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A new strategy to assure compliance with soil loss tolerance at a regional scale

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ABSTRACT

The relevant erosive effects of extraordinary rainfall events due to climate change require establishing soil conservation strategies to prevent damages due to hydrogeological instability. The “tolerable” soil loss, i.e., the maximum soil loss compatible with sustainable soil use, represents a quantitative target to establish the effectiveness of actions to control soil erosion. In this paper, a new approach to defining the condition corresponding to a tolerable soil loss is proposed. At first, using the statistical analysis of the measured annual values of the rainfall erosivity factor, the cover and management factor C_T, for which the maximum tolerable soil loss is equal to the annual soil loss of given return period T, is defined. Then, for the Sicilian region, a relationship between the C_T factor obtained for $T = 1000$ years and the mean annual value of the rainfall erosivity factor, R, is established. For a given value C of the cover and management factor, this relationship allows for the establishment of the corresponding mean annual rainfall erosivity factor, named $R_{land\text{-}use}$. The result $C \leq C_T$ for $T = 1000$ years is obtained for areas with $R \leq R_{land\text{-}use}$, and the compliance with soil loss tolerance is then assured. Conversely, for areas characterized by $R > R_{land\text{-}use}$, the reduction of C to a value less than C_T for $T = 1000$ years is required to obtain a tolerable soil loss condition. Finally, for the Sicilian region, the overlay between the C spatial distribution for arable lands (mainly cereals and legumes) and areas covered by vineyards, derived from the land use map, and the C_T spatial distribution allowed to define areas in which tolerable soil loss conditions occur or soil conservation strategies are required.

1. Introduction

The soil thematic strategy of the European Commission (EC 2006) stated that “*soil is essentially a non-renewable resource and a very dynamic system which performs many functions and delivers services vital to human activities and ecosystems survival*”. Soil is considered a non-renewable resource as its formation process is very slow, requiring from hundreds to thousands of years to form a few centimeters of topsoil when normal agricultural conditions occur (Kendall and Pimentel, 1994) (Bazzoffi, 2009), while its destruction due to erosion phenomena is rapid and can be accelerated by human activities (Dregne, 1987; Raclot et al., 2018).

Soil loss by water erosion is a key element as degraded lands are sensitive to the loss of nutrients and organic matter in the topsoil, soil productivity reduction, pollution of water bodies, destruction of wildlife habitats, and decrease of biodiversity (Food and Agricultural Organization, 2019). Soil erosion rate is greater than that of soil formation in many areas across the European Union and the economic loss associated with soil erosion processes, estimated assuming a restoration cost of $20 per tonne, is almost equal to $20 billion per year (Panagos et al., 2015). Panagos et al. (2018) estimated that in Europe a land area of 12 million hectares is affected by soil erosion phenomena, which result in a loss in crop productivity of €1.25 billion. Land degradation costs (Sartori et al., 2019) are generally estimated considering productivity loss due to soil erosion. This ’first-order’ cost evaluation derives from the land productivity loss (in-site effect) (Martínez-Casasnovas and Ramos, 2006; Erkossa et al., 2015; Hein, 2007) and is estimated by the product of loss in crop production (tonnes) and the average market price ($/tonnes). This approach neglects the ’second round’, which considers other effects beyond the primary resource (off-site effects) such as the transport and deposition of eroded particles and associated nutrients.

In the period 2000–2010, the Common Agricultural Policy adopted soil conservation strategies that reduced soil loss by 20 % in cultivated areas (Panagos et al., 2015). However, the increased frequency of

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extraordinary rainfall events, such as those recently occurred in Luxembourg and Germany (Fekete and Sandholz, 2021), demonstrated that landscape protection strategies are often inadequate and interventions to limit and prevent damages due to hydrogeological instability have been insufficient. Soil conservation strategies mitigate agricultural productivity losses in the areas which are affected by soil erosion phenomena ("on site" effects) and reduce indirect effects ("off-site") related to the transport and deposition of sediments and associated nutrients along the hillslopes and in the river network. As a consequence, the reduction of soil erosion processes protects agriculture and forestry production and reduces the use of fertilizers and other auxiliary substances (Grunewald and Naumann, 2015; Syrbe et al., 2018).

In this context, the introduction of regulating services, named Ecosystem Services (ES) and defined as (Kandziora et al., 2013) "*the benefits people obtain due to the regulation of natural processes such as water purification and erosion control*", plays a relevant role. The ES are related to the main ecosystem processes (such as water regulation, soil loss, and pesticide control), aim to control or modify abiotic and biotic factors, and are associated with the need to mitigate the impacts of current and future environmental hazards (Xu et al., 2020). The payment of subsidies, designed to limit soil erosion and restore degraded soils, is a controversial topic as it is not clear if their application over decades has determined net benefits (Kumar, 2010; Robinson et al., 2012; van Leeuwen et al., 2019). Payment for ecosystem service (PES) continues to attract attention of scholars and stakeholders as an efficient tool to design policies and incentive mechanisms (Swinton et al., 2007; Jónsson and Davíðsdóttir, 2016) able to encourage environmental conservation strategies (Papanastasis et al., 2015; Smith et al., 2015; Zhang et al., 2013). PES can be considered as a cost associated with environmentally respectful soil use and as a tool to compensate for the reduced crop productivity to assure an environmental service. The implementation of ecosystem services may be limited by legal property rights, and private land users should be encouraged to adopt sustainable land uses targeted at reducing the on-site and off-site effects of their activities (Mikša et al., 2020). As an alternative, the cost-benefit analysis of soil conservation strategies requires quantifying the costs due to the depleted resources and their unavailability for other uses and the negative impacts of soil erosion control such as the increased use of herbicides applied in reduced-tillage systems. The need of estimating the costs associated with soil conservation strategies requires a spatial targeting of soil loss reduction to establish where to pay (Guo et al., 2020). In other words, spatial targeting requires the selection of the most suitable areas where soil conservation strategies and PES policies should be implemented.

In the framework of the European Green Deal (EC COM, 2019), a soil loss target is required to adopt soil conservation strategies, evaluating the effectiveness of the adopted strategies, and stating PES policies. The efficiency of the soil conservation strategy associated with the ecosystem service can be measured by the distance of the actual soil loss to the target value (Johnson, 2005; Bagarello and Ferro, 2006; Li et al., 2009). The need for soil conservationists to determine this quantitative target led to establishing a tolerable soil loss, *TSL* (Schmidt et al., 1982; Johnson, 1987). The use of this concept started in 1962 when the U.S. Soil Conservation Service defined *TSL* as "*the maximum level of soil erosion that will permit a high level of crop productivity to be sustained economically and indefinitely*". Values of soil loss tolerance ranging from 4.5 to 11.2 t ha^{-1}year^{-1} were proposed for many American soils (Wischmeier and Smith, 1978; Renard et al., 1997; Lal, 2001). The U.S. Soil Conservation Service defined, by a sharing action among agronomists, geologists, and soil conservationists, a *TSL* value of 12.5 t ha^{-1} year^{-1}. Wischmeier and Smith (1978) suggested a value of 11.2 t ha^{-1} year^{-1}. In the European context (European Environment Agency, 1998) a value of *TSL* ranging from 1 t ha^{-1} year^{-1}, for shallow sandy soils, to 5 t ha^{-1} year^{-1}, for deep well-developed soil, is suggested. The variability of tolerable soil loss values can be related to the specific factors driving the soil formation and Huber et al. (2008) proposed higher values of *TSL* for southern Europe than those for northern Europe. Specific values of *TSL* are adopted in Switzerland (1–2 t ha^{-1} year^{-1}) and in Norway (2 t ha^{-1} year^{-1}) (Verheijen et al., 2009). The Organisation for Economic Co-operation and Development (OECD) suggested a soil erosion rate lower than 6 t ha^{-1} year^{-1}. Bazzoffi (2009) proposed a soil loss tolerance value that balances the need of reducing soil erosion rate with that of allowing agricultural land use. The *TSL* values proposed by Bazzoffi (2009) never exceed 3 t ha^{-1} year^{-1}. For considering the in-site and off-site effects, Larson (1981) proposed a two-level definition in which the lower limit TSL_1 is related to the need for maintaining the in-site soil productivity level while the upper limit TSL_2 is associated with the control of off-site effects such as water pollution, gully erosion, and reservoir sedimentation.

Even if the soil loss tolerance is a generally accepted concept and a useful conservation-planning tool (Toy et al., 2002), its quantitative estimation continues to be debated since a robust scientific method for its estimation is still lacking. A shared value or a common procedure for its estimation is not available in the European environment. Panagos et al. (2015) and Syrbe et al. (2018) assessed soil loss in the European Union by the Revised Universal Soil Loss Equation (RUSLE) and suggested using crop rotation and applied agricultural management practices as factors useful to estimate the soil coverage conditions.

In this paper, a new method to estimate soil loss tolerance at a regional scale is proposed. A procedure for defining a target value of the crop and management factor of the Universal Soil Loss Equation (USLE) corresponding to a tolerable condition is proposed. The proposed method is applied to the Sicily region (south Italy) for two significant land uses (arable land, i.e., mainly cereals and legumes, and vineyards). The overlay between the land use map and the spatial distribution of the target value of the crop and management, originated from the iso-erosivity map, is used to define the areas in which soil loss tolerable conditions occur or soil conservation strategies are required.

2. Methods

2.1. Evaluating tolerable soil loss (TSL)

According to Wischmeier and Smith (1978), the combined use of USLE and *TSL* value (t ha^{-1} year^{-1}) is useful to establish soil conservation strategies:

$$\frac{TSL}{RKS} = LCP \tag{1}$$

in which R (MJ mm ha^{-1}h^{-1} year^{-1}) is the mean annual value of the rainfall erosivity factor, K (t ha^{-1} per unit of R) is the soil erodibility factor, S is the slope steepness factor, L is the slope length factor, C is the cover and management factor, and P is the support practice factor. Eq. (1) establishes that L, C, and P factors can be modified to obtain a soil loss equal to the *TSL* value. Use of Eq. (1) to design soil conservation strategies implies representing the climatic variability by a mean value, such as the rainfall erosivity factor R, and this choice is acceptable in the years in which non-intensive rainfall erosive events occur, while it is critical when relevant erosive events produce intolerable sediment yield values (Gonzales-Hidalgo e al., 2010, 2012). To go beyond a soil conservation design using a "mean climatic condition", many authors (Bagarello et al., 2010; Strohmeier et al., 2016) suggested considering large storms. This last choice must overtake the difficulties related to the availability of historical sequences of soil loss measurements, characterized by a sample size relevant to develop a frequency analysis and have significant soil loss estimates of a given return period (Mannaerts, 1992; Hession et al. 1996; Baffaut et al. 1998; Mannaerts and Gabriels 2000; Bagarello et al. 2010, 2011). Larson et al. (1997) proposed as target the soil loss corresponding to a return period ranging from 10 to 20 years.

Bagarello et al. (2011) used soil loss measurements carried out on plots of different lengths, λ (11, 22, 33, and 44 m) to investigate the statistical distribution of soil loss. These authors proposed to estimate the annual soil loss of given return period T for a bare soil ($C = 1$), $SL_{a,T}$, by the following equation:

$$SL_{a,T} = R_{a,T}KLS = x_T RKLS \tag{2}$$

in which $R_{a,T}$ is the annual rainfall erosivity factor, R_a, having a return period of T years and $x_T = R_{a,T}/R$ is the quantile of the given return period of the variable $x = R_a/R$.

Bagarello et al. (2010, 2011, 2015) performed the frequency analysis of soil loss measurements collected in the period 1999–2012 in the Sparacia (Sicily) experimental area. The test bare plots varied in length from 0.25 m to 44 m, had steepness of 14.9 %, and at least two replicates for each length were available. The variable x_{SL}, which was equal to the ratio between the maximum measured event soil loss for each plot length and recording year and the mean soil loss measured for that event in all plots having the same length, was considered to develop the frequency analysis. However, the findings of Bagarello et al. (2010, 2011) ensured that the parameters of the probability distribution of the annual maximum soil loss and the quantiles of given return period T could be estimated using all the event soil loss measurements. This result allowed the authors to enlarge the sample size to develop a more robust frequency analysis. For example, the database used by Bagarello et al. (2105) consisted of 1,649 event soil loss values from a monitoring period of 14 years. These soil loss measurements varied greatly, as they ranged from 0.0001 kg m^{-2} to 27.3 kg m^{-2}, and resulted both from events with only interrill erosion and with rill and interrill erosion. The frequency distribution of x_{SL} was well fitted by a two-component probability distribution, whose basic and outlying components were discriminated by $x_{SL,To} = 2$, being $x_{SL,To}$ the x_{SL} value with a return period T_o of 25 years. The event soil loss of given return period T was estimated as the quantile, $x_{SL,T}$, of given T of the variable x_{SL} by the mean soil loss measured in the highest erosion event for a given length. Considering that $x_{SL,To} = 2$ discriminates between the relatively low and frequent x_{SL} values and the high and rare ones, the tolerable event soil loss can be set as two times the mean soil loss measured in the highest erosion event for a given plot length. Considering that a great amount of yearly soil loss measured at Sparacia was due to the annual maximum value of event soil loss (Bagarello et al., 2015), differences between the two temporal scales (event in which the maximum soil loss is measured, annual) were considered negligible. Consequently, the annual tolerable soil loss was set equal to the tolerable event soil loss, i.e., two times the mean soil loss measured in the highest erosion event for a given plot length.

Here, considering $RKLS$ as an estimation, according to the USLE/RUSLE scheme, of the mean soil loss for the highest erosion event for bare plots without control practices ($C = 1$ and $P = 1$), the following equation is obtained:

$$TSL = x_{SL,To}RKLS \tag{3}$$

where $x_{SL,To} = 2$ for the Sparacia area.

Under the hypothesis that the C annual value is constant for a given crop and different from the unit, Eq. (2) can be generalized for the condition in which only the effects of control practices are neglected ($P = 1$), resulting in

$$SL_{a,T} = R_{a,T}KLSC \tag{4}$$

Imposing that the tolerable soil loss, expressed by Eq. (3), is equal to the annual soil loss of a given T calculated by Eq. (4), the following equation is obtained:

$$x_{SL,To}RKLS = R_{a,T}KLSC_T \tag{5}$$

in which $C = C_T$ is a specific crop and management factor having the following expression:

$$C_T = \frac{x_{SL,To}R}{R_{a,T}} = \frac{x_{SL,To}}{x_T} \tag{6}$$

Areas with $C > C_T$ are characterized by soil loss values greater than tolerance, in which interventions to reduce the cover and management factor or field length or apply support practices should be carried out.

The C-factor considers the effect of land cover and management measures (Wischmeier and Smith, 1965) and ranges from 0 to 1. C-factor values close to zero are typical of areas with a 100 % ground cover whereas values close to one are typical of a bare plot (no vegetation) with till up and down the slope, which is taken as a reference condition ($C = 1$) (Borrelli et al., 2018). The change in land cover and management factor in 28 Member States of the European Union from 2010 to 2016 was also used as an indicator of the effectiveness of soil conservation measures/strategies supported by the Common Agricultural Policy to reduce soil erosion in Europe (Borrelli and Panagos, 2020).

2.2. Determination of C_T

For the Sicilian region, Ferro et al. (1991) carried out a statistical analysis of the measured annual rainfall erosivity factor R_a, calculated by the procedure of Wischmeier and Smith (1978) using the data of 41 recording rain-gauges, and demonstrated that the Weibull's distribution can be applied to this hydrological variable.

Using the Weibull's distribution (Weibull, 1951) to estimate $R_{a,T}$, Ferro et al. (1991) obtained the following frequency factor x_T:

$$x_T = \frac{R_{a,T}}{R} = \frac{\beta(\ln T)^{1/\varepsilon}}{R} \tag{7}$$

in which β and ε are the two parameters of the Weibull's law.

Introducing Eq. (7) into Eq. (6), the following equation is obtained:

$$C_T = \frac{x_{SL,To}R}{\beta(\ln T)^{1/\varepsilon}} \tag{8}$$

in which β and ε have to be estimated by the following equations:

$$R = \beta\Gamma\left(1 + \frac{1}{\varepsilon}\right) \tag{9}$$

$$\sigma(R_a) = \beta\left[\Gamma\left(1 + \frac{2}{\varepsilon}\right) - \Gamma^2\left(1 + \frac{1}{\varepsilon}\right)\right]^{1/2} \tag{10}$$

where Γ is the gamma function and σ(R_a) is the standard deviation of the annual rainfall erosivity factor R_a.

The mean annual value R can be estimated for a Sicilian recording rain-gauge by the following equation (Ferro et al., 1991):

$$R = 183.82 + 1.3956\left(I_{1,2}I_{6,2}I_{24,2}\right) \tag{11}$$

in which R is expressed in SI units (MJ mm ha^{-1}h^{-1}year^{-1}) and $I_{1,2}$, $I_{6,2}$, $I_{24,2}$ (mm h^{-1}) are the rainfall intensities with 1, 6, and 24 h duration and a return period of two years, respectively.

For a Sicilian non-recording rain gauge, the following equation is available:

$$R = 8.92F_F^{1.59} \tag{12}$$

in which F_F is the mean annual value of the Modified Fournier Index (Bagarello and Ferro, 2006):

$$F_F = \sum_{i=1}^{12}\frac{p_{i,j}^2}{P_j} \tag{13}$$

in which $p_{i,j}$ is the monthly rainfall of the month i of the year j and P_j is the annual rainfall of the year j.

For the Sicilian region, the standard deviation σ(R_a) is estimated by the following equation (Ferro et al., 1991):

$$\sigma(R_a) = -521.63 + 1.38R \tag{14}$$

in which R is expressed in SI units.

3. Results

The R values calculated for 276 Sicilian rain-gauges were used to plot the distribution of the mean annual rainfall erosivity (isoerosivity map) (Fig. 1) and estimate the coefficients β and ε of the Weibull's distribution by Eqs. (9), (10), and (14). Eq. (8) shows that C_T decreases as T increases; in other words, more cover or protection is needed as the return period increases (i.e., heavier storms). The choice of C_T values corresponding to a return period of 1000 years, C_{1000}, can be considered conventional and guarantees high-safety conditions for soil conservation. The C_{1000} values, estimated by Eq. (8) with $T = 1000$ and $x_{SL,To} = 2$, are related to the mean annual value of the erosivity factor by the following relationship (Fig. 2):

$$C_{1000} = 0.198 + \left(\frac{417}{R}\right)^2 \tag{15}$$

Eq. (15) allows for the establishment of a mean annual rainfall erosivity factor, named $R_{land\text{-}use}$, corresponding to a specific value of the cover and management factor depending on land use. In other words, $R_{land\text{-}use}$ is the R value calculated by Eq. (15) for $C_{1000} = C$ (for example, $C = 0.5$ for vineyards and $C = 0.35$ for arable land). The result $C \le C_{1000}$ is obtained for areas in Sicily that are characterized by R smaller than or equal to $R_{land\text{-}use}$, and the compliance with soil loss tolerance is then assured. Instead, for areas characterized by $R > R_{land\text{-}use}$ the reduction of C to a value less than or equal to C_{1000} is required to obtain a tolerable soil loss condition. Fig. 2 demonstrates, for example for the case in which the land use is cereals and legumes ($C = 0.35$), that areas in which $R \le 1070$ are characterized by soil loss values less than or equal to TSL. On the contrary, for $R > 1070$, areas cultivated with cereals and legumes are exposed to soil loss values greater than TSL as 0.35 is greater than C_{1000}.

Fig. 3 shows, for the case of arable land (mainly cereals and legumes) (Fig. 3**a**) and vineyards (Fig. 3**b**), the overlay of the C_T spatial distribution, originated from the isoerosivity map, with C spatial distribution, obtained by Corinne-land cover for Sicily (Bagarello et al., 2016), to identify areas (marked in red) where the soil loss tolerance is exceeded. In these areas, in which $R \ge 1070$ (Fig. 3**a**) and $R \ge 758$ (Fig. 3**b**), respectively, land use should be changed to obtain $C \le C_{1000}$.

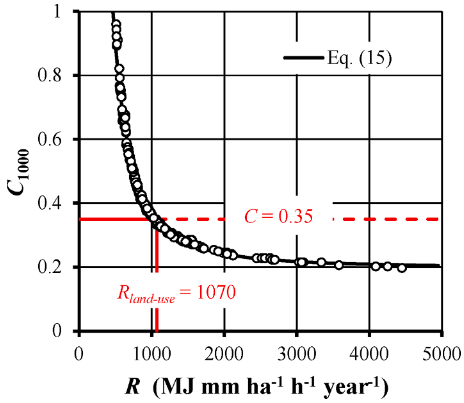

Fig. 2. Relationship between C_{1000} and the mean annual value of the erosivity factor R and graphical example of the determination of the mean annual rainfall erosivity factor $R_{land\text{-}use}$ corresponding to the cover and management factor C for arable land (mainly cereals and legumes).

4. Discussion

In the case $R > R_{land\text{-}use}$, the applied soil conservation strategy must determine a reduction CR of the cover and management factor:

$$CR = C - C_{1000} = C - 0.198 - \left(\frac{417}{R}\right)^2 \tag{16}$$

However, changing land use to another crop or a crop rotation can be unlikely. This is the case, for example, for areas covered by traditional crops (e.g., wheat in the Sicilian hinterland) in which farmers could be not inclined to change traditional cropping due to cultural heritage and soil agricultural vocation. In addition, other land uses (for example vineyards, olive groves, etc.), are much more unlikely to change due to their high profitability. Alternatively, the current land use (represented by C) can be maintained adopting a soil conservation strategy which has the same effect on the soil loss of given return period as the reduction of C to, at least, C_{1000}, e.g., by limiting field length.

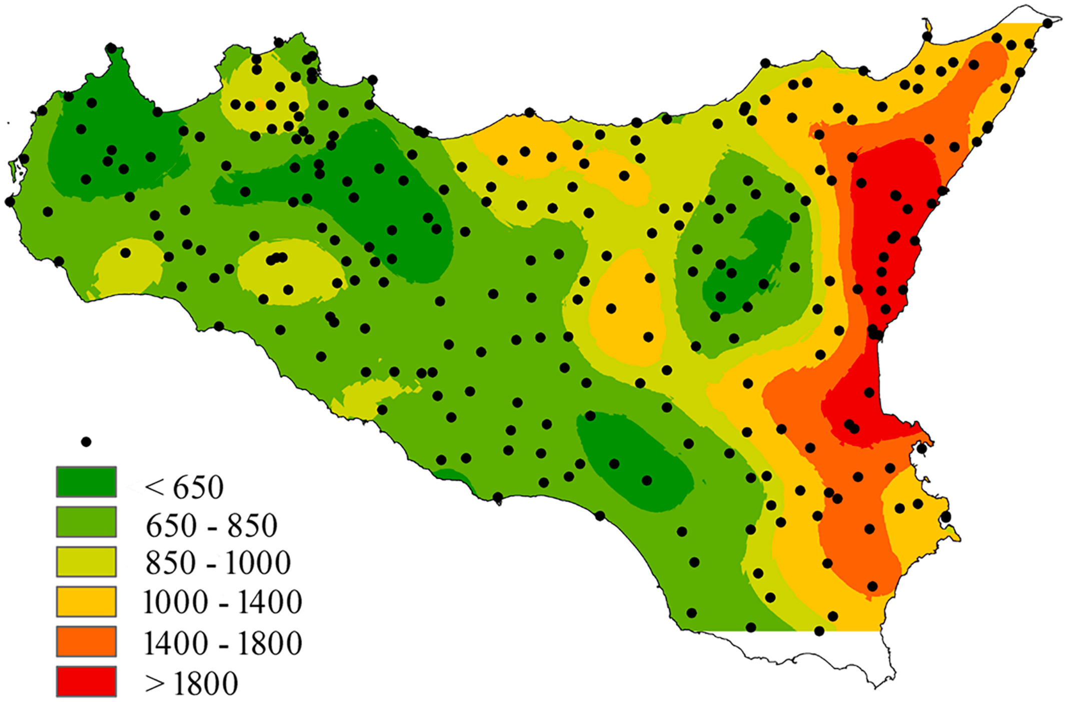

Fig. 1. Sicilian rainfall erosivity map and rain-gauges used to calculate the mean annual values of the rainfall erosivity factor R.

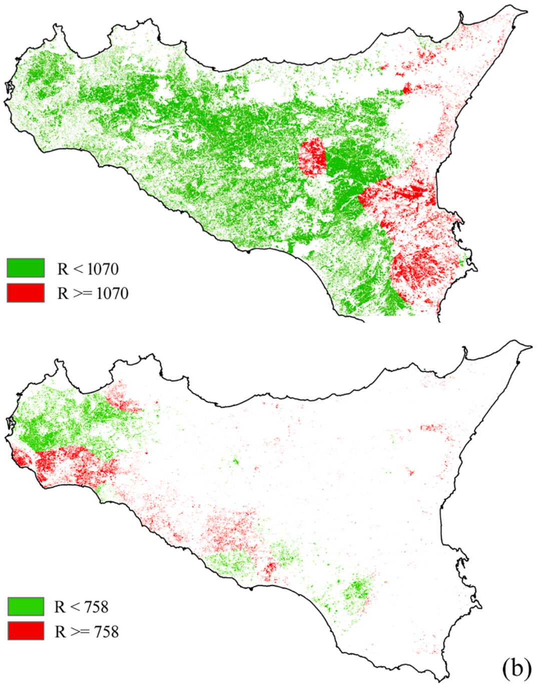

Fig. 3. Overlay of the spatial distributions of C and C_T for arable land (mainly cereals and legumes) (a) and vineyards (b). In the red areas, the annual soil loss $SL_{a,T}$ of $T = 1000$ years exceeds the tolerable soil loss TSL, while in the green areas $SL_{a,T}$ is under TSL. (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

In this case, the changed value, L_c, of the slope length factor for a given C_{1000}, can be obtained by the following relationship:

$$R_{a,T}KL_cSC = R_{a,T}KLSC_{1000} \tag{17}$$

from which

$$L_C = \left(\frac{C_{1000}}{C}\right)L \tag{18}$$

where C_{1000}/C is less than 1.

A reduced value, S_c, of the slope steepness factor can be obtained by a similar relationship to Eq. (17)

$$R_{a,T}KLS_cC = R_{a,T}KLSC_{1000} \tag{19}$$

resulting in

$$S_C = \left(\frac{C_{1000}}{C}\right)S \tag{20}$$

Eqs. (18) and (20) underline that if a farmer wishes to maintain a land use that would determine a soil loss higher than the tolerance value ($C > C_{1000}$ for $R > R_{land\text{-}use}$), some different soil conservation actions to limit the slope length or steepness should be done. Reasonably, it is less expensive reducing the slope length factor by using, for example, contour ditches than realizing terraces for reducing slope steepness. Substantial investment of human and financial resources is required to establish terraces (Food and Agricultural Organization, 2019), but they are widely used in agriculture as they have relatively higher crop productivity.

From previous studies, the generally accepted maximum limit of TSL value is 11.2 t ha^{-1} year^{-1} (Wischmeier and Smith, 1978), but significantly lower values were proposed for the US (Renard et al., 1997; Lal, 2001) and European environments (European Environment Agency, 1998, Verheijen et al., 2009; Bazzoffi 2009). A quantitative model for assessing site-specific TSL values for a region of India (Bhattacharyya et al., 2008) was used by integrating various soil functions (e.g., water entry and transport into soils, physical and bio-chemical degradation of soils, functions related to sustain plant growth), obtaining TSL values varying from 2.5 to 12.5 t ha^{-1} yr^{-1} instead of the single value of 11.2 t ha^{-1} yr^{-1}. The proposed approach goes beyond the simple choice of TSL values suggested in the literature as it considers the effect of R, K, L, S factors on the TSL value, and allows designing conservation systems for limiting soil loss using a target value of normalized soil loss corresponding to a specific return period T_0 (for the Sicilian region equal to 25 years).

This method can be applied in any geographical area if the result (applicability of a two-component probability distribution for x_{SL} and individuation of a specific value $x_{SL,To}$) by Bagarello et al. (2010, 2011, 2015) holds, R is known, the probability distribution of the annual rainfall erosivity factor R_a has been established and its parameters have been estimated. While the availability of rain-gauge networks allows for widespread rainfall data acquisition, long historical sequences of maximum annual soil loss data are a rather uncommon occurrence. In Italy, for example, only two experimental stations for soil erosion measurement, in addition to that of Sparacia in Sicily, are currently operating (Bagarello et al., 2018) and are located at Bagnara (Calabria region, south Italy) and Masse (Umbria region, central Italy). The rainfall data allow determining the spatial distribution of R and the probability distribution of R_a within a region, while the soil loss data have to be used to develop a reliable frequency analysis of x_{SL}.

According to the USLE scheme, an annual value of the C factor applies for a particular cropping and management system (Wischmeier and Smith, 1978), which can vary within a multi-year rotation and results in a single value to be applied at the mean annual scale. In other words, the C factor generally has an inter-annual variability. Additionally, the original procedure to calculate this factor is based on the interrelation between the particular cropping system and the rainfall distribution through the 12 months of the year. As the latter varies with location, the C value for a particular cropping system differs between locations of the same region, thus it also has a spatial variability. According to the proposed approach, this would imply a varying $R_{land\text{-}use}$ with location, as it is obtained by equating the C value and the second member of Eq. (15). Here, Eq. (4), from which the definition of C_T (Eq. (6)) is obtained, is based on the hypothesis that the C annual value is a constant, thereby the temporal variability is neglected. Moreover, although the present approach could be applied in a given site to account for the original C calculation, the aim of Fig. 3 was to provide an overview at a larger spatial scale, i.e., the regional one, under the simplifying hypothesis of an invariant C factor between locations for a given land use.

The USLE/RUSLE predicts soil erosion rates due to rill and interrill erosion processes, while it does not consider other erosion forms, such as ephemeral gully and gully erosion. Therefore, the proposed approach cannot be applied to areas where these erosive forms are present.

Incentive mechanisms to encourage soil conservation strategies should be higher the closer the post-intervention soil loss is to TSL. For example, the payment for the provided ecosystem service PES (€ ha^{-1}) could be determined as

$$\frac{PES}{PES_{max}} \propto \frac{TSL}{SL_{a,T}} \tag{21}$$

in which PES_{max} (€ ha^{-1}) is the maximum payment for the condition $SL_{a,T} = TSL$ (e.g., $C = C_T$). According to Eq. (21), the payment reduces proportionally to $TSL/SL_{a,T}$, which is less than 1 (e.g., $C > C_T$), for soil loss conditions exceeding tolerance.

Soil erosion models and risk maps are useful tools to assist public authorities and political decision-makers establishing land use and soil conservation strategies (Bagarello et al., 2016). A large part of the Sicilian region is cultivated with cereals, especially wheat, and legumes (Fig. 3a). Fig. 3a shows that most of the areas with this land use are characterized by an annual soil loss $SL_{a,T}$ of $T = 1000$ years under tolerable soil loss TSL (green areas), while the eastern part and a spot in the center of the island have $SL_{a,T}$ exceeding TSL (red areas). Vineyards are mainly spread in western Sicily (Fig. 3b), and marginally in the south and close to the Etna mount. For the western and southern areas, the extents of green and red areas are comparable, while in the Etna area the red ones sharply prevail.

These results were obtained under the conservative hypothesis that no control practices are applied ($P = 1$).

5. Conclusions

Considering that soil conservation strategies, designed by the mean annual value of the climatic variable, as the rainfall erosivity factor, allow an appropriate erosion control only in the years in which non-intensive rainfall erosive events occur, a new approach to define the condition corresponding to a tolerable soil loss was proposed. This approach is based on the statistical analysis of the measured annual values of the rainfall erosivity factor and allows determining the cover and management factor C_T for which the tolerable soil loss is equal to the annual soil loss of a given return period T.

For example, using the available measurements for the Sicilian region, a relationship between the factor C_{1000} and the mean annual value of R was established. For a given land use characterized by a specific value C of the cover and management factor, the corresponding mean annual rainfall erosivity factor $R_{land\text{-}use}$ was determined by the proposed regional relationship. Areas having mean annual values R less than or equal to $R_{land\text{-}use}$ feature C_{1000} values greater than or equal to C, while for areas characterized by $R > R_{land\text{-}use}$, the reduction of C to a value less than C_{1000} is required to obtain a tolerable soil loss condition. For example, for vineyards and arable lands, the overlay between the spatial distributions of C and C_T was used to define the areas in which soil tolerable conditions ($R \leq R_{land\text{-}use}$) occur or soil conservation strategies are required ($R > R_{land\text{-}use}$). Finally, for the condition $R > R_{land\text{-}use}$, the land use (represented by C) can be maintained using a reduced field length or slope steepness.

Declaration of Competing Interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

Data availability

Data will be made available on request.

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References

Baffaut, C., Nearing, M.A., Govers, G., 1998. Statistical distributions of soil loss from runoff plots and WEPP model simulations. Soil Sci. Soc. Am. J. 62, 756–763.

Bagarello, V., Di Stefano, C., Ferro, V., Pampalone, V., 2010. Statistical distribution of soil loss and sediment yield at Sparacia experimental area, Sicily. Catena 82, 45–52.

Bagarello, V., Di Stefano, C., Ferro, V., Pampalone, V., 2011. Using plot soil loss distribution for soil conservation design. Catena 86, 172–177.

Bagarello, V., Di Stefano, C., Ferro, V., Pampalone, V., 2015. Establishing a soil loss threshold for limiting rilling. J. Hydrol. Eng. ASCE 20:C5014001-1-C5014001-12.

Bagarello, V., Carollo, F.G., Di Stefano, C., Ferro, V., Giordano, G., Iovino, M., Pampalone, V., 2016. Twenty years of scientific activity at Sparacia experimental area. Quad. Idronom. Mont. 33, 1–142.

Bagarello, V., Ferro, V., 2006. Erosione e conservazione del suolo. McGraw-Hill, Milano, Italy.

Bagarello, V., Ferro, V., Pampalone, V., Porto, P., Todisco, F., Vergni, L., 2018. Predicting soil loss in Central and South Italy with a single USLE-MM model. J. Soil. Sediment. 18, 3365–3377.

Bazzoffi, P., 2009. Soil erosion tolerance and water runoff control: minimum environmental standards. Reg Environ Change 9, 169–179. https://doi.org/10.1007/s10113-008-0046-8.

Bhattacharyya, P., Bhatt, V.K., Mandal, D., 2008. Soil loss tolerance limits for planning of soil conservation measures in Shivalik-Himalayan region of India. Catena 73, 117–124.

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. https://doi.org/10.1002/ldr.2898.

Borrelli, P., Panagos, P., 2020. An indicator to reflect the mitigating effect of Common Agricultural Policy on soil erosion. Land Use Policy 92, 104467. https://doi.org/10.1016/j.landusepol.2020.104467.

EC COM (2019) 640, The European Green Deal.

European Commission. 2006. A strategy to keep Europe's soils robust and healthy. Available from: http://ec.europa.eu/environment/soil/index_en.htm.

Dregne, H.E., 1987. Soil erosion: cause and effect. Land Use Policy 4 (4), 412–418. https://doi.org/10.1016/0264-8377(87)90063-9.

Erkossa, T., Wudneh, A., Desalegn, B., Taye, G., 2015. Linking soil erosion to on-site financial cost: lessons from watersheds in the Blue Nile basin. Solid Earth 6, 765–774. https://doi.org/10.5194/se-6-765-2015.

European Environment Agency. 1998. Europe's environment: the second assessment. Available from: http://www.eea.europa.eu//publications/92-828-3351-8.

Fekete, A., Sandholz, S., 2021. Here Comes the Flood, but Not Failure? Lessons to Learn after the Heavy Rain and Pluvial Floods in Germany. Water 13 (21), 3016. https://doi.org/10.3390/w13213016.

Ferro, V., Giordano, G., Iovino, M., 1991. Isoerosivity and erosion risk map for Sicily. Hydrol. Sci. J. 36 (6), 549–564. https://doi.org/10.1080/02626669109492543.

Food and Agricultural Organization (FAO), 2019. Soil erosion: the greatest challenge to sustainable soil management. Rome, 100 pp. https://www.fao.org/documents/card/en/c/ca4395en.

Gonzales-Hidalgo, J.C., Batalla, R.J., Cerdà, A., De Luis, M., 2010. Contribution of the largest events to suspended sediment transport across the USA. Land Degrad. Dev. 21, 83–91.

Gonzales-Hidalgo, J.C., Batalla, R.J., Cerdà, A., De Luis, M., 2012. A regional analysis of the effects of largest events on soil erosion. Catena 95, 85–90.

Grunewald, K., Naumann, S., 2015. Valuation of Ecosystem Services Regarding the Water Framework Directive on the Example of the Jahna River Catchment in Saxony (Germany).

Guo, Y., Zheng, H., Wu, T., Wu, J., Robinson, B.E., 2020. A review of spatial targeting methods of payment for ecosystem services. Geography and Sustainability 1 (2), 132–140. https://doi.org/10.1016/j.geosus.2020.04.001.

Hein, L., 2007. Assessing the costs of land degradation: a case study for the Puentes catchment, southeast Spain. Land Degrad. Dev. 18 (6), 631–642. https://doi.org/10.1002/ldr.802.

Hession, W.C., Storm, D.E., Haan, C.T., 1996. Two-phase uncertainty analysis: an example using the universal soil loss equation. Trans. ASAE 39, 1309–1319.

Huber, S., Prokop, G., Arrouays, D, 2008. Environmental assessment of soil for monitoring. Volume I indicators and criteria. Office for the Official Publications of the European Communities, Luxembourg, Luxembourg.

Johnson, L.C., 1987. Soil loss tolerance: fact or myth? J. Soil Water Conserv. 42, 155–160.

Johnson, L.C., 2005. Soil loss tolerance: fact or myth. J. Soil Water Conserv. 60, 52.

Jónsson, J.O.G., Davíðsdóttir, B., 2016. Classification and valuation of soil ecosystem services. Agr. Syst. 145, 24–38. https://doi.org/10.1016/j.agsy.2016.02.010.

Kandziora, M., Burkhard, B., Müller, F., 2013. Mapping provisioning ecosystem services at the local scale using data of varying spatial and temporal resolution. Ecosyst. Serv. 4, 47–59. https://doi.org/10.1016/j.ecoser.2013.04.001.

Kendall, H.W., Pimentel, D., 1994. Constraints on the Expansion of the Global Food Supply. Ambio 23 (3), 198–205. http://www.jstor.org/stable/4314199.

Kumar, P., 2010. The Economics of Ecosystems and Biodiversity: Ecological and Economic Foundations. Taylor & Francis.

Lal, R., 2001. Soil degradation by erosion. Land Degrad. Develop. 12, 519–539.

Larson, W.E., 1981. Protecting the soil resource base. J. Soil Water Conserv. 36, 13–16.

Larson, W.E., Lindstrom, M.J., Schumacher, T.E., 1997. The role of severe storms in soil erosion: a problem needing consideration. J. Soil Water Conserv. 52, 90–95.

Li, L., Du, S., Wu, L., Liu, G., 2009. An overview of soil loss tolerance. Catena 78, 93–99.

Mannaerts, C.M., 1992. Assessment of transferability of laboratory rainfall-runoff and rainfall-soil loss relationship to field and catchment scales: A study in the Cape Verde Islandes. Ph.D. Thesis, Faculty of Applied Biological and Agricultural Sciences, Ghent University, Belgium.

Mannaerts, C.M., Gabriels, D., 2000. A probabilistic approach for predicting rainfall soil erosion losses in semiarid areas. Catena 40, 403–420.

Martínez-Casasnovas, J.A., Ramos, M.C., 2006. The cost of soil erosion in vineyard fields in the Penedès-Anoia Region (NE Spain). Catena 68 (2–3), 194–199. https://doi.org/10.1016/j.catena.2006.04.007.

Mikša, K., Kalinauskas, M., Inácio, M., Gomes, E., Pereira, P., 2020. Ecosystem services and legal protection of private property. Problem or solution? Geography and Sustainability 1 (3), 173–180. https://doi.org/10.1016/j.geosus.2020.08.003.

Panagos, P., Borrelli, P., 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., Standardi, G., Borrelli, P., Lugato, E., Montanarella, L., Bosello, F., 2018. Cost of agricultural productivity loss due to soil erosion in the European Union: From direct cost evaluation approaches to the use of macroeconomic models. Land Degrad. Dev. 29 (3), 471–484.

Papanastasis, V.P., Bautista, S., Chouvardas, D., Mantzanas, K., Papadimitriou, M., Mayor, A.G., Koukioumi, P., Papaioannou, A., Vallejo, R.V., 2015. Comparative assessment of goods and services provided by grazing regulation and reforestation in degraded Mediterranean rangelands. Land Degrad. Dev. https://doi.org/10.1002/ldr.2368.

Raclot, D., Le Bissonnais, Y., Annabi, M., Sabir, M., Smetanova, A., 2018. Main issues for preserving Mediterranean soil resources from water erosion under global change. Land Degrad. Dev. 29, 789–799. https://doi.org/10.1002/ldr.2774.

Renard, K.G., Foster, G.R., Weesies, G.A., McCool, D.K., Yoder, D.C., 1997. Predicting soil erosion by water. A guide to conservation planning with the revised universal soil loss equation RUSLE. USDA Agriculture Handbook No 703. U.S. Department of Agriculture, Washington, DC, USA.

Robinson, D.A. Hockley, N., Dominati, E., Lebron, I., Scow, K.M., Reynolds, B., Emmett, B.A., Keith, A.M., de Jonge, L.W., Schjønning, P., Moldrup, P., Jones, S.B., Tuller, M., 2012. Natural Capital, Ecosystem Services, and Soil Change: Why Soil Science Must Embrace an Ecosystems Approach. Vadose Zone Journal, 11 (1): vzj2011.0051. doi: https://doi.org/10.2136/vzj2011.0051.

Sartori, M., Philippidis, G., Ferrari, E., Borrelli, P., Lugato, E., Montanarella, L., Panagos, P., 2019. A linkage between the biophysical and the economic: Assessing the global market impacts of soil erosion. Land Use Policy 86, 299-312. https://doi.org/10.1016/j.landusepol.2019.05.014.

Schmidt, B.L., 1982. Determinants of soil loss tolerance. ASA Publication No. 45. American Society of Agronomy, Madison, WI, USA.

Smith, P., Cotrufo, M.F., Rumpel, C., Paustian, K., Kuikman, P.J., Elliott, J.A., McDowell, R., Griffiths, R.I., Asakawa, S., Bustamante, M., House, J.I., Sobocká, J., Harper, R., Pan, G., West, P.C., Gerber, J.S., Clark, J.M., Adhya, T., Scholes, R.J., Scholes, M.C., 2015. Biogeochemical cycles and biodiversity as key drivers of ecosystem services provided by soils. Soil 1, 665–685. https://doi.org/10.5194/soil-1-665-2015.

Strohmeier, S., Laaha, G., Holzmann, H., Klik, A., 2016. Magnitude and occurrence probability of soil loss: a risk analytical approach for the plot scale for two sites in lower Austria. Land Degrad. Develop. 27, 43–51.

Swinton, S.M., Lupi, F., Robertson, G.P., Hamilton, S.K., 2007. Ecosystem services and agriculture: Cultivating agricultural ecosystems for diverse benefits. Ecol. Econ. 64 (2), 245–252. https://doi.org/10.1016/j.ecolecon.2007.09.020.

Syrbe, R., Schorcht, M., Grunewald, K., Meinel, G., 2018. Indicators for a nationwide monitoring of ecosystem services in Germany exemplified by the mitigation of soil erosion by water. Ecol. Ind. 94 (2), 46–54. https://doi.org/10.1016/j.ecolind.2017.05.035.

Toy, T.J., Foster, G.R., Renard, K.G., 2002. Soil erosion: processes, prediction, measurement, and control. Wiley, New York, NY, USA.

van Leeuwen, C.C.E., Cammeraat, E., de Vente, J., Boix-Fayos, C., 2019. The evolution of soil conservation policies targeting land abandonment and soil erosion in Spain: A review. Land Use Policy 83, 174–186. https://doi.org/10.1016/j.landusepol.2019.01.018.

Verheijen, F., Jones, R., Rickson, R., Smith, C., 2009. Tolerable versus actual soil erosion rates in Europe. Earth Sci. Rev. 94, 23–38.

Weibull, W., 1951. A statistical distribution function of wide applicability. J. Appl. Mech. 18 (3), 293–297.

Wischmeier, W.H., Smith, D.D., 1965. Prediction Rainfall Erosion Losses from Cropland East of the Rocky Mountains: A Guide for Selection of Practices for Soil and Water Conservation. Agricultural Handbook No. 282, 47 p.

Wischmeier, W.H., Smith, D.D., 1978. Predicting rainfall erosion losses: a guide to conservation planning. USDA Agriculture Handbook No 537. U.S. Department of Agriculture, Washington, DC, USA.

Xu, C., Kohler, T.A., Lenton, T.M., Svenning, J., Scheffer, M., 2020. Future of the human climate niche. Proc. Natl. Acad. Sci. 117 (21), 11350–11355. https://doi.org/10.1073/pnas.1910114117.

Zhang, J.J., Fu, M.C., Zeng, H., Geng, Y.H., Hassani, F.P., 2013. Variations in ecosystem service values and local economy in response to land use: a case study of Wu'an. China. Land Degrad. Dev. 24 (3), 236–249. https://doi.org/10.1002/ldr.1120.