Assessing the potential effect of different land cover management practices on sediment yield from an abandoned farmland catchment in the Spanish Pyrenees

Manuel López-Vicente · Noemí Lana-Renault · José M. García-Ruiz · Ana Navas

M. López-Vicente (✉) · A. Navas

Department of Soil and Water, EEAD-CSIC. Avda. Montañana 1005, 50059 Zaragoza, Spain

N. Lana-Renault

Área de Geografía, Departamento de Ciencias Humanas, Universidad de La Rioja. 26004 Logroño, Spain

J. M. García-Ruiz

Department of Geo-environmental Processes and Global Change, IPE-CSIC. Avda. Montañana 1005, 50059 Zaragoza, Spain

(✉) **Corresponding author:** Manuel López-Vicente; Tel.: 34 976 716 161; e-mail address: mvicente@eead.csic.es

----- Received	23 March 2011
----- Accepted	3 September 2011
----- Uncorrected proof (DOI: 10.1007/s11368-011-0428-2) in Springer-Verlag	6 September 2011
----- Published online (corrected proof) in Springer-Verlag	16 September 2011
----- Published in <i>Journal of Soils and Sediments</i> * 11(8): 1440–1455	21 November 2011

*Responsible editor: Nikolaus Kuhn

Abstract

Purpose: Sediment delivery from headwater catchments to reservoirs is a serious threat to reservoir sustainability, and is a critical issue in Mediterranean environments where water resources are scarce. In this study we assessed the consequences of two landscape management scenarios (natural vegetation recovery and scrub clearance) on soil erosion and sediment yield. The results were analyzed in relation to predicted and measured rates of soil erosion and sediment yield, with the aim of promoting better management practices.

Materials and methods: The study area was the Arnás River catchment (284 ha), which is located in the central Spanish Pyrenees; the area includes abandoned and poorly managed fields. The combination of the RUSLE and SEDD models of soil erosion and sediment delivery was evaluated in terms of its ability to predict annual rates of sediment yield, using field measurement data for seven water years at the gauging station. The consequences of natural plant succession in other areas of the Spanish Pyrenees, and scrub clearance practices implemented by certain regional governments to increase grazing meadow areas and reduce the incidence of wildfires, were spatially analyzed using GIS. The main sediment source areas were identified and their specific and total sediment yield rates were calculated.

Results and discussion: The predicted soil loss under existing conditions was $2.6 \text{ Mg ha}^{-1} \text{ yr}^{-1}$, with 5% of the surface area affected by rates greater than 2 tons. The measured sediment yield (SY) range was 69–534 Mg yr^{-1} . The maximum SY detected was associated with an extraordinary debris flow. The predicted rates were strongly correlated to measured rates (Nash–Sutcliffe coefficient = 0.72). The main sources of were alluvial deposits (Specific Sediment Yield = $51 \text{ Mg ha}^{-1} \text{ yr}^{-1}$), bare soil (SSY = 12), unpaved trails (SSY = 11), lots (SSY = 4) and pastures (SSY = 1). Under a scenario of vegetation recovery, decreases of 3, 17 and 16% in soil loss, and sediment delivery and yield (respectively) are predicted, whereas increases of 15, 5 and 2% are predicted following scrub clearance practices.

Conclusions: Coupling the RUSLE and SEDD models enabled estimation of annual values of soil erosion and delivery in monitored and unmonitored catchments of small and medium size, making this approach a useful tool for risk analysis. Management practices that combine fire-risk control, by the implementation of scrub clearance practices, with the effects of plant succession on the sediment production are suggested as the best management strategy.

Keywords Pyrenees • RUSLE model • SEDD model • Sediment yield • Soil erosion • Suspended sediment

1 Introduction

Although sediment has a variety of diverse roles, and its regulation and management are complex and differ among countries (Apitz 2008), it is well accepted that reservoir siltation (RS) is one of the worst off-site consequences of soil erosion and sediment delivery (Korsching et al. 2001; Navas et al. 2004). RS is a serious threat to the economic sustainability of irrigated areas and the provision of water supplies for human consumption (Hao et al. 2004). It reduces the water storage capacity of reservoirs, reduces reservoir lifespan, affects electricity production and flood control capacities, and increases maintenance costs (Garbrecht and Garbrecht 2004). Approximately 880 million tons of agricultural soil is deposited in reservoirs in the USA and other aquatic systems each year. As a result of human activities, erosion and runoff from croplands contribute about 25 billion metric tons of soil, sediment and suspended solids to the world's surface waters each year (Biswas 1997). In response to concerns about these threats the European Union (EU) issued the Water Framework Directive (WFD; EPC 2000) in October 2000, which defines management objectives, and set water quality objectives and standards to avoid water pollution and preserve water resources and human health.

During the 20th century the number of dams increased markedly, and in 2000 the World Commission on Dams (WCD) reported that more than 47,000 large dams and reservoirs were in operation worldwide (WCD 2000). Apart from regional differences, there are almost no river catchments that are unaffected by large dams and their continuous or intermittent influence on sediment yield (Schäuble et al. 2008). In Mediterranean areas and semiarid environments, where water resources are scarce and variable in time, dams located in the headwaters are the main source of water for irrigation, hydroelectricity production, and human supplies. Hence, information on water-induced soil erosion and sediment export is essential to the implementation of management practices that are better designed to prevent reservoir siltation (Molino et al. 2007). Moreover, spatially distributed values of specific sediment yield can be used to assess the risk of

desertification in Mediterranean areas ([Vanmaercke et al. 2011](#)). Several studies in the Spanish Pyrenees have addressed various processes and scenarios involved in soil erosion, and sediment delivery and supply to reservoirs, including the Barasona (e.g. [Valero-Garcés et al. 1999](#) and [Navas et al. 2004](#)) and Yesa reservoirs ([Navas et al. 2009](#)).

There has been a general shift in the geomorphological and sedimentological behavior of headwaters in the Mediterranean region, dictated by an increase in vegetation cover in Mediterranean mountains. Since the middle of the 20th century, abandonment of farmland on hillslopes and in many overgrazed areas has been the most outstanding feature of Mediterranean mountains, associated with generalized depopulation ([García-Ruiz and Lana-Renault 2011](#)). This has resulted in vegetation recolonization (expansion of shrubs and forest), which has been associated with a reduction of soil erosion and implementation of reforestation policies by regional and national offices. However, sheet-wash erosion, piping and gullyng are affecting abandoned fields in semiarid environments. The EU Agrarian Policy and the strengthening of national and international markets have encouraged the expansion of croplands into marginal lands, including steep, stony hill slopes, which has led to increased soil erosion, particularly during intense rainstorms ([García-Ruiz 2010](#)). [García-Ruiz et al. \(2011\)](#) reported that most rivers in the Mediterranean region have been subject to decreased streamflows because of the combined effects of decreased precipitation, increased temperature and, particularly, plant recolonization following farmland abandonment. This confirmed that the evolution of soil erosion is closely related to land use/land cover changes, and suggests a progressive shrinkage of the sediment sources. Since approximately 1970 a decrease in sediment siltation has occurred in several reservoirs of the Pyrenees, and this has been attributed mainly to a decrease in the geomorphological activity of south-facing slopes following farmland abandonment ([Beguería 2006](#)). However, the subsequent expansion of forests and shrubs has had negative consequences; in particular it is considered the principal cause of extensive wildfires ([MacDonald et al. 2000](#)).

The pressures on soil and water resources are increasing because of the existing socio-economic and environmental conditions, which are affected by land use changes, increasing temperature, uncertain precipitation trends, population growth, and an increasing proportion of urban inhabitants generating greater energy and food demands ([López-Moreno et al. 2008](#)). To ensure good management of water resources/reservoirs it is necessary to estimate the rates of sediment delivery from upstream areas, and to assess the impact of various land management practices on sediment delivery. In this study annual soil erosion, and sediment delivery and yield rates were predicted for the Arnás River catchment (central Spanish Pyrenees) by coupling the revised universal soil loss equation (RUSLE) and sediment delivery distributed (SEDD) models. The Arnás River is a tributary of the Lubierre River (Aragón River basin), which contributes most of the inflow to the Yesa reservoir (one of the largest reservoirs in the Pyrenees). The Arnás River catchment is a small headwater catchment that has been subject to farmland abandonment and subsequent plant recolonization.

Following validation of the models for estimated rates of occurrence of suspended sediments at the outlet of the catchment, based on comparison with field measurements, the models were run under two management regimes that have been implemented by regional governments in other areas of the Spanish Pyrenees and the Iberian Range, as part of their environmental policies. The successful adaptation of the RUSLE and SEDD models to a similar catchment in the external ranges of the Pyrenees was recently tested

by López-Vicente and Navas (2009 and 2010a). These past mountain agricultural areas are today marginal territories with poorly defined management strategies, and are therefore subject to a high degree of uncertainty in relation to the evolution of water availability and quality, soil degradation, and productivity. In this context the present study aimed to promote medium- to long-term sustainable soil and water conservation practices focused on soil erosion and degradation processes in catchments affected by land abandonment, and to increase the lifespan of reservoirs located in their headwaters.

2 Material and methods

2.1 Study area

The Arnás research catchment (284 ha) is located in the central part of the Spanish Pyrenees, in the basin of the upper Aragón River, a northern tributary of the Ebro River (Fig. 1a, b). The altitude of the highest divide is 1341 m a.s.l., and the outlet is at 910 m a.s.l. The substratum of the entire area is Eocene flysch, and is composed of thin, alternating layers of northward-sloping sandstones and marls. The east–west orientation of the stream results in a marked contrast in slope between the north-facing and south-facing hillslopes (average slopes 29% and 51%, respectively). The north-facing hillslope is characterized by an undulating topography with old scars and stable mass movement deposits covered by well-developed haplic kastanozems and haplic phaeozems. The south-facing hillslope is characterized by severe erosion and mobilization of soil particles (Navas et al. 2005; Navas et al. 2011a). The soils are mostly shallow carbonate-rich regosols, show signs of major degradation, and have low water retention capacity and infiltration rates (Navas et al. 2008). The recent and relatively active slumps that are evident on the upper parts of this hillslope are disconnected from the drainage network, and consequently their net contribution to the total sediment budget of the catchment is limited. The three gully systems of the study area are of limited extent, and they lack vegetation only in the inner parts.

The average annual precipitation from 1999 to 2008 was 926 ± 182 mm. Precipitation mostly occurs in autumn and spring, although intense short-duration convective storms are relatively frequent in summer. The average annual temperature during this period was 9.8 °C. The catchment is representative of mid-altitude Mediterranean mountain environments and land management evolution. It was completely cultivated with cereal crops in terraced fields until the middle of the 20th century, then progressively abandoned and naturally colonized by *Genista scorpius*, *Echinospartum horridum*, *Juniperus communis*, *Rosa* gr. *canina* and *Buxus sempervirens*. The north-facing hillslope now has dense vegetation cover, with large areas of *Pinus sylvestris* and *Quercus faginea* (Fig. 1c). In contrast, the steeper and more sun-exposed south-facing hillslope is characterized by poorly developed open shrub cover. The main areas contributing sediment are: i) the bare sections of the taluses flanking the main stream where the flysch lithology crops out; ii) patches of land affected by severe sheet-wash erosion and trails related to present grazing activity, spread mostly across the steep south-facing hillslope; and iii) an unpaved and deteriorated road on the north-facing hillslope (Lana-Renault and Regüés 2009).

The Yesa reservoir is one of the largest in the Pyrenees, and was constructed in 1959 to supply water to 60,701 ha of irrigated fields in the lowlands/Ebro Depression (northeast Spain) via the Bardenas Canal (Fig. 1b). The reservoir drains an area of 2181 km² in the upper Aragón River basin. Downstream of the reservoir the Aragón River runs southward and flows into the Ebro River. Human activities in the area

changed substantially during the 20th century. Most of the cultivated fields were abandoned (except in the Inner Depression) and are now colonized by shrubs or reforested with pines. Natural forests occur above 900 m a.s.l. on the north-facing hillslopes, and above 1400 m a.s.l. on both the north- and south-facing hillslopes. Subalpine grasslands dominate above 1600–1700 m. The rapid transformation of land use largely explains the different sedimentation rates identified in the history of sediment accumulation in the Yesa reservoir (Navas et al. 2009). The original capacity of the reservoir (470 hm³) was recalculated by bathymetry in 1986 to be 450 hm³ (CEDEX 1988), which implies an annual average siltation rate of 0.79 hm³ over the period 1959–1986. The Yesa reservoir is being enlarged (Spanish Hydrological Plan) to support an increase in irrigation area and to provide water for human consumption in the metropolitan area of the city of Zaragoza (almost 800,000 inhabitants in 2010).

2.2 The RUSLE model of soil erosion

The revised universal soil loss equation (RUSLE) model (Renard et al. 1997) predicts annual soil loss (E ; Mg ha⁻¹ yr⁻¹; Equation 1) by sheet and rill water erosion as the product of the factors of rainfall and runoff erosivity (R ; MJ mm ha⁻¹ h⁻¹ yr⁻¹), soil erodibility (K ; Mg h MJ⁻¹ mm⁻¹), slope steepness and length (LS), cover management (C) and support practices (P):

$$E = R K L S C P \quad (1)$$

The R factor assesses the effect of rainfall impact on the soil surface as well as the magnitude of runoff, and is defined mathematically as the sum of the storm erosivity indexes (EI_{30} ; MJ mm ha⁻¹ h⁻¹) for all the erosive storm events during the year. An erosive event has a rainfall volume > 12.7 mm or a peak rainfall intensity > 6.35 mm in 15 min. The R factor is usually calculated from records of precipitation with a time interval of ≤ 30 min, but as in the case for our study site there is frequently insufficient data to compute R . However, it can be estimated using the approach of De Santos Loureiro and De Azevedo Coutinho (2001), who analyzed 17 rain gauges in southern Portugal using the relationship between the calculated EI_{30} values and two parameters: the monthly rainfall for days with ≥ 10 mm ($rain_{10}$; mm) and the monthly number of days with rainfall ≥ 10 mm ($days_{10}$; n). They reported a strong correlation ($r^2 = 0.84$) for this relationship, and the resulting regression equation (Equation 2) has been widely applied in Mediterranean areas (e.g. Diodato and Bellocchi 2010; Grauso et al. 2010; Catari et al. 2011):

$$EI_{30month} = 7.05 \, rain_{10} - 88.92 \, days_{10} \quad (2)$$

Using the approach of Renard et al. (1997) and equation (2), López-Vicente et al. (2008) found very similar values of the R factor for a ten-year period in the Spanish Pre-Pyrenees, demonstrating the reliability of the selected approach. Soil erodibility is a complex property, but can be defined as the ease with which soil is detached by rain splash or runoff, or these factors in combination. The K factor is a lumped parameter that represents an integrated average annual value of the soil profile reaction to the processes of soil detachment and transport by raindrop impact and surface flow, localized deposition as a consequence of the topography and tillage-induced roughness, and rainwater infiltration into the soil profile (Renard et al. 1997). This factor (Equation 3) is assessed as a function of the soil organic content or organic matter (SOC or OM ; %), the product of the percentages of modified silt (2–100 μm) and sand (100–2000 μm) (M), the classes of aggregate structure (s), and the soil permeability (p):

$$K = \frac{[2.1 \cdot 10^{-4} (12 - OM) M^{1.14} + 3.25 (s - 2) + 2.5 (p - 3)]}{100} 0.1317 \quad (3)$$

The RUSLE model was used to establish four soil structure classes and six permeability classes (Renard et al. 1997). The latter property was calculated from field measured values of saturated hydraulic conductivity (K_{fs} , mm day⁻¹), and modified by the effect of rock fragments on soil infiltration, following Equation 4. In previous studies of stony soils, López-Vicente et al. (2006) and López-Vicente and Navas (2009) estimated a higher K factor value in abandoned fields in the Estaña catchment when the high content of coarse fragments was accounted for. Although the percentage of coarse fragments in a soil can vary within a particular area, rocks appear in the soil profile as a frame, especially in inter-rill areas where runoff cannot move them. Moreover, rock fragments larger than 2 mm are excluded when K_{fs} values are estimated using Equation 3. To account for the effect of rocks on soil erodibility the RUSLE model includes the approach defined by Equation 4:

$$K_b / K_{fs} = (1 - R_w) \quad (4)$$

where K_b (mm day⁻¹) is the modified saturated hydraulic conductivity after accounting for the effect of rock fragments, and R_w (%) is the weight percentage of coarse fragments. As the effect of rock fragments on runoff production and soil loss is very complex and time-dependent (Smets et al. 2011), Equation 4 provides an effective solution for stony Mediterranean soils. The LS factor describes the combined effect of slope length and steepness, and can be thought of as a measurement of the capacity for sediment transport by runoff. In this study the LS factor was calculated (Equation 5) as a function of the net contributing area ($A_{s,i}$, m) and the slope angle (α_i , radians) (Moore and Burch 1986; Moore and Wilson 1992):

$$LS_i = \left(\frac{A_{s,i}}{22.13} \right)^p \left(\frac{\sin \alpha_i}{0.0896} \right)^q \quad (5)$$

where p and q are two empirical exponents having values ($p = 0.4$ and $q = 1.3$) assigned by Moore and Wilson (1992). As the RUSLE model was designed to estimate soil loss from planar slopes with uniform characteristics, the original equation of the model is not suitable at a catchment scale (Kinnell 2010). Equation 5 enables the model to be applied to hillslopes that are not uniform, and it is easy to incorporate it into a GIS application. This approach has been successfully used in studies of other Mediterranean areas including northeast Spain (Martínez-Casasnovas and Sánchez-Bosch 2000) and southern Italy (Di Stefano et al. 2000). In estimating the LS factor in this study, particular attention was paid to including the high values of slope steepness that occur in the scarps of gullies and the river course. The scarps in the study area only occupy 0.02% of the total surface area, but as they are located next to the Arnás River they represent important sources of sediments. Although it may not be possible in the RUSLE model to precisely represent the processes that take place in river taluses, gullies and scarps, an attempt was made to represent these conditions (bare soil surface and steep slope). The cover management (C) factor reflects the effects of cropping and management practices on erosion rates. The soil loss ratio (SLR_i) is an estimate of the ratio of soil loss under actual conditions to losses occurring under reference conditions (clean-tilled continuous fallow). An individual SLR_i value was calculated for each time period i using Equation 6:

$$SLR_i = PLU_i CC_i SR_i SC_i SM_i \quad (6)$$

where the sub-factors for each time period i are the prior land use (PLU_i), the canopy cover (CC_i), the surface roughness (SR_i), the surface cover (SC_i), and the antecedent soil moisture (SM_i). Each SLR_i value was then weighted by the fraction of rainfall and runoff erosivity (EI_{30i} , %) associated with the corresponding time period, and these weighted values were combined into an overall C factor value using Equation 7:

$$C = \frac{1}{EI_{30t}} \sum_{i=1}^n EI_{30i} SLR_i \quad (7)$$

where EI_{30t} (%) is the sum of the EI_{30i} percentages for the entire time period, and n is the total number of time periods i . The values of the C factor range from 0 (total control of the erosion) to 1 (no effectiveness of cover/management practices). The sub-factors are defined by equations 8–11:

$$PLU_i = C_f C_b \exp\left[-\left(c_{ur} B_{ur}\right) + \left(c_{us} B_{us} / C_f^{C_{uf}}\right)\right] \quad (8)$$

$$CC_i = 1 - F_c \exp(-0.1 H) \quad (9)$$

$$SR_i = \exp[-0.66(R_u - 0.24)] \quad (10)$$

$$SC_i = \exp\left[-b S_p \left(\frac{0.24}{R_u}\right)^{0.08}\right] \quad (11)$$

where C_f is a surface soil consolidation factor, C_b represents the relative effectiveness of subsurface residue in consolidation, B_{ur} (lb acre⁻¹ in⁻¹) is the mass density of live and dead roots in the upper inch of the soil, B_{us} is the mass density of incorporated surface residue in the upper inch of the soil (lb acre⁻¹ in⁻¹), c_{uf} represents the impact of soil consolidation on the effectiveness of incorporated residue, c_{ur} and c_{us} are calibration coefficients indicating the impacts of subsurface residues, F_c (%) is the fraction of land surface covered by canopy, H (ft) is the distance that raindrops fall after striking the canopy, R_u (in) is the surface roughness at initial conditions and immediately prior to tillage, b is an empirical coefficient that indicates the effectiveness of surface cover in reducing soil erosion, and S_p (%) is the percentage of land area covered by surface cover. Antecedent soil moisture is an inherent component of continuous-tilled fallow plots, the effects of which are reflected in the soil erodibility factor. Hence, no adjustment was made for changes in soil moisture when calculating the C factor. For the elaboration of C factor maps in this study, those pixels associated with the main flow paths (the river and the permanent gullies) were considered to be areas of bare soil. The support practices (P) factor is the ratio of soil loss with a specific support practice on croplands to the corresponding loss with upslope and downslope tillage. In the Arnás catchment the P factor was set to a value of 1.0 because of the absence of any support practice.

2.3 Sediment delivery and sediment yield

Various numerical approaches are available for estimating the sediment fluxes according to the spatial and temporal scales considered in the equations as well as the number and types of processes. For large river basins the specific sediment yield index (SSY index) facilitates the integration of mass movement, hillslope erosion processes, transfer of sediments, and deposition processes (Delmas et al. 2009). For catchments of medium and small size the sediment delivery distributed (SEDD) model discretizes the study area into morphological units, and calculates the sediment delivery ratio for each unit (Ferro and Porto 2000). In a spatially distributed approach, each cell i of the catchment is considered to be a morphological unit (areas of

defined aspect, length and steepness), and thus the sediment delivery ratio for cell i (SDR_i) is the fraction of the gross soil loss from cell i that reaches a flow pathway that is a continuous stream system or a change in steepness or slope aspect. The SDR_i was estimated as a function of travel time (t_i , hr) according to equations 12–14 (Ferro 1997):

$$SDR_i = \exp(-\beta t_i) \quad (12)$$

$$t_i = \frac{1}{3600} \sum_{i=1}^{N_p} \frac{l_i}{v_i} \quad (13)$$

$$v_i = d_i s_i^{0.5} \quad (14)$$

where β is the catchment-specific parameter (which primarily depends on watershed morphology and is assumed constant for a given basin), N_p is the number of cells through which the flow path traverses from cell i to the channel, l_i (m) is the length of segment i in the flow path (and is equal to the side or diagonal of a cell, depending on the flow direction in the cell), v_i (m s⁻¹) is the flow velocity, d_i (m s⁻¹) is a coefficient that is a measure of surface roughness characteristics, and s_i (m m⁻¹) is the slope of cell i . Equation 14 is derived from Manning's equation, and the d_i coefficient of each grid cell is influenced by land cover. This effect on the d_i coefficient is measured by the value of Manning's roughness coefficient and the hydraulic radius (Fu et al. 2006). Three maps of the d_i coefficient were generated to reflect the various patterns of runoff dynamics for each scenario of land management considered for the Arnás River catchment.

In modeling sediment yield using the RUSLE and SEDD models we assumed that $\beta = 1$, following the method of Fernandez et al. (2003) and Fu et al. (2006) for catchments in the USA. Jain and Kothyari (2000) tested β between 0.1 and 1.6 (increments of 0.1), and reported that sediment yield was not very sensitive to the value of β . We assumed that SDR is time-dependent only, and that over a long period an ephemeral or permanent stream must transport all of the sediment to the outlet. The specific sediment yield for each cell i (SSY_i , Mg ha⁻¹ yr⁻¹) was estimated (Equation 15) using values of gross soil erosion obtained by the RUSLE model and the map of SDR_i :

$$SSY_i = SDR_i E_i \quad (15)$$

The predictive ability of coupling the RUSLE and SEDD models for an annual cycle was demonstrated at the catchment scale under Mediterranean conditions by Di Stefano and Ferro (2007). The sediment transport effectiveness of a catchment decreases as t_i increases, and consequently as catchment area (A_w , ha) increases (López-Vicente and Navas 2010a). The annual intra-basin stored sediment (ISS_w , %) is the total amount of sediment that temporarily accumulates in the basin during the year and does not reach the basin outlet (Beach 1994). To improve assessment of the processes of soil redistribution in the study area, ISS_w was calculated (Equation 16) as the proportion of the total eroded particles (E_w) that temporarily accumulated in the catchment.

$$ISS_w = 100 - \left(\frac{100 SSY_w}{E_w} \right) \quad (16)$$

2.4 Field data

We included annual data on rainfall and suspended sediment yield in the catchment for seven water years (from 1 October to 30 September) during the period 1999–2008. The water years 2004/05 and 2006/07 were not included because of significant gaps in the datasets. Values of daily rainfall were collected using three tipping buckets located along the main stream; the buckets are connected to data loggers that record 0.2 mm precipitation increments. The gauging station was located at the outlet of the catchment (42°38'19" N, 0°35'10" W), and provided continuous recordings of water discharge and sediment transport (Fig. 1c). The water level was measured using an ultrasound sensor and a water pressure sensor. Discharge was determined using established stage rating curves. The suspended sediment concentration (SSC) was determined using a turbidity meter that had previously been calibrated in the laboratory using material from the catchment. The calibration was performed three times following installation of the turbidity meter to test its robustness through time. All sensors were connected to data loggers recording average data every 5 min. In addition, suspended sediments were recorded during floods using an automatic water sampler. Further details of the instrument set up are provided by Lana-Renault and Regüés (2009), and Lana-Renault et al. (2010) provides a summary of the research carried out in this catchment.

A total of 78 soil samples were collected in a regular grid of 200 × 200 m (Fig. 1c). The samples were air dried, ground, homogenized, quartered and passed through a sieve (2 mm) and the percentages of rock fragments, clay, silt, sand and soil organic content (*OM*) were estimated. Maps of the parameters required in equations 3 and 4 were created using the Splines Interpolator Method, which fits a minimum curvature surface through the input points. The method is based on a family of continuous, regular, and derivable functions. Splines are similar to the equations obtained from trend surfaces or regression-based methods. This method creates a new function for each location or sampling point with no loss of continuity properties among the curves (Mitasova et al. 1995). Consequently, it is adequate for mapping the soil properties and parameters involved in soil erosion processes. The predominant soil aggregate type was medium or coarse granular, and only three soil samples had a blocky structure. These data and the saturated hydraulic conductivity measured for each soil type (range 130–1500 mm day⁻¹) were used in equations 3 and 4 for calculating the soil erodibility factor. The parameters of net contributing area and slope steepness (*LS* factor) were calculated from the digital elevation model (5 × 5 m of raster spatial resolution) and a combined flow accumulation algorithm that provides an accurate spatially distributed description of the cumulative overland flow. A threshold value was derived from the map of flow accumulation to discriminate those areas dominated by non-concentrated overland flow (using a multiple flow algorithm from mountain divides to the beginning of the main overland flow pathways) from those with concentrated overland flow (using a single flow algorithm downwards from the beginning of the main overland flow pathways; more details are provided by López-Vicente and Navas 2010b). Three maps of the average annual *C* factor and flow velocity (Equation 14) were calculated for the various land use scenarios modeled in this study, and were incorporated into the RUSLE and SEDD models, respectively. All the input and output maps and mathematical operations were generated and run at a spatial scale of 5 × 5 m using ArcGIS 9.3[®] software.

2.5 Land use scenarios

A map of the land uses in the catchment was prepared based on visual interpretation of an aerial photograph taken in 2000 (0.5 m of raster cell spatial resolution; freely available at SIGPAC:

<http://sigpac.mapa.es/feqa/visor/>) and field surveys (Fig. 2). The scenario of vegetation recovery (VR-sc) was based on the process of plant colonization that commenced when the fields were abandoned. It is a complex process involving the interaction of a range of variables including: (i) time, which allows plant succession and an increase in the complexity of the structure of the plant community; (ii) the ecological conditions (both physical and biotic) of each abandoned field; and (iii) human activity, including the agricultural history of the field, and subsequent management by grazing, fire and clearing. In general, the abandoned fields of the central Pyrenees have passed through the following successional stages following abandonment (Molinillo et al. 1997): (1) invasion by herbaceous plants during the years immediately following abandonment, with some woody shrubs appearing after 10–15 years; (2) spreading and generalized coverage by woody shrubs between 10 and 35 years; (3) decline of woody shrubs and a new expansion of herbaceous plants between 35 and 60 years; and (4) the appearance of young trees in fields approximately 60 years following abandonment. In the latter stages, particularly in relatively undisturbed areas, the fields begin to take on the appearance of natural hillslopes, the process of soil deposition predominates over net soil loss, and the soil nutrient conditions (including total organic carbon and total nitrogen) approach those measured in cultivated fields (see Navas et al. 2011b in relation to abandoned fields in the Aisa valley, central Spanish Pyrenees). The regeneration of plant cover leads to economic and environmental benefits including increased absorption of CO₂ (Adams and Piovesan 2002), reduced soil erosion, higher concentrations of sediments in some places, and improved quality of runoff water (García-Ruiz and Lana-Renault 2011).

The scenario of scrub clearance (SC-sc) was based on programs implemented by certain regional governments to enlarge the grazing meadow areas and to avoid an increase in the incidence of fires, which has been one of the negative outcomes of land abandonment in the Spanish Mediterranean mountains. Scrub clearing has had positive results in the La Rioja region (northeast Spain; Lasanta et al. 2009), and has led to a reduced incidence of wildfires in the past decade, relative to their incidence during the 1970s and 1980s. Mediterranean scrub species pose a high fire risk (Papio and Trabaud 1991). Scrub clearing appears to be an efficient (not requiring large investments of time and money) means of management for marginal mountainous areas. There has been a significant increase in fire risk over large areas as a result of the spread of scrub and woodland, which is dominated by conifers. In addition, the expansion of thorny scrubs has reduced the potential to increase livestock numbers and reorganizing the extensive cattle management. In this context the regional administrations of Spanish autonomous communities including Andalusia, Asturias, Galicia and La Rioja have introduced specific measures that favor the regeneration of pasture land. These include the clearing of scrub in areas where slope steepness does not exceed 35% (to avoid intense soil erosion), with the aim of supporting extensive grazing, and the simultaneous creation of fire-breaks to reduce the risk of fire and limit fire extent. The conversion of abandoned fields and scrubland to grassland has produced good results in other Mediterranean areas (e.g. Stagliano et al. 1999; Lécirvain and Beylier 2004), and is a relatively simple process involving mechanical clearing of the scrub and periodic introduction of livestock to prevent its re-establishment (Fernández-Alés et al. 1997). From the middle of the 17th century (when the greatest stock numbers occurred) until the middle of the 20th century the number of sheep declined continuously. However, in the 1990s a slight change in this trend occurred as a consequence of subsidies under the CAP (Common Agricultural Policy). While scrub clearance could help to increase the available area for cattle grazing, the consequences of such programs on soil erosion and sediment yield are not well

understood (Lasanta et al. 2001). Consequently, evaluation of the *SC-sc* in this study may provide valuable information for policy makers and management authorities.

3 Results and discussion

3.1 Current land use scenario and results validation

The average value of the *R* factor was $1686 \text{ MJ mm ha}^{-1} \text{ h}^{-1} \text{ yr}^{-1}$ for the period 1999/00–2007/08, and $1860 \text{ MJ mm ha}^{-1} \text{ h}^{-1} \text{ yr}^{-1}$ for the seven water years analyzed in the study. However, there was marked interannual variability, with values ranging from 783 to $2740 \text{ MJ mm ha}^{-1} \text{ h}^{-1} \text{ yr}^{-1}$ (Fig. 3a). These values and their temporal variability are consistent with measurements of the *R* factor in other Mediterranean areas (e.g. Ramos 2006; López-Vicente and Navas 2009). The homogeneity of the parent materials provided a common starting point for the *K* factor, which averaged $0.0136 \text{ Mg h MJ}^{-1} \text{ mm}^{-1}$ (standard deviation 0.0026) for the entire catchment. Nonetheless, the different degrees of soil degradation described above (section 2.1) for the southwest-facing (SW-f) and northeast-facing (NE-f) hillslopes explains the higher value of soil erodibility found in the SW-f half of the catchment ($0.0148 \text{ Mg h MJ}^{-1} \text{ mm}^{-1}$) relative to the value obtained for the NE-f part ($0.0130 \text{ Mg h MJ}^{-1} \text{ mm}^{-1}$) (Fig. 3b). The *LS* factor map clearly identifies areas with the greatest runoff detachability and transport capacity, which result from the process of overland flow concentration and extreme steepness of the slopes. The SW-f hillslope had higher values for *LS* (average = 16.5) than did the NE-f hillslope (average = 13.8) (Fig. 3c). The average value of the *C* factor for the existing land use scenario was 0.0055, with the highest values (low protection) found for the river, the main gully systems, areas of bare soil, the alluvial deposits and the unpaved trails ($C > 0.08$). Based on Equation 1 the average estimated value for soil erosion for the entire catchment was $2.63 \text{ Mg ha}^{-1} \text{ yr}^{-1}$, which is greater than the tolerable soil loss ($1.4 \text{ Mg ha}^{-1} \text{ yr}^{-1}$) suggested by Verheijen et al. (2009) for conditions prevalent in Europe and estimated for the European Commission's Thematic Strategy for Soil Protection. However, the spatial distribution of soil erosion was highly variable, with low ($0\text{--}1 \text{ Mg ha}^{-1} \text{ yr}^{-1}$) and tolerable ($1\text{--}2 \text{ Mg ha}^{-1} \text{ yr}^{-1}$) levels of soil loss occurring in 93.4% and 1.6%, respectively, of the study area (Fig. 4); only 5% of the surface area of the Arnás catchment was affected by high erosion rates ($> 2 \text{ Mg ha}^{-1} \text{ yr}^{-1}$). Areas of bare soil on hillslopes, trails, lots and areas with low vegetation cover adjacent to the river (alluvial deposits and the river talus) were affected by intense soil erosion. For these areas the annual average rates found ($22\text{--}97 \text{ Mg ha}^{-1} \text{ yr}^{-1}$) are consistent with the values previously reported by Lana-Renault and Regüés (2009) for the same study site. Although these values indicate the occurrence of intense soil remobilization, the maximum predicted rates in the Arnás catchment are six-fold lower than the quantified values of soil loss reported by Nadal-Romero and Regüés (2010) for badlands located in the central part of the Spanish Pyrenees (approximately 570 and $600 \text{ Mg ha}^{-1} \text{ yr}^{-1}$, in different catchments).

Wood good, brush good, brush poor and wood-grass good areas had low and tolerable values of soil erosion ($< 2 \text{ Mg ha}^{-1} \text{ yr}^{-1}$), whereas meadows and pasture good areas were affected by moderately high values of soil erosion (average rates $2.4\text{--}5.8 \text{ Mg ha}^{-1} \text{ yr}^{-1}$). For the same study area Navas et al. (2005) reported that north-facing slopes (forest) were characterized by soil stability and only minimal soil loss (quantified using inventories of ^{137}Cs), whereas high rates of soil loss ($12\text{--}30 \text{ Mg ha}^{-1} \text{ yr}^{-1}$) were found for a south-facing hillslope with low vegetation cover; these values are in good agreement with the average values predicted using the RUSLE model. However, in contrast to the predictions of the model they found very high rates of

net soil loss (approximately $45 \text{ Mg ha}^{-1} \text{ yr}^{-1}$) in some sections of the Arnás River valley and in some patches of very sparse vegetation within areas of brush poor. These spatial differences highlight the shortcomings of the RUSLE model in those areas of Mediterranean landscape where patches of bare soil typically occur interspersed in scrublands. Consequently, a more detailed map of the *C* factor at higher spatial resolution is required to improve the model predictions, because this factor largely controls the predicted rates of soil loss and their spatial location.

The sediment delivery ratio (*SDR*; average value 0.18) progressively increased from the mountain divides to the outlet of the catchment (Fig. 4). The spatial variation of the *SDR* can be explained by the location of the main overland flow pathways and the river, and the fact that the mathematical procedure of the SEDD model justifies the presence of some linear artifacts. The combined RUSLE and SEDD models estimated sediment flux rates ranging from 100 Mg yr^{-1} (2001/02) to 284 Mg yr^{-1} (2000/01), with an average rate of 193 Mg yr^{-1} for the entire study period. Thus, each year an average of 76% of the total eroded soil was temporarily stored within the catchment (*ISS_w*; Equation 16), and was able to be delivered to the Aragón River in subsequent years. The highest specific sediment yield rates (*SSY*) were found for alluvial deposit ($51 \text{ Mg ha}^{-1} \text{ yr}^{-1}$), areas of bare soil ($12 \text{ Mg ha}^{-1} \text{ yr}^{-1}$), unpaved trails ($11 \text{ Mg ha}^{-1} \text{ yr}^{-1}$) and lots ($4 \text{ Mg ha}^{-1} \text{ yr}^{-1}$). Spatial analysis of the results showed that the amount of eroded sediment exported from the catchment from the wood good, brush good and brush poor areas (total sediment yield 93 Mg yr^{-1}) was similar to the amount exported from areas of bare soil (gullies and river talus), alluvial deposits and trails (70 Mg yr^{-1}), even though the latter areas occupied 63-fold less of the total surface area (Table 1). This demonstrates that the bare soil, alluvial deposit and trail areas (which are legacies of past human activities) were highly connected to the drainage network, and thus contributed substantially to the total sediment yield despite their smaller extent. This finding is consistent with radionuclide analysis (Gaspar et al. 2011), which showed a trend of intense modification of the natural dynamics of soil redistribution in areas subject to human disturbance, relative to forested areas. These values are also consistent with the results obtained for disturbed catchments where anthropogenic disturbance has played a major role in overall sediment flux (e.g. Matthew et al. 2008). Moreover, the main sediment source areas predicted by the models quite accurately reflect the main sediment contributing areas identified by field survey (Lana-Renault and Regüés 2009). The rates of sediment yield predicted in this study are comparable to and within the range of those calculated in areas with similar climatic and physiographic conditions, including the drainage catchment of the Barasona reservoir (central Spanish Pyrenees; $90\text{--}250 \text{ Mg yr}^{-1}$; Alatorre et al. 2010) and many other reservoirs in Spain with similar levels of specific sediment yield (Avendaño Salas et al. 1997). We predicted lower levels of sediment yield than reported in studies performed in gullied Spanish catchments (e.g. De Vente et al. 2005) because the area of the Arnás River catchment affected by gullies is very small. In addition, a significant extent of the river bed is covered with medium sized and large rock fragments, and the slumps located on the south-facing hillslope are disconnected from the drainage network, with the result that their net contribution to the total sediment budget is very limited.

Analysis of the predicted rates of sediment yield indicated a moderately strong relationship between the observed and predicted values which fitted a multiplicative (non-linear) model (Pearson's *r* correlation = 0.78; *p* < 0.05; standard deviation of the residuals = 0.22) that was statistically significant at the 95% confidence level (Fig. 5). For values of measured sediment yield < 200 Mg yr^{-1} the RUSLE and SEDD

models overestimated the total amount of sediment exported; conversely, for high rates of sediment yield the models underestimated the actual yield. During the water year 1999/00 a large rainfall event occurred that caused a debris flow (Dr. S. Beguería, EEAD-CSIC, pers. comm.), which explains the highest *SY* value recorded at the gauging station during the study period. The 1999/00 water year was a significant outlier in the correlation between the predicted and measured *SY* values; the correlation was improved significantly when this particular rainfall event was excluded from the analysis (linear correlation; Pearson's *r* correlation = 0.88). Moreover, a recent study that included data from the Arnás catchment demonstrated that most of the hillslope debris flows are triggered in the central Spanish Pyrenees by rainfall events with a 25–30 year return period (García-Ruiz et al. 2002). As sediment yield is a non-normal variable, we also estimated the Nash–Sutcliffe model efficiency coefficient, and derived a value of 0.72 when the water year in which the extraordinary debris flow occurred was excluded. This indicates that the model predictions are accurate for normal years, and highlights the usefulness of integrated approaches to analysis of soil erosion at the hillslope scale and sediment delivery through the main flow paths. Such integration leads to improved understanding of the entire soil and particle redistribution cycle, will aid sustainable catchment management and development, and helps address the increasing demand by the scientific community and policy makers for information of this nature (e.g. Owens and Collins 2006). As the predicted average sediment yield was 19.5% less than the average yield measured at the gauging station, and because the predicted rates of soil erosion were within the same range as those quantified in previous studies, it is recommended that further research on the permanent water streams of the study area be undertaken using a specific hydrological model of sediment delivery.

3.2 Sediment yield under the scenarios of natural plant succession and scrub clearance

For the vegetation recovery (*VR-sc*) and scrub clearance (*SC-sc*) scenarios the average values of the *C* factor were 0.0054 and 0.0072, respectively, and the average predicted soil erosion rates were 2.55 and 3.01 Mg ha⁻¹ yr⁻¹, respectively (Fig. 4). These values represent a decrease of 3% in soil erosion under the *VR-sc* scenario and a significant increase (15%) under the *SC-sc* scenario. For each scenario these changes were strongly associated with the *C* factor, which reflects differences in canopy cover and soil properties. Improvements in soil properties occur in abandoned fields following plant recovery (e.g. Navas et al. 2008; Wang et al. 2011), and soil loss areas increase with decreasing vegetation cover (e.g. Rivera et al. 2011). Mirroring these patterns, the average sediment delivery ratio for the *VR-sc* was 17% lower than the average value under existing conditions, and the average *SDR* for the *SC-sc* was 5% higher. The combined effects of changes in soil erosion and sediment delivery led to a 16% decrease in the average specific sediment yield (0.54 Mg ha⁻¹ yr⁻¹) and the total sediment yield (150 Mg yr⁻¹) predicted for the *VR-sc* in relation to the average existing values of *SSY* and *SY* (Fig. 6). These results are consistent with the findings of previous studies, which have demonstrated that vegetation regrowth after land abandonment reduces: (i) runoff production; (ii) the connectivity between hillslopes and fluvial channels because of the increasing presence of dense vegetation patches, which reduce the accessibility of sediment (García-Ruiz et al. 2010); and (iii) sedimentation in water bodies, thereby extending their lifespan (Valero-Garcés et al. 1999; Morellón et al. 2009). The relative contribution of each land use to the total amount of exported sediments (calculated as the ratio of the percentages of *SY* and soil surface area) indicated changes in the *VR-sc* in relation to the existing scenario, with increasing relative contributions from the brush poor, bare soil and trail areas, and decreasing

contributions from the alluvial deposits (Table 1). The increase in soil erosion predicted in the *VR-sc* for new forest areas (previously brush good) can be explained by the slightly reduced protection against splash detachment provided by the branches and leaves of trees relative to the protection provided by a canopy of shrubs located near the soil surface.

Conversely, a small increase in the *SSY* (2%) was predicted under the *SC-sc* scenario, with an average value of $0.65 \text{ Mg ha}^{-1} \text{ yr}^{-1}$ and a total sediment yield of 183 Mg yr^{-1} . The small increase in the *SDR* predicted under this scenario explains the modest rise in the levels of exported sediments instead of the significant increase predicted for the values of soil erosion. The increase in the total sediment yield in this scenario can largely be explained by the marked increases in the *SSY* value ($0.75 \text{ Mg ha}^{-1} \text{ yr}^{-1}$) and the area of meadows (38%) relative to their levels under existing conditions (*SSY*: $0.36 \text{ Mg ha}^{-1} \text{ yr}^{-1}$; area: 0.8%) (Table 1). However, reactivation of the gullies, mass movements and an advance of the river talus could be expected under the *SC-sc* scenario, triggered by an increase in runoff as a consequence of the decrease in total vegetation cover. These results highlight the importance of coupling a soil erosion model with an equation for sediment delivery, which enables more accurate predictions of sediment yield rates because the same changes in land use trigger different degrees of change in the average and total values of soil loss and sediment transport. The spatially distributed changes in the value of the *SSY* indicate a relocation of the main sources of sediments, with a general increase in rates associated with hillslopes, and a general decrease associated with the main overland flow pathways and the banks of the Arnás River (Fig. 6). These changes can be explained by the spatially distributed changes in the *SDR*, which showed an increase in the transport capacity for 71% of the study surface area, while only 26% of the surface area retained the same delivery ratio as that predicted for existing conditions. Although changes in soil erosion, sediment delivery and yield were evident between the land management scenarios, the estimated percentages of intra-basin stored sediments (*ISS_w*) remained similar, with only small increases (4.4% and 3.4%, respectively) predicted under the *VR-sc* and *SC-sc* scenarios relative to the existing scenario. Analysis of the consequences of the two land management scenarios on soil erosion processes and sediment production suggests that the best management practice may be to merge the positive effects of natural plant succession (reducing the amount of exported sediments) and the fire-risk mitigation effects of scrub clearance practices, as used in other Spanish regions contributing to the maintenance of cattle and of a density of human population necessary for improving biodiversity and landscape quality.

4 Conclusions

Coupling of the *RUSLE* and *SEDD* models of soil erosion and sediment transport facilitated prediction of the rates of sediment yield at the outlet of the Arnás River catchment over seven water years, as evidenced by the significant correlation coefficient with rates measured in the field. This approach appears to be suitable for estimating annual values of soil erosion and delivery in monitored and unmonitored catchments of small and medium size in mountainous environments. The results can be used to assess the relative effects of land cover changes on soil erosion, and will be useful in risk analysis. The main shortcomings of the approach used in the study are related to (i) application of the *C-RUSLE* factor for those areas where small patches of bare soil are interspersed in scrublands, (ii) the specific processes that take place in the gullies and river talus, and (iii) the sediment delivery capacity of the permanent water streams. Analysis of the maps of the spatial distribution

of soil erosion and sediment yield showed that only a small percentage of the study area was affected by high rates of soil loss. The main source areas for detached particles were the alluvial deposits, the areas of bare soil, and the unpaved trails and pastures. In contrast, the forest, scrubland and meadow areas gave effective protection to the soil against detachment and transport.

Under the scenario of natural plant recovery a decrease is expected in soil loss, sediment delivery and sediment yield. Conversely, these factors are predicted to increase under the scrub clearance scenario. We suggest that scrub clearance combined with recovery of the natural vegetation along contour lines, to create buffer strips, would be effective in retaining eroded soil particles. This represents an additional approach to reorganization of land management systems, with the aim of improving the sustainability of large infrastructure, such as the Yesa reservoir, while providing a framework of effective support practices to maintain the human population and cattle production. Future research should focus on monitoring and modeling the effects of land use, management changes and climate change on the processes of soil erosion and sediment yield in the Arnás River catchment.

Acknowledgements This research was financially supported by the projects: “Soil erosion and carbon dynamic in Mediterranean agroecosystems: radioisotopic modelling at different spatial and temporal scales” (MEDEROCAR, CGL2008-00831/BTE), and “Processes and sediment balances at different spatial scales in Mediterranean environments: Effects of climate fluctuations and land use changes” (PROBASE, CGL2006-11619/HID), both funded by the Spanish Ministry of Science and Innovation. Monitoring of the catchment was supported by an agreement between the CSIC and the Spanish Ministry of Environment (RESEL).

References

- Adams JM, Piovesan G (2002) Uncertainties in the role of land vegetation in the carbon cycle. *Chemosphere* 49(8):805–819
- Alatorre LC, Beguería S, García-Ruiz JM (2010) Regional scale modeling of hillslope sediment delivery: A case study in the Barasona Reservoir watershed (Spain) using WATEM/SEDEM. *J Hydrol* 391(1-2):109–123
- Apitz SE (2008) Is risk-based, sustainable sediment management consistent with European policy? *J Soils Sediments* 8(6):461–466
- Avendaño Salas C, Sanz Montero E, Rayan C, Gomez Montana JL (1997) Sediment yield at Spanish reservoirs and its relationship with the drainage area. *Proceedings of the 19th Symposium of Large Dams*, Florence, International Committee on Large Dams, Paris: 863–874
- Beach T (1994) The fate of eroded soil: sediment sinks and sediment budgets of agrarian landscapes in southern Minnesota, 1851–1988. *Ann Assoc Am Geogr* 84(1):5–28
- Beguería S (2006) Identifying erosion areas at basin scale using remote sensing data and GIS: a case study in a geologically complex mountain basin in the Spanish Pyrenees. *Int J Remote Sens* 27(20):4585–4598
- Beguería S (January 2011) Personal communication. Address: Estación Experimental de Aula Dei (EEAD-CSIC), Avda. Montañana 1005, 50059 - Zaragoza, Spain
- Biswas AK (Editor) (1997) *Water Resources: Environmental Planning, Management and Development*. McGraw-Hill, New York.

- Catari G, Latron J, Gallart F (2011) Assessing the sources of uncertainty associated with the calculation of rainfall kinetic energy and erosivity - Application to the Upper Llobregat Basin, NE Spain. *Hydrol Earth Syst Sc* 15(3):679–688
- CEDEX (1988) Reconocimiento Batimétrico del embalse de Yesa. Centro de Estudios Hidrográficos, Ministerio de Obras Públicas y Urbanismo, Madrid. 63 pp.
- De Santos Loureiro N, De Azevedo Coutinho M (2001) A new procedure to estimate the RUSLE EI_{30} index, based on monthly rainfall data and applied to the Algarve region, Portugal. *J Hydrol* 250(1-4):12–18
- De Vente J, Poesen J, Verstraeten G (2005) The application of semi-quantitative methods and reservoir sedimentation rates for the prediction of basin sediment yield in Spain. *J Hydrol* 305(1-4):63–86
- Delmas M, Cerdan O, Mouchel J-M, Garcin M (2009) A method for developing a large-scale sediment yield index for European river basins. *J Soils Sediments* 9(6):613–626
- Di Stefano C, Ferro V (2007) Evaluation of the SEDD model for predicting sediment yield at the Sicilian experimental SPA2 basin. *Earth Surf Proc Land* 32(7):1094–1109
- Di Stefano C, Ferro V, Porto P (2000) Length slope factors for applying the revised universal soil loss equation at basin scale in southern Italy. *J Agric Eng Res* 75:349–364
- Diodato N, Bellocchi G (2010) MedREM, a rainfall erosivity model for the Mediterranean region. *J Hydrol* 387(1-2):119–127
- EPC (The European Parliament and the Council of the European Union) (2000) DIRECTIVE 2000/60/EC OF THE EUROPEAN PARLIAMENT AND OF THE COUNCIL of 23 October 2000 establishing a framework for Community action in the field of water policy. *Official Journal of the European Communities L* 327:1–72
- Fernandez C, Wu JQ, McCool DK, Stöckle CO (2003) Estimating water erosion and sediment yield with GIS, RUSLE, and SEDD. *J Soil Water Conserv* 58(3):128–136
- Fernández-Alés R, Leiva MJ, García-Gordo J (1997) Regeneración de matorral y producción de pasto tras el desbroce con grada y con cadena en una dehesa de Sierra Morena. *SEEP* 37:295–300
- Ferro V (1997) Further remarks on a distributed approach to sediment delivery. *Hydrolog Sci J* 42(5):633–647
- Ferro V, Porto P (2000) Sediment delivery distributed (SEDD) model. *J Hydrol Eng* 5(4):411–422
- Fu G, Chen S, McCool DK (2006) Modeling the impacts of no-till practice on soil erosion and sediment yield with RUSLE, SEDD, and ArcView GIS. *Soil Till Res* 85(1-2):38–49.
- Garbrecht JD, Garbrecht GKH (2004) Siltation behind dams in antiquity. In: Rogers JR, Brown GO, Garbrecht JD (eds) *Water Resources and Environmental History*. ASCE, Reston, VA: American Society Of Civil Engineers, pp. 35–43
- García-Ruiz JM (2010) The effects of land uses on soil erosion in Spain: A review. *Catena* 81(1):1–11
- García-Ruiz JM, Lana-Renault N (2011) Hydrological and erosive consequences of farmland abandonment in Europe, with special reference to the Mediterranean region – A review. *Agr Ecosyst Environ* 140:317–320
- García-Ruiz JM, Lana-Renault N, Beguería S, Lasanta T, Regüés D, Nadal-Romero E, Serrano-Muela P, López-Moreno JJ, Alvera B, Martí-Bono C, Alatorre LC (2010) From plot to regional scales: Interactions

of slope and catchment hydrological and geomorphic processes in the Spanish Pyrenees. *Geomorphology* 120(3-4):248–257

García-Ruiz JM, López-Moreno JI, Vicente-Serrano SM, Lasanta T, Begueria S (2011) Mediterranean water resources in a Global Change scenario. *Earth-Sci Rev* 105(3-4):121–139

García-Ruiz JM, Martí-Bono C, Lorente A, Beguería S (2002) Geomorphological consequences of frequent and infrequent rainfall and hydrological events in Pyrennez Mountains of Spain. *Mitigation and Adaptation Strategies for Global Change* 7(3): 303–320

Gaspar L, Navas A, Walling DE, Machín J, Gómez Arozamena J (2011) Using ^{137}Cs and $^{210}\text{Pb}_{\text{ex}}$ to assess soil redistribution on slopes at different temporal scales. *Catena*, doi:10.1016/j.catena.2011.01.004

Grauso S, Diodato N, Verrubbi V (2010) Calibrating a rainfall erosivity assessment model at regional scale in Mediterranean area. *Environ Earth Sci* 60(8):1597–1606

Hao FH, Chang Y, Ning DT (2004) Assessment of China's economic loss resulting from the degradation of agricultural land in the end of 20th century. *J Environ Sci-China* 16(2):199–203

Jain MK, Kothyari UC (2000) Estimation of soil erosion and sediment yield using GIS. *Hydrol Sci J* 45(5):771–786

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

Korsching PF, Hoiberg EO, Bultena GL, Padgett SC (2001) Soil erosion as a community issue: Public perceptions of off-site impacts. *Society & Natural Resources* 14(1):67–76

Lana-Renault N, Regüés D (2009) Seasonal patterns of suspended sediment transport in an abandoned farmland catchment in the central Spanish Pyrenees. *Earth Surf Proc Land* 34(9):1291–1301

Lana-Renault N, Regüés D, Nadal-Romero E, Serrano Muela P, García-Ruiz JM (2010) Streamflow response and sediment yield after farmland abandonment: results from a small experimental catchment in the central Spanish Pyrenees. *Pirineos* 165:97–114

Lasanta T, Arnáez J, Oserín M, Ortigosa L (2001) Marginal Lands and Erosion in Terraced Fields in the Mediterranean Mountains A Case Study in the Camero Viejo (Northwestern Iberian System, Spain). *Mt Res Dev* 21(1):69–76

Lasanta T, Arnáez J, Errea MP, Ortigosa L, Ruiz-Flaño P (2009) Mountain pastures, environmental degradation, and landscape remediation: The example of a Mediterranean policy initiative. *Appl Geogr* 29(3):308–319

Lécrivain E, Beylier B (2004) Technique de réouverture d'une colline embroussaillée adaptée au comportement des troupeaux. *Options Méditerranéennes* 62:233–238

López-Moreno JI, Beniston M, García-Ruiz JM (2008) Environmental change and water management in the Pyrenees: Facts and future perspectives for Mediterranean mountains. *Global Planet Change* 61(3-4):300–312

López-Vicente M, Navas A (2009) Predicting soil erosion with RUSLE in Mediterranean agricultural systems at catchment scale. *Soil Sci* 174(5):272–282

López-Vicente M, Navas A (2010a) Relating soil erosion and sediment yield to geomorphic features and erosion processes at the catchment scale in the Spanish Pre-Pyrenees. *Environ Earth Sci* 61(1):143–158

- López-Vicente M, Navas A (2010b) Routing runoff and soil particles in a distributed model with GIS: implications for soil protection in mountain agricultural landscapes. *Land Degrad Dev* 21(2):100–109
- López-Vicente M, Navas A, Machín J (2006) Variation of soil erodibility in abandoned fields: A case study in the Carrodilla Range (Spanish Pyrenees). In: Martínez-Casasnovas JA, Pla Sentís I, Ramos Martín MC, Balasch Solanes JC (Eds.). *Soil and Water Conservation Under Changing Land Use*. pp. 167–170
- López-Vicente M, Navas A, Machín J (2008) Identifying erosive periods by using RUSLE factors in mountain fields of the Central Spanish Pyrenees. *Hydrol Earth Syst Sc* 12(2):523–535
- MacDonald D, Crabtree JR, Wiesinger G, Dax T, Stamou N, Fleury P, Gutiérrez Lazpita J, Gibon A (2000) Agricultural abandonment in mountain areas of Europe: Environmental consequences and policy response. *J Environ Manage* 59, 47–69
- Martínez-Casasnovas JA, Sánchez-Bosch I (2000) Impact assessment of changes in land use/conservation practices on soil erosion in the Penedès-Anoia vineyard region (NE Spain). *Soil Till Res* 57(1-2):101–106
- Matthew CR, Odhiambo BK, Church JM (2008) Spatial Analysis of Soil Erosion and Sediment Fluxes: A Paired Watershed Study of Two Rappahannock River Tributaries, Stafford County, Virginia. *Environ Manage* 41(5):766–778
- Mitasova H, Mitas L, Brown WM, Gerdes DP, Kosinovsky I, Baker T (1995) Modelling spatially and temporally distributed phenomena: new methods and tools for GRASS GIS. *Int J Geogr Inf Syst* 9(4):433–446
- Molinillo M, Lasanta T, García-Ruiz JM (1997) Managing mountainous degraded landscapes after farmland abandonment in the Central Spanish Pyrenees. *Environ Manage* 21(4):587–598
- Molino B, Viparelli R, De Vincenzo A (2007) Effects of river network works and soil conservation measures on reservoir siltation. *Int J Sediment Res* 22:273–281
- Moore ID, Burch GJ (1986) Physical basis of the length-slope factor in the universal soil loss equation. *Soil Sci Soc Am J* 50:1294–1298
- Moore ID, Wilson JP (1992) Length-slope factors for the Revised Universal Soil Loss Equation: simplified method of estimation. *J Soil Water Conserv* 47(5):423–428
- Morellón M, Valero-Garcés B, González-Sampériz P, Vegas-Vilarrúbia T, Rubio E, Rieradevall M, Delgado-Huertas A, Mata P, Romero O, Engstrom DR, López-Vicente M, Navas A, Soto J (2009) Climate changes and human activities recorded in the sediments of Lake Estanya (NE Spain) during the Medieval Warm Period and Little Ice Age. *J Paleolimnol*, doi:10.1007/s10933-009-9346-3
- Nadal-Romero E, Regüés D (2010) Geomorphological dynamics of subhumid mountain badland areas - weathering, hydrological and suspended sediment transport processes: A case study in the araguás catchment (central pyrenees) and implications for altered hydroclimatic regimes. *Prog. Phys Geog* 34(2):123–150
- Navas A, Gaspar L, López-Vicente M, Machín J (2011a) Spatial distribution of natural and artificial radionuclides at the catchment scale (South Central Pyrenees). *Radiat Meas* 46(2):261–269
- Navas A, Gaspar L, Quijano-Gaudes L, López-Vicente M, Machín J (2011b) Patterns of soil organic carbon and nitrogen in relation to soil movement under different land uses in mountain fields (South Central Pyrenees). *Catena*, doi: 10.1016/j.catena.2011.05.012

- Navas A, Machín J, Beguería S, López-Vicente M, Gaspar L (2008) Soil properties and physiographic factors controlling the natural vegetation re-growth in a disturbed catchment of the Central Spanish Pyrenees. *Agroforest Syst* 72(3):173–185
- Navas A, Machín J, Soto J (2005) Assessing soil erosion in a Pyrenean mountain catchment using GIS and fallout ^{137}Cs . *Agr Ecosyst Environ* 105:493–506
- Navas A, Valero-Garcés BL, Gaspar L, Machín J (2009) Reconstructing the history of sediment accumulation in the Yesa reservoir: an approach for management of mountain reservoirs. *Lake Reserv Manage* 25(1):15–27
- Navas A, Valero-Garcés B, Machín J (2004) An approach to integrated assessment of reservoir siltation: the Joaquín Costa reservoir as a case study. *Hydrol Earth Syst Sc* 8(6):1193–1199
- Owens PN, Collins AJ (2006) *Soil Erosion and Sediment Redistribution in River Catchments: Measurement, Modelling and Management*. CAB International, Wallingford, UK.
- Papio C, Trabaud L (1991) Comparative study of the aerial structure of five shrubs of Mediterranean shrublands. *Forest Sci* 37(1):146–159
- Ramos MC (2006) Soil water content and yield variability in vineyards of Mediterranean northeastern Spain affected by mechanization and climate variability. *Hydrol Process* 20(11):2271–2283
- Renard KG, Foster GR, Weesies GA, McCool DK, Yoder DC (1997) *Predicting Soil Erosion by Water: A Guide to Conservation Planning with the Revised Universal Soil Loss Equation (RUSLE)*. Handbook #703. US Department of Agriculture, Washington, DC
- Rivera S, Ferreira OI, de Anguita PM, Espinal FM (2011) Soil and economic loss evaluation on small hillside farms in the central mountains of Honduras. *J Sustain Forest* 30(1):57–78
- Schäuble H, Marinoni O, Hinderer M (2008) A GIS-based method to calculate flow accumulation by considering dams and their specific operation time. *Comput Geosci* 34:635–646
- Smets T, López-Vicente M, Poesen J (2011) Impact of subsurface rock fragments on runoff and interrill soil loss from cultivated soils. *Earth Surface Processes and Landforms*, doi: 10.1002/esp.2220
- Stagliano N, Argenti G, Pardini A, Piemontese S, Talamucci P (1999) Restoring of an abandoned farm with trees and herbaceous resources by fallow deer grazing. *Options Méditerranéennes* 39:265–268
- Valero-Garcés BL, Navas A, Machín J, Walling D (1999) Sediment sources and siltation in mountain reservoirs: a case study from the Central Spanish Pyrenees. *Geomorphology* 28(1-2):23–41
- Vanmaercke M, Poesen J, Maetens W, de Vente J, Verstraeten G (2011) Sediment yield as a desertification risk indicator. *Sci Total Environ* 409(9):1715–1725
- Verheijen FGA, Jones RJA, Rickson RJ, Smith CJ (2009) Tolerable versus actual soil erosion rates in Europe. *Earth-Sci Rev* 94(1-4):23–38
- Wang B, Liu GB, Xue S, Zhu B (2011) Changes in soil physico-chemical and microbiological properties during natural succession on abandoned farmland in the Loess Plateau. *Environ Earth Sci* 62(5):915–925
- WCD – World Commission on Dams (2000) *Dams and development: a new framework for decision making*. Earthscan Publications Ltd. (now part of the Taylor & Francis Group Ltd), Oxford, UK

Fig. 1 Geographic situation of the Arnás catchment in the province of Huesca (Spain) (a) and within the drainage basin of the Yesa reservoir (b); aerial image of the study area with location of the sampling points (c) and picture of the gauging station at the outlet of the catchment (d)

Fig. 2 Current land uses and land uses under the scenarios of plant recovery and scrub clearance

Fig. 3 Annual values of precipitation and of the rainfall and runoff erosivity (R) factor (a), and maps of the soil erodibility (K) (b) and topographic (LS) (c) factors in the Arnás catchment

Fig. 4 Maps of soil erosion (RUSLE model) and of the sediment delivery ratio (SEDD model) estimated for the different scenarios of land uses

Fig. 5 Predicted versus measured total annual sediment yield at the outlet of the Arnás catchment during eight water years. The significance of R^2 is $p < 0.05$

Fig. 6 Changes in soil erosion, sediment delivery (SDR) and yield (SSY) in the Arnás catchment for the different land use scenarios. Much less erosion and yield: lower than 2 Mg / ha yr; less erosion and yield: between 2 and 0.5 Mg / ha yr; lightly less erosion and yield: between 0 and 0.5 Mg / ha yr; less delivery: lower than 0.05; lightly less delivery: between 0 and 0.05

Plant succession vs. Current scenario

Scrub clearance vs. Current scenario

Table 1 Specific sediment yield (SSY; Mg ha⁻¹ yr⁻¹) and total sediment yield (SY; Mg yr⁻¹) calculated for the different land uses in the Arnás catchment under the different scenarios of land management. %*: percentage from total area; %**: percentage from total exported sediments

Land use	Current scenario				Vegetation recovery scenario				Scrub clearance scenario			
	Area	SSY	SY		Area	SSY	SY		Area	SSY	SY	
	%*	Mean	Value	%**	%*	Mean	Value	%**	%*	Mean	Value	%**
Wood good	19.4	0.4	23.3	13.1	65.8	0.4	65.6	43.8	19.2	0.4	23.9	13.1
Brush good	45.7	0.3	44.2	24.8	29.0	0.3	24.2	16.2	14.6	0.2	6.5	3.6
Brush poor	28.1	0.3	25.9	14.5	3.7	0.9	9.0	6.0	22.8	<0.1	2.3	1.3
Pasture good	3.7	1.3	13.9	7.8	----	----	----	----	3.7	1.4	14.5	8.0
Wood-grass good	0.7	0.1	0.3	0.1	----	----	----	----	0.1	<0.1	<0.1	<0.1
Meadow	0.8	0.4	0.8	0.5	----	----	----	----	38.0	0.8	79.7	43.8
Bare soil	0.6	12.2	19.0	10.7	0.6	13.1	20.4	13.7	0.6	15.3	23.6	13.0
Alluvial deposit	0.2	51.4	29.3	16.4	0.2	8.7	5.0	3.3	0.2	8.4	4.9	2.7
Unpaved trail	0.7	10.9	21.7	12.1	0.7	12.7	25.3	16.9	0.7	13.4	26.6	14.6
Lots	<0.1	4.4	0.1	<0.1	<0.1	5.9	0.1	0.1	<0.1	6.6	0.1	0.1