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ORIGINAL ARTICLE

Mapping erosion risk at the basin scale in a Mediterranean environment with opencast coal mines to target restoration actions

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Abstract River basin restoration and management is crucial for assuring the continued delivery of ecosystem services and for limiting potential hazards. Human activity, whether directly or indirectly, can induce erosion processes and drastically change the landscape and alter vital ecological functions. Mapping erosion risk before future restoration-management projects will help to reveal the priority areas and develop a hierarchy ordered according to need. For this purpose, we used the Revised Universal Soil Loss Equation (RUSLE) erosion model. We also applied a novel technique called GPVI (Genetic Programming Vegetation Index) in the Martín River basin in NE Spain (2,112 km^2), which has a large coalfield located in the southern part of the basin. Approximately two-thirds (69%) of the area of the Martín basin presents low and medium soil loss rates, and one-third (31%) of the area presents high (18%), very high (10%), and irreversible (3%) erosion rates. The southern part of the basin is the most degraded and is strongly influenced by the topography. This work allows us to locate areas prone to erosional degradation processes to help create a buffer around the river and locate "spots" in need of restoration. We also checked the error estimation of the methodology because our soil maps do not include rock and bare rock areas. The usefulness of applying RUSLE for predicting degraded areas and the consequent directing of soil conservation–restoration actions at the basin scale is demonstrated. We highly recommend a field survey of the selected areas to prove the goodness of the model estimations.

Keywords Water erosion · Rehabilitation · Semi-arid · Revised Universal Soil Loss Equation (RUSLE) · GIS · Spain

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Introduction

Restoring eroded lands is a major objective to give back value to large parts of the world where erosion is a major environmental problem (Pimentel et al. 1995).

However, defining areas for restoration in a vast territory requires establishing the magnitude of the problem and the benefits of the solutions at an adequate spatial scale (Boardman 2006).

Soil is often lost through erosion, a natural process that can be fostered by inappropriate land use and intense precipitation, among other factors (Garcia-Ruiz 2011). The European Union considers soil to be a non-renewable resource, and soil degradation has strong impacts on soil and water resources (Montanarella 2000). The loss of topsoil and changes in its properties will cause the decline in the ecological processes that rely on it. Soil erosion increases the impact on streams through high sediment delivery, which has been identified as a leading cause of river degradation (USEPA 2000). Consequently, soil erosion causes the loss of the services provided by ecosystems (Van Wilgen et al. 1996), and knowing the spatial distribution of erosion rates is a primary step for planning restoration at the watershed scale.

In Mediterranean areas, developing efficient tools for decision making regarding land use management is a major objective (Simoncini 2009) because of the multiple environmental problems arising from the intensive use of the land since long ago (Tabara and Ilhan 2008), particularly problems related to erosion (Boardman et al. 2003; Bazzoffi 2009). Opencast mining is one such activity, which contributes mostly to erosion (Wu and Wand 2007). Opencast mines are sources of high sediment yield to rivers if restoration is not properly carried out (Balamurugan 1991; Taylor and Owens 2009). Subsequently, human intervention in failed reclamation areas, especially opencast mines with highly eroded slopes connected with the river network is necessary to prevent water pollution and to slow irreversible erosion (Pimentel et al. 1995; Palmer et al. 2010).

Mapping ecological processes and restoring areas with high sediment delivery would help avoid irreversible degradation that removes nutrients and reduces fertility (DeFries and Eshleman 2004), thus limiting the sedimentation and eutrophication of nearby rivers, which would represent a potential hazard for the long-term sustainability of agriculture and ecosystem services at the basin scale (Krauze and Wagner 2007). For this reason, the number of projects on sediment-related river restoration at the River basin scale is increasing (Kondolf 1998; Ward and Tockner 2001; Pizzuto 2002; Pennisi 2004). Successful restoration projects on river basins require an understanding of sediment transport processes. This understanding is achieved by identifying the suspended sediment sources on the basis of sediment monitoring and modeling (Gao 2008).

Environmental planning faces uncertainty when decisions on resource allocation must be made, especially on large-scale projects. The aims of this study are to identify erosion-prone areas at the basin scale and to produce a map with a hierarchy of zones degraded by erosion to be useful to prioritize the spatial distribution of restoration actions. The results of this application to the opencast coal mining practices in the basin are shown here as a qualitative planning and demonstration tool for soil protection. The application of our methodology is discussed as a resource for managers to identify critical soil source areas and the distribution of restoration actions at the basin scale. This should be also a relevant tool for planning land use management by individual land owners and extractive companies.

Study area

The Martín River basin is a 2,112 km^2 basin located in the south-central part of the Ebro River basin (NE Spain) (Fig. 1). The elevation ranges between 143 and 1,620 m a.s.l. The basin shows two major "regions": the highlands in the south (mean elevation 1,100 m, 765 km^2) and the lowlands in the north (mean elevation 750 m, 1,347 km^2), where most agricultural lands are established. The differences between these two zones are also marked by the presence of two dams, Escuriza and Cueva Foradada (maximum water storage capacities 6 and 22 Hm^3, respectively), which intercept sediments from the entire upstream area and disturb the natural river flow regime, creating a completely human-activity-altered environment downstream. The climate is Mediterranean, with continental influence. The summers and winters are usually dry, and the annual average precipitation of 360 mm is heterogeneously distributed in both space and time. The water deficit ranges between 530 and 758 mm, extending the dry period from May until October. The mean annual temperature range is 13–16°C, with minimum and maximum average temperatures of 5 and 25°C, respectively. Dryness, which has increased in recent years (Moreno-de las Heras et al. 2009), is the main limitation for natural plant development in the region and for the development of agriculture, which is the major socioeconomic activity in the lowland part of the basin, covering 53% of this part of the basin. This land is mostly used for dry cereal farming, except in the narrow belts along the river's sides near the villages, where an old canal network is still in use to irrigate vegetable and fruit tree fields (Fig. 2). The meso-Mediterranean garrigue (*Quercus ilex*), accompanied by sabine (*Juniperus sabina*) in a few zones in the southern sector, is replaced northward by Kermes oak (*Quercus coccifera*), rosemary (*Rosmarinus officinalis*) formations, and steppe with small species (*Macrochloa tenacissima, Stipa tenacissima, Ligeum spartum, Tamarix africana, Juniperus phoenicea*). The only significant forests are located in the central part of the basin, and they consist mainly of Aleppo pine (*Pinus halepensis*). Riparian vegetation is extremely degraded because of the extensive cover of agricultural practices and regulated river flows.

Regosol is the most widespread soil type in the Martín River basin, covering 41% of the total area. This soil is composed of medium and fine-textured materials derived from a wide range of rocks, which are normally extensive in eroding lands (FAO-UNESCO 1988), particularly in arid

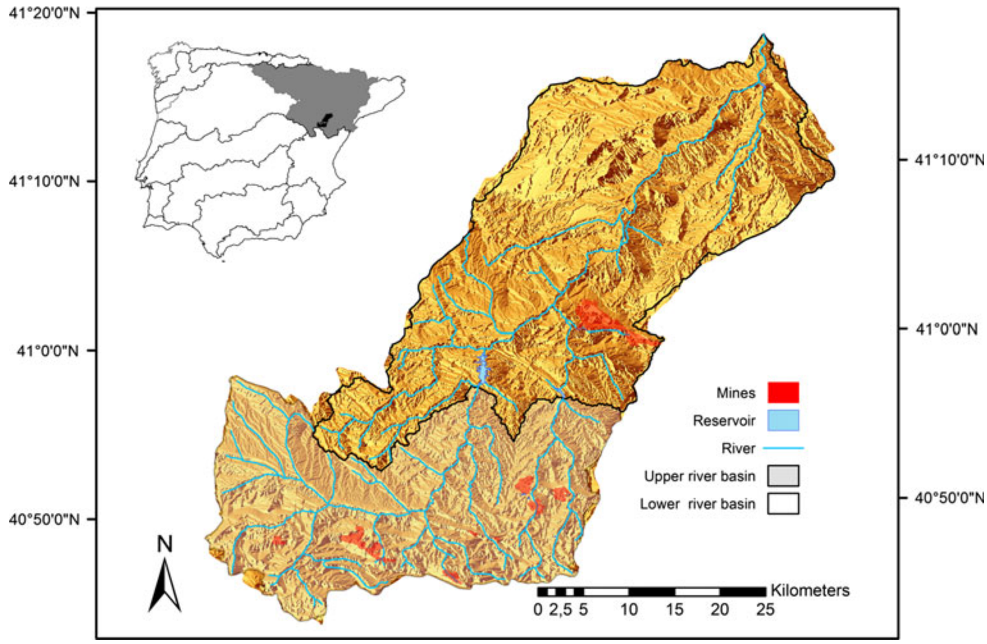

Fig. 1 Maps of the Martín River basin showing its hydrological network, and the upper (South) part and the lower (North) part of the basin

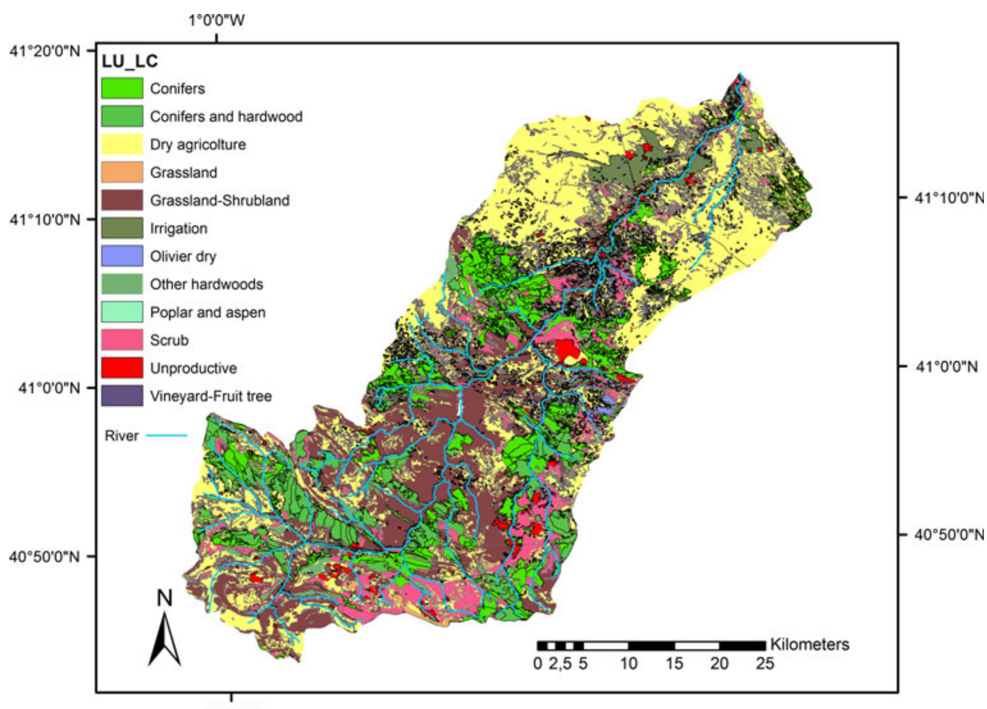

Fig. 2 Map of the Martín River basin showing the different land cover units

and semi-arid areas and in mountain regions (Sánchez-Andrés et al. 2010). Rendzina-lithosol and cambisol, which are shallow soils with medium and fine-textured materials, cover 11.7 and 12.6% of the Martín basin, respectively. Calcic yermosol, defined as a surface horizon that usually consists of surface accumulations of rock fragments (“desert pavement”) embedded in a loamy vesicular crust and covered by a thin eolian sand or loess layer, extends over 8% of the study area. These qualities make these soils prone to erosion if combined with land cover-management misuse and steep slopes.

A large coalfield is located in the southern part of the basin (Fig. 1). Mining was the main socioeconomic activity for people living in this region from 1960 to 1990. After a period of great prosperity of opencast mining during the 1980s with 17 active opencast mines (27 km^2), the activity has strongly declined and only three mines are currently operating (Comín et al. 2009). The mining zones (Fig. 3)

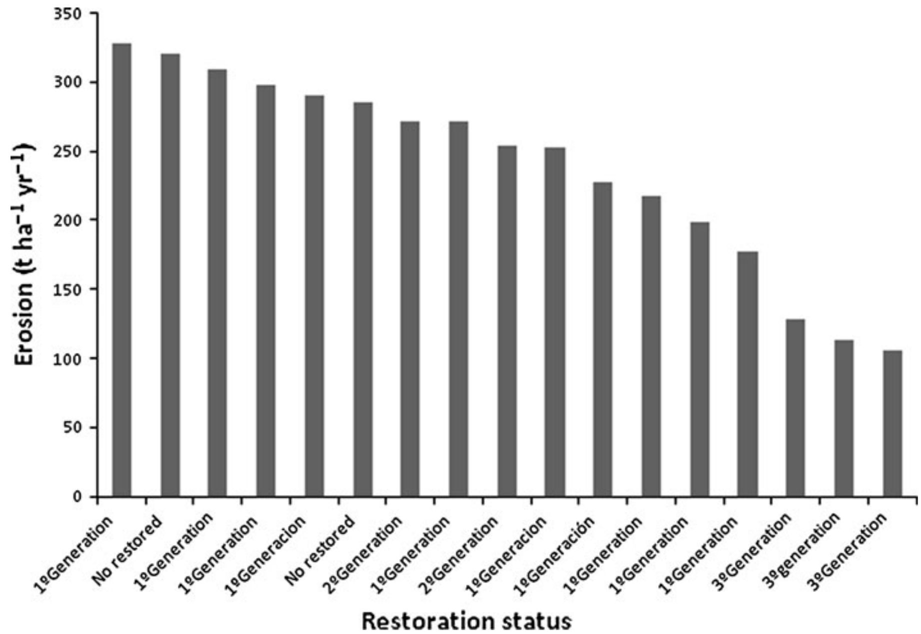

Fig. 3 Erosion rates by mine restoration status, the lower values correspond with the third generation restoration status

contribute to the emission of sediment according to their restoration status. These mining zones are classified as first generation, second generation, and third generation. The first generation mines were restored by depositing materials following the platform-slope-ditch model during the 1980s. These mines have large areas of steep slopes (>22°), with ditches formed from rill erosion connected to the river network and an absence of pits. The second generation group of mines was restored with the same model but with slopes of 15° and deep pit zones that accumulate runoff discharges. Moreover, these mines have received extensive application of soil and plant material to restore the plant community. The third generation mines were subject to a topographic restoration model that tries to simulate natural landforms not connected with the natural drainage. In addition to the three groups described above, a few mines abandoned after exploitation remain as non-restored mine zones in the region.

Materials and methods

Erosion rates have been estimated at the regional scale using the RUSLE model (Fu et al. 2005; Onori et al. 2006; Pizzuto 2002; Pennisi 2004). European environmental researchers (Panagos et al. 2011) have recently mapped a soil erodibility dataset at the European scale. The objective was to overcome problems of limited data availability for the application of the USLE (Universal Soil Loss Equation) model and to present a high-quality resource for modelers who aim to estimate soil erosion at the local/regional, national, or European scale. Following this direction, the location of eroded areas and the estimation of the average annual soil loss from rill and sheet erosion in the Martín basin (Ebro basin, north-east Spain) were determined by using the RUSLE (Renard et al. 1997) and an updated version of USLE (Wischmeier and Smith 1978), coupled with GIS (geographic information system).

Many authors have used GIS/RUSLE models to estimate sheet wash erosion and non-point source material discharges in watersheds (Fu et al. 2005; Lim et al. 2005; Smith et al. 2007) and for environmental assessment (Boellstorff and Benito 2005; Erdogan et al. 2007; Ozcan et al. 2008). An increasing number of studies on restoration ecology are using this model to identify potential restoration areas (Güneralp et al. 2003; Vellidis et al. 2003) and to design reclamation plans for degraded areas such as opencast mines (Toy et al. 1999; Martín Moreno et al. 2008; Moreno-de las Heras et al. 2009).

Despite some uncertainties regarding RUSLE, such as the overestimation of soil loss on plots with low erosion rates and the underestimation of soil loss on plots with high erosion rates (Nearing 1998; Risse et al. 1993), we decided to use this model because it requires data that are relatively common and inexpensive to be processed with GIS. One of the highlights is the formulation of results that can be used for comparative or complementary future studies (Millward and Mersey 1999; Wang et al. 2003; Beguería 2006).

The RUSLE model

We used GIS commercial software (using a spatial analysis tool) to examine spatial variations in erosion using elevation data at a 20-m grid scale within the study area. Digital land cover data are available as shape files at the Aragon Territorial Information Centre (ATIC 2006). The Universal Soil

Loss Equation (USLE) was used for this study because it is the most used empirical model that assesses long-term averages of sheet and rill erosion. This model is based on plot data collected in the USA (Wischmeier and Smith 1978). The USLE and its adapted version RUSLE (Renard et al. 1991) have been applied to various spatial scales and region sizes in different environments worldwide (Vrieling et al. 2008).

The USLE and RUSLE are statistically based water erosion models related to six erosion factors (for a detailed description of the factors and data collection methods, see the Electronic Supplementary Material at points 1 and 2, respectively):

$$A = R \times K \times L \times S \times C \times P$$

where A is the average soil loss from sheet and rill erosion, reported here in tons per hectare per year (t ha^{-1} year^{-1}).

R is the rainfall-runoff factor and represents the erosion energy in MJ mm ha^{-1} h^{-1} year^{-1} based on the methodology of Renard et al. (1991), and it represents the average annual summation (EI) values in a normal year's rainfall. The R-factor map for the area was implemented by Angulo-Martínez and Beguería (2009)

K is the soil erodibility factor, which represents both the susceptibility of soil to erosion and the rate of runoff, as measured under the standard unit plot condition expressed in t h MJ^{-1} mm^{-1} Renard et al. (1991).

Only R and K have units; those units, multiplied together, give erosion in units of mass per area and time. Each of the other terms scales the erosion relative to specified experimental conditions (>1 is faster than erosion under those experimental conditions, and <1 is slower). The remaining factors are non-dimensional scaling factors.

LS is the topographic factor describing the combined effect of slope length and steepness and is calculated with the approach of Moore and Wilson (1992).

From the standpoint of soil conservation planning, the C factor is the most essential factor because land use changes that characterize, reduce, or increase soil erosion are represented by this factor (Khana et al. 2007); however, the C factor is also the most costly (in time, at least) to estimate locally and then to extrapolate from the local measurements to the entire system of interest. Vegetation cover acts as a buffer layer between the atmospheric elements and the soil, absorbing most of the energy of raindrops and surface water to decrease the volume of rain reaching the soil surface (Khana et al. 2007). Soil constantly tilled or disturbed has a maximum potential for erosion ($C = 1$). Soil not recently disturbed has a nominal value of 0.45. Live or dead vegetation and rocks reduce C, reaching a maximum of 1.0 in constantly tilled soil. In places where total ground cover by live or dead material remains, C is taken as 0. In this study, several field samples were collected to determine the C factor following the approach of González Miguel A. and Bullock S.H. (unpublished). The next step was to extrapolate the punctual C factor values to the entire study area using the Genetic Programming methodology described by Puente et al. (2011) to obtain Vegetation Indices (VI's) designed exclusively for our area. For a detailed description of the methodology used to calculate the C factor, see the ESM.

Connectivity

Connectivity means the physical linkage of sediment through the channel system, which is the transfer of sediment from one zone or location to another and the potential for a specific particle to move through the channel system (Hooke 2003). In an attempt to evaluate the sediment connectivity in the Martín River basin, we created a buffer of 500 m at the sides of the main channel and its effluents. The area directly connected to the conveyor belt varies over different timeframes or under various flow conditions. We used this buffer size because it reflects a situation of moderate magnitude (Fryirs et al. 2007) over which sediments can readily reach the water without being intercepted by depositional areas. Then, we visually identified (color graduation) the higher eroded areas included in the buffer and marked them. In an effort to locate the areas and test the prediction of the model, we conducted a field and photographic survey in the degraded areas included within the buffer described by the model (Fig. 4).

Statistical analysis methodology

To assess the relationship between erosion and the available covariates, a generalized linear model (GLM, McCullagh and Nelder 1997) with a Gaussian response was selected. Among the various relevant factors that normally influence erosion, we chose cover, slope (LS), and rain (R) because they result in the best fit with erosion values. The response (erosion) and one of the covariates (LS) were log-transformed to reach normality. The regression models were fitted with the open-source R software (R Development Core Team 2010). For model selection, an all-subset regression with K-fold cross-validation was performed (Miller 2002), with Bayesian Information Criterion (BIC) as selection criteria. The one-standard deviation rule was applied for making the model selection more stable and for selecting the most parsimonious and adequate model (Hastie et al. 2009).

Results

Erosion at the basin scale

Based on the pixel resolution of the RUSLE model used, the mean erosion value for the Martín River basin was 13.8 t ha^{-1}

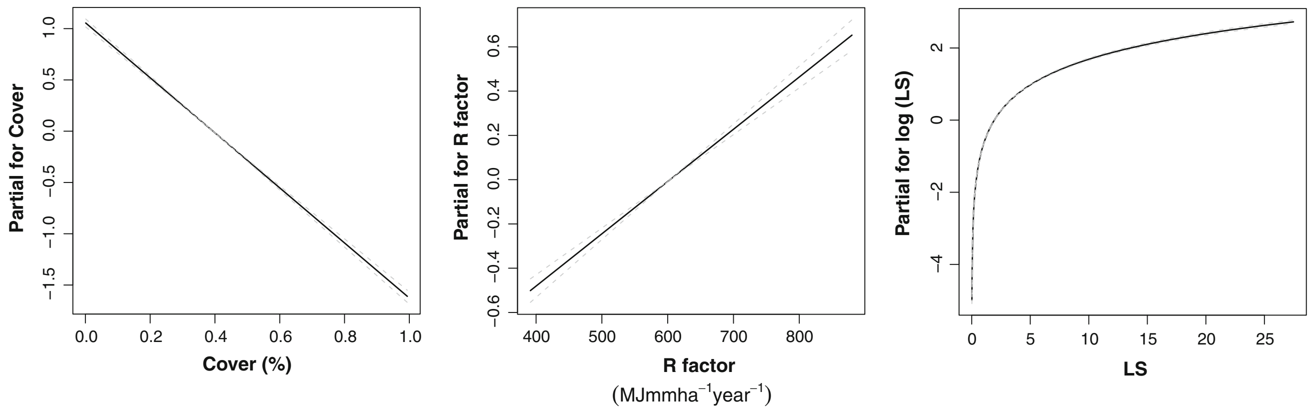

Fig. 4 Estimated effects of the covariates, with standard errors (SE). 95% confidence R factor is interval cover %, *LS* factor is used R factor

year^{-1}, which is just over the maximum tolerable soil erosion that can occur and still permit crop productivity to be sustained economically (2.2–11.2 t ha^{-1} year^{-1}) according to the RUSLE model of soils in the United States.

The spatial distribution of potential soil loss rates predicted by RUSLE and the watershed area–related erosion rates are shown in Figs. 5 and 6, respectively. Two-thirds (69%) of the area of the Martín basin has low and medium soil loss rates (less than 20 t ha^{-1} year^{-1}), and one-third (31%) of the area, mostly located in the central and southern parts of the basin, has high (18% of the watershed area with 20–40 t ha^{-1} year^{-1}) and very high (over 40 t ha^{-1} year^{-1} in 13% of the area) erosion rates. A detailed description of the data estimated for each factor in the RUSLE equation is given in the ESM.

The soil loss is at a maximum in rendzina-lithosol, with and area-weighted average (w.a.) of 23.3 t ha^{-1} year^{-1}, and in regosol, with a loss of 15 t ha^{-1} year^{-1} (w.a). This soil distribution covers the greatest part of the steep slope areas in the Martín basin ($0 \leq LS \geq 49$, ESM).

Annual soil losses corresponding to the different land covers are shown in Table 1. Dry farming, which occupies 38.6% of the basin area, has a moderate value of potential soil loss of 10.1 t ha^{-1} year^{-1}. Grassland-"shrubland" formations occupy 24.9% of the basin area, with a mean soil loss of 20.2 t ha^{-1} year^{-1}. The mean estimates for conifers (12% of the basin) and the formations of conifer and hardwood (8%) are 12 and 12.2 t ha^{-1} year^{-1}, respectively. Scrub, irrigated agricultural, and unproductive land (mines, quarries, urban) cover 9.9, 2.8, and 1.5% of the basin area, respectively. Other cover (grassland, olive grove and vineyard, other hardwoods, poplar and aspen, vineyards, fruit trees) occupies 4.6% of the basin area.

The final model selected, according to percentage of explained deviation and Akaike (1974) Information Criteria (AIC) (Konishi and Kitagawa 2008), with values of 92% and 473.8, respectively, was log (Erosion) = log (*LS*) + *R* factor + Cover (Table 2). The log (*LS*) topographic factor explained 78% of the total explained deviance (Fig. 4) and contributed most of the variability of the values of predicted soil erosion. The percentages of plant cover explained only 21%.

For modeling purposes, the variable *C* factor was deleted because it was highly correlated with some covariates and its inclusion would cause colinearity. In the full model, all possible two-term interactions were added.

Results obtained from all-subset regression with $K = 10$ (Fig. 7) shown as best models:

$$\text{Model 1} : \log(\text{Erosion}) \sim \log(LS) + \text{Cover} + \text{Rain} + \text{Cover} : \log(LS)$$

$$\text{Model 2} : \log(\text{Erosion}) \sim \log(LS) + \text{Cover} + \text{Rain}$$

According to the one-standard deviation rule, Model 2 was the most parsimonious (Fig. 8), with best cross-validation score inside the interval $CV \pm s/\sqrt{K}$, being *s* the standard deviation of *CV* and *K* the validation samples ($CV = 0.04$, SD $= 0.014$, $K = 10$).

Erosion in the coal mines

In the Martín basin, 8 mines are in good ecological status, as they are restored and preserved, and 9 are in bad ecological status, as they are either non-restored (3) or restored and degraded (6) (Comín et al. 2009). Five mines are closed basins; they have a surface design simulating natural geomorphology. The RUSLE estimates of soil loss in the mines ranged between 1.4 and 328 t ha^{-1} year^{-1}. The lowest rates correspond with flat areas created for restoration that are used for dry farming purposes and with wetland areas created in the old exploitation pit, which receive all the drainage of the surrounding areas. Maximum values were registered in very steep ditches, on hill slopes

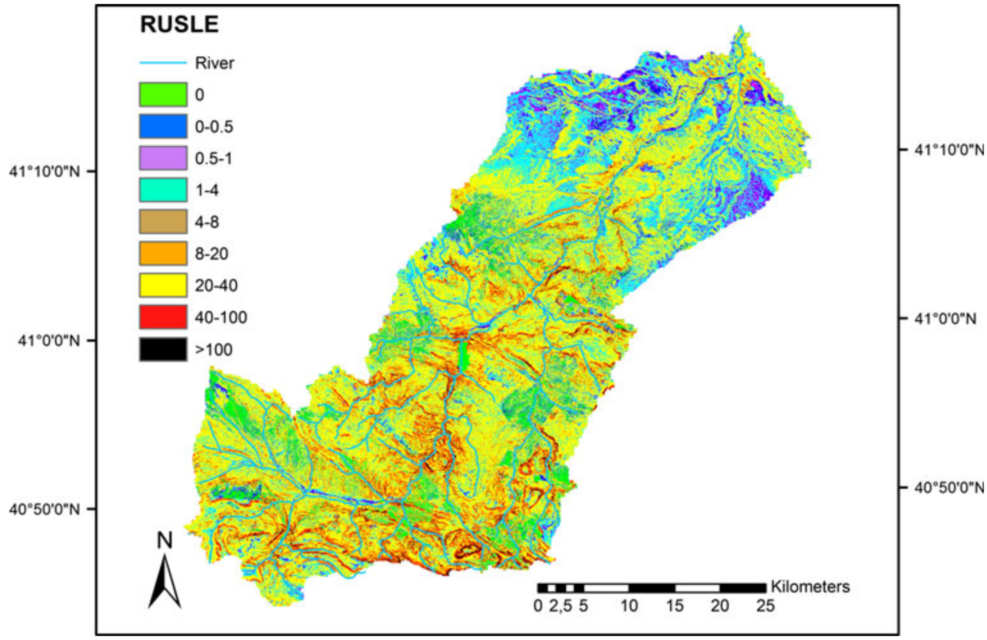

Fig. 5 Map of predicted soil erosion with the RUSLE model in the Martín River basin

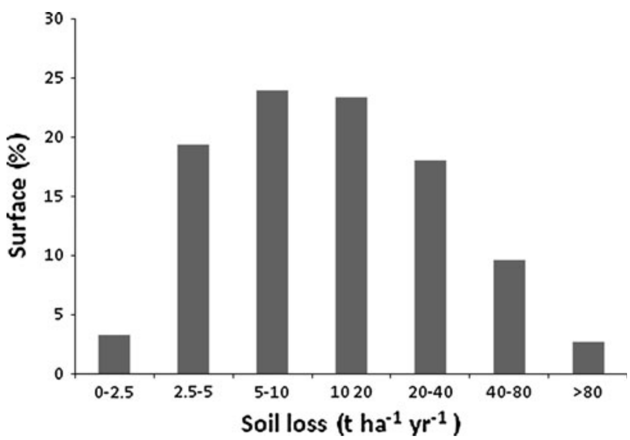

Fig. 6 Histogram of predicted soil erosion with the RUSLE model in the Martín River basin

Table 1 Statistic value of annual soil loss for the different land uses at the Martín River basin

Land use and land cover	Area %	Min	Max	Mean	Standard deviation
Dry farming	38.6	0	403	10	15
Grassland-shrubland	25	0	650	20	22
Grassland	1	0	290	25	29
Olivier dry	2	0	299	18	22
Vineyard-fruit tree	1	0	191	12	16
Unproductive	1.5	0	354	23	30
Irrigation	3	0	260	7	12
Scrub	10	0	603	24	28
Poplar and aspen	0.5	0	232	15	21
Other hardwoods	1	0	241	13	20
Conifers	8	0	482	12	20
Conifers and hardwood	8	0	370	12	19

and, overall, in abandoned, non-restored or deficiently restored mines, where it was not possible for plants to colonize because of steep zones and the use of overburden top soil material (Fig. 6). These areas are directly exposed to the eroding power of rainfall, generating high runoff.

Old, first generation mine restorations following sequences of platform bank with a slope angle of 22° have a range of 177–328 t ha^{-1} year^{-1} for maximum values and mean values ranging between 17 and 54 t ha^{-1} year^{-1}. Abandoned mines have a range of 116–320 t ha^{-1} year^{-1} as maximum values and 17–44 t ha^{-1} year^{-1} as mean values (Fig. 3). The second generation corresponds to mines where restoration was performed following the same practices as in the first generation with lowered bank slopes (15°). Intermediate erosion rates were estimated in these mine zones that still in exploitation-restoring process

Table 2 Estimated effects of the covariates, with standard errors (SE)

	Estimate	SE	*t* value	*p* value
(Intercept)	1.0141	0.0698	14.52	<2e-16
log(*LS*)	1.0252	0.0096	105.82	<2e-16
RAIN	0.0023	0.0001	18.87	<2e-16
COVER	−2.6843	0.0493	−54.36	<2e-16

Where used *LS* factor, *R* factor, and *C* factor

(17–25 t ha^{-1} year^{-1}) recording maximum soil loss of 184 t ha^{-1} year^{-1} with medium value of 174 t ha^{-1} year^{-1}. Micro-watersheds with gentle slopes and a drainage

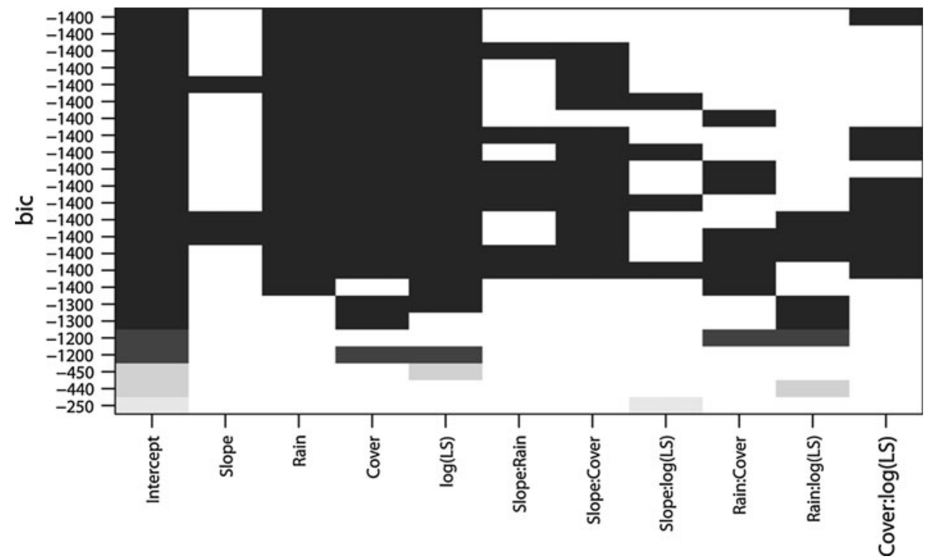

Fig. 7 BIC of the different models obtained by all-subset regression (":" indicating interaction between the covariates)

network were created for the mines restored under third generation concepts. In these areas, maximum soil loss estimates range between 106 and 98 t ha^{-1} year^{-1}, while the mean values range from 16 to 23 t ha^{-1} year^{-1}. It is clear that applying improved restoration techniques reduces soil loss in mine zones and that non-restored and deficiently restored mines are sites contributing the highest soil loss.

Discussion

This study demonstrates that the RUSLE model used with appropriate values for each factor is a powerful tool. Using the GP (Genetic Programming) methodology proposed by Puente et al. (2011) was proven as a reliable approach to generating specifically designed indices to estimate the *C* factor in contrast to traditional indices, such as those of the NDVI and SAVI family (Puente et al. 2011). We identified high-risk areas where soil conservation–restoration practices are needed. In the Martín River basin, major efforts should be dedicated to retaining soil in its southern high relief part and, especially, in the no restored mines to prevent the irreversible degradation of these zones. For this purpose, the results of this study are useful for identifying different zones of erosion risk at the watershed scale and at lower scales (e.g., subwatershed). The identification of high erosion risk areas is crucial for soil conservation and restoration planning to reduce or avoid irreversible degradation.

The average annual soil loss rate estimated using RUSLE and GIS for the Martín River basin was 13.8 t ha^{-1}year^{-1}. This estimation exceeds the estimated tolerable limits for soil formation of between 2 and 12 t ha^{-1} year^{-1} in Mediterranean environments (Rojo 1990). These results compare well with other studies in similar areas (Renschler et al. 1999; Van Rompaey 2003; Capolongo et al. 2008), confirming that the RUSLE-generated estimates of soil loss in this study appear to be reasonable.

The spatial variation of erosion in the Martín basin appears to be dominated by slope. The higher mean values of potential erosion were associated with zones located in the highlands with steep areas, including opencast coal mines that had the highest erosion rates even though large areas of many coal mine zones have been submitted to a restoration process. Although erosion varies greatly depending on the type of mine restoration, the steepest zones in the opencast mines match the highest erosion rates in the Martín River basin because of the creation of large (sometimes 1 or more km^2) hillslope areas inside and surrounding the mines by means of excavation. The scale of the mined areas (0.14–7.2 km^2) in comparison with the pixel size of the DEM (400 m^2) supports our assumption. Rill and gully networks in these reclaimed systems can markedly limit water availability and modify the spatial distribution of soil moisture at the slope scale by reducing the opportunities for downslope runoff re-infiltration and by concentrating the water flow along the channeling network (Biemelt et al. 2005; Moreno-de las Heras et al. 2010).

During the photographic field survey to evaluate the connectivity and eroded area prediction along the created buffer, we observed that some of the areas, appearing in the model analysis as high erosion areas, corresponded to bare rock and rock landslide phenomena; however, in the monitored areas, the model generally recognizes riverside degraded areas, as shown in Fig. 9. We also identified some mining areas that were degraded in the year of the

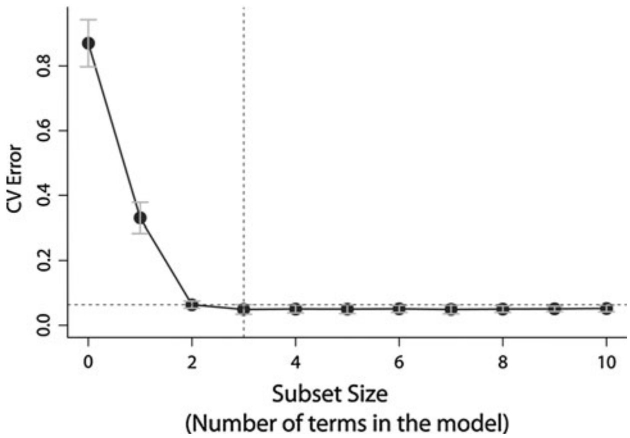

Fig. 8 Model selection with the one-standard deviation rule. *CV* cross-validation

creation of the DEM used here and that are now restored, because of the dynamics of the extractive areas.

Road embankments have not been considered with a special focus in this paper, but during the photographic survey, we realized the magnitude of their impact on the river system. These slopes are often directly connected by channels to the river network without having any way to intercept the sediment through depositional areas. Bochet et al. (2009) argued that the stabilization of road slopes is doomed to failure if the ecological knowledge of the topographic thresholds that limit vegetation establishment is not taken into account at the time of road building. This argument is supported by the existence of road cuts with a slope gradient exceeding 45°, where intense erosion occurs, generating very high soil loss and impacts that other studies have already highlighted in Mediterranean sites near the study area (Bochet and García-Fayos 2004). We also highlight that some areas are highly degraded but are disconnected from the fluvial channel or are intercepted by depositional areas. These areas are not a direct threat to water bodies because they are not significant contributing areas. Management plans for a watershed should take into account the need to evaluate the importance of these areas with respect to different uses and the potential benefits of restoring these areas, assessing the effective value for the production of ecosystem services and the mechanical and

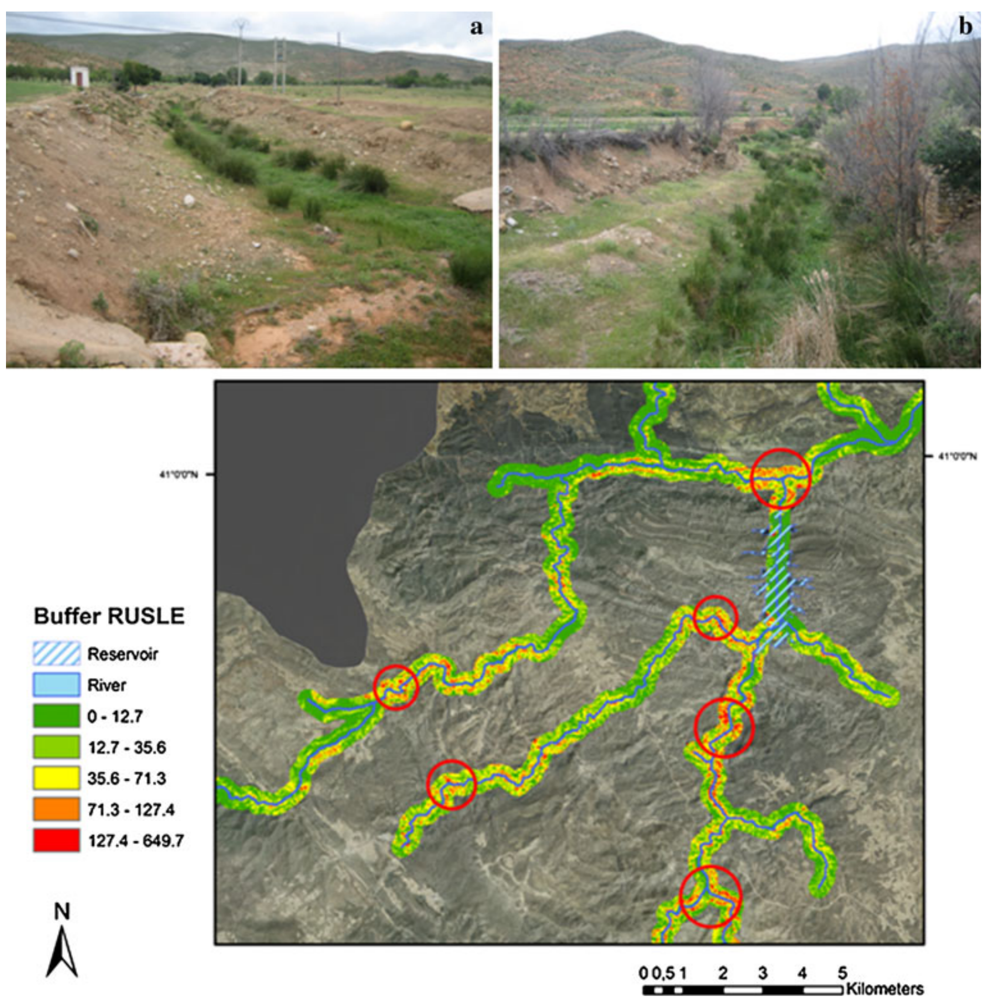

Fig. 9 Example of highly degraded riversides, founded using RUSLE-buffer map. In evidence on the *bottom left* (**a**), concrete ditch discharging straight in the river

monetary possibility of action (usually steep slopes) to restore it.

Plant colonization and reclaimed slopes

Moreno-de las Heras et al. (2011) suggest that natural plant colonization in Mediterranean-continental reclaimed environments requires vegetation cover of at least 30% and rill erosion rates below 17 t ha^{-1} year^{-1}. In our case, 59% of the river basin has less than 30% plant cover, and 41% of the basin area has a vegetation cover over 30%; this result is due to the very slow rate of plant recolonization and forest expansion, which occupies approximately 21% of the mountainous southern part of the basin.

Fifty-six percent of the mine areas are included in the acceptable soil loss range for plant colonization, but the rate is higher than 17 t ha^{-1} year^{-1} in 44% of the mine zones in the Martin basin. In these latter zones, plant colonization is difficult, enabling the formation of rill networks depending on the degree of disturbance, slope length and available water, among other factors (Moreno-de las Heras et al. 2010). The consequence is a high erosion rate that endangers the life span of these newly created habitats and the wetlands created in the pit of the restored mines, which were established by being filled with high loads of mined materials but are filled with eroded sediments. This siltation process also reduces other key ecological processes (e.g., sediment–water column exchanges, organic matter enrichment) and the biological structure of this type of ecosystem (Mitsch and Gosselink 2007; Gell et al. 2009).

In most of the (north) lowland and relatively flat part of the basin, which is dominated by agriculture, the estimated erosion rates are much lower (in general, <10 t ha^{-1} year^{-1}). The high rates in this part of the basin are associated with river dynamics (bank erosion) and land use (Fig. 2), prevalent cereal crops, and scrublands. In the southern and central zones of the basin, which are covered by conifers and hardwoods, the estimated values of the *C* factor (the vegetation-related variable in the RUSLE equation; see part 2 of the ESM) were, as expected, low because of the relatively high cover density. The *C* factor for vineyards and olive trees had typical intermediate values because of the vegetation-free zones between the rows of plants, which are common in this type of land use. However, for scrubland, the *C* values obtained reflect the low vegetation density of this land cover. Grassland was expected to show lower values than those obtained, but these values, again, depend on vegetation density, which is widely spaced. In any case, grasslands occupy only 0.5% of the whole area of Martín basin.

Grassland-shrubland was found to be more susceptible to soil losses by water erosion than cropland, forests, and plantations. A high erosion rate seems unlikely to occur in conifer plantations, but the relatively high rate observed in this land cover in Martin basin is probably due to these artificial plantations being established with the highly regular spatial distribution of the trees in hillslope areas. These results are similar to those observed in other semi-arid areas labeled as poor soil environments with past human overexploitation (Erdogan et al. 2007). These results are also partially a consequence of the anthropogenic displacement of shrubs and forest from low slopes (Smith et al. 2007). Past agricultural practices in these zones have eliminated natural vegetation from the steep zones, leaving a difficult terrain for agriculture (Garcia-Ruiz 2011). Other studies in Spain showed that reforestation followed by insufficient forest management may negatively affect both soil properties and the ecosystem's response to the erosive action of rainfall (Pardini et al. 2003).

Restoration planning to counteract erosion was approached with general reforestation actions extensively applied to large areas for most of the second half of the twentieth century. Now, more specific and autochthonous species are used for plant reforestation in Mediterranean areas (Pausas et al. 2004). Because slope plays a key role in erosion in the Martín basin, restoration actions must focus on the mitigation of slope-based erosion impacts, which requires a more comprehensive restoration planning than just revegetating by planting trees.

The most efficient place from which to remove pollutants and nutrients from watershed discharges is the riparian zone (Welsch and Management 1991), before the water flows enter a stream channel. As most steep zones are located in the upper parts of the basin, the most important locations for protecting and restoring riparian buffers are along these headwater streams. Buffers disrupt lateral linkages within catchments, and they may include alluvial pockets of floodplains, fans, or piedmont zones that occur at breaks in slopes along valley margins, disconnecting lateral connectivity in catchments (Fryirs et al. 2007). Solutions include low-cost erosion control techniques such as contour hedgerows across the slope in cropped fields or regenerated on the base of steeper inaccessible areas, where restoration actions are impossible or too expensive, to reduce runoff velocity and prevent pollution of the river network.

Lasanta et al. (2001) and other studies showed that in Spain, the main process following the abandonment of hillslope cropping is the collapse of the terrace walls by landslides. Many areas identified in the highlands are affected by this problem. Where possible, the recovery of decaying cropping terraces in the steep slopes will be a good soil conservation practice (Dunjó et al. 2003). Other techniques are stone terracing, where a stone embankment

(Marienfeld 1994) around a hillside intercepts overland flow, enhances infiltration, and safely guides runoff off field. These are some of the major recommended engineering structures for controlling soil erosion.

Stimulating extensive livestock forage in depleted soil using leguminous forage crops (*Medicago sativa L.*) would improve the soil conditions in the valley floors (Prosperi et al. 2006). Because the shortage of nutrients in the Aragon region is the first limiting factor for plant colonization (Ries et al. 2000; Lasanta et al. 2000), an enormous step forward will be the creation of a management plan for the use of organic waste as compost. This action would improve soil structure with organic matter and nutrients, taking advantage of this precious resource that is currently lost in landfills. This action will help plant colonization and consequently soil cover, which, when exceeding 60%, can significantly reduce soil erosion in semi-arid environments (Sauer and Ries 2008). These combined benefits will result in increased and sustained crop yields as well as enhancements to multiple ecosystem services.

Conclusion

In this paper, we show a detailed mapping of soil erosion rates in the Martin basin. On the basis of this mapping, we are able to locate erosion-prone areas, where the concentrated water flow creates irreversible soil losses, by applying novel methodologies for collecting representative data for the RUSLE-GIS model. In particular, we demonstrate that the data sampling methodology and the GP-based methodology designed by González and Bullock (unpublished) and Puente et al. (2011), respectively, are useful techniques for estimating the RUSLE C factor. On the basis of our experimental results, we believe that these two techniques applied together could improve the prediction of soil erosion rates and soil conservation–restoration planning at the basin scale.

Detailed site-specific restoration actions should be planned at the watershed scale, focusing on implementing buffer strip zones in the steep zones of the watersheds and in the river shores to decrease soil loss from the upper valleys to the rivers.

Future studies on mine areas will require the latest available information to avoid erroneous estimation and, consequently, erroneous future restoration plans. We strongly recommend that, after the application of the model, a field survey be conducted to verify the predicted degraded areas that can be masked by bare rock and rock accumulation, among others.

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