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Soil loss at sub-basin scale in the Alt Penedès – Anoia region in 2000s

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Abstract

This paper evaluates soil loss due to water erosion in an area of 32,362 ha with a predominant land use of vineyards (Alt Penedès – Anoia region, Catalonia, Spain). The Soil and Water Assessment Tool (SWAT) was used incorporating daily climatic data for the period 2000-2010 and also detailed soil and land use maps. Particular attention was given to the USLE cover and management factor (C factor) of vineyards, with a minimum value of 0.15 being determined for this crop. The model was calibrated using daily flow data for the year 2010, which yielded satisfactory results. Even so, significant differences were obtained on days with high intensity rainfall events, when the model overestimated runoff and peak discharge. In these vineyards, the simulated average soil losses per sub-basin ranged between 0.13 and 9.73 Mg ha^{-1} y^{-1}, with maximum values of between 26.32 and 42.60 Mg ha^{-1} y^{-1} registered in fine-loamy soils developed on unconsolidated Tertiary marls. Other findings were related to problems associated with SWAT calibration under Mediterranean conditions characterised by major climate variability and high intensity rainfall events.

Keywords: Soil erosion, vineyards, SWAT, regional scale

INTRODUCTION

The intensification and mechanization of agriculture in recent decades has been identified as one of the main causes of the acceleration of erosion processes (Caraveli 2000; Cerdà *et al.*, 2009; García-Ruiz, 2010; Li *et al.*, 2012; Olang *et al.*, 2012). The Alt Penedès - Anoia region (Catalonia, Spain), which forms part of the Penedès Designation of Origin for wine and cava, offers a clear example of this problem. Since 1991, various types of erosion processes and their consequences have been studied in this area at the field scale. These studies have measured such different phenomena as soil loss due to sheet and rill erosion and ephemeral gully development (Meyer & Martínez-Casasnovas, 1999; Martínez-Casasnovas *et al.*; 2002), as well as evaluating the on-site effects of concentrated flows resulting from high intensity rainfall (Martínez-Casasnovas *et al.,* 2005; Ramos & Martínez-Casasnovas, 2010a).

Other works have specifically assessed nutrient losses due to runoff in vineyards and their relationship with rainfall erosivity (Ramos & Martínez-Casasnovas, 2006) and extreme precipitation events (Ramos & Martínez-Casasnovas, 2009). They revealed maximum N and P losses of 8.5 and 8.4 kg ha^{-1} respectively, which were equivalent to between 3.9% and 7.1% of annual N intakes and between 16.9% and 33.81% of annual P intakes and therefore supposed economic losses. Soil losses and variations in soil water content, which were influenced by field reorganisation and land levelling, were also studied in other, more specific, research (Ramos & Martínez-Casasnovas, 2006; 2007; 2010). Water deficits, which are frequent in this area, increased in levelled plots, even in wet years. Differences in vine grape yield of up to 53% were observed between wet and dry years, while average differences of about 15% were observed between levelled and non-levelled areas within a given year (Ramos & Martínez-Casasnovas, 2010b).

In this area, erosion has been scientifically recognized as a significant problem and one conditioned by tectonic processes at the regional scale (Gallart, 1981; Martínez-Casasnovas & Ramos, 2009b). This is mainly determined by the local lithology (marls and unconsolidated Tertiary sandstones), rainfall characteristics (with frequent high intensity events in spring and autumn, including rainfall of > 100 mm h^{-1} for 5 min periods), and land uses (with reduced vegetation cover in the form of vineyards and olive trees). To date, however, there has been no regional approach to land use planning in this area. There is therefore a need to use modelling tools to analyse the effects of changes in land use, management practices and climatic variation on non-point pollution problems at the regional scale.

Several attempts have been made to apply erosion models at the regional scale. Average erosion rates have been estimated for entire catchment areas using either the universal soil loss equation (USLE) (Wischmeier & Smith, 1978) or its revised version (RUSLE) (Renard *et al.,* 1991). In this respect, the Soil and Water Assessment Tool (SWAT) is a well-known model which is much used to quantify soil erosion and sediment yield (Arnold *et al.,* 1998) at the regional scale. SWAT utilizes the modified USLE (MUSLE) (Williams, 1975) to estimate soil losses, using runoff as an indicator of erosive energy (Neitsch *et al.,* 2011). There is then no need for a delivery ratio to be used, as in the case of applying USLE or RUSLE. Moreover, to date, SWAT has been applied in few regions with vineyards as one of the main crops. The most significant of these cases is the work of Potter & Hiatt (2009). These authors introduced into the land-cover SWAT plant database a specific cover class for vineyards to distinguish them from orchards, also adjusting the USLE C factor to simulate the effects of increased ground cover.

Within this context, this paper evaluates water flow generation and soil loss due to water erosion in an area of 32,362 ha of the Alt Penedès – Anoia region, with a predominant vineyard use. The modelling tool SWAT (Arnold *et al.,* 1998) was used for this purpose and for regional planning.

MATERIAL AND METHODS

Study area

The study area (32,362 ha) is part of the Alt Penedès and Anoia regions (Catalonia, Spain) (Figure 1). It forms part of the Vallès – Penedès Tertiary Depression, which is mainly covered by unconsolidated sedimentary rocks (marls, sandstones and conglomerates). The predominant soils belong to the Soil Taxonomy subgroups (Soil Survey Staff, 2010) *Typic Calcixerepts* and *Typic Xerorthents* (*Haplic Calcisols* and *Haplic Regosols*) (IUSS Working Group WRB, 2007).

The climate is Mediterranean, with average annual rainfall of 550 mm (ranging between 380 mm and 900 mm) and frequent high-intensity events in spring and autumn (> 100 mm h^{-1}). The average rainfall erosivity factor (R = kinetic energy x maximum intensity in 30-min period) is about 1200 MJ mm ha^{-1} h^{-1} y^{-1}. However, in the decade 2000-2010, some of these values ranged between 1350 – 3900 MJ mm ha^{-1} h^{-1} y^{-1} based on 1-min intervals (Ramos & Martínez-Casasnovas, 2009). The main agricultural land use in the region is grape production for high quality wine (*Vitis vinifera*), which occupies 30.9% of the land. Although vineyards are the most extended crop in this catchment, other crops like cereals (mainly barley, *Hordeum vulgare*) (8.3%) and grasslands (2.8%) are also present. The arboreal vegetation mainly consists of *Pinus halepensis*, *Quercus ilex* and *Quercus faginea* (31.2%), while other scrubland species are also present (10.8%).

In this area, deep ploughing (0.6-0.7 m) before vine planting is common to favour root penetration (Martínez-Casasnovas & Ramos, 2009). After the plantation is established, the soil is usually maintained free of weeds with cultivator tillage several times during the growing season, to avoid competition for water. Land levelling has also been a frequent practice in order to create more easily machineable plots. Studies conducted in this region have reported significant changes in soil properties after levelling operations (Ramos & Martínez-Casasnovas, 2006). Another related problem is an increase in soil erosion, with a 26.5% increase in average annual soil loss associated with land transformation and the removal of traditional broad terraces (Martínez-Casasnovas & Ramos, 2009).

Model interface and input data

Water flow generation, runoff and soil loss in the study area were modelled using the SWAT (Soil and Water Assessment Tool) (Arnold *et al.,* 1998). The ArcSWAT

2009.93.7b extension for ArcGIS 9.3 was used as the SWAT interface. SWAT calculates these parameters for hydrological response units (HRU). These are produced by spatially overlapping soil, land use and slope degree data in each sub-basin of the study area. This was done using the following input data: a 15x15 m digital elevation model produced by the Cartographic Institute of Catalonia; a soil map (1:25,000) of the Penedès area, produced by the Ministry of Agriculture (Generalitat de Catalunya) (DAR, 2008); and the land cover map of Catalonia (1:5,000), produced by the Centre for Ecological Research and Forestry Applications (3rd edition).

In the first steep, a minimum threshold of 75 ha was considered in the definition of sub-basins. This threshold was established to avoid very large sub-basins for further soil conservation planning purposes. This allowed the generation of 231 sub-basins within the 32,362 ha, with an average area of 140 ha per sub-basin.

The soil map of Catalonia available for the study area contained 77 soil series belonging to 37 different soil families (Soil Survey Staff, 1999) (S1). The soil family was the level adopted for introducing data into the SWAT soil database, which included 20 different parameters: number of horizons, hydrological group, porosity fraction, textural fractions, depth, bulk density, available water content, coarse element content, electrical conductivity, organic carbon content, soil erodibility factor and saturated hydraulic conductivity (Neitsch *et al.,* 2010). The soil erodibility factor (K factor) was computed for each soil unit using the equation (1) proposed by (Wischmeier *et al.,* 1971).

$$(1)\quad K_{USLE} = \frac{0.00021 \cdot M^{1.14} \cdot (12 - OM) + 3.25(Csoilstr - 2) + 2.5 \cdot (Cperm - 3)}{100}$$

where M = Sand (%); OM = organic matter (%); Csoilstr = soil structure parameter; Cperm= soil permeability parameter.

The legend of the land use/cover map of Catalonia, with more than 100 categories at its most detailed level of definition, was aggregated to adapt to the crop and urban categories available in the SWAT database. Following this aggregation, the main vegetation types and crops considered were: forest and scrubland (13,573 ha), mainly formed by *Pinus pinea*, *Pinus halepensis*, *Quercus ilex* and *Quercus faginea,* amongst others, and scrubs and brushes in abandoned agricultural fields and border areas between streams, gullies and fields; pasture lands and abandoned fields developing to grasslands (916 ha); almond (*Prunus amygdalus*) and olive tree plantations (*Olea europaea*) (1,541 ha); vineyards producing grapes destined for winemaking (*Vitis vinifera*) (9,984 ha); and winter cereals, mainly barley (*Ordeum vulgare*) (2,679 ha).

Particular attention was given to the USLE cover and management factor (C factor) for the vineyards. SWAT updates the C factor on a daily basis, expressing it as a function of the minimum C factor and the amount of residue on the soil surface (Neitsch *et al.,* 2011). Based on this, Potter & Hiatt (2009) established three different values for the C factor: the default value of 0.1 and values of plus 0.03 and 0.003 to simulate increased vineyard ground cover in California. Also Novara *et al.* (2011) observed C factor values between 0.18 and 0.23 in Sicilian vineyards with different cover crops between rows.

These values were not directly applicable to the case study area because the local farmers do not maintain herbaceous vegetation between rows as this would increase competition for scarce water resources. Based on the vegetation cover observed in the vineyards in each year, we applied the C factor values described in Table 1: the minimum C factor value was 0.15 and the maximum was 0.35. This last value is of the same order of magnitude as reported by Lieskovský & Kenderessy (2012) in Slovakian vineyards (0.389). For the rest of the land uses/covers, the minimum C factors adopted were those obtained from the SWAT Crop database.

In addition, to soil and land use, five slope percentage classes were considered for the definition of the HRU: 0 – 7%, 7 – 15%, 15 – 25%, 25 – 45% and >45%. Other inputs considered included daily climatic data for the period 2000-2010 obtained from four observatories belonging to the Meteorological Service of Catalonia (Els Hostalets de Pierola, Sant Martí Sarroca, la Granada and Font-Rubí). The data included maximum and minimum temperatures, precipitation, relative humidity, solar radiation and wind speed. Table 2 summarizes the annual rainfall registered at the four observatories during the study period and also the weighted average for the study catchment.

Model calibration and application

Because of the high rainfall variability observed during the last two decades in the study area (Ramos *et al.*, 2012), confirmed as well by Reiser & Kutiel (2011), it was difficult to select a representative or average year for calibration purposes. The year 2010 was selected to calibrate the model because it presented the typical rainfall distribution pattern of the Mediterranean region along the year. In addition, it could be also representative of future climate characteristics due to the presence of extreme events and the high irregular rainfall distribution. The water flow results were then calibrated using daily flow data for the year 2010 measured at the Sant Sadurní d'Anoia gauging station (Catalan Water Agency); this covered 41% of the study area.

The hydrographs generated from the SWAT for the calibration year were compared with the data obtained from the gauging station. Calibration was only possible for daily flow. Runoff and sediment production were not calibrated due to a lack of data for calibration at the control station.

For calibration, a sensitivity analysis was conducted based on an LH-OAT process (Latin Hypercube - One factor At a Time) (van Griensven *et al.,* 2006). Following the application of the input data, the model parameters were changed according to their degree of influence on known results.

The following indicators were used to evaluate the accuracy of the model: the coefficient of determination (R^2); the coefficient of efficiency, or Nash-Sutcliffe Efficiency (NSE) (Nash & Sutcliffe, 1970); the percentage of bias (PBIAS) (Gupta *et al.,* 1999); and the mean square error rate (RSR) (Equations 2, 3, 4 and 5). The resulting values for these indicators were evaluated according the criteria proposed by Moriasi *et al.* (2007).

(2) $$R^2 = \left(\frac{\sum_{i=1}^{n}\left(Y_m - \overline{Y}_m\right)\left(Y_s - \overline{Y}_s\right)}{\sqrt{\sum_{i=1}^{n}\left(Y_m - \overline{Y}_m\right)^2 \sum_{i=1}^{n}\left(Y_s - \overline{Y}_s\right)^2}} \right)^2$$

(3) $$NSE = 1 - \frac{\sum_{i=1}^{n}\left(Y_m - Y_s\right)^2}{\sum_{i=1}^{n}\left(Y_m - \overline{Y}_m\right)^2}$$

(4) $$PBIAS = \frac{\sum_{i=1}^{n}\left(Y_m - Y_s\right) * 100}{\sum_{i=1}^{n}\left(Y_m\right)}$$

(5) $$RSR = \frac{\sqrt{\sum_{i=1}^{n}\left(Y_m - Y_s\right)^2}}{\sqrt{\sum_{i=1}^{n}\left(Y_m - \overline{Y}_m\right)^2}}$$

where Y_m is the measured value and Y_s is the value simulated with SWAT, and $\overline{Y}_m$ is the mean of the measured values of each of the parameters analysed.

Once a good fit for water flow estimations had been obtained for the calibration year, SWAT was applied to predict water flow for the period 2002 - 2009. It is worth underlining that the data for the years 2000 and 2001 were used to give a period of adjustment for beginning the water cycle model (Zhang *et al.,* 2008).

Sediment yield loads were directly assessed from SWAT outputs since the suspended sediment concentration was not measured at the control gauging station. The results presented should therefore be considered for comparative purposes but not in absolute terms.

RESULTS

Model calibration

Table 3 shows how the model parameters varied according to their degree of influence on the water flow estimation at the control station for the calibration year (2010). The method best suited for calculating runoff and evapotranspiration was the one based on the calculation of the daily curve number using daily crop or vegetation growth. The based on the antecedent soil moisture produced excessively high estimates. In addition, the value of the plant ET curve number coefficient (CNCOEF) was set to 0.5 (values between 0.5 and 2 are permitted), to limit surface runoff and to give a better fit. Another parameter used to control surface runoff was the soil evaporation compensation factor (ESCO), which was adjusted to 0.115 in order to reduce runoff.

The model also required the adjustment of the parameters that monitored the subsurface flow into the aquifer from the shallow aquifer to the root zone and from the aquifer to the drainage network. The GW_REVAP and REVAPMN parameters were therefore modified to 0.199 and 2.775, respectively. The first of these values modifies monitors the movement of water between the aquifer and the root zone. Values of GW_REVAP of close to 0.2 (as in this case) allow the transfer of water to the root zone and increase evapotranspiration (Neitsch *et al.,* 2011). Along with these parameters, other factors that monitor the groundwater flow to the drainage network were also modified: ALPHA_BF and Ch_K(2). The former (ALPHA_BF) is a direct index of groundwater flow response to changes in recharge. Its low value indicated a slow soil response in the study area in terms of aquifer recharge (Neitsch *et al.,* 2011). With respect Ch_K (2), this value indicated only minor flow losses from the main drainage channels during aquifer recharge.

Figure 2 shows the results of water flow calibration at the Sant Sadurní d'Anoia gauging station at daily time scale in the year 2010, following sensitivity analysis. According to the statistical indicators (NSE = 0.51, R^2 = 0.74, PBIAS = 15.29 and RSR = 0.69), the fit between simulated and observed data was good for R^2 and satisfactory for NSE, PBIAS and RSR (Moriasi *et al.,* 2007). The largest differences were observed on days on which high rainfall totals were registered. In these cases the model tended to overestimate runoff and, as a consequence, to increase the simulated peak discharge.

Water flow estimation for the period 2002-2009

According to the goodness of fit indicators of the simulated data for the period 2002-2009 (Table 4), only 2002 could be considered satisfactory (NSE 0.53, R^2 0.57 and RSR 0.69), and good with respect to PBIAS (10.86%). 2008 presented either satisfactory or good fits, according to the R^2 and PBIAS values, but not with respect to NSE or RSR. 2004 was the year with the poorest results.

In 2004, there was a marked difference between the observed and simulated data, both in terms of the base and peak flows (Figure 3). However, in both 2008 and 2009, although the statistical indicators were not good, there was a high level of concordance between the observed and simulated hydrographs, but with runoff being overestimated in the case of high rainfall events (Figure 3).

Soil loss simulation

Figure 4 shows the simulated spatial and temporal soil losses in the study area for the period of analysis. The years in which the greatest soil losses were generated were 2002 and 2010, followed by 2009 and 2006. The fact that the spatial distribution pattern was different was attributed to the spatial variability of precipitation associated with the location of the four weather stations used.

Table 5 shows the average soil loss per land cover unit simulated by SWAT for the period 2002 – 2010. In the case of vines (30.9% of the total land area), differences in soil losses were observed between years with different climatic characteristics. In the years analysed, the greatest soil losses occurred in 2002 and 2010, with average values of 5.28±5.43 and 9.73±6.67 Mg ha^{-1} y^{-1}, respectively. However, the maximum values recorded were 42.60 and 26.32 Mg ha^{-1} y^{-1}; these occurred in HRU fine-loamy soils that had developed on unconsolidated Tertiary marls. 2002 and 2010 were two of the wettest years in the series, with weighted averages of 611 mm and 689.7 mm of precipitation, respectively. The main differences between them and the other wet years in the series (2004 and 2008, with 616 mm and 668.6 mm respectively) were the distribution of rainfall throughout the year and the soil water content in the previous year. For example, in 2004, the rainfall was particularly concentrated in spring rather than in autumn. In 2008, the previous years (2005-2007) could be considered dry, with annual rainfall totals of 405.5 mm, 343 mm and 464.8 mm, respectively, which caused an average fall in soil water content across the catchment area of up to 33.7 mm. The 2008 rainfall would therefore have mainly filled the water retention capacity of the soils (to 73.9 mm) rather than have generated much runoff.

DISCUSSION

Model calibration and application

Model calibration results were considered as satisfactory, with main differences concentrated on days on which high rainfall totals were registered. This problem is referred to in the literature and most frequently occurs when base flows are low (Potter & Hiatt, 2009). To compensate for the excess runoff associated with major precipitation events, the CN2 factor (curve number for moisture condition II soil) can be reduced (Piniewski & Okruszko, 2011). The values of the other parameters that control the flow generation process can also be reduced (Rostamian *et al.,* 2008). These same authors also reported great uncertainty concerning the calibration of extreme events because of the excess runoff estimated by SWAT.

Differences between water flow estimates, after applying the model to other years of the series (Figure 3), and real measured values were also reported by other researchers (Rostamian *et al.,* 2008). Such marked differences could have been due to the location and number of meteorological observatories in relation to the study area, as they would not have been able to record some local high-intensity rainfall events, or they could have been the result of the intra-annual variability of rainfall during the calibration year.

In this respect, Tuppad *et al.* (2010) highlighted the variability of responses in SWAT hydrological modelling in relation to the spatial resolution of the precipitation data. Ramos & Martínez-Casasnovas (2006) and Ramos *et al.* (2012) confirmed the changes in the pattern of rainfall distribution over the year in the study area and a trend towards a greater concentration of rainfall events. This variability may, however, change the pattern of the hydrological response of the basin. This, in turn, could lead to the recommendation that years with similar rainfall distribution patterns should be grouped together. Then, different calibration parameters should be applied to each group in order to improve water flow simulations. This hypothesis was confirmed by the obtained results. The best fits were observed for years in which rainfall amount and distribution were more similar to that of the calibration year. This can be observed in the better fit between the observed and simulated water flows in the years 2002 and 2008, which were also wet years, like 2010 (the calibration year) and with a similar distribution pattern. Those were the years more important from the erosion point of view, because of the high soil losses generated. For dry years a specific calibration should be done separately.

Soil loss simulation

The area with the most frequent and important problems of soil loss was the north-east sector. This was the area in which major problems of erosion had been described in previous studies (Meyer & Martínez-Casasnovas, 1999; Martínez-Casasnovas *et al.,* 2009). This area has the combination of soils developed on poorly consolidated marls of Tertiary origin, vineyards as the predominant land use, and moderate slopes (10-15%), all of which increase the risk of soil loss. Furthermore, recent land transformations have eliminated traditional protective measures such as bench terraces, contour farming and broadbase and/or drainage terraces and favoured soil erosion (Martínez-Casasnovas & Ramos, 2009). Another study conducted by Farguell & Sala (2005) confirmed that the southern part of the Anoia river basin, which presented the highest soil losses in 2002, was particularly affected by high intensity rainfall events. This contributed more to the sediment load in the river than the northern part, where rainfall fell with less intensity.

Regarding soil loss per land use type, the main soil losses in the catchment were produced in vineyards. The average soil loss rates for the whole study area were slightly above the range reported by Kosmas *et al.* (1997), of between 0.67 and 4.6 Mg ha^{-1} y^{-1}, relating to soil losses in a number of Mediterranean countries (Portugal, Spain, France, Italy and Greece). However, other studies have cited higher erosion rates: up to 7-21 Mg ha^{-1} y^{-1} in Alsatian vineyards (Schwing, 1978), 35 Mg ha^{-1} y^{-1} in the Mid Aisne (France) (Wicherek, 1991), 30 Mg ha^{-1} in the vineyards of Navarra (Casalí *et al.,* 2009) and 8-36 Mg ha^{-1} y^{-1} in the Languedoc region (France) (Paroissien *et al.,* 2010). In other studies addressed to measure the effects of different tillage methods and/or vegetation cover in reducing soil losses in vineyards, also high erosion rates are reported. For example, Novara *et al.* (2011), in Sicilian vineyards, found that conventional tillage yielded on average 102.2 Mg ha^{-1} y^{-1}. In that case, different cover crops reduced erosion by 39.6 to 69.8%. Lieskovský & Kenderessy (2012), who evaluated the effects of tillage, hoeing, rotavating and grass cover in Slovakian vineyards, observed rates of between 0.28 and 19.1 Mg ha^{-1} y^{-1}, being the first the average of soil loss in vineyards with grass cover and the last the average under conventional tillage. In the same study area, other authors have reported higher rates of soil loss than those estimated in the present study: 18-22 Mg ha^{-1}, measured only during the period from September to November (Ramos & Porta, 1997) and rates of between 15 and 25 Mg ha^{-1} y^{-1}, from vineyard plots (Ramos & Martínez-Casasnovas, 2009). However, the land levelling and the management practices carried out in the new vineyards are incrementing soil erosion rates (Ramos and Martínez-Casasnovas, 2010a).

Soil loss estimates in rain fed fruit-tree orchards (almond and olive trees) (2.88 Mg ha^{-1} y^{-1} in average) were lower as compared with soil losses in olive orchards reported by Gómez *et al.* (2003): 8.5 Mg ha^{-1} y^{-1} with the herbicide treatment; 4.4 Mg ha^{-1} y^{-1} with conventional tillage; but higher than under herb cover (1.2 Mg ha^{-1} y^{-1}); or Van

Wesemael et al. (2006): 5.5 Mg ha^{-1} y^{-1} (net soil loss) in almond orchards ploughed several times per year. Kosmas *et al.* (1997) and Taguas *et al.* (2010) reported even lower soil loss values in Mediterranean olive tree plantations (0 – 1.0 Mg ha^{-1} y^{-1}). In those cases, grass covers and plant residues drastically reduced soil losses. In the present case study area, olive and almond tree plantations remain with bare soil most part of the year, but they are usually located in the most stable geomorphological positions. This fact could influence the moderate soil losses of the rain fed tree plantations in relation to other study areas as well as in relation to vineyards.

Other agricultural lands in the study area, cultivated with cereals (mainly winter barley), produced average soil losses of 0.98 Mg ha^{-1} y^{-1}. These losses are in the upper limit of the range reported by Kosmas *et al.* (1997). As well as these authors report, there was a clear trend of increasing soil losses in this land use with increasing annual precipitation.

This review of rates of soil loss, and in particular of the rates registered in the vineyards of the study area, could suggest that SWAT estimated default soil losses. However, the rates measured in previous research works mainly referred to measures made at the plot scale using various different methodologies (USLE, sediment traps and vine-stock benchmarks), and not at sub-basin or catchment scale. This is in accordance with different works that have recently addressed the issue of scale and erosion, which is one of the most poorly understood components of the catchment sediment system (Cerdà *et al.*, 2013; Rodríguez-Blanco *et al.*; 2013, Sadeghi *et al.*, 2013). As example, the research by Rodríguez-Blanco *et al.* (2013) reveal that soil erosion rates measured at one scale are not representative of sediment yield at another more generalized scale. Soil losses measured at the Corbeira catchment outlet (NW Spain) (28.53 Mg) were indeed more than 5 times lower than the measured on the fields (140.5 Mg). These differences are attributed to sediment deposition along the route from field to catchment outlet. Sadeghi et al. (2013) also suggest that measures in small plots, although are practical for calibration of models, may not result in accurate watershed-scale estimates of runoff and erosion; and upscaling of results from small plots needs special considerations.

Despite these differences, the soil losses estimated by SWAT could be employed for comparative purposes, rather than being considered in absolute terms. This would be useful when prioritising sub-basins for the adoption of soil conservation measures. In this respect, the average soil loss at the sub-basin scale calculated for the period 2002-2010 minimised the effects of spatial and temporal variability from year to year. This proved useful for establishing the prioritization of soil conservation measures within the catchment area.

SWAT is a powerful model which includes not only soil and climate characteristics but also land use management practices. It can provide valuable information for planning purposes. One of the main difficulties for their application in a Mediterranean environment is the high variability of climate characteristics. This made necessary the use of different parameters to find a better fit of the model. In addition, the daily scale results are not always suitable to estimate the hydrological response to the common high intensity rainfall events that occur in a short time. Nevertheless, the application of the model, for conditions where more erosion in expected, could give satisfactory results.

CONCLUSIONS

The present work constitutes a new step in research carried out in the Penedès vineyard region to map soil loss at the regional scale for land use planning purposes. In this respect, the application of the model identified the spatial distribution of the sub-basins that were most affected by erosion.

The average erosion rates were lower than those reported in previous works conducted in this and other study areas. This can be attributed to scale effects in soil erosion, as suggested by other researchers. The most important soil losses from vineyards occurred in autumn and spring. This led to proposals for soil protection throughout the year and for placing specific emphasis on these stations.

Another conclusion that can be drawn from this research is that it is not possible to calibrate the model for individual years and then to standardize the parameters for the whole period of analysis (8-10 years). The different years should be separated into groups according to rainfall distribution patterns and the amounts of rainfall received throughout the year; different calibration parameters should then be applied for each group.

Finally, the present work suggests that, in the Mediterranean region, with high rainfall variability from year to year, the SWAT model could be employed to estimate soil losses for comparative purposes at sub-basin scale. This constitutes a useful tool for prioritizing the areas in which soil and water conservation measures should be established.

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Table 1. Soil cover in vineyard fields in different vegetation periods and the USLE C factor.

Dates	Period name	% arboreal vegetation cover	% herbaceous vegetation cover	USLE C factor
1 Nov – 15 Mar	Dormant period	0 – 10	20 – 30	0.20
15 Mar – 15 May	Bud break to bloom	0 – 40	0 – 10	0.35
15 May – 20 Jul	Bloom to veraison	30 – 40	0 – 10	0.15
20 Jul – 17 Aug	Veraison to harvest	30 – 40	0 – 10	0.15
15 Aug – 1 Nov	Post-harvest	20 – 30	10 – 20	0.18

Table 2. Annual rainfall (mm) registered by the observatories in the period 2000-2010.

Year	Els Hostalets de Pierola	Sant Martí Sarroca	La Granada	Font-Rubí	Weighted average
2000	266.0	437.4	380.0	362.7	361.5
2001	447.8	474.2	371.2	277.4	392.7
2002	612.6	726.2	744.9	445.4	611.0
2003	496.0	664.2	586.8	575.2	542.4
2004	785.5	520.3	504.3	430.8	616.0
2005	365.0	480.3	523.2	339.6	405.5
2006	329.8	357.3	404.7	294.0	343.0
2007	548.0	398.9	387.9	400.4	464.8
2008	751.5	570.9	575.6	625.2	668.6
2009	541.9	499.8	499.6	555.8	532.0
2010	729.4	734.1	571.2	756.2	689.7

Table 3. SWAT parameters adjusted after the sensitivity analysis according to their influence on the water flow estimation.

Modified SWAT parameter	Description	SWAT default value	Parameter used after sensitivity analysis
ICN	Daily curve number calculation method (determines runoff and evapotranspiration)	Soil Moisture	Plant ET
PET Method	Potential evapotranspiration calculation method	Penman-/ Monteith	Hargreaves
CNCOEF	Plant ET curve number coefficient (influences evapotranspiration)	2	0.5
ALPHA_BF	Base flow recession constant (days) (direct index of groundwater flow response to changes in recharge)	0.048	0.008
CANMX (mm)	Maximum canopy storage of rainwater (mm) (modifies infiltration and evapotranspiration)	0	0.412
Ch_K(2) (mm/h)	Effective hydraulic conductivity in the main alluvial cannel (mm h^{-1}) (modifies groundwater and base flow)	0	0.015
EPCO	Plant uptake compensation factor (modifies the water available for infiltration)	0.7	0.009
ESCO	Soil evaporation compensation factor (modifies surface runoff)	0.95	0.115
GW_DELAY (days)	Groundwater delay time (days)	31	31.08
GW_REVAP	Groundwater demand for evapotranspiration. "revap" coefficient (indicates the recharge of the soil unsaturated zone from the shallow aquifer)	0.02	0.199
REVAPMN (mm)	Threshold depth of water in the shallow aquifer for percolation to the deep aquifer to occur (mm) (modifies subsurface flow)	1	2.775
GWQMIN (mm)	Threshold depth of water in the shallow aquifer required for return flow to occur (mm) (modifies subsurface flow)	0	50.04
FFCB	Initial soil water storage expressed as a fraction of water content at field capacity (modifies lateral flow and groundwater)	0	0.6

Table 4. Goodness of fit indicators for the simulated water flow during the period 2002-2009.

Year	Nash-Sutcliffe Efficiency (NSE)	R^2	Percentage of bias (PBIAS)	Mean square error rate (RSR)
2002	0.53	0.57	10.86	0.69
2003	0.20	0.34	35.38	0.89
2004	-0.37	0.49	59.46	1.17
2005	-0.20	0.12	67.84	1.09
2006	-0.26	0.14	41.64	1.12
2007	0.19	0.41	51.41	0.90
2008	-0.64	0.59	1.35	1.28
2009	-0.85	0.40	10.67	1.36

Table 5. Average soil loss per land cover / crop simulated by SWAT for the period 2002-2010 (Mg ha^{-1} y^{-1}).

Year	2002	2003	2004	2005	2006	2007	2008	2009	2010
Forest & Scrubland	0.29	0.19	0.16	0.00	0.20	0.01	0.23	0.82	0.56
Grasslands	4.41	1.14	1.56	0.02	1.60	0.01	0.47	1.63	3.07
Almonds & Olive trees	3.84	1.39	3.29	0.29	1.77	0.25	2.91	2.89	9.27
Vineyards	5.29	4.19	3.89	0.13	3.09	1.14	4.59	5.13	9.73
Winter Barley	1.27	0.48	0.59	0.50	1.01	0.10	0.95	1.20	2.76

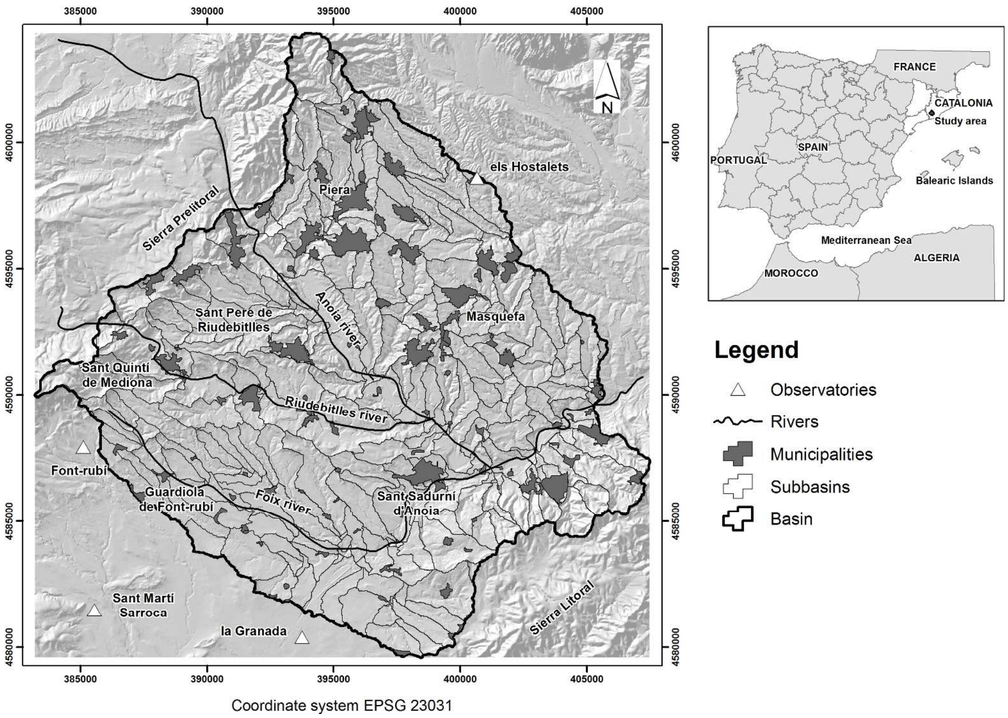
Figure captions

Figure 1. Location of the study area.

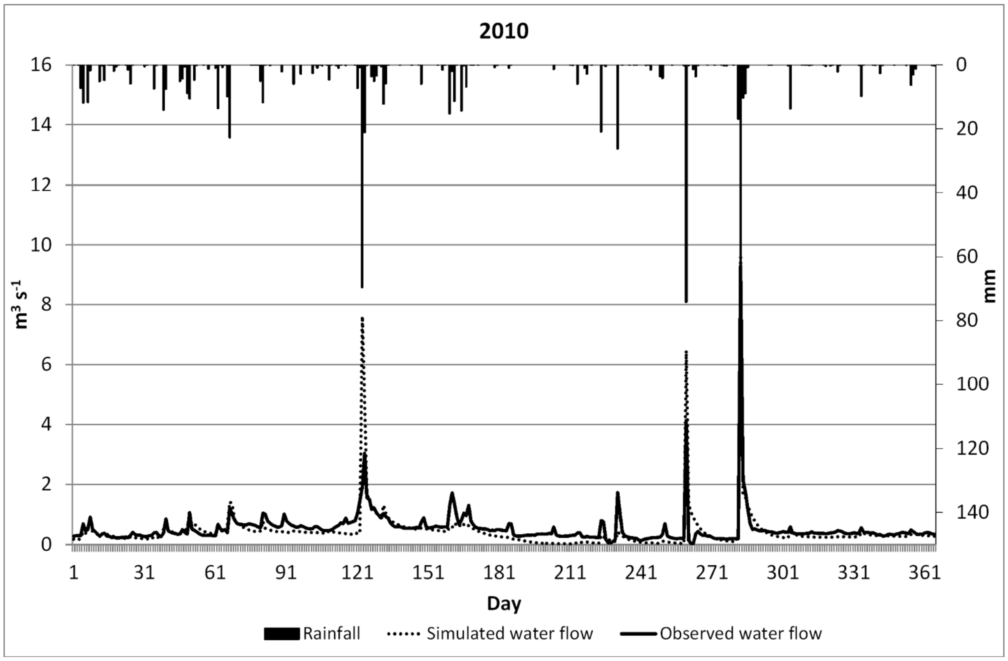
Figure 2. Comparison of daily flows at the gauging station during the calibration period (2010) after the sensitivity analysis.

Figure 3. Hydrographs of selected years in the series 2002-2010 that show different patterns of simulated and observed water flow in the control area: 3a) years 2002 and 2004. 3b) years 2008 and 2010.

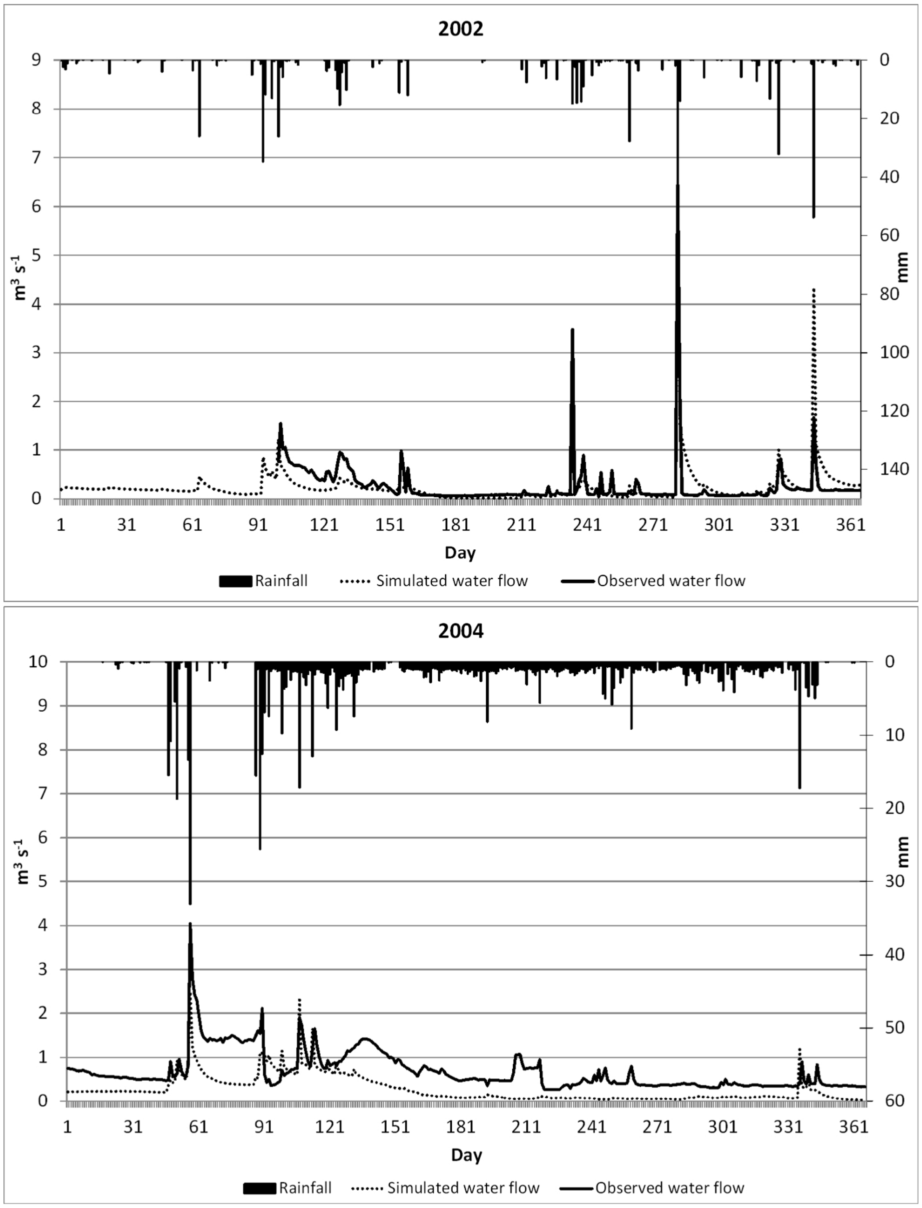
Figure 4. Spatial and temporal comparison of soil loss in the period 2002-2010 as simulated by SWAT.



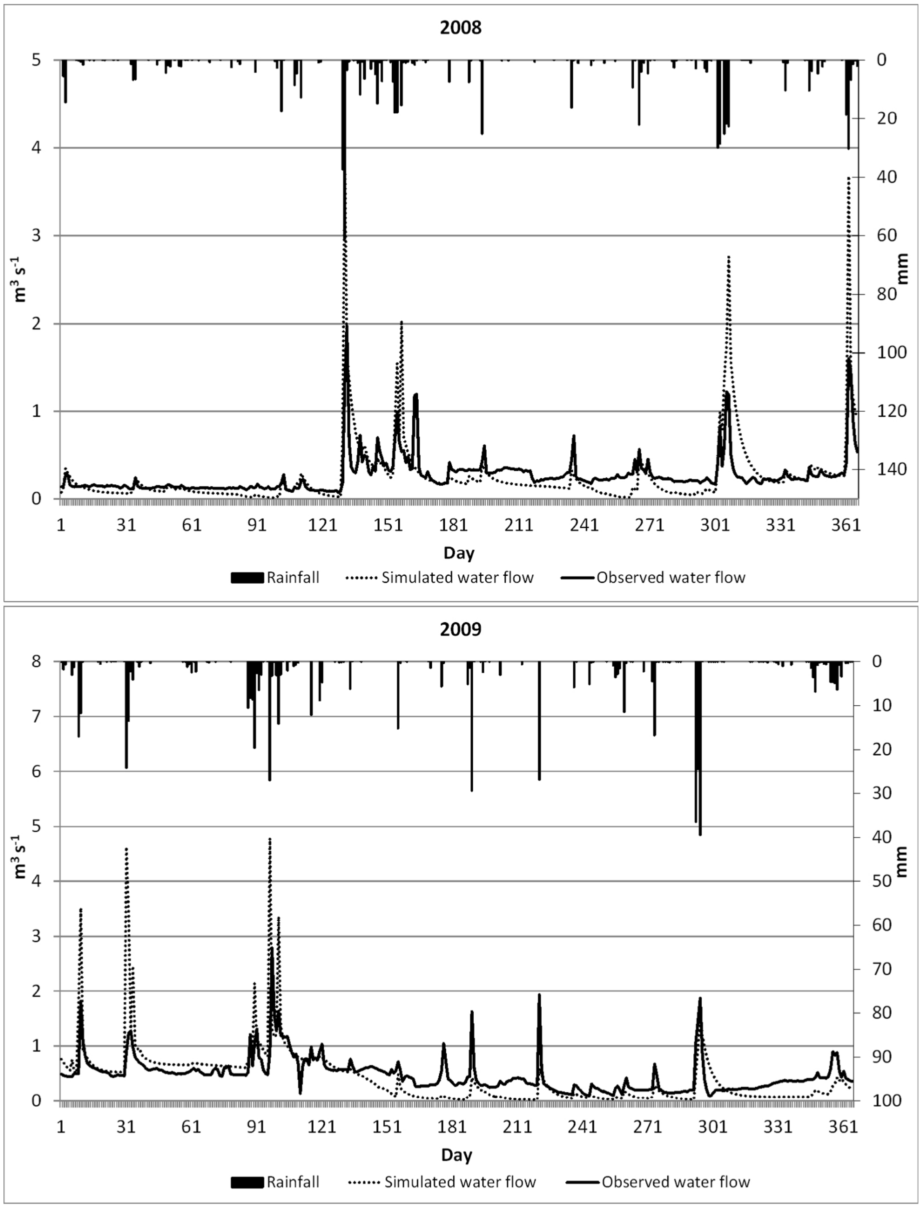
Location of the study area
276x195mm (300 x 300 DPI)



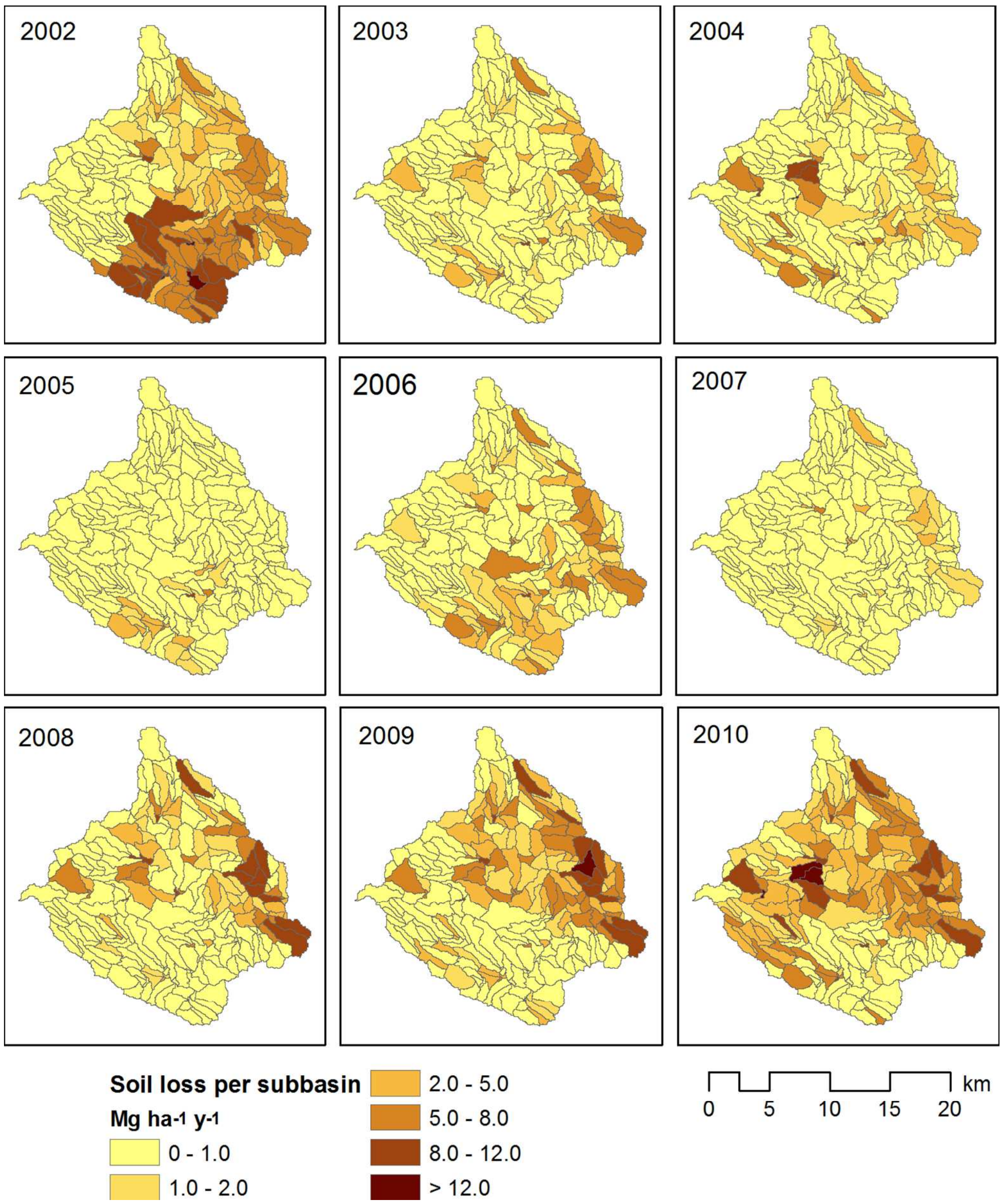
Comparison of daily flows at the gauging station during the calibration period (2010) after the sensitivity analysis
284x186mm (200 x 200 DPI)



Hydrographs of selected years in the series 2002-2010 that show different patterns of simulated and observed water flow in the control area: 3a) years 2002 and 2004
198x260mm (200 x 200 DPI)



Hydrographs of selected years in the series 2002-2010 that show different patterns of simulated and observed water flow in the control area: 3b) years 2008 and 2010
198x260mm (200 x 200 DPI)



Spatial and temporal comparison of soil loss in the period 2002-2010 as simulated by SWAT
192x229mm (200 x 200 DPI)