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Prevention of erosion in mountain basins: A spatial-based tool to support payments for forest ecosystem services

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Abstract: This paper presents a spatial-based decision support system (DSS) to assist public and private forest managers in the analysis of potential feasibility in payments for forest ecosystem services (PES) for the prevention of soil erosion. The model quantifies the maximum willingness to pay (*WTP*) of managers of a reservoir to prevent soil loss. The minimum willingness to accept (*WTA*) of forest owners for the activation of a private market is also computed. The comparison of *WTP* and *WTA* identifies the forest area where PES are ideally feasible with additional potential for compensation to enable the schemes. The DSS highlights forest idiosyncrasies as well as local socio-economic and geomorphological characteristics influencing PES suitability at a geographic level. The potential applications and future improvements of the model are also discussed.

Keywords: environmental payments; forest management; funding mechanism; soil erosion; spatial analysis

Soil erosion processes are well known and represent a global concern. In Europe, findings from the European Sediment Research Project (SedNet) (Förstner, Salomons 2010) suggested that soil and water policies could fail if governments do not seriously address sediment issues. Deforestation and land use conversion are major causes of soil erosion due to increased runoff, flood risk, and sediment loads in water bodies. Adequate forest management is of primary importance to stabilise soil and reduce erosion, especially in the proximity of artificial lakes. Protection from soil erosion is an important forest ecosystem service (ES) (MEA 2005; Jones-Walters, Mulder 2009; Haines-Young, Potschin 2012), where if this service were to be reduced or altered, water basins may be severely damaged. Unfortunately, there are contrasting objectives between forest management and reduction of erosion, because conserving the soil through forest protection services is only attainable within the limits of forest exploitation. Therefore, forest owners may incur economic losses if they agree to maintain adequate levels of protection against forest soil erosion.

In situations of payments for contrasting stakes for ecosystem services (PES), one solution is to balance the productive objective of forest management and the quality of water basins. PES are market-based tools that help with the valorisation of environmental externalities using market approaches, where buyers and sellers of ecosystem services meet to achieve reciprocal benefits (Engel et al. 2008). In a PES scheme, the ES seller (i.e., forest owner) and ES buyer (artificial lake manager) agree to a number of management actions that the seller undertakes

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to assure the provision of soil erosion protection and a monetary compensation for this service is provided. Sone et al. (2019) described an example of PES developed in Brazil for soil conservation practices, which consisted of terracing a sub-basin area to restore riparian vegetation, which led to a 25% reduction in soil erosion. Havinga et al. (2020) applied a spatial model to boost the efficiency of PES schemes through an economic quantification of the value for the prevention of soil erosion in Costa Rica. The ancillary benefits of diminished siltation in forestry using PES are also found in the literature: for example, Ranjan (2019) described its impact on fishery in Chilika Lake (India), Jones et al. (2017) examined the reduction of wildfire risk in Colorado, and Lu and He (2014) explored the effects on water quality in the Shaying River watershed (China).

PES have some distinctive characteristics that should always be accurately described when planning a scheme (Wunder 2005). Some of these characteristics are: (1) the scheme is voluntary, (2) the service is accurately described, (3) the area subject to PES is geographically identified, (4) the payment is conditional on the actual provision of the service, and (5) the amount of payment should be at least equal to the economic loss incurred by forest owners to provide the service and not higher than the total economic value of the service.

To account for all these PES characteristics, the scheme requires a detailed description of the environmental, morphological, and economic variables. At present, the literature lacks procedures that systematically account for all PES variables. Many studies have focused on the spatial dimension and risk assessment of forest protection from an erosion perspective (Cotler, Ortega-Larrocea 2006; Schmidt, Wei 2006; Nasiri 2013; Borrelli, Schütt 2014; Panagos et al. 2015a, b; Borrelli et al. 2017; Didoné et al. 2017; Lang et al. 2017). The role of forests in sediment retention was also evaluated from a geographic perspective by Paudyal et al. (2019) in a case study in Nepal. The author of this paper assesses the flow of this service in relation to land use and changes in land coverage.

The economic aspects of PES are often assessed using non-market valuation procedures and willingness to pay (*WTP*). Coastal (e.g., Alves et al. 2015; Dribek, Voltaire 2017; Enriquez-Acevedo et al. 2018) and soil (Asrat et al. 2004; Colombo et al. 2005) erosion has been broadly studied; however, the value of forest protection from erosion is less frequently examined in the literature. A Mexican study has found that the avoided cost of erosion gained from forests amounted to 100 $USD\cdot ha^{-1}$ (Adger et al. 1995). In a study in Catalonia, Brey et al. (2007) included avoided erosion as an attribute in a choice experiment and found that people are willing to pay for forest protection against erosion for an annual fee of 1 216 EUR. In Arizona, Yoo et al. (2014) combined a sediment delivery model with a hedonic price function that returned an average value of forest protection in the range of 140–330 $USD\cdot t^{-1}$ of sediment.

The cited papers suggest that geomorphological and economic analyses of erosion are often undertaken separately, and little attention is given to their integration. Therefore, an accurate analysis of erosion should comprise both the area potentially affected by erosion and the related economic effects.

In this study, we fill this gap with a methodology that detects suitable forest management to prevent soil erosion as well as a base mechanism to implement PES in a basin. Our approach uses spatial data to identify the following: (*i*) areas potentially affected by erosion, (*ii*) a risk analysis to reveal potential economic damage from erosion, (*iii*) economic losses for providers of the ecosystem service, and (*iv*) a system of compensation between participants in the scheme in case of a lack of financial feasibility. The procedure was tested in a case study in the Municipality Union of the Sieve and Arno valleys in Tuscany (Italy).

The remainder of this paper is structured as follows. In the second section, the general framework of the study, the examined area, the hypothesised scenario, and the methodology are briefly described. The third section is dedicated to the results. The fourth section discusses the results, and the final section suggests the policy and practical implications of the method and offers additional conclusions.

MATERIAL AND METHODS

General framework for PES. The model analyses the contribution of forests to diminish the erosion of surface soil and the potential areas for PES schemes (Figure 1). Avoided erosion from business-as-usual (BAU) activities (i.e., current forest management) to a PES scenario is computed for different forest types and management strategies. Prevention of soil loss at a basin level is then converted into economic value in relation to the cost

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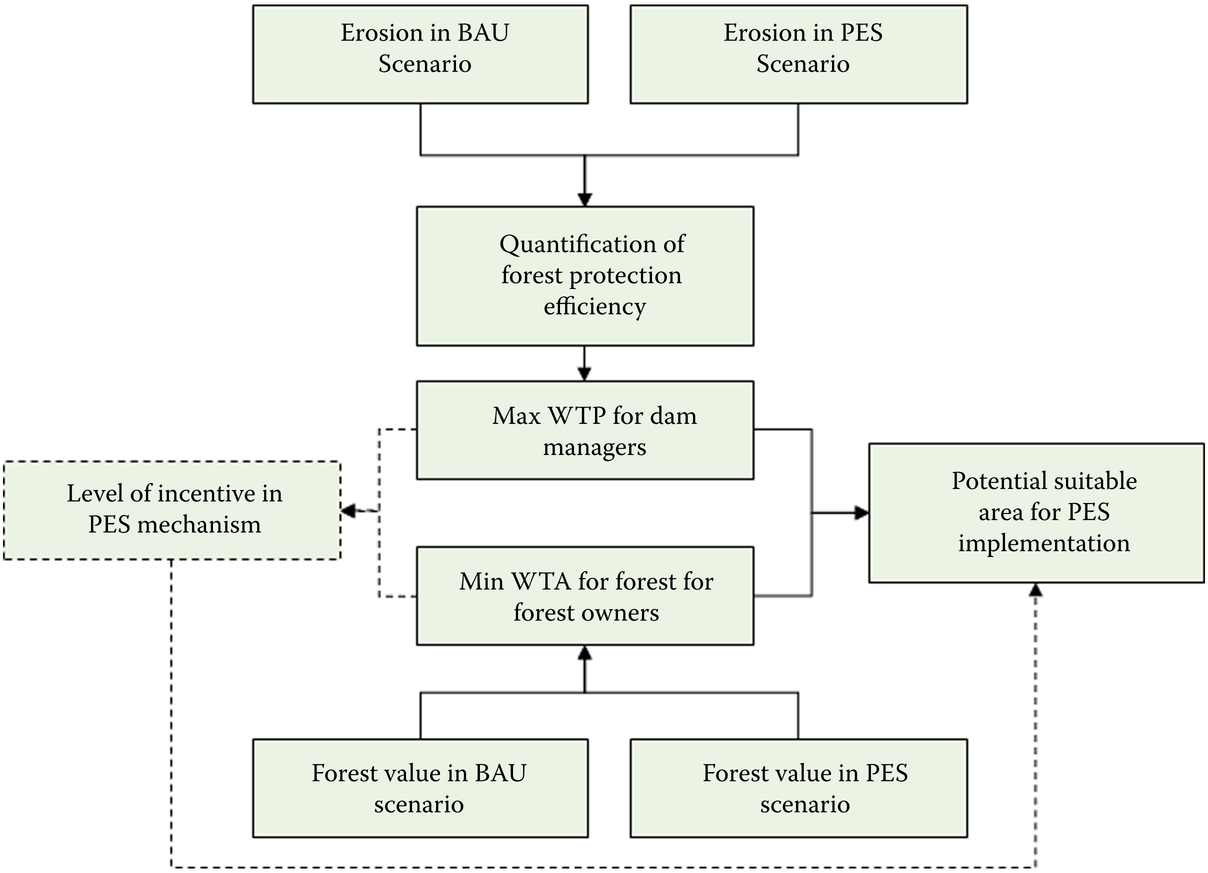

Figure 1. Logical flow for the quantification of *maxWTP*, *minWTA*, potential area for PES suitability, and incentives for PES activation

maxWTP – maximum willingness to pay; *minWTA* – minimum willingness to accept; PES – payments for forest ecosystem services

of artificial lake maintenance (emptying). The feasibility of PES is eventually evaluated based on two parameters: (*i*) the difference between maximum willingness to pay (*maxWTP*) for reservoir emptying and minimum willingness to accept (*minWTA*) alternative forest management carried out by forest owners (from BAU to PES), and (*ii*) the potential level of public/private entities' contribution in favour of the PES mechanism.

The overall approach considers the conditionality concept (the subject who maintains the reservoir pays if an intervention on the forest is made – an input-based approach) as well as the additionality notion that the payment is accomplished if and only if an improvement in respect to the BAU scenario is achieved (Smith et al. 2013). The model is implemented on a spatial basis (GRASS GIS software v. 7.6, 2019).

Smith et al. (2013) identified five phases to implement a PES market: (1) identify a saleable ecosystem service and prospective buyers and sellers, (2) establish PES scheme principles and resolve technical issues, (3) negotiate and implement agreements, (4) monitor, evaluate, and review implementation, and (5) consider opportunities for multiple-benefit PES. The present work concerns phase 1 and focuses on the determination of a geographic area where a PES is potentially suitable and the level of economic feasibility for providers and beneficiaries of the environmental benefit. Thus, the DSS can be considered as a support in the preliminary step of the PES mechanism.

Study area and territorial informative system (TIS). The study area is located in the Tuscan sector of the Italian Apennines, specifically in the Municipality Union of the Sieve and Arno valleys in the province of Florence. The territory has a surface area of 49 500 ha and has a low population density (approximately 1.3 inhabitants·ha^{-1}). About 62% of the surface is covered by forest,

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mainly broadleaf forests (52%), followed by conifers (6%), and mixed forests (4%). An artificial lake (dam) located on the Moscia River was considered for the case study (Figure 2). The analysed dam has a height of 7.5 m and was completed in 1973. Huge solid loads have always characterised the Moscia River's torrents. Emptying operations are needed to reduce the flood risk of the river during calamitous events. The dam on the Moscia River was designed with the dual purpose of stopping the solid flow and preventing it from settling downstream, as well as to lighten the flow to cause a natural deepening of flooded riverbed sections. The dataset is composed of a digital terrain model (DTM) and current forest typologies derived from the TIS and the land use monitoring project of the Tuscany

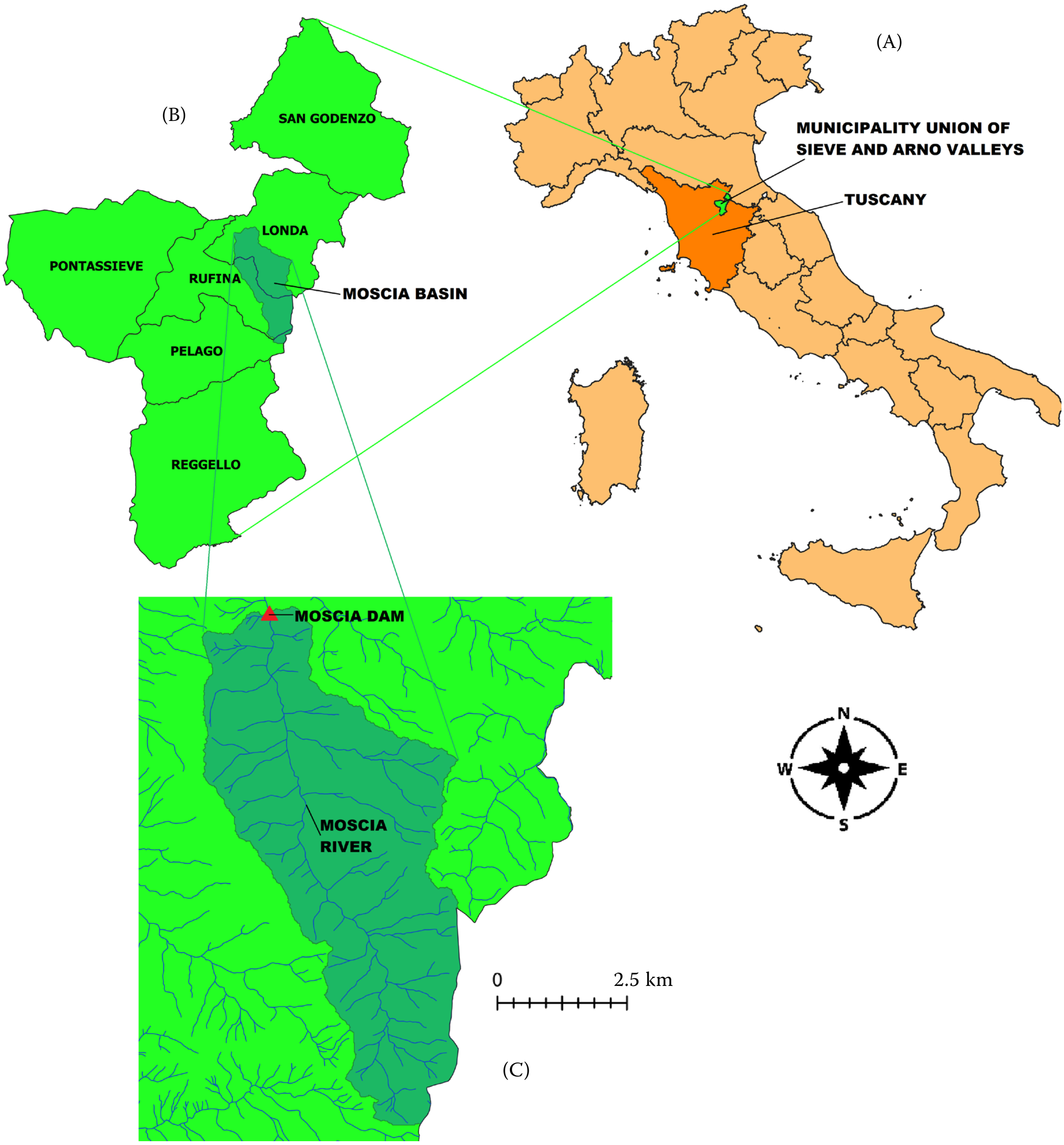

Figure 2. Localization of the case study: (A) Tuscany region and Municipality Union of Sieve and Arno valleys, (B) municipalities and Moscia basin, (C) focus on rivers of the Moscia basin and localisation of the dam; scale bar refers to Figure 2C

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region (http://www502.regione.toscana.it/geoscopio/cartoteca.html). Forest management characteristics are introduced based on regional forest inventory cartography (http://www502.regione.toscana.it/geoscopio/cartoteca.html). The adopted resolution is 10 × 10 m with a squared pixel.

The multifunctional role of local forests is important, particularly in terms of public properties, despite non-productive functions that are not usually monetized. Private stand management is usually focused on productive function (particularly in coppices of small-sized private woodlands); therefore, forest owners are not incentivised to undertake additional interventions, other than those legally prescribed, to improve multifunctionality. Due to morphological conditions that are typical of mountainous areas and the presence of artificial lakes, a PES scheme could represent a solution to both maintain forest protective functions and allow for additional income for forest owners.

Scenario analysis. The potential suitability of forest areas for a PES scheme is based on a comparison of two scenarios: a BAU scenario based on current forest practices and a PES scenario based on the optimisation of interventions for the prevention of erosion. The BAU scenario is defined by the ordinary silvicultural systems applied in the forests of the Municipality Union of the Sieve and Arno valleys. Current practices as well as a PES scenario were defined through interviews with technicians of the Municipality Union and experts in the forest sector (researchers). The results describing the two scenarios for each forest management type (coppice or high forest) and forest typology are summarised in Table 1.

Table 1 shows that in a BAU scenario, high forests are managed with an intermediate thinning (in general at 20 or 30 years) with the cover maintained at 80%. Final harvesting in a conifer forest (or a prevalence of conifers) is carried out by means of strip cutting with a rotation period of 70 years and a prescribed final yield of 65%. High forests of broadleaf trees are managed through shelterwood cutting with a rotation age ranging from 60 to 90 years. In a BAU scenario, thinning is not applied to coppices. Final harvesting is developed as clearcutting with standards release (from 30 to 60 years according to the species and based on regional forest norms), apart from European beech forests that are managed with a coppice selection system.

Table 1. Silvicultural characteristics for the scenario, forest management and forest typology

		Forest typology					
		Norway spruce/ silver firs	Mixed forest with a prevalence of chestnuts	Mixed forest with a prevalence of Turkey and other oaks	European beech	Mixed forest with a prevalence of conifers	Mixed forest with a prevalence of other broadleaf trees
BAU scenario	coppices	not present	FH; CC; R = 8; S = 30	FH; CC; R = 18; S = 60	FH; CS; R = 20; S = 60	not present	FH; CC; R = 20; S = 60
	high forests	T; Y = 20; Cov = 80% FH; SC; R = 70; P = 65%	not present	T; Y = 30; Cov = 80% FH; ShC; R = 80; P = 65%	T; Y = 20; Cov = 80% FH; ShC; R = 90; P = 65%	T; Y = 20; Cov = 80% FH; SC; R = 70; P = 65%	T; Y = 30; Cov = 80% FH; ShC; R = 60; P = 65%
PES scenario	coppices	not present	FH; CC; R = 16; S = 30	FH; CC; R = 36; S = 60	FH; CS; R = 40; S = 60	not present	FH; CC; R = 40; S = 60
	high forests	T; Y = 40; Cov = 80% FH; SC; R = 120; P = 65%	not present	T; Y = 30; Cov = 80% FH; ShC; R = 140; P = 65%	T; Y = 40; Cov=80% FH; ShC; R = 160; P = 65%	T; Y = 40; Cov = 80% FH; SC; R = 120; P = 65%	T; Y = 30; Cov = 80% FH; ShC; R = 100; P = 65%

T – thinning; Y – age of thinning (years); Cov – cover after thinning (%); FH – final harvesting; SC – strip cutting; ShC – shelterwood cutting; CC – clear cutting with standards release; CS – coppice selection system; R – rotation age (years); S – standards after final harvesting (n); P – prescribed final yield (%)

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The PES scenario is based on the assumption of an augmented rotation period, an increase in thinning years, and an increased number of released standards. All the above practices lead to higher forest cover and a decrease in soil erosion, as described in the following section.

Maximum willingness to pay (*maxWTP*). The first step for *maxWTP* quantification is the evaluation of avoided erosion by means of differences in erosion between the BAU and PES scenarios. The model applies a modified version of the revised universal soil loss equation (RUSLE2015) (Panagos et al. 2015a). RUSLE2015 estimates soil loss (*E*, expressed in $t \cdot ha^{-1} \cdot y^{-1}$) by applying five input factors: rainfall erosivity (*R*), soil erodibility (*K*), cover management (*C*), topography factor (*LS*), and support practices (*P*) (Equation 1).

$$E = R \times K \times LS \times C \times P \tag{1}$$

Rainfall erosivity is influenced by the geography of the area. The soil erodibility factor represents the average loss of soil expressed in tons per hectare and measures the susceptibility of the soil particles to be transported in the case of rain and surface flow. The texture of the soil is the main factor influencing *K*, but the structure, organic matter, and permeability also contribute. The cover management factor is used to determine the efficiency of soil management practices in relation to soil loss. The topography factor – a combination of slope length and slope steepness – represents the portion of soil lost in standard conditions of altitude difference (22.13 m in altitude with a slope of 9%): the longer and the steeper is the slope, the greater the risk of erosion (Stone, Hilborn 2001). The support practices factor is a coefficient linked to soil management and reflects the effects of management practices that affect the share of water that reaches the soil, which in turn influences erosion. The factor *P* represents the portion of soil lost in the case of cultivation, which increases along the lines of the maximum slope. Geodata related to *R*, *K*, *LS*, *C*, and *P* are freely available from the European Soil Data Centre (ESDAC) repository (https://esdac.jrc.ec.europa.eu/resource-type/soil-threats-data).

In our work, the *R*, *K*, and *LS* factors are held as constants because of their long-term variation and low influence on the forest rotation period. The *P* factor is not included because of its small impact on forested areas (Panagos et al. 2015a).

Erosion in the BAU and PES scenarios is based on potential variation in the *C* factor. *C* was quantified using Equation (2) (Panagos et al. 2015b):

$$\begin{aligned} C_s = {} & \min(C_{land\_use}) + \\ & + \left[\max(C_{land\_use}) - \min(C_{land\_use})\right] \times \\ & \times \left(1 - F_{cover,s}\right) \end{aligned} \tag{2}$$

where:

$\min(C_{land\_use})$ – minimum *C* factor for forests (Panagos et al. 2015b);

$\max(C_{land\_use})$ – maximum *C* factor for forests (Panagos et al. 2015b);

$F_{cover,s}$ – average forest cover in the rotation period (%) in the *s*-th scenario.

Forest cover is not reported in available digital maps; thus, its value was elicited from a focus group with the involvement of local experts of the Municipality Union and researchers. For each forest typology, the cover percentage is calculated for both the BAU and PES scenarios considering woodland age and silvicultural interventions as reported in Table 1 (e.g., thinning in year *m*, final harvesting in year *t*; Figure 3). Theoretically, forest cover in the PES scenario can be incremented with the adaptation of silvicultural interventions with respect to the BAU scenario (e.g., in PES_a in Figure 3), augmented rotation period (e.g., PES_b in Figure 3), or increased year of thinning, as suggested in Table 1.

The average cover is computed as the ratio between the total cover for each forest typology and the rotation period (Equation 3).

$$F_{cover,s} = \frac{\int_0^{t_s} f(s)}{t_s} \tag{3}$$

where:

s – scenario (BAU, PES_a, or PES_b);

t – rotation period;

f – trend of forest cover in the scenario, as graphically expressed in Figure 3.

Based on this approach, the *C* factor reaches its minimum value when F_{cover} is equal to 1 (the soil is fully covered by trees and crowns) and vice versa for F_{cover} equal to 0. Avoided erosion (*AE*) in the *i*-th pixel will be:

$$AE_i = R_i \times K_i \times LS_i \times \left(C_{i,BAU} - C_{i,PES}\right) \tag{4}$$

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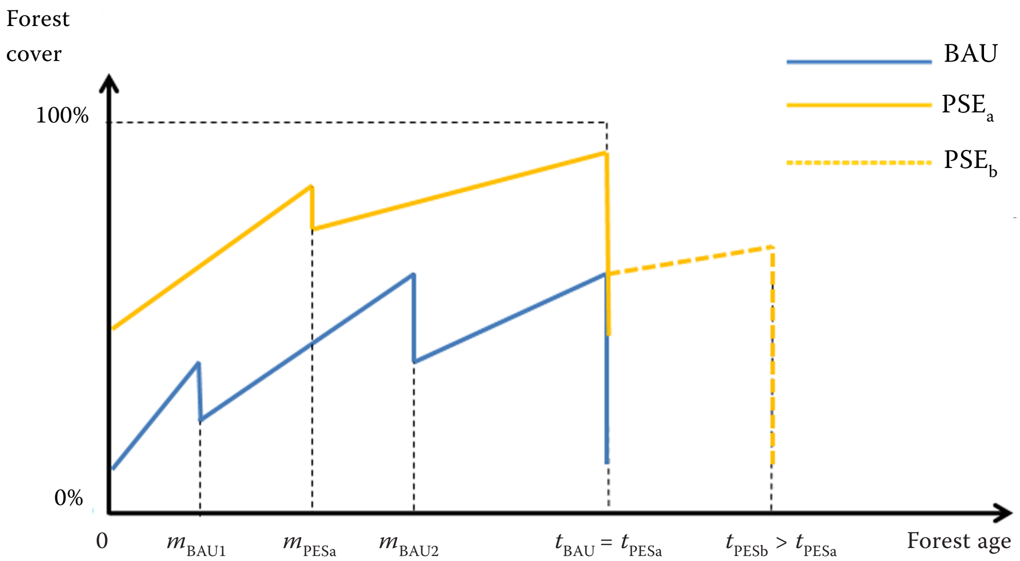

Figure 3. Schematisation of the hypothesised trend for forest cover in BAU and PES scenarios

BAU – business-as-usual

The estimate obtained by means of the RUSLE2015 model provides indications of the quantity of material that is handled overall over the entire surface of the basin. The assessment of sediment collected by the dam also requires the determination of the sediment delivery ratio (*SDR*). The *SDR* considers the deposit phenomena that occur along the hillside. This coefficient, defined as the ratio between the suspension transport detected in measurement stations and the average erosion value assessed through the model in the same period and in the same area, is a dimensionless number and can be formally expressed as:

$$SDR = \frac{Y}{E} \times 100 \tag{5}$$

where:
SDR – sediment delivery ratio;
Y – the suspension transport detected;
E – calculated annual loss of soil.

The value of the *SDR* estimates the percentage of eroded soil that effectively leaves the basin. This estimate indicates that a certain amount of the material eroded on the slopes does not arrive at the closure of the basin (dam), but is also deposited as sediments on the hillside following variations in steepness, surface roughness, presence of vegetation, and variations in soil permeability.

SDR is quantified using the simplified formula of De Rosa et al. (2016):

$$SDR = 0.4724 \times A^{-0.125} \tag{6}$$

where:
A – total area of the basin expressed in square kilometres (De Rosa et al. 2016).

The value *A* is computed by means of GIS operations based on *r.watershed* and *r.water.outlet* modules of GRASS GIS; *r.watershed* allows for the identification of drainage and stream maps of the area; and *r.water.outlet* uses drainage and stream maps to define the basin boundary for a particular artificial dam (GRASS Development Team 2019).

The analysis of *maxWTP* relates to the amount of avoided soil loss, *SDR*, and the cost of soil removal from the dam (Equation (7)). Either actual values, if available, or bibliographic data can be applied to the model. In this study, a cost equal to 29.29 EUR·t^{-1} for sediment removal was used (Palmieri et al. 2014).

$$WTP_i = AE_i \times SDR \times \sigma \tag{7}$$

where:
σ – cost for sediment removal (EUR·t^{-1}).

Minimum willingness to accept (*minWTA*). The methodology to compute the minimum willingness to accept (*minWTA*) follows a recent work focused on payments for the prevention of shallow landslides (Grilli et al. 2020). The approach is based on the concept that forest owners must be compensated for the decrease in the value of the forest because it is assumed that optimised interventions

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for the provision of the ecosystem services (here the reduction of erosion) involve the lengthening of the rotation period or the modification of silvicultural intervention. The consequence is a decrease in the forest financial value due to either longer immobilisation of capital or smaller volume of harvested wood.

The value of the forest is annualised for comparison with the *maxWTP*. Annualised forest expected value (*aFEV*) includes bare land and stands. The *aFEV* is calculated according to the formula for future income (Merlo 1991) for the BAU and PES scenarios. We assumed an intermediate year of the rotation period due to the absence of an updated database relating to the age of forest stands; *aFEV* is calculated as follows in Equation 8 (see below).

$$aFEV_s = \frac{SV_s + \sum_m T_s \times q^{t_s - m_s} + (v_s - e_s) \times \left(\frac{q^{t_s - n_s} - 1}{r} \right) + LEV_s}{q^{t_s - n_s}} \times r \quad (8)$$

where:
aFEV – annualized forest expected value;
s – scenario (BAU or PES);
SV – stumpage value of final harvesting;
T – the stumpage value of thinning;
t – rotation age;
m – year of thinning;
n – age of stand (hypothesised equal to t/2 in our case study);
v – annual revenues;
e – annual costs;
$q = 1 + r$ with *r* interest rate;
LEV – value of bare land according to the classical Faustmann formula (Faustman 1894):

$$LEV_s = \frac{SV_s + \sum_m T_s \times q^{t_s - m_s}}{q^{t_s} - 1} + \frac{v_s - e_s}{r} \quad (9)$$

The stumpage values *SV* and *T* are quantified on a spatial basis using the *r.green.biomassfor* model, a GIS tool allowing for the financial assessment of a forest and an amount for obtainable wooded resources (Sacchelli et al. 2013). The model has been applied and validated in different case studies from a national to an international level (Grilli et al. 2017; Sacchelli et al. 2018; Sacchelli et al. 2021). The *r.green.biomassfor* DSS enables the quantification of woody assortments and their financial value calculated in a multistep process. The potential availability of assortments is quantified based on the average annual increment of wood, partitioning of increment in assortments, rotation age, and prescribed yield in thinning and final harvesting. The technical availability of assortments is quantified by the introduction of limits to the extraction process: constraints are based on mechanisation typology and level (e.g., low/medium/high power cable crane extraction, extraction with a forwarder/skidder/tractor etc.), and geomorphological and logistic restrictions (slope, roughness, and distance from the forest and main roads). Financial evaluation considers the revenueand costs of the entire production process. Revenue is computed based on the unitary prices and total amount of each assortment; costs are derived from the product of productivity in each step of the forest process and the unitary cost of machinery and workers. Additional costs are also included in the modellisation (direction costs, administrative costs, and interests; Sacchelli et al. 2013). Stumpage value is defined as the difference between total revenue and total costs. As the model is applied in the same area of the work as Grilli et al. (2020), the starting data to quantify *aFEV* in the BAU and PES scenarios are the same. Modifications to the model regard rotation age as well as prescribed yield in thinning and final harvesting for the PES scenario, as reported in Table 1. Therefore, the main difference between the quantification of *minWTA* in the work of Grilli et al. (2020) and the present study is depicted in the scenario analysis: the PES scenario is optimised for protection from landslide in Grilli et al. (2020) and for protection from erosion in the present research. For additional details about the model, as well as the geodata and variables applied in the study area see Grilli et al. (2020).

The *minWTA* will be given as:

$$minWTA = aFEV_{\mathrm{BAU}} - aFEV_{\mathrm{PES}} \quad (10)$$

Localisation of potential PES activation and incentives. Suitable areas (γ) for PES activation are defined as follows:

$$\gamma \in \left[\left(maxWTP_i - \lambda_i \right) - \left(minWTA_i + \varphi_i \right) \right] > 0 \quad (11)$$

where:
λ – transaction costs for the buyer (here set at 2.5%; Phan et al. 2017);

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$$\omega = \left[\text{if} \left(maxWTP_i - \lambda_i \right) - \left(minWTA_i + \varphi_i \right) \geq 0 \text{ then } 0 \text{ else } \left| \left(maxWTP_i - \lambda_i \right) - \left(minWTA_i + \varphi_i \right) \right| \right] \tag{12}$$

φ – transaction costs for the seller (here set at 2.5%; Phan et al. 2017).

Equation (11) indicates that the PES scheme activation is feasible if payment from dam managers at least covers the losses of the forest owners.

The results of Equation (11) may highlight a negative gap between *maxWTP* and *minWTA*. In the latter case, it is necessary to introduce mixed payment mechanisms in which, in addition to a market between managers of the artificial lake and forest owners, other public or private entities could finance part of the payment. The compensation (ω) is quantified in Equation (12).

RESULTS

Figure 4 shows the difference between *maxWTP* and *minWTA* net of transaction costs. The difference ranges from 6 EUR·ha^{-1}·y^{-1} to –107 EUR·ha^{-1}·y^{-1}.

The case study shows a very low suitability for a PES mechanism if only based on a demand and offer agreement. A positive difference (γ) is highlighted for 4.2 ha on a total of 478 ha of potential forest area in the basin (0.9%). In this first application of DSS, the valuation of the potential PES market is concentrated in forests (478 ha) where $aFEV_{\text{BAU}}$ is positive. In other words, to maintain the additionality concept, we focus on woodland that currently is, or should be, actively managed for the production of traditional assortments. Managers of the artificial lake on the Moscia River are willing to pay an amount of 439 EUR·y^{-1} (sum of *maxWTP*), while the sum of *minWTA* is 23 795 EUR·y^{-1} (average loss of 49.8 EUR·ha^{-1}·y^{-1}). The total *aFEV* moves from 110 902 EUR·y^{-1} in the BAU scenario to 87 687 EUR·y^{-1} in the PES scenario, with an average decrease of 21% in financial value. These results confirm that the implementation of the PES mechanism in the study area can be mainly promoted by means of compensation from other sources. Average incentives (ω) to activate the market amount to 49.3 EUR·ha^{-1}·y^{-1}.

The focus on average *minWTA* per forest category reveals how the stands most impacted seem to be the woodland of European beech, followed by areas covered with Norway spruce/silver firs, and oaks (Table 2). A lower value is depicted for mixed forests with prevalence of Turkey oak. This forest category (together with stands of Norway spruce and silver fir) also reported the smallest average *maxWTP*. The 'other oaks' category has higher average *maxWTP*.

The increase in forest cover from BAU to PES is evident in European beech stands (28.5%); other forests have a variation of the cover ranging from 8.5% of mixed forest with prevalence of Turkey oak to 18.6% of mixed forest with prevalence of chestnut.

WTP can be influenced not only by forest characteristics, but also by the morphological condition of stands. Table 3 analyses the variation in *maxWTP* based on the slope and distance from the Moscia River.

Both parameters seem to impact *maxWTP*; in particular, a direct and an indirect trend appears for slope and distance from the river, respectively. Forest management influences *maxWTP* and *minWTA*: *maxWTP* results of 0.30 and 1.00 EUR·ha^{-1}·y^{-1} in coppices and high forest, respectively. *MinWTA*

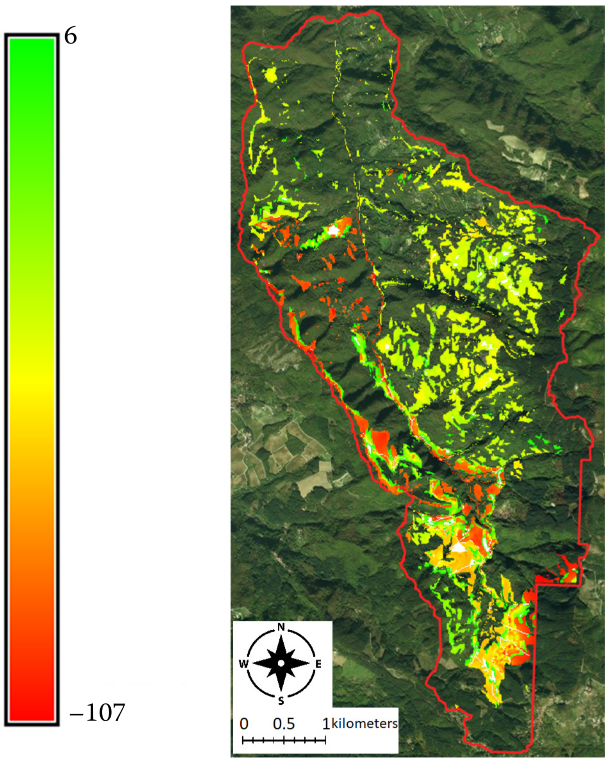

Figure 4. Difference between *maxWTP* and *minWTA* (EUR·ha^{-1}·y^{-1})

maxWTP – maximum willingness to pay; *minWTA* – minimum willingness to accept

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Table 2. *MinWTA*, *maxWTP*, and cover variation per forest category

Forest category	Average *minWTA*	Average *maxWTP*	Forest cover BAU	Forest cover PES	ΔForest cover (PES–BAU)
	(EUR·ha^{-1}·y^{-1})			(%)	
Norway spruce/silver fir	71.44	0.32	72.5	83.1	10.6
European beech	81.97	0.98	39.4	67.9	28.5
Turkey oak	56.59	0.92	61.1	79.4	18.2
Other oaks	59.00	1.19	61.2	79.3	18.1
Mixed forest with a prevalence of Turkey oak	31.78	0.32	78.1	86.6	8.5
Mixed forest with a prevalence of chestnut	51.83	0.46	64.5	83.1	18.6
Mixed forest with a prevalence of other broadleaf trees	43.32	0.68	63.0	81.4	18.4
Mixed forest with a prevalence of conifers	46.68	0.66	76.1	85.5	9.4

minWTA – minimum willingness to accept; *maxWTP* – maximum willingness to pay; BAU – business-as-usual; PES – payments for forest ecosystem services

Table 3. Variation of the average *maxWTP* in relation to morphological conditions

Class of slope (%)	Average *maxWTP* (EUR·ha^{-1}·y^{-1})	Class of distance from stream (m)	Average *maxWTP* (EUR·ha^{-1}·y^{-1})
0–10	0.58	0–100	1.37
10.1–20	0.77	100.1–300	1.56
20.1–30	0.98	300.1–500	1.07
30.1–40	0.93	500.1–700	0.86
40.1–50	1.03	700.1–1 000	0.92
> 50	1.20	> 1 000	0.68

varies from coppices (70.00 EUR·ha^{-1}·y^{-1}) to high forests (48.3 EUR·ha^{-1}·y^{-1}).

DISCUSSION

The dimension of the examined basin (1 812 ha of total forested area, 478 ha of forest with positive $aFEV_{BAU}$) can be one of the causes for the low suitability of PES activation without incentives. In fact, a PES mechanism similar to the proposed one was activated in 2000 for the Ridracoli Dam (central Italian Apennines), 21 km away from the Moscia reservoir (Gagliоppa et al. 2017). However, the Ridracoli basin is approximately 5 200 ha of forested area and represents the first positive national case of PES to avoid erosion in forested areas. Although the amount of erosion depends on the factors described in the RUSLE2015 equation, PES feasibility is strictly correlated with the total amount of sediment delivered to the artificial lake. The involved basin surface could therefore be considered of primary importance to define the potentiality of PES activation, even if the literature does not currently report any potential trends due to basin dimensions. The influence of the basin area is intuitive because buyers' *maxWTP* increases with larger benefits derived from avoided erosion.

The current forest management practices developed in the study area are another reason for the small number of potential stands involved in the PES market: the majority of woodlands present a high average level of cover throughout the rotation period (generally greater than 60%, Table 2). Forest owners rarely observe the minimum rotation period prescribed by regional regulations. The age of trees at final harvest, the number of released standards, or the percentage of canopy cover are, in fact, usually higher than those defined by regional laws. This is mainly due to the progressive shift from an intensive use of the forests towards a protective policy typical in the Italian Apennines (Coppini, Hermanin 2007). In the PES scenario, the cover increased by approximately 16% on average, highlighting a potentially low level of avoided erosion.

The decrease in the financial value of the forest (on average 21%) is mainly concentrated in European beech, fir, and oak stands. One reason could be the significant increase in cover from the BAU

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to PES scenario in forests of European beech (relative to other forest categories), causing a valuable reduction in final harvesting. An additional motivation is the high economic value of the other mentioned formations. Norway spruce, silver fir, and oaks – Turkey oaks in particular – show relevant *aFEV* in the BAU scenario; consequently, even a small reduction in harvesting in the PES scenario can lead to major financial losses with respect to other forest categories. A good value of *aFEV* is mainly related to the stumpage value given by the difference in revenue and costs of production processes. Unitary prices of wood assortments (linked to the revenue) are significant for Norway spruce and silver fir, trees that are mainly delivered as timber to sawmills. In addition, Turkey oak presents the relevant value for its assortments, in general high-quality firewood (Sacchelli et al. 2018). Like in other areas of the Italian Apennines (Marchi et al. 2018), in the Municipality Union territory conifer stands come from artificial reforestation; accessible areas were often chosen for reforestation, resulting in a reduction in production costs related to logistic and geomorphologic characteristics.

MinWTA and – as a consequence of the case study – the level of compensation are higher for coppices with respect to high forests. The increase in forest cover in coppices causes a significant difference from $aFEV_{BAU}$ to $aFEV_{PES}$.

MaxWTP does not seem to be primarily influenced by species, but trends are evident for both slope and distance from the river, as recently confirmed by Maltsev and Yermolaev (2020).

The level of compensation (49.3 EUR·ha^{-1}·y^{-1} on average) and *minWTA* are in line with the few examples available in the literature and empirical case studies. Sone et al. (2019) reported a value of 21.30 USD·ha^{-1}·y^{-1} to allow for soil conservation greater than 70%. Alarcon et al. (2017) stressed a range from 116.53 USD·ha^{-1}·y^{-1} to 185.56 USD·ha^{-1}·y^{-1} for forest conservation or restoration in a PES test in Brazil. In the case study of the Ridracoli Dam, forest owners were compensated 100 EUR·ha^{-1}·y^{-1} to carry out the best forest management practices.

Future research should explore potential sources of incentives and/or compensation and their relationship with the results of the model. For example, funding from specific measures of the Rural Development Plan (RDP) of Tuscany can be investigated. The current scheduling of the RDP provides – among measures that can be applied – measure 8.1 'Funding for reforestation', measures 8.3 and 8.4 for both the prevention and recovery of damage to forests due to fires and other extreme events, and measure 8.5 'Funding to increase resilience and environmental value of forest ecosystems'. Further investigations should focus on downscaling the RUSLE2015 variables to support a local analysis. Also, a specific analysis of the *SDR* factor is recommended, in which a unique value for the whole basin is applied, while the micro-variability impacting sedimentation of eroded soil can be considered by means of specific modellisation. The current version of the model excludes the role of land cover types other than forests. Agricultural areas can be evaluated in future applications thanks to the flexibility and fitting of the RUSLE2015 equation. However, the validity of the approach is still maintained because of the innovative focus on the forest PES scheme, even if the contribution of the forest area to erosion can be limited (in relative terms) with respect to other rural areas.

As mentioned in the description of the general framework, the model represents a tool to support decision-makers in the first phase of the PES market. Additional phases of PES implementation need to be further investigated to complete the process. Questions concerning stakeholders' involvement, effectiveness of silvicultural interventions, trade-offs with other ecosystem services, property rights, and risk management are crucial for establishing PES and must be evaluated with proper techniques and methods (Hausknost et al. 2017).

In future studies, a multiscale approach can be considered by applying the model in large areas (e.g., a regional or national analysis) or larger basins, taking into account the required level of generalisation and potential loss of detail and lack of spatial information.

In the proposed DSS, the additionality concept is based on the hypothesis that forest owners are compensated for reduced *aFEV* from the BAU to the PES scenario due to a decrease in harvested timber or a longer rotation period. However, future investigations could consider forests that are not currently managed because the harvesting costs are higher than the expected profits (i.e., negative stumpage value).

The conditionality principle is grounded in input-based mechanisms (Smith et al. 2013); payments are centred on the implementation of management practices rather than the effective quantification of erosion reduction. Further field assessments of effectively avoided soil delivery can be quantified us-

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ing an output-based approach to validate the results of the model. In other words, future evaluations can verify the model in real natural conditions for real forest sites using original measurements.

CONCLUSION

In this paper, a GIS-based DSS to analyse the preliminary steps of a PES scheme was presented and discussed. The DSS is focused on PES for the prevention of erosion in forests. The procedure was applied to an Italian case study (Municipality Union of Sieve and Arno valleys, province of Florence). The empirical application highlights the low suitability of the study area for PES based on an exclusive agreement between buyers and sellers of environmental services. However, this procedure offers an interesting tool for future PES analyses. The proposed spatial DSS has the main advantage of being created in an open-source platform with respect to available tools, allowing for flexibility and replicability. Flexibility facilitates the scenario analysis, for example, the iteration of DSS with different input coefficients (price of raw material, unitary costs of the production process, cost of dam emptying, etc.) or geodata. Replicability allows for an investigation of PES markets in other areas. The geographical basis of the model improves the ability to present and discuss the results to decision-makers and local stakeholders, as well as the participative process for potential improvement of DSS. The preliminary area for testing the PES mechanism can be depicted through the model.

The application of DSS in the case study and comparison with existing literature demonstrates how PES suitability depends on forest characteristics, socio-economic variables, size of the basin, and geomorphological conditions. Either the price range between *maxWTP* and *minWTA* or funding in the case of compensation can be evaluated by considering the above variables.

The application of the model, considering the limits and potential improvements stressed in the discussion section, can avoid the risk of inflation of compensation (in the case of implausibly large areas subjected to PES) or the threatened provision of environmental benefit in the case of the smaller than necessary PES zone.

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