

Research papers

Establishing the payment for the ecosystem service of soil erosion control

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ABSTRACT

Soil conservation strategies based on the mean annual value of the rainfall erosivity factor could not guarantee appropriate erosion control for intense rainfall events. In this paper, a new method to define soil loss tolerance at a regional scale, using the crop and management factor of the USLE, C , was applied for estimating the payment for ecosystem service (PES) of soil erosion control. PES is characterized by a target value PES_{tg} corresponding to a post-intervention condition in which soil loss is equal to the tolerance value. The PES was estimated by three different scenarios: (1) only reducing the current C factor, or combining its reduction with that of the (2) support practice factor P or (3) slope length factor L . For *Scenario 1*, the analysis demonstrated that a drastic reduction of the C factor determines $PES/PES_{tg} \geq 1$ regardless of the investigated climatic conditions. For *Scenario 2*, when C and P factors are equally weighted, the payment is higher than the target value for slope-steepness $\leq 15\%$. *Scenario 2* always yields a higher payment compared to *Scenario 1*. For *Scenario 3*, the reduction of the topographic factor can lead the PES over the target value. Although PES is generally considered effective for environmental policy, it should be allocated by targeting erosion-prone areas, where soil conservation is a primary objective. These areas experience soil loss greater than tolerable value, which usually lacks a solid scientific basis. By contrast, the proposed method, developed for the Sicilian region, is grounded in a scientific framework and is useful to support policymakers and public authorities in the informed design of economic incentives for farmers who convert their activities in order to preserve soil resources.

1. Introduction

The European Union (EU) soil thematic strategy (EC, 2006) defined soil as “essentially a non-renewable resource and a very dynamic system, which performs many functions and delivers services vital to human activities and ecosystems survival”. This definition highlights that soil formation is extremely slow – centuries to millennia for a few centimeters of topsoil under normal agricultural conditions (Bazzoffi, 2009) – whereas degradation from erosive processes is rapid and often intensified by human activities (Lal, 2014; Raclot et al., 2018). The European Green Deal acknowledges the key contribution of soils to achieving the 2030 Sustainable Development Goals (Bouma et al., 2019; Montanarella and Panagos, 2021). Within this framework, the EU Soil Strategy for 2030 aims to mitigate climate change, preserve biodiversity, prevent desertification, and enhance soil monitoring and restoration to ensure healthy soil ecosystems in Europe (Panagos et al., 2022). The actions adopted by the European Commission respond to the evidence that soil erosion in

European agricultural areas proceeds at rates roughly twice as high as soil formation (Panagos et al., 2016; Montanarella and Panagos, 2021). The associated economic impacts are substantial: Panagos et al. (2015) estimated annual restoration costs of about \$20 billion, while Panagos et al. (2018) reported 12 million hectares affected and €1.25 billion in lost crop productivity.

Land degradation costs (Sartori et al., 2019) are commonly estimated by considering productivity losses caused by soil erosion. These “first-order” costs rely on crop production losses (tons) (in-site effect) (Erkossa et al., 2015) and corresponding average market price (\$/tons). “Second-round” costs derive from sediment transport along hillslopes and river networks, including nutrient export (Serio et al., 2020). In the EU and United Kingdom, water-driven erosion potentially mobilizes 135 million m^3 of sediment each year towards river networks, with removal costs from reservoirs exceeding €2.3 billion (Panagos et al., 2024). Considering all erosion processes (gullies, landslides, etc.), sedimentation in the EU’s large reservoirs exceeds 1 billion m^3 annually, with removal

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cost estimated at €5–8 billion.

Reducing soil erosion enhances agricultural and forest productivity, lowers fertilizer demand and promotes more sustainable soil use (Grunewald and Naumann, 2015; Syrbe et al., 2018). During 2000–2010, the Common Agricultural Policy promoted, through economic incentives for landowners, Best Management Practices to reduce soil loss on agricultural lands (Ricci et al., 2020) and mandatory soil conservation measures, named Good Agricultural Environmental Conditions (GAEC) (Panagos et al., 2016). In particular, the economic incentives were related to good agricultural practices such as minimum tillage, erosion-limiting land management, and the conservation of soil organic matter (Schomers and Matzdorf, 2013; Borrelli et al., 2016; Panagos et al., 2022). These measures reduced soil loss by 20% in cultivated areas. However, the increasing recurrence of extreme rainfall events (Carollo and Ferro, 2015; Carollo et al., 2016, 2018, 2023a; Serio et al., 2019a,b) across Europe (Fekete and Sandholz, 2021; Moccia et al., 2024; Rombeek et al., 2025) highlighted the limitations of existing protection measures and gaps in hydro-geomorphological risk mitigation (Ferro et al., 2020).

In this context, Ecosystem Services (ES), defined as “the benefits people obtain due to the regulation of natural processes such as water purification and erosion control”, play a key role (Kandziora et al., 2013; Lal, 2014; Ferrarini et al., 2018; Hasan et al., 2020). ES are linked to major ecosystem processes including water regulation, soil erosion, soil organic matter decline, soil biodiversity loss, and pesticide control (Steinhoff-Knopp et al., 2021). Ecosystem Services help buffer abiotic and biotic pressures and reduce the impacts of current and future environmental hazards (Xu et al., 2020). “Erosion control” is a soil management-related ecosystem service listed in the Common International Classification of Ecosystem Services (CICES) (Paul et al., 2020).

Soil, as a dynamic three-dimensional system, supports multiple functions that underpin ES delivery providing benefits that justify economic incentives for soil conservation and its ecosystem services (Adhikari and Hartemink, 2016; Bartkowski et al., 2018).

Payment for Ecosystem Services (PES) offers a policy instrument that compensates land users for implementing environmentally sound practices (Van Hecken and Bastiaensen, 2010). Scholars and stakeholders consider PES an effective tool for environmental policy and incentive mechanisms (Jónsson and Davíðsdóttir, 2016; Smith et al., 2015; Zhang et al., 2013). The basic idea supporting PES is that “rather than to punish farmers for “bad behavior”, beneficiaries of “good behavior” should be made to pay” (Van Hecken and Bastiaensen, 2010). According to Wunder (2005), PES is “a voluntary transaction where a well-defined ES (or a land-use likely to secure that service) is being “bought” by an ES buyer from an ES provider if and only if the ES provider secures ES provision (conditionality)”.

Nonetheless, subsidies for erosion control and soil restoration remain debated because long-term benefits are not always evident (Breure et al., 2012; van Leeuwen et al., 2019). PES thus represents both a cost related to sustainable soil use and compensation for reduced crop productivity. Because farmers’ property rights often constrain the adoption of ES-enhancing measures, PES can encourage their adoption, reducing onsite and offsite impacts of agricultural activities (Mikša et al., 2020).

Cost estimation for soil conservation requires spatial targeting to identify priority areas for soil loss reduction and PES allocation (Guo et al., 2020).

The need to determine a quantitative target to evaluate erosion mitigation strategies led to the concept of tolerable soil loss, *TSL* (Johnson, 1987; Di Stefano and Ferro, 2016). In 1962, the U.S. Soil Conservation Service first defined *TSL* as “the maximum level of soil erosion that will permit a high level of crop productivity to be sustained economically and indefinitely”. Suggested values for many US soils range between 4.5 and 11.2 t ha⁻¹ year⁻¹ (Wischmeier and Smith, 1978; Renard et al., 1997). Larson (1981) introduced a two-level definition of the soil loss tolerance distinguishing on-site productivity protection from off-site impacts, including water pollution, gully erosion, and

reservoir sedimentation. Verheijen et al. (2009) defined *TSL* as “any actual soil erosion rate at which a deterioration or loss of one or more soil function does occur” and proposed to define the unsustainable conditions by a soil loss-formation balance. Although *TSL* is a generally accepted concept (Li et al., 2009), no robust scientific method for estimating it is currently shared.

The Universal Soil Loss Equation (USLE) and USLE-type models (Wischmeier and Smith, 1965, 1978; Renard et al., 1997; Bagarello et al., 2017) have the ability to reduce a very complex system to a quite simple one for predicting erosion by considering the main environmental and anthropogenic factors that affect soil erosion process. Conceptually, these models apply the same principle of more complex process-oriented models, with a driving force (rainfall erosivity), a resistance term (soil erodibility), and other factors representing landscape and farming effects (i.e., topographical conformation of the field (*LS*), cropping system (*C*) and conservation practices (*P*)). While the original expressions to calculate the slope length *L* and slope steepness *S* factors were suitable for plots with a uniform slope, *LS* has to be corrected to describe a non-uniform slope condition (e.g., concave and convex profiles) and account for both soil erosion and deposition processes (Di Stefano et al., 2000; Fernandez et al., 2003). The USLE was originally developed to predict the mean annual soil loss but was also applied at the event scale. For example, at the Sparacia experimental station, in Sicily, the USLE scheme performed better with reference to highly erosive events as compared with low erosivity events (Bagarello and Ferro, 2004) and reproduced better the annual maxima than plot soil losses of the complete dataset available for that site (Bagarello et al., 2017). Despite its empirical basis, the USLE formulation can be derived through dimensional analysis and self-similarity theory (Ferro, 2010), using the same variables representative of soil erosion and reference condition indicated by Wischmeier and Smith (1978). The widespread use of USLE-type models stems from acceptable accuracy and relative simplicity, modest data requirements, and the distinguishing availability of a worldwide distributed dataset (Risse et al., 1993; Rosewell, 1993; Liu et al., 2001; Hann and Morgan, 2006; Salvador Sanchis et al., 2008; Bagarello et al., 2017). For these reasons, these models have become standard tools in soil conservation (Morgan, 2005).

Carollo et al. (2023b) proposed a regional-scale method that identifies areas where soil loss exceeds *TSL*, defined through the statistical analysis of plot soil loss measurements collected in a multi-year period in the Sparacia (Sicily) experimental area (Bagarello et al., 2015). The main novelty was a different definition of *TSL* compared to the constant value generally applied (10–13 t ha⁻¹). The method, developed for the Sicilian region, is also based on the probability distribution of the annual rainfall erosivity factor and the crop and management factor *C* of the USLE (Wischmeier and Smith, 1978).

The present study relates PES and the ratio between *TSL*, as defined above, and post-intervention soil loss. It aims to parameterize the payment with respect to a tolerance value deduced on a scientific basis rather than assumed as a conventional value, as usually done. In particular, this study aimed to propose a new approach to estimate PES considering how far the soil loss will be after a land use change or a soil conservation strategy from a target condition, corresponding to a post-intervention soil loss equal to *TSL* and a target payment PES_{tg} . In detail, the method by Carollo et al. (2023b) is combined with PES estimate through examination of three scenarios: (1) reducing the current *C* factor to have a more conservative condition; (2) combining *C* reduction with a reduction in the support practices factor; and (3) combining *C* reduction with a reduction in the slope-length factor.

2. Methods

2.1. Evaluating tolerable soil loss conditions

Combining the USLE and the choice of a *TSL* value (t ha⁻¹ year⁻¹) is a useful method to identify appropriate soil conservation strategies and

results in [Wischmeier and Smith \(1978\)](#):

$$\frac{TSL}{RKS} = LCP \quad (1)$$

in which R ($\text{MJ mm ha}^{-1} \text{ h}^{-1} \text{ year}^{-1}$) is the mean annual value of the rainfall erosivity factor, K (t ha^{-1} per unit of R) is the soil erodibility factor, S and L are the topographic factors, and P is the support practice factor of the USLE. Eq. (1) states that a soil loss equal to the tolerable value can be obtained by modifying the L , C , and P factors. The design of soil conservation works or strategies by using Eq. (1) implies the choice of representing the climatic forcing of the erosion process by R , which is a mean annual value. This choice is unsuitable for the years in which relevant erosive rainfalls occur, thereby producing annual values of the rainfall erosivity factor higher than R ([González-Hidalgo et al., 2010, 2012](#)).

Recently, some authors ([Bagarello et al., 2010; Strohmeyer et al., 2016; Carollo et al., 2023b](#)) suggested designing soil conservation practices using large storm events and soil loss values of a given return period T . The latter require a relation between the event-based soil loss and rainfall to assess the soil loss temporal distribution using long-term rainfall data or a sample size of historical soil loss measurements large enough to permit a direct and reliable frequency analysis of soil loss data ([Larson et al., 1997; Mannaerts and Gabriels, 2000; Bagarello et al., 2010, 2011](#)). Using soil loss measurements carried out on bare plots without support practices ($C = 1$, $P = 1$) of different lengths λ (11, 22, 33, and 44 m) at the Sparacia experimental area (Sicily) ([Fig. 1](#)), [Bagarello et al. \(2011\)](#) proposed to estimate the annual soil loss of a given return period T , $SL_{a,T}$, as follows:

$$SL_{a,T} = R_{a,T} KLS = x_T R KLS \quad (2)$$

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

Measurements of soil loss, collected from 1999 to 2012 in the Sparacia experimental area on bare plots varying in length from 0.25 m to 44 m and with a steepness of 14.9%, were used by [Bagarello et al. \(2010, 2011, 2015\)](#) to develop frequency analyses. Specifically, for each plot length and the most severe event of a recording year, these analyses were carried out by using the dimensionless variable x_{SL} that is equal to the ratio between the individual soil loss and the mean soil loss, for that event, across all plots having the same length. Accordingly, the event soil loss of a given return period T was estimated by multiplying this mean soil loss by the quantile $x_{SL,T}$. Using a database consisting of 1649

event soil loss measurements from a monitoring period of 14 years, [Bagarello et al. \(2015\)](#) established that x_{SL} is distributed according to a two-component probability law. The basic component is distinguished from the outlying one by $x_{SL,T_0} = 2$, being x_{SL,T_0} the quantile of x_{SL} with a return period $T = T_0 = 25$ years. Considering that $x_{SL,T_0} = 2$ distinguishes between the relatively low and frequent x_{SL} values (basic component) and the high and rare ones (outlying component), it represents the upper limit of the x_{SL} values having a relatively low magnitude. Therefore, the corresponding event soil loss, equal to two times the mean soil loss measured in the most severe event of the year for a given plot length, was assumed as tolerable event soil loss. For the Sparacia area, [Bagarello et al. \(2015\)](#) concluded that as the annual maximum event soil loss accounts for a great part of annual soil loss.

The circumstance that few severe events control total annual soil loss and effective soil conservation strategies should be planned based on these events rather than average soil loss, surely holds for Mediterranean environments dominated by rare and high-intensity storms ([Bagarello et al., 2010](#)). For example, [González-Hidalgo et al. \(2007\)](#) demonstrated that in western Mediterranean basin, more than 50% of the annual erosion is concentrated in the five most severe daily erosion events. Many studies suggest that the majority of soil loss is often associated with an extremely limited number of observations (i.e., days) ([González-Hidalgo et al., 2009](#)) also outside the Mediterranean area. For the U.S., [Wischmeier \(1962\)](#) concluded that 1/3 of the total soil loss in a multi-year period was due to extreme storms having return periods greater than 2 years. [Hjelmfelt et al. \(1986\)](#) stated that the seven largest events of 513 events over 37 years account for 25% of the total soil loss in Missouri. For the Northeast Oregon (U.S.) area, [Zuzel et al. \(1993\)](#) established that nearly all the total erosion over the 1978–1990 period was produced by only 50% of runoff events. [González-Hidalgo et al. \(2009\)](#) investigated the contribution of the most severe daily rainfall events to total soil erosion using the USLE database, which includes 27,857 daily erosion events monitored at 310 erosion plots in the U.S. They concluded that the five most severe events account for 50.9% of total soil loss, independently of plot soil characteristics, plant cover, and tillage practices. Therefore, according to the previous literature results, the hypothesis of neglecting the difference between the annual maximum event and annual soil loss could be applicable in different environmental contexts.

Considering that the difference between the two temporal scales (event, year) can be neglected, the annual tolerable soil loss TSL was set equal to two times the mean soil loss measured in the most severe erosion event for a given plot length. Following the USLE/RUSLE scheme applied to a bare plot with no control practices ($C = 1$ and $P = 1$) to estimate the latter, one obtains:

$$TSL = x_{SL,T_0} R KLS \quad (3)$$

in which $x_{SL,T_0} = 2$ for the Sparacia area. This value was obtained under specific experimental conditions (bare clay soil of a typical Sicilian inland area and Mediterranean climate) and should be tested across different soil types and climates. The bare soil condition underlies the proposed approach and must still be maintained for testing elsewhere, while soil type and rainfall regime could affect the x_{SL,T_0} value. More in general, the occurrence of a two-component frequency distribution for x_{SL} should be tested and the x_{SL,T_0} value then determined, by using long-term soil loss data that, unfortunately, are rarely available. For the Sparacia area, [Bagarello et al. \(2015\)](#) demonstrated that a commonly accepted TSL value of 11.5 t ha^{-1} can be converted to a threshold $x_{SL,T} = 0.91$ with $T = 2$ years, which is lower than $x_{SL,T_0} = 2$, thus resulting in a comparatively reduced soil loss threshold. However, as the occurrence of rilling for x_{SL} values under the two thresholds was comparable, $x_{SL,T_0} = 2$ was proposed as reference value for soil conservation interventions able to limit rilling in the studied area.

For 28 member States of the European Union, the change of the C factor from 2010 to 2016 was used to estimate the effectiveness of soil



Fig. 1. Aerial views of the erosion plots located at the Sparacia experimental area.

conservation measures/strategies supported by the Common Agricultural Policy to reduce soil erosion in Europe (Borrelli and Panagos, 2020). The C-factor assumes values ranging from 0 to 1; values close to zero are typical of areas with 100% ground cover, while $C = 1$ is typical of a bare plot (no vegetation) with till up and down the slope, which was also considered as the reference condition by Wischmeier and Smith (1965).

For a given crop, represented by a constant C annual value that, similarly to KLS , does not alter the hypothesized correspondence between the return periods of SL_a and R_a , Eq. (2) assumes the following form:

$$SL_{a,T} = R_{a,T} KLS C \quad (4)$$

Imposing the equality between the annual tolerable soil loss TSL , expressed by Eq. (3), and the annual soil loss of a given return period T , $SL_{a,T}$, calculated by Eq. (4), one obtains:

$$x_{SL,T} R_{a,T} KLS = R_{a,T} KLS C \quad (5)$$

in which $C = C_T$ is a specific crop and management factor corresponding to a soil loss, of given T , equal to the tolerance. From Eq. (5), considering that $x_T = R_{a,T}/R$, the following expression of C_T is obtained:

$$C_T = \frac{x_{SL,T} R}{R_{a,T}} = \frac{x_{SL,T}}{x_T} \quad (6)$$

Areas with $C > C_T$ are characterized by soil losses greater than tolerance and need soil conservation interventions and strategies that can be implemented by reducing the cover and management factor or the field length, or by applying support practices.

Di Stefano et al. (2023) applied the results of the x-frequency analysis, developed by Ferro et al. (1991), to estimate C_T by Eq. (6). In particular, for the Sicilian region, Ferro et al. (1991) carried out a frequency analysis of R_a , calculated by the procedure of Wischmeier and Smith (1978), using data from 41 recording rain gauges. Their analysis demonstrated that R_a is distributed according to Weibull's law (Weibull, 1951), and x_T can be expressed as:

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

where β and ε are the two parameters of Weibull's law. These parameters are estimated by the following equations derived by the method of moments (Ferro et al., 1991):

$$R = \beta \Gamma \left(1 + \frac{1}{\varepsilon} \right) \quad (8)$$

$$\sigma(R_a) = \beta \left[\Gamma \left(1 + \frac{2}{\varepsilon} \right) - \Gamma^2 \left(1 + \frac{1}{\varepsilon} \right) \right]^{1/2} \quad (9)$$

where Γ is the gamma function and $\sigma(R_a)$ is the standard deviation of R_a . Introducing Eq. (7) into Eq. (6) yields to:

$$C_T = \frac{x_{SL,T} R}{\beta (\ln T)^{1/\varepsilon}} \quad (10)$$

For the Sicilian region, the mean annual value R ($\text{MJ mm ha}^{-1} \text{h}^{-1} \text{year}^{-1}$) can be estimated using recording rain gauge data and the following equation (Ferro et al., 1991):

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

where $I_{1,2}$, $I_{6,2}$, $I_{24,2}$ (mm h^{-1}) are the rainfall intensities with 1, 6, and 24 h duration and a return period of two years. For a Sicilian non-recording rain gauge, R ($\text{MJ mm ha}^{-1} \text{h}^{-1} \text{year}^{-1}$) can be estimated by the following equation:

$$R = 0.524 F_F^{1.59} \quad (12)$$

where F_F is the mean annual value of the Modified Fournier Index $F_{a,j}$ (Ferro et al., 1991)

$$F_{a,j} = \sum_{i=1}^{12} \frac{p_{ij}^2}{P_j} \quad (13)$$

in which p_{ij} 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 $\sigma(R_a)$ is estimated by the following equation (Ferro et al., 1991):

$$\sigma(R_a) = -521.63 + 1.38R \quad (14)$$

in which R is expressed in $\text{MJ mm ha}^{-1} \text{h}^{-1} \text{year}^{-1}$.

In a previous paper (Carollo et al., 2023b), the R values calculated for 276 Sicilian rain gauges were used to map the R spatial distribution (iserosivity map) that, with Eq. (14), allowed us to estimate β and ε by Eqs. (8) and (9).

Eq. (10), which shows that C_T decreases as T increases, highlights that more cover or protection using soil conservation strategies and interventions is required as the return period increases (i.e., heavier storms). Di Stefano et al. (2023) and Carollo et al. (2023b) conventionally proposed to use C_T values corresponding to a return period of 1000 years, C_{1000} , to guarantee high-safety conditions for soil conservation. For the Sicilian region, Carollo et al. (2023b) also obtained that C_{1000} , estimated by Eq. (10) with $T = 1000$ and $x_{SL,T} = 2$, is related to the mean annual value of the erosivity factor by the following strong relationship (Fig. 2):

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

which is applicable in the C range 0.2–1 and is characterized by relative mean square error of 0.018. In spite of the limitation due to the application of a local $x_{SL,T}$ value, equal to 2, to a regional scale, this appears counterbalanced by the idea to propose an approach where TSL is determined by an objective criterion rather than with the use of conventional values that, although generally accepted, are based on qualitative considerations.

Due to the $x_{SL,T} - C_T$ relationship expressed by Eq. (6), the uncertainty on $x_{SL,T}$ is directly reflected on C_T . For example, for $x_{SL,T}$ ranging from 1.5 to 2.5, C_T (and C_{1000} of Eq. (15)) varies from 0.75 to 1.25 times the C_T value corresponding to $x_{SL,T} = 2$. The two scaled values correspond to more and less restrictive conditions to ensure compliance with

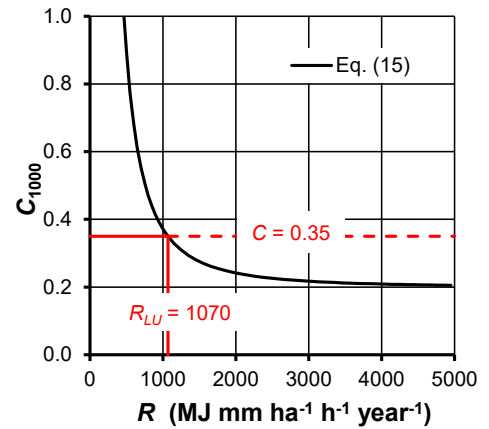


Fig. 2. Plot of the crop and management factor corresponding to soil loss of 1000-year return period and equal to the tolerance, C_{1000} , against the mean annual erosivity factor R . Embedded is a graphical example of the determination of the R value, R_{LU} , for which soil loss, for a land use characterized by $C = 0.35$, corresponds to the tolerance (cover and management factor $C = 0.35 = C_{1000}$).

soil loss tolerance, respectively.

Fig. 2 clearly shows that, for a given soil use (for example, $C = 0.35$), the rainfall erosivity factor R_{LU} (in the example, $R_{LU} = 1070$) that defines the crop-climate pair corresponding to the soil loss tolerance can be established. For a given crop, R_{LU} discriminates between conditions in which soil conservation interventions are needed ($R > R_{LU}$) or not ($R \leq R_{LU}$).

2.2. Evaluation of the PES

The payment of incentives to promote soil conservation strategies/interventions should be established considering how much the post-intervention soil loss is close to the TSL . In particular, the payment for a specific ecosystem service, PES (€ ha^{-1}), could be determined as:

$$\frac{PES}{PES_{ig}} = \frac{TSL}{SL_{a,T}} \quad (16)$$

in which PES_{ig} (€ ha^{-1}) is the target payment corresponding to a post-intervention condition characterized by $SL_{a,T} = TSL$. According to Eq. (16), the payment reduces proportionally to $TSL/SL_{a,T}$ for soil loss exceeding tolerance ($TSL/SL_{a,T} < 1$), while it increases if more conservative solutions ($TSL/SL_{a,T} > 1$) apply. From Eq. (5) with $T = 1000$ and Eq. (4) with $C = C_d$, in which C_d is the factor corresponding to the designed intervention, it follows:

$$TSL = R_{a,T} KLSC_{1000} \quad (17)$$

$$SL_{a,T} = R_{a,T} KLSC_d \quad (18)$$

and

$$\frac{TSL}{SL_{a,T}} = \frac{C_{1000}}{C_d} \quad (19)$$

Therefore, the following equation (*Scenario 1*) is obtained from Eqs. (16) and (19):

$$\frac{PES}{PES_{ig}} = \frac{C_{1000}}{C_d} = \frac{1}{C_d} \left[0.198 + \left(\frac{417}{R} \right)^2 \right] \quad (20)$$

In this scenario, interventions reducing C raise the PES , which reaches the target value when $C_d = C_{1000}$. The PES_{ig} value must account for the distance between the current C factor, named C_0 , and C_{1000} . In other terms, it should increase as the difference between C_0 and the optimal crop and management factor C_{1000} rises. If the target payment PES_{ig} is hypothesized proportional to the distance between C_0 and C_{1000} , it can be calculated by the following expression:

$$PES_{ig} = CEC(C_0 - C_{1000}) \quad (21)$$

where CEC is the Cost of Erosion Control (€ ha^{-1}). According to Kuhlman et al. (2010), CEC is the cost of the application, in areas of severe erosion ($>10 \text{ t ha}^{-1} \text{ year}^{-1}$), of measures such as the conversion of arable land into forest/pasture, terracing, buffer strips, residue management, cover crops, and conservation tillage. They quantified this cost at 296 € ha^{-1} . Considering that this value was quantified in 2010, it was adjusted here to account for the inflation and increasing costs of living, using a coefficient of increased value (<https://rivaluta.istat.it/>) equal to 1.312 and obtaining an updated value of 388 € ha^{-1} .

The intervention zones within a region should be established by policymakers using the actual land use map, defining the distribution of the current crop and management factor C_0 , and establishing crop factor values C_d corresponding to designed interventions assuring an expected soil loss less than or equal to TSL ($C_d \leq C_{1000}$). Within the region, priority could be given to areas characterized by the highest distance between C_0 and C_{1000} (i.e., the difference between the actual land use and that corresponding, for the erosivity condition of the area, to TSL). For areas with the same difference ($C_0 - C_{1000}$), priority could be reserved to

farmers who implement the most conservative strategies (the smallest C_d values less than C_{1000}). In other words, within areas characterized by a given PES_{ig} (Eq. (21)), farmers implementing more conservative solutions ($C_d \ll C_{1000}$), who also will receive a payment greater than the target value ($C_{1000}/C_d \gg 1$), could be favored.

The procedure for estimating C_{1000} by Eq. (10) requires the estimate of R and Weibull's parameters that, according to Eqs. (8) and (9), depend on R and $\sigma(R_d)$. Given the increasing evidence of non-stationarity in rainfall regimes under climate change, it is expected that Eqs. (11), (12) and (14), which were empirically obtained (D'Asaro and Santoro, 1983; Ferro et al., 1991) using recording raingauge data collected in the period 1951–1970, can be affected by temporal changes. According to the results by Wischmeier and Smith (1978), the R factor should be calculated by rainfall data corresponding to a period of 20–25 years, therefore the invariance of the relationships (11), (12) and (14) should be tested using more recent recording raingauge data measured in the period 2000–2025. Even if this analysis is beyond the scope of the present investigation, if we hypothesize that in the same range of the product $I_{1,2} I_{6,2} I_{24,2}$ the R values for 2000–2025 are greater than those for 1951–1970, in the more recent period we could obtain lower values of C_{1000} and, as a consequence of the increased difference between C_0 and C_{1000} , higher values of PES_{ig} according to Eq. (21). Hypothesizing that $C_d = C_{1000}$, according to Eq. (20), PES also increases.

If farmers are not willing to adopt a land use that alone ensures soil loss does not exceed the tolerable threshold, but are willing to shift from the current land use (C_0) to a more conservative one ($C_{1000} < C_d < C_0$), additional soil conservation practices, represented by the P factor of the USLE (*Scenario 2*), can be implemented simultaneously. When sloping soils have to be cultivated, the covering action of a crop can be coupled with practices that slow runoff, reducing the amount of transported soil particles. According to Wischmeier and Smith (1978), “the most important of these supporting cropland practices are contour tillage, stripcropping on the contour, and terrace systems”.

Accordingly, in *Scenario 2*, the PES/PES_{ig} calculation can be obtained as:

$$\frac{PES}{PES_{ig}} = a \frac{C_{1000}}{C_d} + \frac{b}{P_d} = \frac{a}{C_d} \left[0.198 + \left(\frac{417}{R} \right)^2 \right] + \frac{b}{P_d} \quad (22)$$

in which P_d is the designed P factor, PES_{ig} is calculated by Eq. (21), and a and b are two coefficients. They represent the weights of the changes in the crop and management factor and the support practice factor. Indeed, $C_d < C_0$ and P_d is lower than the initial value of the support practice factor. Considering that the condition $a + b = 1$ must be satisfied, Eq. (22) reduces to Eq. (20) (*Scenario 1*) for $a = 1$.

Alternatively, *Scenario 3* relies on the simultaneous change of the current land use and slope length. Specifically, it can be implemented by reducing C_0 to a C_d value higher than C_{1000} and the slope-length factor to a designed value, $L_{c,d}$, lower than the current one, L . By imposing the coincidence between $SL_{a,T}$ (Eq. (4)) and TSL , one obtains:

$$R_{a,T} KL_{c,d} SC_d = R_{a,T} KLSC_{1000} \quad (23)$$

where, according to Eq. (5) with $T = 1000$, the second member equals TSL . Eq. (23) leads to the following expression of the designed length factor:

$$L_{c,d} = \frac{C_{1000}}{C_d} L \quad (24)$$

Similarly to Eq. (22), the following expression can be applied to calculate the payment:

$$\frac{PES}{PES_{ig}} = a \frac{C_{1000}}{C_d} + \frac{b}{L_{c,d}} \quad (25)$$

where a and b ($= 1 - a$) are the weights of the two terms of the second member of Eq. (25). Considering that when $SL_{a,T} = TSL$, $PES/PES_{ig} = 1$

and Eq. (24) is valid, introducing Eq. (24) into Eq. (25) leads to:

$$1 = a \frac{C_{1000}}{C_d} + \frac{(1-a)C_d}{LC_{1000}} \quad (26)$$

The a coefficient, satisfying Eq. (26), has the following expression:

$$a = \frac{\frac{C_d}{C_{1000}} - \left(\frac{C_d}{C_{1000}}\right)^{\frac{1}{L}}}{1 - \frac{1}{L} \left(\frac{C_d}{C_{1000}}\right)^{\frac{1}{L}}} \quad (27)$$

Eqs. (25) and (27) define the payment for *Scenario 3*, which is applicable when the a values calculated by Eq. (27) are positive and lower than 1, as $0 < a < 1$ by definition. Conversely, when a falls outside this range, *Scenario 3* cannot be applied to calculate the payment.

Since Eqs. (20), (22), and (25) include a term depending on R , the effect of R variability on PES should be considered. Within this variability, the effect of the interpolation error due to the applied kriging method (Carollo et al., 2023b) on R estimate is negligible compared to the spatial variability of R . As an example for *Scenario 1*, the variability of PES with R , is analyzed by the following derivative of Eqs. (20) with PES_{ig} calculated by Eq. (21):

$$\frac{dPES}{dR} = \frac{CEC}{C_d} \left(-2 \cdot \frac{417^2}{R^3} \right) \left(C_0 - 2 \cdot 0.198 - 2 \cdot \frac{417^2}{R^2} \right) \quad (28)$$

3. Results and discussion

For a given crop, in *Scenario 1*, the soil conservation strategy is implemented by changing the land use to more conservative crops, characterized by smaller C_d values. Fig. 3 shows, as an example, the application of Eq. (20) for arable land characterized by different C_d values with the increase in PES/PES_{ig} for decreasing C_d values. Fig. 3 also shows that the R_{LU} value, for which soil loss is equal to tolerance and $PES/PES_{ig} = 1$, increases from 1070 to 1800 as C_d lowers from 0.35 to 0.25. For $C_d = 0.15$ $PES/PES_{ig} > 1$ regardless of the climatic condition investigated. In other words, the rainfall forcing is not a limiting factor and conditions characterized by $SL_{a,T} < TSL$ always occur, determining a PES higher than PES_{ig} . This is a realistic condition as C_d even lower than 0.15 regards, for example, land use characterized by fruit trees regardless of the type of tillage, or grain corn, spring and winter cereals, seasonal vegetable crops, all with row tillage or no tillage (Bagarello and Ferro, 2006).

With respect to arable land (mainly cereals and legumes, $C_0 = 0.35$) and vineyards ($C_0 = 0.5$) in Sicily, Carollo et al. (2023b) identified areas with soil loss exceeding tolerance ($C_0 > C_{1000}$) and provided maps at the regional scale. The map related to arable land shows that areas with soil

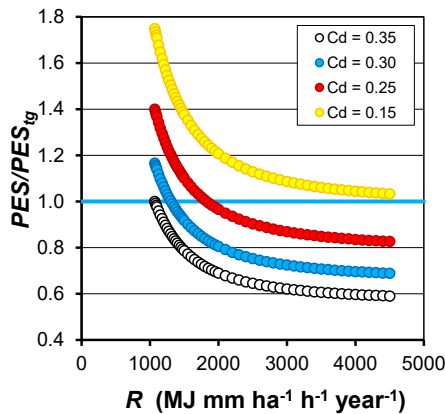


Fig. 3. Variation of the ratio between the payment and its target value, PES/PES_{ig} , over the range of mean annual erosivity factor R , for different reductions from the initial C factor to designed C_d values (*Scenario 1*, Eq. (20)).

loss exceeding tolerance are located in the eastern part and in the center of the island, while that concerning vineyards depicts that they are located in western Sicily, in the south and close to Mount Etna. In all these areas, following *Scenario 1* (Eqs. (20) and (21)), the reduction from C_0 to $C_d \leq C_{1000}$ determines a PES (€ ha^{-1}) varying with R and C_d , as shown in Fig. 4. Due to the higher C_0 for vineyards than arable lands, for the same designed crop and management factor C_d , a higher payment is made in the former case than in the second case.

The application of Eq. (22) (*Scenario 2*) needs a criterion to estimate the designed P_d factor. For example, Table 13 of the USLE Handbook N.537 (Wischmeier and Smith, 1978) lists the P values with the slope-length limits and slope ranges for contouring. The slope-length limits were established to ensure that, for a given slope steepness s (%), varying from 1 to 25%, slope is short enough for the full effectiveness of the control practice. Using these (s , P) data pairs, the following relationship was obtained (Fig. 5):

$$P_d = 0.3708 + 0.0228s \quad (29)$$

Fig. 6 shows, as an example for $C_d = 0.35 < C_0$, the application of Eq. (22) by setting $a = 0.9$ and $b = 0.1$ (Fig. 6a) or $a = 0.5$ and $b = 0.5$ (Fig. 6b). When the weight of the component of Eq. (22) due to the P factor is low ($b = 0.1$, Fig. 6a), the range of R in which PES/PES_{ig} is greater than or equal to 1 is very limited. In particular, PES/PES_{ig} is higher than 1 for low R values (no severe rainfalls), while it is less than 1 in a wide range of mean annual rainfall erosivity. Conversely, when the two components of Eq. (22) have the same weight ($a = b = 0.5$, Fig. 6b), $PES/PES_{ig} > 1$ for slope-steepness less than or equal to 15%. In any case,

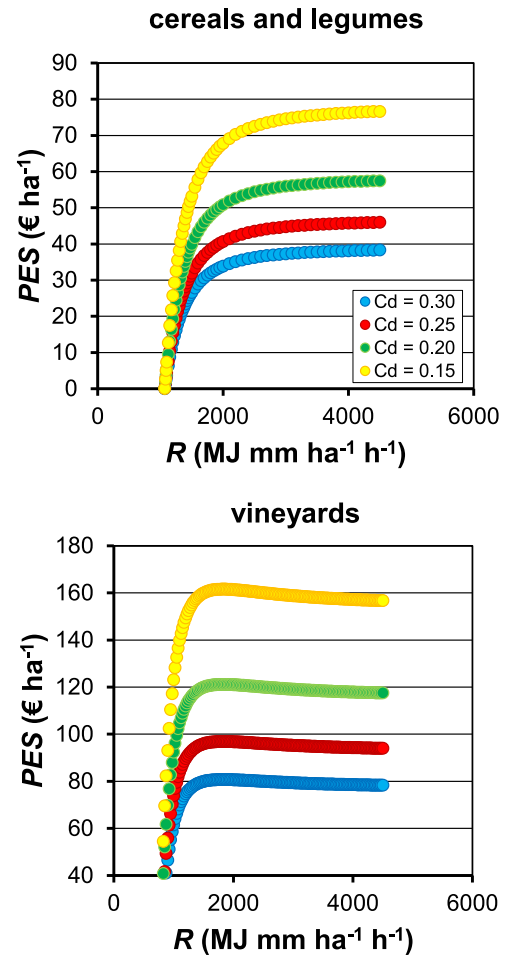


Fig. 4. Variation of the payment PES across the range of mean annual rainfall erosivity R for different values of the designed crop and management factor C_d and two land uses (cereals and legumes and vineyards).

Table 1

Values, as an example for $C_d = 0.35$, of the rainfall erosivity factor R_{LU} for which $PES = PES_{ig}$ under *Scenario 2* (reduction from C_0 to C_d and contouring implemented for different slopes s) for different a values.

a	s (%)	R_{LU} (MJ mm ha ⁻¹ h ⁻¹)
1		1070
0.9	5	1253
	15	1129
	25	1078
0.75	5	2491
	15	1287
	25	1096
0.5	5	*
	15	3980
	25	1157
0.25	5	*
	15	*
	25	1423

* Absence of R_{LU} means that PES is higher than PES_{ig} over the entire range of rainfall erosivity R .

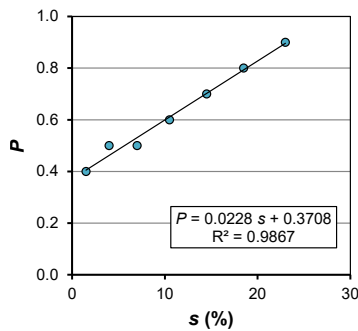


Fig. 5. Plot of the (s, P) pairs, where s = slope-steepness s (%) and P = support practice factor for contouring, and Eq. (28).

Fig. 6 allows for quantifying the expected increases in PES/PES_{ig} relative to *Scenario 1* (white dots in Fig. 6) deriving from the use of the control practice combined with the reduction of the crop and management factor. In other words, under the same C_d value, *Scenario 2* allows for a higher payment under the considered slope range as it includes a double intervention of soil conservation. Table 1 lists, for an example concerning $C_d = 0.35$, the values of the rainfall erosivity factor R_{LU} for

which $PES = PES_{ig}$ under *Scenario 2* and different a weights. Considering that PES is higher than the target for $R < R_{LU}$, it is evident that these conditions are favoured by slope and a decrease. Additionally, PES is higher than PES_{ig} over the entire range of rainfall erosivity for the lower slopes and $a = 0.25$ and 0.5 , and it is confirmed the beneficial effect of a double intervention compared to the single one of *Scenario 1* ($a = 1$). Indeed, this last case is characterized by the minimum R_{LU} value (1070 MJ mm ha⁻¹ h⁻¹ year⁻¹) and minimum range of R in which soil loss is less than TSL and no conservation strategies are strictly required.

Finally, *Scenario 3* (Eqs. (25) and (27)) is tested for different values of C_d , L , and $L_{c,d}/L$. The latter expresses the reduction of the slope length factor. For example, for $C_d = 0.45$ and $L = 1.1$, Fig. 7 shows that PES/PES_{ig} is almost systematically lower than 1 for $L_{c,d}/L \geq 0.7$, while it can be higher than 1 under this threshold within different ranges of R . The extent of these ranges increases as $L_{c,d}/L$ decreases and spans the entire R domain for $L_{c,d}/L = 0.45$.

Scenarios 1 and *3* can yield the same payment. For example, from Fig. 7 it is evident that the payment under *Scenario 3* for $C_d = 0.45$, $L = 1.1$, and $L_{c,d}/L = 0.8$ equals that under *Scenario 1* and $C_d = 0.3$ for $R > 2500$. This means that a 20% reduction of the slope length factor offsets the adoption of a less conservative cover and management factor. For $1300 < R < 2500$, the two scenarios provide the same payments if $0.7 < L_{c,d}/L < 0.8$.

Under the hypothesized C_d (0.45) and L (1.1) values, the a is positive and less than 1 over the entire explored range $1070 \leq R \leq 4500$. This implies that Eq. (25) is always applicable. Instead, some $C_d - L$ combinations yield (Eq. (27)) unacceptable a values and the possibility to apply Eq. (25) within limited ranges of R . For example, Fig. 8 demonstrates that, for $C_d = 0.45$ and $L = 2$ (Fig. 8a), Eq. (25) can be applied for $R > 2600$, while for $C_d = 0.55$ and $L = 2$ (Fig. 8b), its application is limited to $R > 1600$. For the common range of $R > 2600$ and fixed $L_{c,d}/L$, the lower C_d value assures a higher payment.

Contour ditches are a practically affordable intervention aimed at reducing the slope length factor. A more expensive intervention concerns bench terraces that, although they require remarkable investment, are widely used in agriculture to enhance crop productivity (Food and Agricultural Organization, 2019).

The relative magnitude of payments for the three different scenarios can be sensitive to the $x_{SL,T0}$ value and, as a consequence, C_{1000} . As reported above, for $x_{SL,T0}$ ranging from 1.5 to 2.5, C_{1000} varies from 0.75 to 1.25 times the C_{1000} value corresponding to $x_{SL,T0} = 2$. Both *Scenario 2* and *3* rely on C_d values (here denoted as $C_{d,2}$ and $C_{d,3}$) that, reasonably, are higher than that adopted in the *Scenario 1* (here denoted as $C_{d,1}$).

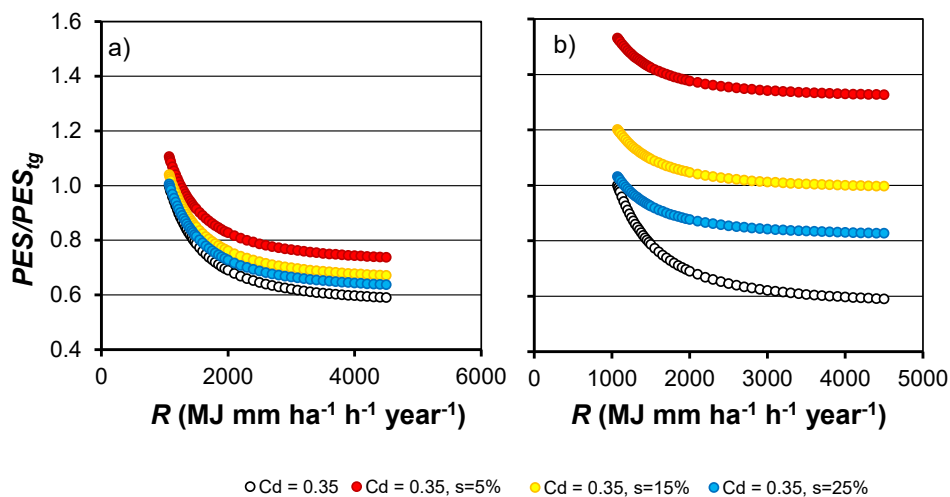


Fig. 6. Relationship between the mean annual erosivity factor R and ratio between the payment and its target value, PES/PES_{ig} , under the soil conservation intervention implemented, for example, by both the reduction of C factor to the designed $C_d = 0.35$ and contouring modeled with the slope s – designed support practice P_d relationship (*Scenario 2*, Eqs. (22) and (29)) for (a) $a = 0.9$ and $b = 0.1$, (b) $a = 0.5$ and $b = 0.5$.

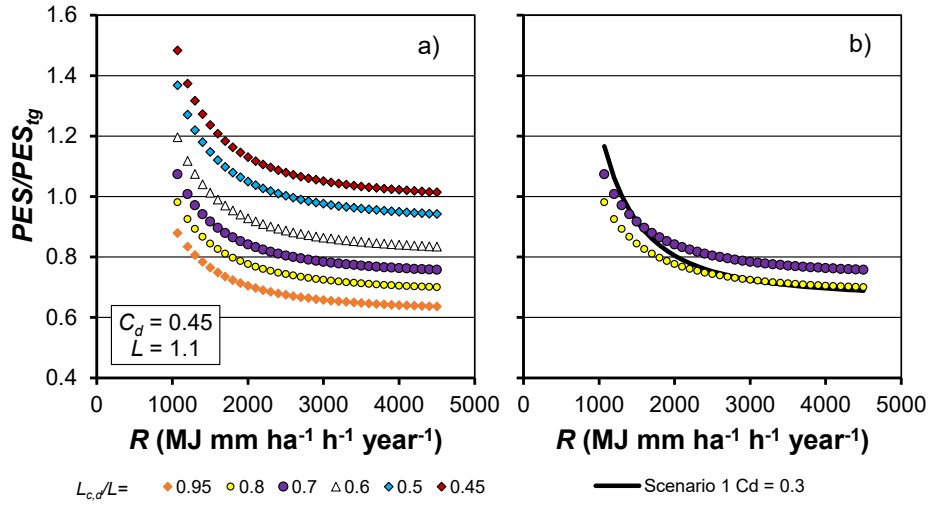


Fig. 7. (a) Variation of the ratio between the payment and its target value, PES/PES_{tg} , over the range of mean annual erosivity factor R , under the soil conservation intervention implemented, for example, by both the reduction of C factor to the designed $C_d = 0.45$ and the reduction of the initial slope length factor $L = 1.1$ to the designed $L_{c,d}$ (Scenario 3, Eqs. (25) and (27)). The comparison between two curves from Scenario 3 and that related to Scenario 1 with $C_d = 0.3$ is also shown in panel (b).

This occurs because Scenarios 2 and 3 are structured to compensate a less conservative (i.e., higher) C_d with a second intervention by the support practice factor or slope length factor. From Scenario 2 (Eq. (22)) and Scenario 1 (Eq. (20)) one obtains:

$$\frac{PES_{scenario2}}{PES_{scenario1}} = \frac{a}{\beta} + \frac{b}{P_d} \frac{C_{d1}}{C_{1000}} \quad (30)$$

where $\beta = C_{d,2}/C_{d,1}$ is higher than 1 and C_{1000} is the value corresponding to $x_{SL,T0} = 2$. The right-side member of Eq. (30) is higher than 1 (i.e., $PES_{scenario2} > PES_{scenario1}$) when

$$C_{1000} < \frac{C_{d1}b}{\left(1 - \frac{a}{\beta}\right)P_d} \quad (31)$$

The substitution of C_{1000} with $0.75 C_{1000}$ implies that, for maintaining C_{1000} as a single left-side term of the inequality (31), its right-side member has to be divided by 0.75. Accordingly, the adjusted inequality continues to be valid and $PES_{scenario2}$ is confirmed to be higher than $PES_{scenario1}$. Similarly, the right-side member of Eq. (30) is lower than 1 (i.e., $PES_{scenario2} < PES_{scenario1}$) when

$$C_{1000} > \frac{C_{d1}b}{\left(1 - \frac{a}{\beta}\right)P_d} \quad (32)$$

Dividing the right-side member of the inequality (32) by 1.25, the adjusted inequality continues to be valid, and $PES_{scenario2}$ is confirmed to be lower than $PES_{scenario1}$. In other cases, the sign of the adjusted inequality (right-side member of (31) divided by 0.75 or of (32) divided by 1.25) is not analytically determinable *a priori* as depending on the specific values of different parameters ($C_{d,1}$, b , a , β , and P_d). Similar reasonings and conclusions apply when comparing the relative magnitude of $PES_{scenario3}$ and $PES_{scenario1}$ (setting in the inequality (31) and (32), $\beta = C_{d,3}/C_{d,1}$, and changing P_d with $L_{c,d}$). Summarizing, a multiplicative factor of 0.75 for C_{1000} does not determine changes on the relative magnitude between $PES_{scenario1}$ and those of each alternative scenario when, in the reference condition (i.e., C_{1000} obtained for $x_{SL,T0} = 2$), the latter are higher than $PES_{scenario1}$. Additionally, a correction factor of 1.25 for C_{1000} does not determine changes on the relative magnitude when, in the reference condition, PES from each alternative scenario is lower than $PES_{scenario1}$.

In accordance with Eq. (28), being CEC a constant, $dPES/dR$ depends

on C_0 , C_d and R , thus the analysis of the effect of R variability on PES requires a numerical evaluation by fixing the values of C_0 and C_d . As an example, for $C_0 = 0.35$ (cereals and legumes) and $C_d = 0.1$ and 0.2 , Fig. 9 shows that $dPES/dR$ is positive, inversely related to R and tends to 0 for R greater than about 3000; in other words, PES increases weakens for increasing R values and PES tends to become constant, confirming the pattern depicted in Fig. 4. This tendency is common for all the three scenarios because C_{1000} , and PES_{tg} consequently (Eq. (21)), tend to be invariant for R over 3000 (Fig. 2). Therefore, Eqs. (20) and (22), and Eq. (25) with a expressed by Eq. (27) give an almost constant PES for these higher values of the mean annual rainfall erosivity.

In conclusion, the suggested modeling of the PES , related to climatic, support practice, and slope length factors, could be a useful tool to assist policymakers and public authorities in economically supporting farmers who choose to adopt more environmentally sound land use practices. A point deserving further attention is establishing the weights (a and b coefficients) of the two components that determine the payment according to Scenario 2. This choice should be adopted by considering specific territorial conditions. For example, for guaranteeing the conservation of agricultural soil vocation (e.g., wheat in the Sicilian hinterland) or cash crops (e.g., vineyards), which significantly support the local economy, more weight should be given to b . Conversely, if changing land use is considered a priority because it allows other challenging objectives to be achieved (e.g., water saving), then more weight should be given to a . High slope steepness is another constraint that requires lowering of the b weight. Indeed, areas steeper than 25% limit contouring and P_d cannot be rigorously determined by Eq. (29). Finally, if there are no specific needs like those reported above, a more balanced weight distribution ($a \approx b \approx 0.5$) can be applied.

Within a zone characterized by C_0 and C_{1000} values, i.e., a fixed value of PES_{tg} , the policymakers could state priority by choosing the most conservative strategies in any case, leaving the farmers free to implement to obtain the lowest C_d using Scenario 1 or to combine a reduction of the crop factor with additional soil conservation practices (Scenarios 2 and 3). The farmers will choose the type of soil conservation intervention depending on local morphological conditions and individual choices such as conserving agricultural soil vocation.

4. Conclusions

This paper focused on the payment for the ecosystem service of soil erosion control under three different scenarios. The proposed approach

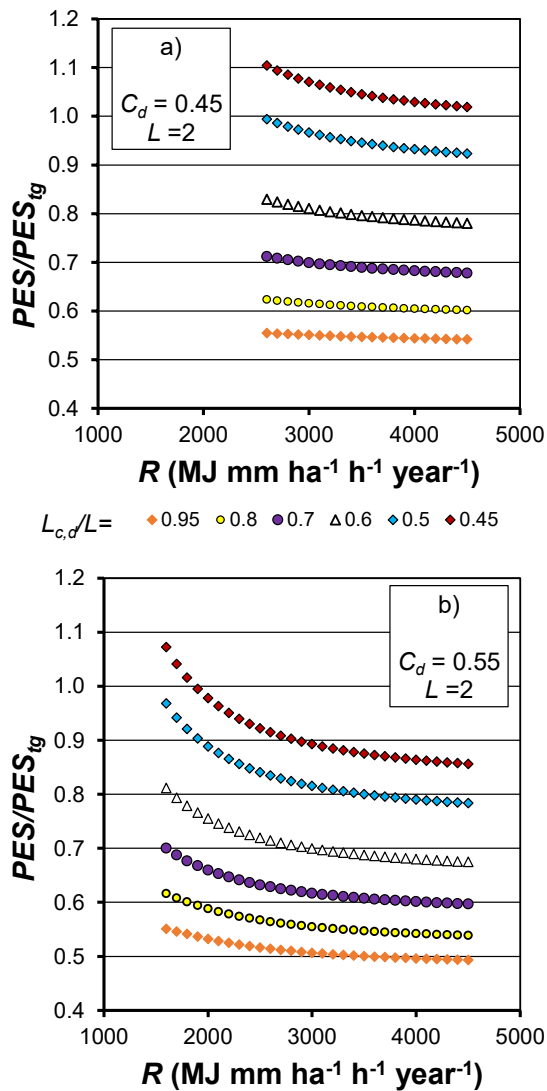


Fig. 8. Effect of different designed C_d values on the relationship of the ratio between the payment and its target value, PES/PES_{ig} , versus the mean annual erosivity factor R under the soil conservation intervention implemented by Scenario 3 (Eqs. (25) and (27)). The two panels depict the curves featured by varying ratios between the designed slope length factor $L_{c,d}$ and the initial one L , as an example, for $L = 2$ and (a) $C_d = 0.45$ and (b) 0.55 .

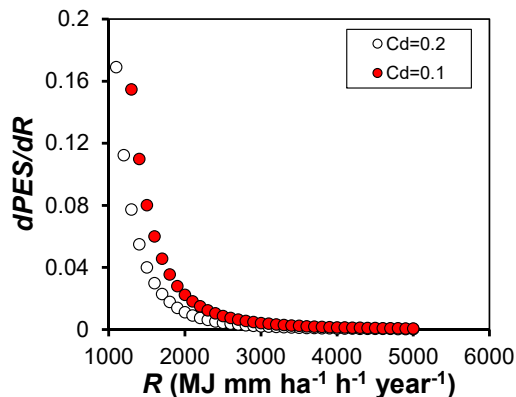


Fig. 9. Effect of the variability of the mean annual erosivity factor R on PES variability, expressed by Eq. (28), across the explored R range, as an example for initial crop and management factor $C_0 = 0.35$, and designed C_d values of 0.1 and 0.2.

relies on the soil loss tolerance defined by a new method using the crop and management factor of the USLE.

Using Scenario 1, the analysis demonstrated that a drastic reduction of the C factor can determine PES/PES_{ig} values greater than 1 for any climatic condition investigated. The results obtained for Scenario 2 suggested that, when the two components due to C and P factors are equally weighted, the use of a control practice, such as contour tillage, yields a higher payment than the target value for slope steepness $\leq 15\%$. Under the same C_d value, Scenario 2 always yields a higher payment compared to Scenario 1. Finally, the analysis developed for Scenario 3 pointed out that the reduction of the topographic factor can lead the PES over the target value. Moreover, this reduction can offset a less marked reduction of the C factor compared to Scenario 1, resulting in the same payment between the two scenarios.

The proposed method for estimating the payment for ecosystem services related to climatic conditions (R factor), C , P and L factors could be useful to address the choices of policymakers and public authorities and define impartial criteria to establish economic aid for farmers who decide to realize environmentally sound land use.

The proposed method requires that policymakers identify zones within a region where conservation strategies are required defining the distribution of C_0 , and C_d values equal to or lower than C_{1000} . Within each zone, characterized by a constant value of PES_{ig} , policymakers could prioritize the most conservative strategies and leave the farmers free to implement one of the three available scenarios in regard to local morphological conditions and individual farmer choices, such as conserving agricultural soil vocation. This framework and methodological aspects can be transferred to different regions, characterized by different climate, land use, and morphology, where the specific context could guide the choices of policymakers and farmers differently than in the Sicilian region.

CRedit authorship contribution statement

Costanza Di Stefano: Writing – review & editing, Writing – original draft, Visualization, Validation, Methodology, Investigation, Formal analysis, Data curation, Conceptualization. **Alessio Nicosia:** Writing – review & editing, Writing – original draft, Visualization, Validation, Methodology, Investigation, Formal analysis, Data curation, Conceptualization. **Vincenzo Palmeri:** Writing – review & editing, Writing – original draft, Visualization, Validation, Methodology, Investigation, Formal analysis, Data curation, Conceptualization. **Vincenzo Pampalone:** Writing – review & editing, Writing – original draft, Visualization, Validation, Methodology, Investigation, Formal analysis, Data curation, Conceptualization. **Maria Angela Serio:** Writing – review & editing, Writing – original draft, Visualization, Validation, Methodology, Investigation, Formal analysis, Data curation, Conceptualization. **Vito Ferro:** Writing – review & editing, Writing – original draft, Visualization, Validation, Supervision, Methodology, Investigation, Formal analysis, Data curation, Conceptualization.

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.

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Data availability

Data will be made available on request.

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