main fields for different crops.

Accordingly, the mapped soil loss rates, which additionally captured erosion in the headlands and due to water inflow, would have to be higher than the modelled soil loss where this could not be accounted for. On the other hand, rills often occur only in certain areas of a field and not across the whole field, and do not occur every year. In contrast, with the RUSLE-model rill and sheet erosion are calculated across the whole landscape (Evans and Boardman, 2016b). Therefore, the comparison between mapped and modelled actual soil loss based on the 203 fields shows only a weak to moderate correlation. However, the field size is the ultimate area for decision making by the land users and the unit for enforcement of legal requirements.

The results of this study illustrates that the modelled actual soil loss is drastically higher than the mapped soil loss. Numerous studies have revealed that the USLE / RUSLE tends to overestimate both the severity and the extent of erosion rates. Our finding is also supported by the literature, which suggests that low erosion rates tend to be overestimated and high erosion rates are actually partly underestimated. Risse et al. (1993) found early on that "USLE usually overestimates at sites with relatively low erosion rates and underestimates at sites with higher erosion rates […]. The accuracy in terms of the difference between measured and observed data is better at higher erosion rates." Rapp (1994) confirmed the results for the data set used by Risse et al. (1993), although calculated with the RUSLE. Nearing (1998) listed further examples from the literature that confirm this trend. Di Stefano et al. (2017) tested three different USLE approaches, compared measured and modelled soil loss rates and found that all three USLE approaches tended to overestimate low event soil losses (< 10 t/ha), while two of the approaches tended to underestimate high (> 10 t/ha) annual soil losses. Furthermore, Kinnell (2010) showed that when soils have a low runoff coefficient, USLE overestimates low event soil losses and underestimates high event soil losses. In a study similar to his study, Steinhoff-Knopp and Burkhard (2018) compared 1,355 mapped erosion forms in 86 fields in Germany over 17 years with USLE-based modelling. The mean of the measured actual soil loss was significantly lower than the mean of the modelled actual soil loss. Evans and Brazier (2005), also found a discrepancy between predicted erosion and actual erosion for a number of localities in lowland England and Wales. Abu Hammad (2011) observed in the Central

Palestinian Highlands that the RUSLE-GIS model overestimated the measured soil loss by 21%. Fernandez et al. (2010) investigated post-fire soil losses predicted by the RUSLE in NW Spain and found that RUSLE model predictions overestimated actual annual soil losses without multiplying the R- and C-factors by 0.7 and 0.865, respectively. Finally, Rymszewicz et al. (2015) compared RUSLE application on a national scale against measured sediment yield values in different catchments in Ireland and reported an overestimation of modelled sediment yield values for most (8 from 12) of the selected catchments ranging from 220 - 2839% difference.

In contrast, some studies observed a good agreement between mapped and modelled erosion. Alewell et al. (2019) concluded on the basis of their literature review that "soil loss estimation with USLE-type models are within the order of magnitude compared to measured soil loss rates". Napoli et al. (2016) compared predicted soil loss versus field data measuring soil erosion on 566 fields over six years in Chianti (Italy). They found a good accuracy, with a predicted average soil loss of 13.8 t/ha/yr in comparison to the field measured soil loss of 14.9 t/ha/yr. Fischer et al. (2017) compared predicted event soil loss using the official prediction system in Bavaria (Germany), based on the USLE, and validated the predictions with aerial photo erosion classifications of 8,100 fields. In their study, visually classified and predicted soil loss correlated very highly. Bagarello et al. (2017) tested USLE-derived models to predict the annual maxima of event soil loss. They found evidence that the USLE-based approach was very useful for estimating high soil loss rates. Van Oost et al. (2005) conducted experiments in two catchment areas in Belgium and reported that the total sediment export, derived from erosion surveys, was substantially higher (about 30%) than the measured sediment export at the catchment outlet. Onnen et al. (2019), meanwhile, discovered an underestimation of the modelled sediment yield compared to the measured rill erosion in Denmark. However, in these last two studies, the measurement of erosion in the field and sediment exports and the comparison of the two raises additional difficulties.

Our investigations have shown that the modelled soil loss resulted in much higher rates of soil loss compared to the mapped soil loss, both in the analysis of the classified pixels. (mapped 0.77 vs modelled 6.20 t/ha/yr = factor 8.0). This mismatch is significantly higher than reported by Steinhoff-Knopp and Burkhard (2018) in a comparable study (mapped 0.9 vs modelled 2.94 t/ha/yr = factor 3.3). Other geographical settings and environmental conditions between the study in Lower Saxony and our study are probably responsible for these differences. On the one hand, in Lower Saxony the field size area is larger and the loess soils are more erodible, on the other hand, rainfall erosivity is lower, slopes are less steep and crop rotations are more intensive (no temporary ley grass). Taking into account all uncertainties and errors in mapping and modelling, the huge difference in this study cannot be explained. But even supposing a maximum one-directional error in mapping of minus 30%, i.e. mapped erosion rates were 30% lower, a very conservative estimate always results in an overestimation of factor 6, which is still significantly higher than in the other studies cited.

Since models always deviate from reality, calibration and validation is important. There are no guidelines for appropriate application of models such as USLE; each user applies a different model variation based on the available data. Fiener et al. (2020) compared three different USLE applications and observed substantial differences in the modelled soil loss, with up to 75% difference in the results. Thus, there are also problems and limitations with harmonization and standardization procedures in the application of USLE. Thus, calibration and validation of erosion models remain difficult. Favis-Mortlock (1998) already stated: "Very few models have been validated in any scientifically acceptable sense". This is still true today. Batista et al. (2019) therefore concluded that "calibration seems to be the main mechanism of model improvement".

Models tend to overestimate soil loss when used uncalibrated (Saggau et al., 2019) or do not lead to satisfactory model performance (Bernet et al., 2018).

A general reduction of all modelled soil loss values by factor 8 – or even conservatively by factor 6 – is not appropriate, because the overestimation is not equally distributed across all soil loss classes. High soil losses, which are the relevant losses with regard to soil protection, offsite damage or exceeding of reference values, are reproduced quite well by the model. Ledermann et al. (2010) and Prasuhn et al. (2013) have already demonstrated through plausibility checks that certain fields with a high potential erosion risk often suffer high soil losses in reality. In particular, linear erosion in slope depressions (thalweg erosion) was relatively well captured by the model. These findings have also been confirmed by other studies (Kotremba et al., 2016; Steinhoff-Knopp and Burkhard, 2018). Thus, reducing modelled erosion in general leads to an underestimation of the soil losses in these areas. This is not desirable. On the other hand, the model predicts high erosion for uniformly stretched or convex slopes, which in reality often produce very little erosion. Furthermore, in the study by Steinhoff-Knopp and Burkhard (2018), the difference in the class "no to very low" (<0.2 t/ha/yr) was extraordinarily high for the area proportion, with 1.7% modelled and 59.8% mapped. This is disadvantageous for policy and enforcement; the credibility of the modelled erosion maps decreases, as farmers know their own fields well regarding soil erosion.

From a political point of view, a moderate overestimation of the modelled erosion rates is quite reasonable or even preferable. For raising public awareness, an overestimation is better than an underestimation, especially if only risk maps (e.g. low, moderate, high risk) are presented and absolute soil loss values are omitted. Models are often used by stakeholders to predict soil erosion, and are tools for political decision-makers to design mitigation measures and provide policy advice. In terms of soil erosion prevention, it is of course beneficial to predict soil loss rates that are slightly too high. However, too-high soil erosion rates can also lead to misguided management decisions about where and which mitigation measures should be implemented (van Oost et al., 2005). In contrast, for the implementation of legal guidelines, reliable absolute erosion rates are necessary, because this is ultimately linked to requirements for receiving direct support in form of subsidies for sustainable management or disincentives, including financial sanctions for farmers for management practices resulting in repeated high erosion events. Thus uncalibrated modelled soil loss rates are inadequate in the context of policy advice, planning and decision making (Alewell et al., 2019).

Conclusion

In a study in the Swiss Midlands, we compared mapped soil loss with RUSLE-based modelled soil loss values for 203 fields over 10 years. An extensive amount of input data for both mapping and modelling was available. This was crucial, as the type and spatial resolution of the input data had a significant impact on the output of the envisaged comparison. Our results show a substantial mismatch of soil loss rates between modelling and mapping. The modelled soil erosion was higher than the mapped one by a factor of 8. Even taking into account various uncertainties in soil erosion damage mapping, a more conservative evaluation results in an overestimation of approximately factor 6. Thus, our study supports numerous investigations demonstrating that USLE / RUSLE-based erosion models generally tend to overestimate both the severity and the extent of soil loss rates. However, none of the studies showed the modelled soil erosion rate to be so much higher than the mapped assessment as our study did. Yet, this substantial difference did not occur equally in all areas. Areas with relatively high mapped soil loss rates (> 4 t/ha/yr), which are above the tolerable limit, were adequately covered by the model. However, these areas are comparatively rare in the Swiss Midlands due to the widespread use of conservation tillage practices and mixed crop rotations. In particular, linear erosion by concentrated runoff in slope depressions, which was mapped several times in the same fields at the same locations, was accurately captured by the model using multiple flow algorithms and the contributing area concept. However, on many uniformly stretched or convex slopes with low mapped soil loss rates (< 4 t/ha/yr), the model predicted higher erosion rates than what was assessed and mapped in the field.

The overestimation of the modelled soil losses compared to long-term field verification is mainly driven by the LS-factor and the C-factor calculation – or a combination of both. Therefore, there is a potential or need to improve the model predictions. However, we could also demonstrate that it is difficult to improve the USLE / RUSLE in a generic way, e.g. by reducing the modelled soil loss values by factor 6, because the USLE / RUSLE does not capture some of the factors responsible for mapped soil loss (e.g. traffic lanes, compacted headlands, plough furrows) in a complex landscape. Probably process-oriented models could overcome some of the shortcomings of the USLE / RUSLE, but such models are complex and time consuming for parameterization and therefore not applicable for larger areas or whole regions or countries. Our results only pertain to the study area, which covered a wide range of topographical parameters and a typical crop rotation practised in the Swiss Midlands. The determined factors of overestimation cannot be transferred to other regions without adjustment. Any regionalization of the USLE / RUSLE must be verified. However, our findings, which are based on long-term surveys of erosion assessment within complex landscapes, confirm several plot studies showing that USLE / RUSLE-type models overestimate small soil losses and need to be calibrated.

Mapping can only be done in retrospect after erosive events have occurred, while modelling also allows for predictions or land management scenario analyses. This is the major advantage of the USLE / RUSLE and also the main practical application of the model. As long-term mapping is demanding and only feasible for selected sites, USLE / RUSLE modelling can be applied within a short period of time and with reasonable inputs. On the other hand, if the results of the USLE / RUSLE modelling were calculated and used for practical recommendation to farmers without field assessment, tolerable soil losses in our study area would be exceeded on most fields and almost all farmers would have to draw up mitigation plans and significantly change their farming practices. Yet, the field assessment showed, that the land use on most fields is adapted to the location with small field size and extended areas with temporary ley grass, cover crops and conservation tillage.

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References

Abu Hammad A. (2011). Watershed erosion risk assessment and management utilizing revised universal soil loss equation-geographic information systems in the Mediterranean environments. Water Environ. J. 25(2), 149-162.

Aksoy H. & Kavvas M.L. (2005). A review of hillslope and watershed scale erosion and sediment transport models. Catena 64(2-3), 247-271.

Aksoy H., Gedikli A., Yilmaz M., Eris E., Unal N.E., Yoon J., Kavvas M.L., Tayfur G. (2020). Soil erosion model tested on experimental data of a laboratory flume with a pre-existing rill. J. Hydrol. 581, 124391.

Alewell C., Borrelli P., Meusburger K., Panagos P. (2019). Using the USLE: Chances, challenges and limitations of soil erosion modelling. Int. Soil Water Conserv. Res. 7(3), 203-225.

Auerswald K. (1992). Verfeinerte Bewertung von Erosionsschutzmaßnahmen unter deutschen Anbaubedingungen mit dem P-Faktor der allgemeinen Bodenabtragsgleichung (ABAG). Z. Kulturtech. Landentwicklung 33, 137-144.

Auerswald K., Fiener P., & Dikau R. (2009). Rates of sheet and rill erosion in Germany—A meta-analysis. Geomorphology 111(3-4), 182-193.

Auerswald K., Fischer F.K., Kistler M., Treisch M., Maier H., Brandhuber R. (2018). Behavior of farmers in regard to erosion by water as reflected by their farming practices. Sci. Total Environ. 613, 1-9.

Bagarello V., Di Stefano C., Ferro V., Pampalone V. (2017). Predicting maximum annual values of event soil loss by USLE-type models. Catena 155, 10-19.

Baggaley N., Potts J. (2017). Input factors that should be considered when running a regional soil erosion model at a catchment scale - An application in Scotland. Geoderma Reg. 11, 104-112.

Batista P.V., Davies J., Silva M., Quinton J.N. (2019). On the evaluation of soil erosion models: Are we doing enough? Earth-Sci. Rev. 197. https://doi.org/10.1016/j.earscirev.2019.102898

Bernet D.B., Zischg A., Prasuhn V., Weingartner R., (2018). Modeling the extent of surface water floods in rural areas: Lessons learned from the application of various uncalibrated models. Environ. Model. Softw. 109, 134-151.

Bircher P., Liniger H. P. & Prasuhn V. (2019a). Comparing different multiple flow algorithms to calculate RUSLE factors of slope length (L) and slope steepness (S) in Switzerland. Geomorphology 346, 106850. https://doi.org/10.1016/j.geomorph.2019.106850

Bircher P., Liniger H. P. & Prasuhn V. (2019b). *Aktualisierung und Optimierung der Erosionsrisikokarte (ERK2): Die neue ERK2 (2019) für das Ackerland der Schweiz: Schlussbericht* Bern: Bundesamt für Landwirtschaft.

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

Boardman J., Evans R. (2019). The measurement, estimation and monitoring of soil erosion by runoff at the field scale: Challenges and possibilities with particular reference to Britain. Prog. Phys. Geogr., Earth and Environment. https://doi.org/10.1177%2F0309133319861833

Boix-Fayos C., Martínez-Mena M., Arnau-Rosalén E., Calvo-Cases A., Castillo V., Albaladejo J. (2006). Measuring soil erosion by field plots: Understanding the sources of variation. Earth-Sci. Rev. 78(3-4), 267-285.

Borrelli P., Meusburger K., Ballabio C., Panagos P., Alewell C. (2018). Object-oriented soil erosion modelling: A possible paradigm shift from potential to actual risk assessments in agricultural environments. Land Degrad. Dev. 29(4), 1270-1281.

Botschek J., Billen N., Brandhuber R., Bug J., Deumlich D., Duttmann R., Elhaus D., Mollenhauer K., Prasuhn V., Röder C., Schäfer W., Thiermann A., Unterseher E., Wurbs D. (2021). Bodenerosion durch Wasser – Kartieranleitung zur Erfassung aktueller Erosionsformen. DWA-Regelwerk, Merkblatt DWA-M 921, Deutsche Vereinigung für Wasserwirtschaft, Abwasser und Abfall, Hennef, 112 S.

Casalí J., Loizu J., Campo M.A., De Santisteban L.M., Álvarez-Mozos J., (2006). Accuracy of methods for field assessment of rill and ephemeral gully erosion. Catena 67, 128-138.

Cerdan O., Poesen J., Govers G., Saby N., Le Bissonnais Y., Gobin A., Vacca A., Quinton J., Auerswald K., Klik A.,. Kwaad F. J.P.M., Roxo M.J. (2006). Sheet and rill erosion. In: Boardman J. & Poesen J. (editors), Soil Erosion in Europe, Wiley, pp. 501-513. doi: 10.1002/0470859202.ch38

Covelli C., Cimorelli L., Pagliuca D.N., Molino B., Pianese, D. (2020). Assessment of Erosion in River Basins: A Distributed Model to Estimate the Sediment Production over Watersheds by a 3-Dimensional LS Factor in RUSLE Model. Hydrology 7(1), 13. https://doi.org/10.3390/hydrology7010013

Desmet P. J. J., Govers G. (1996). A GIS procedure for automatically calculating the USLE LS factor on topographically complex landscape units. J. Soil Water Conserv. 51(5), 427-433.

DIN 19708 (2017). Bodenbeschaffenheit - Ermittlung der Erosionsgefährdung von Böden durch Wasser mit Hilfe der ABAG, Deutsches Institut für Normung e.V., 2nd ed., Beuth, Berlin, 26 pp.

Di Stefano C., Ferro V., Pampalone V. (2017). Applying the USLE family of models at the Sparacia (South Italy) experimental site. Land Degrad. Dev. 28(3), 994-1004.

Evans R. (2017). Factors controlling soil erosion and runoff and their impacts in the upper Wissey catchment, Norfolk, England: A ten year monitoring programme. Earth Surf. Proc. Landf. 42(14), 2266-2279.

Evans R., Brazier R. (2005). Evaluation of modelled spatially distributed predictions of soil erosion by water versus field-based assessments. Environ. Sci. Policy 8, 493-501.

Evans R., Boardman J. (2016a). The new assessment of soil loss by water erosion in Europe. Panagos P. et al., 2015. Environ. Sci. Policy 54, 438-447 - A response. Environ. Sci. Policy 58, 11-15.

Evans R., Boardman J. (2016b). A reply to Panagos et al., 2016 Environ. Sci. Policy 59, 53-57. Environ. Sci. Policy 60, 63-68.

Evans R., Collins A.L., Zhang Y., Foster I.D., Boardman J., Sint H., Griffith B.A. (2017). A comparison of conventional and 137Cs-based estimates of soil erosion rates on arable and grassland across lowland England and Wales. Earth-Sci. Rev. 173, 49-64.

Fernández C., Vega J. A., Vieira D. C. S. (2010). Assessing soil erosion after fire and rehabilitation treatments in NW Spain: performance of RUSLE and revised Morgan–Morgan–Finney models. Land Degrad. Dev. 21(1), 58-67.

Favis-Mortlock D. (1998). Validation of field-scale soil erosion models using common datasets, in: Boardman, J., Favis-Mortlock, D. (Eds.), Modelling Soil Erosion by Water. Springer, Berlin, Heidelberg, pp. 89-127.

Fiener P., Auerswald K. (2016). Comment on" The new assessment of soil loss by water erosion in Europe" by Panagos et al., 2015. Environ. Sci. Policy 54, 438-447. Environ. Sci. Policy 57, 140-142.

Fiener P., Wilken F., Auerswald K. (2019). Filling the gap between plot and landscape scale - eight years of soil erosion monitoring in 14 adjacent watersheds under soil conservation at Scheyern, Southern Germany. Adv. Geosci. 48, 31-48.

Fiener P., Dostál T., Krása J., Schmaltz E., Strauss P., Wilken, F. (2020). Operational USLE-Based Modelling of Soil Erosion in Czech Republic, Austria, and Bavaria - Differences in Model Adaptation, Parametrization, and Data Availability. Appl. Sci. 10(10), 3647.

Fischer F., Kistler M., Brandhuber R., Maier H., Treisch M., Auerswald K. (2017). Validation of official erosion modelling based on high-resolution radar rain data by aerial photo erosion classification. Earth Surf. Proc. Landf. 43(1), 187-194.

Franzluebbers A.J., Hons F.M., Zuberer D.A. (1995). Tillage-induced seasonal changes in soil physical properties affecting soil CO2 evolution under intensive cropping. Soil Tillage Research, 34, 41-60.

Freeman T.G. (1991). Calculating catchment area with divergent flow based on a regular grid. Comput. Geosci. 17(3), 413-422.

Gobin A., Jones R., Kirkby M., Campling P., Govers G., Kosmas C., Gentile A. R. (2004). Indicators for pan-European assessment and monitoring of soil erosion by water. Environ. Sci. Policy 7(1), 25-38.

Guo Q, Hao Y, Liu B (2015) Rates of soil erosion in China: A study based on runoff plot data. Catena 124, 68-76. https://doi.org/10.1016/j.catena.2014.08.013.

Herweg K. (1996). Field Manual for Assessment of Current Erosion Damage. SCRP Ethiopia, and Centre for Development and Environment, University of Berne: Berne.

Jetten V., Govers G., Hessel R. (2003). Erosion models: Quality of spatial predictions. Hydrol. Process. 17, 887-900. doi:10.1002/hyp.1168.

Kinnell P. I. A. (2010). Event soil loss, runoff and the Universal Soil Loss Equation family of models: A review. J. Hydrol. 385(1-4), 384-397.

Kotremba C., Trapp M., Thomas K. (2016). Hochauflösende GIS-basierte Bodenabtragsmodellierungen für ausgewählte Agrarstandorte in Rheinland-Pfalz. Bodenschutz 21, 46-56.

Ledermann T., Herweg K., Liniger H.P., Schneider F., Hurni H., Prasuhn V. (2010). Applying erosion damage mapping to assess and quantify off-site effects of soil erosion in Switzerland. Land Degrad. Dev. 21(4), 353-366.

Leser H., Meier-Zielinski S., Prasuhn V., Seiberth C. (2002). Soil erosion in catchment areas of Northwestern Switzerland. Methodological conclusions from a 25-year research programme. Z. Geomorphol. N. F. 46, 35-60.

Merritt W. S., Letcher R. A., Jakeman A. J. (2003). A review of erosion and sediment transport models. Environ. Model. Softw. 18(8-9), 761-799.

Mosimann T., Crole-Rees A., Maillard A., Neyroud J.-A., Thöni M., Musy A. and Rohr W. (1990). Bodenerosion im Schweizerischen Mittelland: Ausmass und Gegenmassnahmen. NFP-Bericht, Nr. 51, Liebefeld, Bern, Schweiz.

Mosimann T. & Rüttimann M. (2006). Dokumentation – Berechnungsgrundlagen zum Frucht-folgefaktor zentrales Mittelland 2005 im Modell Erosion CH (Version V2.02). Terragon, Bubendorf, 30 S.

Napoli M., Cecchi S., Orlandini S., Mugnai G., Zanchi C. A. (2016). Simulation of field-measured soil loss in Mediterranean hilly areas (Chianti, Italy) with RUSLE. Catena 145, 246-256.

Nearing M. A. (1998). Why soil erosion models over-predict small soil losses and under-predict large soil losses. Catena 32(1), 15-22.

Nearing M.A., Lane L.J., Lopes, V.L. (2017). Modeling soil erosion. In: Soil erosion research methods. Routledge, 127-158.

Ogermann P., Hebel B., Prasuhn V., Weisshaidinger R. (2006). Erfassung von Bodenerosion in der Schweiz: vergleichende Anwendung verschiedener Methoden und Beurteilung ihrer Eignung für den Vollzug der Bodenschutzgesetzgebung. Geogr. Helv. 61(3), 209-217.

Onnen N., Heckrath G., Stevens A., Olsen P., Greve M.B., Pullens J.W., Kronvang B., Van Oost K. (2019). Distributed water erosion modelling at fine spatial resolution across Denmark. Geomorphology 342, 150-162.

Panagos P., Borrelli P., Poesen J., Ballabio C., Lugato E., Meusburger K., Montanarella L., Alewell C. (2015). The new assessment of soil loss by water erosion in Europe. Environ. Sci. Policy 54, 438-447.

Panagos P., Borrelli P., Poesen J., Meusburger K., Ballabio C., Lugato E., Montanarella L., Alewell C. (2016a). Reply to the comment on "The new assessment of soil loss by water erosion in Europe" by Fiener & Auerswald. Environ. Sci. Policy 57, 143-150.

Panagos P., Borrelli P., Poesen J., Meusburger K., Ballabio C., Lugato E., Montanarella L., Alewell C. (2016b). Reply to "The new assessment of soil loss by water erosion in Europe. Panagos P. et al., 2015. Environ. Sci. Policy 54, 438-447. A response" by Evans and Boardman [Environ. Sci. Policy 58, 11-15]. Environ. Sci. Policy 59, 53-57.

Pandey A., Himanshu S.K., Mishra S.K., Singh V.P. (2016). Physically based soil erosion and sediment yield models revisited. Catena 147, 595-620.

Parsons A.J. (2019). How reliable are our methods for estimating soil erosion by water? Sci. Total Environ. 676, 215-221.

Poesen J. W., Boardman J., Wilcox B., Valentin C. (1996). Water erosion monitoring and experimentation for global change studies. J. Soil Water Conserv. 51(5), 386-390.

Prasuhn V. (2011). Soil erosion in the Swiss midlands: Results of a 10-year field survey. Geomorphology 126(1-2), 32-41.

Prasuhn V. (2012). On-farm effects of tillage and crops on soil erosion measured over 10 years in Switzerland. Soil Tillage Res. 120, 137-146.

Prasuhn V. (2020). Twenty years of soil erosion on-farm measurement: annual variation, spatial distribution and the impact of conservation programmes for soil loss rates in Switzerland. Earth Surf. Proc. Landf. 45(7), 1539-1554. https://doi.org/10.1002/esp.4829

Prasuhn V. (2022). Experience with the assessment of the USLE cover-management factor for arable land compared with long-term measured soil loss in the Swiss Plateau. Soil & Tillage Research 215. https://doi.org/10.1016/j.still.2021.105199

Prasuhn V., Grünig K. (2001). Evaluation der Ökomassnahmen – Phosphorbelastung der Oberflächengewässer durch Bodenerosion. Schriftenreihe der FAL Nr. 37, Zürich-Reckenholz, 152pp.

Prasuhn V., Liniger H. P., Hurni H., Friedli S. (2007). Bodenerosions-Gefährdungskarte der Schweiz. Agrarforschung 14(3), 120-127.

Prasuhn V., Liniger H.P., Gisler S., Herweg K., Candinas A., Clément J.P. (2013). A high-resolution soil erosion risk map of Switzerland as strategic policy support system. Land Use Policy 32, 281-291.

Qin W., Guo Q., Cao W., Yin Z., Yan Q., Shan Z., Zheng F. (2018). A new RUSLE slope length factor and its application to soil erosion assessment in a Loess Plateau watershed. Soil Tillage Res. 182, 10-24.

Risse L. M., Nearing M. A., Laflen J. M., Nicks A. D. (1993). Error assessment in the universal soil loss equation. Soil Sci. Soc. Am. J. 57(3), 825-833.

Rapp J. F. (1994). Error assessment of the Revised Universal Soil Loss Equation using natural runoff plot data. Unpublished M.S. Thesis, University of Arizona, Tucson, Arizona, 90 pp.

Renard K.G., Foster G.R., Weesies G.A., McCool D.K., Yoder D.C. (1997). Predicting Soil Erosion by Water, a Guide for Conservation Planning with the Revised Universal Soil Loss Equation (RUSLE). Agriculture Handbook 703: USDA. ARS. Washington, DC.

Rohr W., Mosimann T., Bono R., Rüttimann M. and Prasuhn V. (1990). Kartieranleitung zur Aufnahme von Bodenerosionsformen und -schäden auf Ackerflächen. Legende, Erläuterungen zur Kartiertechnik, Schadensdokumentation und Fehlerabschätzung. Materialien zur Physiogeographie, Heft 14, Basel, Schweiz.

Rüttimann M., Prasuhn V. (1990). Möglichkeiten der Fehlerabschätzung und Optimierung der Erosionsschadenskartierung. - Materialien z. Physiogeographie 14, Basel, 41-49.

Rüttimann M., Schaub D., Prasuhn V., Rüegg W. (1995). Measurement of runoff and soil erosion on regularly cultivated fields in Switzerland - some critical considerations. Catena 25(1-4), 127-139.

Rymszewicz A., Mockler E., O'Sullivan J., Bruen M., Turner J., Conroy E., Kelly-Quinn M., Harrington J., Lawler D. (2015). Assessing the applicability of the Revised Universal Soil Loss Equation (RUSLE) to Irish Catchments. Proc. Intern. Assoc. Hydrol. Sci. 99-105.

Saggau P., Kuhwald M., Duttmann R. (2019). Integrating Soil Compaction Impacts of Tramlines into Soil Erosion Modelling: A Field-Scale Approach. Soil Systems 3(3), 51.

Schäuble H. (2005). AVErosion 1.0 für ArcView 3.x. Analyse der Bodenerosion nach den Modellen USLE und MUSLE87. Erweiterung für ArcView 3.x und den Spatial Analyst 1.x https://www.terracs.com/infosysteme/gis-software/averosion/

Schaub D., Prasuhn V. (1993). The role of test plot measurements in a long-term soil erosion research project in Switzerland. In: Wicherek (ed.), Farm land erosion in temperate plains environment and hills. Elsevier Science Publishers B. V., pp. 111-123.

Schaub D., Prasuhn V. (1998). A map on soil erosion on arable land as a planning tool for sustainable land use in Switzerland. Adv. GeoEcology 31, 161-168.

Schmidt S., Alewell C., Panagos P., Meusburger K. (2016). Regionalization of monthly rainfall erosivity patterns in Switzerland. Hydrol. Earth Syst. Sci. 20(10), 4359-4373.

Schwertmann U., Vogl W., Kainz M. (1990). Soil erosion by water. Verlag Eugen Ulmer, Stuttgart (in German).

Seibert J., McGlynn B. L. (2007). A new triangular multiple flow direction algorithm for computing upslope areas from gridded digital elevation models. Water Res. Res. 43(4).

Steinhoff-Knopp B., Burkhard B. (2018). Mapping Control of Erosion Rates: Comparing Model and Monitoring Data for Croplands in Northern Germany. One Ecosystem 3, e26382. https://doi.org/10.3897/oneeco.3.e26382

Swerts M., Deproost P., Oorts K., Buyle S., Vandekerckhove L. (2019). 15 years of experience with the use of detailed erosion maps in soil erosion policy in Flanders, Belgium. In: Global Symposium on Soil Erosion (GSER19), 15 - 17 May 2019 at FAO. Working Documents, 502-506.

Swisstopo (2015). swissALTI 3D Ausgabebericht 2015. Bundesamt für Landestopografie Swisstopo, Wabern (2015).

Van Oost K., Govers G., Cerdan O., Thauré D., Van Rompaey A., Steegen A., Nachtergaele J., Takken I., Poesen J. (2005). Spatially distributed data for erosion model calibration and validation: The Ganspoel and Kinderveld datasets. Catena 61(2-3), 105-121.

Wainwright J., Mulligan M., (2013). Environmental Modelling: Finding Simplicity in Complexity: Second Edition, doi:10.1002/9781118351475.

Winchell M.F., Jackson S.H., Wadley A.M., Srinivasan R. (2008). Extension and validation of a geographic information system-based method for calculating the revised universal soil loss equation length-slope factor for erosion risk assessments in large water- sheds. J. Soil Water Conserv. 63, 105-111.

Wischmeier W.H., Smith D.D. (1978). Predicting rainfall erosion losses - a guide to conservation planning, agriculture handbook No. 537. USDA/Science and Education.

Highlights

- We compared mapped soil loss from an area of 258 ha during 10 years with RUSLE-based modelled soil loss values.
- The mean mapped soil loss was 0.77 t/ha/yr, while the modelled was 6.20 t/ha/yr.
- The comparison of mapped and modelled soil loss show a substantial overestimation by modelling in the order of a factor 8.
- Areas with high mapped soil loss rates >4 t/ha/yr were modelled quite accurately by the model.
- Areas with low mapped soil loss rates <4 t/ha/yr were drastically overpredicted by the model.