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A model-based integrated assessment of land degradation by water erosion in a valuable Spanish rangeland

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ABSTRACT

This paper presents an integrated assessment model aimed at evaluating land degradation by water erosion in dehesa rangelands in the Iberian Peninsula. The model is built following the system dynamics approach. The degradation risk is likened to the probability of losing a certain amount of soil within a number of years, as estimated over a great number of stochastic simulations. Complementary indicators are the average times needed to lose different amounts of soil over the simulations. A group of exogenous factors are ranked in order of importance. These factors are mainly climatic and economic and potentially affect soil erosion. Calibration is carried out for a typical dehesa defined over 22 working units selected from 10 representative farms distributed throughout the Spanish region of Extremadura. The degradation risk turns out to be moderate. The importance of climatic factors on soil erosion considerably exceeds that of those linked to human activities.

1. Introduction

Rangelands cover approximately 90,000 km^2 in the central and south-western Iberian Peninsula (Gea-Izquierdo et al., 2006). These rangelands were created from former oak forests, mainly composed by holm oak and cork oak (*Quercus ilex rotundifolia* and *Quercus suber*) as the dominant tree species. By forest thinning, clear-cutting of shrubs, livestock grazing and cultivation, a dynamic mosaic of cover types was created in the form of open or wooded pasturelands and scrublands of variable tree densities. This land use is called dehesa in Spain and montado in Portugal. These systems, most of them held on private ownership (Plieninger et al., 2004), have evolved as an adaptation to poor soils and adverse rainfall conditions that cannot support intensive agricultural use. Their dominant use at present is livestock rearing (sheep, cattle, pigs and goats) and forestry (cork, wood and charcoal). Cultivation is of minor importance and restricted to limited areas with good soil conditions. Similar agro-silvo-pastoral systems can be found in other Mediterranean countries as well (Papanastasis and Mansat, 1996; Pardini, 2007).

Dehesa landscapes are highly interesting from a cultural, economic and environmental point of view (Campos Palacín, 1993; Díaz et al., 1997). However, these valuable rangelands have been suffering localized environmental problems mainly because of overgrazing, undergrazing and/or the lack of tree regeneration (Campos Palacín, 1983; Marañón, 1988; Pinto Correia, 1993; Montero et al., 2000; Pulido et al., 2001; Plieninger et al., 2003; Moreno and Pulido, 2009; Pulido and Picardo, 2010).

Soil degradation by water erosion, the process which is assessed here, is associated with overgrazing. This was favoured by the abandonment of transhumance and the resulting predominance of fencing and permanent grazing over restricted areas. Subsidies paid by the EU's Common Agricultural Policy have also been cited as one of the causes leading to the increase in the number of animals in dehesas (Donázar et al., 1997).

Excessive livestock causes soil degradation, usually understood as a deterioration of the physical, chemical or biological properties of soil that results in the reduction of its potential productive capacity (Imeson, 1988). Indeed, trampling decreases soil porosity thereby reducing water retention capacity and increasing runoff (Gamougoun and Smith, 1984; Mulholland and Fullen, 1991). Also, grazing reduces biomass and thus the protection it provides against erosion, "the sign 'par excellence' of degradation" (van der Leeuw, 1998, p.4).

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Concerns about land degradation and desertification in the Mediterranean are evidenced by the number of research projects that have dealt with the matter over the last few decades. As an illustration, Baartman et al. (2007) report 39 projects specifically focussed on Mediterranean regions. Consequently, scientific publications are also plentiful (e.g. Brandt and Thornes, 1996; Wainwright and Thornes, 2004; Boardman and Poesen, 2006).

In the particular case of dehesas, a soil degradation survey carried out in a large number of farms in the region of Extremadura (SW Spain) evidenced that approximately 23% of them suffered high risk of soil degradation, including soil erosion, while approximately 60% of the region showed high sensibility to degradation processes (Schnabel et al., 2006; Lavado et al., 2009). Sheet erosion is particularly observed on hillsides, gullying takes place at the bottom of small upland valleys and rill erosion occurs mainly in the cultivated areas (Schnabel, 1997; Schnabel et al., 1999).

The Spanish National Action Programme to combat Desertification (SNAPD) (MAGRAMA, 2008) includes dehesas among the vulnerable socio-ecological systems needing integrated assessments of land degradation. In order to make such integration effective, the SNAPD has adopted an assessment methodology based on multidisciplinary models of representative areas, hereafter the SNAPD models. The methodology has been applied so far to five socio-ecological systems. This paper presents the model corresponding to one of these applications and the assessment procedure based on it.

2. Model characterization

The model presented here is an integrated assessment model since it integrates multidisciplinary processes into a single framework aimed at generating useful information for decision-making (Jakeman and Letcher, 2003). Its construction follows the system dynamics approach. A system dynamics model consists in a system of ordinary differential equations that makes a stock-and-flow representation of the studied system. Model's structure as a whole, which is made up of causal feedback loops including nonlinear relationships and delays, constitutes a holistic and easily-overlooked cause of its behaviour (Forrester, 1961; Sterman, 2000).

In what follows, the characterization of the model is deepened mainly by following the useful framework provided by Kelly et al. (2013). These authors also give an updated review of integrated assessment models which illustrates the intense work carried out in this field.

2.1. Purposes of the model

Every SNAPD model has two main and two derived purposes:

Purpose 1: Assessing the risk of degradation that a land-use system is running. **Derived purpose 1:** Early warning about land-use systems that are particularly threatened by degradation.

Purpose 2: Assessing the degree to which different factors would hasten degradation if they changed from the typical values they show at present. **Derived purpose 2:** Evaluating the role of human activities on degradation.

The SNAPD models are not intended for prediction or forecasting, even though they provide outputs over time periods. This is because there are not enough data to validate the models for such purposes (see Section 2.2). Thus, the aim is to get qualitative rather than quantitative outputs, e.g. is degradation risk high or low?, have human activities a strong influence on degradation?

The goal of building useful tools for management and decision-making under uncertainty is kept in mind, though focused on the early stage of identifying degradation problems rather than evaluating different alternatives to cope with them. System understanding and experimentation also constitute important goals of the models, as well as social learning, though this is limited to those cases where the assessment shows a special risk of degradation. Since degradation usually proceeds slowly, stakeholders are not sufficiently aware of its consequences, no matter how they value the land use being degraded. Thus, every SNAPD model is intended as a "means of exploration" (Oxley et al., 2004, p.1008) for helping them to better understand how systems may behave.

2.2. Available data and implications

Time-series data on almost all the endogenous variables of the model presented here are non-existent or too short. Therefore, the model could not be calibrated to reproduce historical observations and its accuracy in this regard could not be tested. Hence prediction and forecasting purposes must be rejected, as already mentioned. Also, data samples are not enough to estimate some relevant relationships by means of regression analysis. In addition, the scarcity of data precludes assessing model performance by means of quantitative tools, such as those reviewed by Bennett et al. (2013). In any case, historical observations are bounded to be insufficient to validate a model whose purpose is to explore long-term states of degradation never observed so far in the studied system.

In facing these difficulties, the system dynamics approach is particularly useful. First, this modelling technique helps in calibrating the model because of its requirement for every parameter to have a real world counterpart (Sterman, 2000). This makes possible to obtain parameter values from the literature or expert opinion when they cannot be estimated from *in situ* data. Therefore, our model is formulated in such a way that every parameter has an easily understandable real world counterpart (see Table 1). To run the model, all these parameters must take values that are representative of the study area. The function of some of these values is to calibrate the functional relationships included in the model. This means that the rate at which the processes represented evolve are ultimately specified by the parameter values. As an illustration, the values of the parameters 'initial topsoil porosity' and 'remaining soil depth at which topsoil porosity is reduced by a half' (Table 1) calibrate the function that distributes porosity along the soil profile (Fig. 2B). In so doing, they contribute to determining the rates at which surface runoff and topsoil bulk density increase as soil is lost, or in other words, the long-term dynamics of erosion.

Second, the system dynamics approach advises on how to qualitatively validate a model (Sterman, 2000). These specifications were thoroughly followed in our case. Thus, the model was tested on: i) its suitability to the purposes of the assessment (see Section 5); ii) its conformance to fundamental formulation principles, e.g. variables are adequately bounded by model structure, not in an *ad hoc* basis; iii) its robustness in facing extreme variations in input conditions (by running exploratory simulations); and iv) its coherence and plausibility. This last point deserves additional comment.

The complete set of parameter values specifies the present or initial state of the modelled dehesa. Model structure is globally coherent and plausible in the sense that its stochastic output will always be consistent with any initial state configured by the parameter values, as long as degradation effects are not significant yet. In other words, while degradation is still unnoticed, the model is structurally bounded to perform well on the direct value comparison method for measuring model's performance, which tests whether the model output shows similar summary statistics to the set of comparison data (Bennett et al., 2013).

Table 1
Parameters.[a]

Name	Definition	Value	Units
$bd_{pasture}$	Topsoil bulk density above which pasture does not grow	1.75	g cm^{-3}
bf_i	Initial number of breeding females (when $GM_{aexp} = \mu_{GMai}$)	0.59	AU ha^{-1}
cv_{PP_y}	Coefficient of variation annual precipitation	0.29	d.u.
$cv_{PR_{mt}}$	Coefficient of variation meat price	0.09	d.u.
$cv_{PR_{sf}}$	Coefficient of variation supplemental feed	0.12	d.u.
$cv_{RC_{wpi}}$	Coefficient of variation runoff coefficient soil at wilting point	0.91	d.u.
dt_{GM_f}	Average length of the delay in forming GM_{fexp}	6	yr
ec_g	Pasture energy content	0.14	FU kg^{-1}
ec_{sf}	Supplemental feed energy content	0.8	FU kg^{-1}
fc	Field capacity (volumetric)	0.262	d.u.
fe_{mr}	Fraction of annual evapotranspiration during mr	0.14	d.u.
fp_{mr}	Fraction of annual precipitation fallen in mr	0.62	d.u.
fs	Ratio of breeding females to stocking rate	0.96	d.u.
gc_f	Normal pasture consumption per breeding female	310.53	kg AU^{-1} yr^{-1}
gp_{hgv1}	Minimum pasture production for GV = 1 without livestock	490	kg ha^{-1} yr^{-1}
gv_{sri}	Fractional pasture cover corresponding to initial stocking rate	0.895	ha ha^{-1}
mr	Months when precipitation > ET_o	5	Month
μ_{ER_i}	Initial mean erosion rate	2.5	tm ha^{-1} yr^{-1}
$\mu_{ER_{bsi}}$	Initial mean bare soil erosion rate	99	tm ha^{-1} yr^{-1}
$\mu_{ET_{oy}}$	Mean annual reference evapotranspiration	112.65	cm yr^{-1}
μ_{GP_h}	Initial mean pasture production per hectare	940.3	kg ha^{-1} yr^{-1}
μ_{MP_f}	Mean meat production per breeding female	337.95	kg AU^{-1} yr^{-1}
μ_{PP_y}	Mean annual precipitation	494.91	mm yr^{-1}
$\mu_{PR_{mt}}$	Mean meat price	2.24	€ kg^{-1}
$\mu_{PR_{sf}}$	Mean price of supplemental feed	0.28	€ kg^{-1}
$\mu_{RC_{fci}}$	Initial mean runoff coefficient soil at field capacity	0.215	cm cm^{-1}
$\mu_{RC_{wpi}}$	Initial mean runoff coefficient soil at wilting point	0.062	cm cm^{-1}
oc_f	Costs per female other than the cost of supplemental feed	395.91	€ AU^{-1} yr^{-1}
pi_{bf}	% Increase in breading females if μ_{GMfi} increased by 10%	5.27	%
$pp_{y_{min}}$	Annual precipitation under which pasture does not grow	158.67	mm yr^{-1}
sb_h	Total subsidies per hectare	181.22	€ ha^{-1} yr^{-1}
sd_i	Remaining soil depth at $t = 0$	23.4	cm
$sd_{0.5}$	Remaining soil depth at which topsoil porosity is $0.5 \times sp_i$	0.05	cm
sf_{fi}	Initial supplemental feed per breeding female	442.1	kg AU^{-1} yr^{-1}
si_f	Secondary income per breeding female	47.84	€ AU^{-1} yr^{-1}
sp_i	Initial topsoil porosity	0.4265	d.u.
wp	Wilting point (volumetric)	0.05	d.u.
wr	Weathering rate of the parent rock	0.003	cm yr^{-1}
Δ	Time step	0.015625	yr

[a] Strictly speaking, the term 'initial' refers to any t at which SD = sd_i. However, this will only happen at $t = 0$, normally, since erosion makes SD to be less than sd_i afterwards.

2.3. Model conceptualization

The SNAPD models are intended to be used, at a late stage, in participatory exercises with stakeholders whenever the assessment carried out with them shows a special risk of degradation. In this way, the findings of the research would be conveyed to them and the goals of social learning and system understanding and experimentation would be fully achieved. Specifically, stakeholders should be convinced that the easiness of the model to reach a degraded state signals the risk the corresponding real system is running. Although this is in no way a straightforward task, the level of transparency and user-friendliness of the model plays a crucial role in success (Oxley et al., 2004). Thus, it was deemed convenient the models to have moderate dimensions, including at most a few dozens of variables, and that only the most indisputable processes were represented. The fact that the models are only fed with meaningful parameters and that they initially reproduce the present state of the modelled system, in whose specification the own stakeholders could take part, should also help in facilitating communication. Additionally, a piece of software has been programmed to assist in carrying out the intended analyses (explained in Section 5). This software interfaces with Vensim® (Ventana Systems Inc.), which is the software used to manage and run the models.

The SNAPD models are conceived of as both research and policy models, according to the characterization made by Oxley et al. (2004). They are policy models because the goal of possibly using them in participatory exercises with stakeholders conditions their levels of detail, complexity, transparency and user-friendliness. However, they also share some features of research models, since they are process oriented, seek improving understanding and try to be interesting, worthwhile and scientifically innovative not only through their output but also by the particular representations they make and the type of assessment they allow to carry out.

The SNAPD models sacrifice precision to generality and realism (Levins, 1966). A good illustration of this is given by the way in which functional forms are chosen. Since our concerns are focused on getting qualitative rather than quantitative outputs, we could choose theoretical functional forms on the basis of their general shape (increasing or decreasing), their economy in terms of the number of parameters required and the plausibility of the bounds they imposed on the corresponding variables, instead of by fitting curves to data (recall that every functional form is calibrated by assigning representative values to a few meaningful parameters, Fig. 2).

Although one of the purposes of the SNAPD models is early warning of degradation risk, our work is barely related with those aimed at early warning catastrophic regime shifts or critical transitions (e.g. Scheffer et al., 2001; Scheffer et al. 2009). We can briefly comment on this by taking the present case study as illustration. The presence of the soil state variable is decisive in configuring the phase space of the model. The system is within the basin of a complete-degradation attractor consisting of no soil, and thus of no vegetation and no livestock, at each time step in which the rate of erosion is greater than that of soil formation. On the contrary, the system is within the basin of a different attractor, determined within the pasture-livestock subsystem, at each time step in which soil formation is greater than erosion. Since the model is stochastic and the values of the exogenous variables vary from time step to time step, the system's attractor could be constantly changing within a simulation. But this by no means imply that the state of the system is also suffering critical transitions, since it is affected by delays in its path towards any attractor. In the present case study, the erosion rate is much greater than the weathering rate of the parent rock, on average (Table 1). Accordingly, the complete-degradation attractor dominates, i.e. it is the attractor of the system more times and its basin of attraction is frequently steeper than others. However, the time until the state of the system reaches such attractor is long (see Section 5.2).

Cilliers (1998) distinguishes between complex and complicated models. The latter are those which can be analysed accurately, no matter how large the number of their components may be, while the former are those "constituted by such intricate sets of non-linear relationships and feedback loops that only certain aspects of them can be analysed at a time" (Cilliers, 1998, p.3). Regarding this distinction, our model is complicated. Indeed, although its structure is made up of several feedback loops including non-linear relationships and delays, and also combines processes working at different time scales (Fig. 1), for all these elements to be at work simultaneously soil has necessarily to undergo erosion. But then, as indicated, the system is doomed to long-term degradation and can be analysed accurately.

Finally, the model is lumped spatial since its outputs are referred to the entire area modelled, which is an ideal, representative dehesa with homogeneous topographical, biophysical and managerial characteristics. Time is treated in a quasi-continuous way, i.e. outputs are provided for each time step (Kelly et al., 2013), and the system is described at an annual time scale. Nevertheless, as already mentioned, the model represents processes working at three different time scales.

3. Model description

The model presented here is part of a larger model constituting the current stage in an ongoing line of work aimed at building an integrated tool to assess land degradation in Mediterranean rangelands (Martínez Valderrama and Ibáñez, 2004; Ibáñez et al., 2007, 2012). At this stage, apart from the variables to be described later, the complete model includes shrub cover. However, this variable has been removed for this particular application on the assumption that shrubs are periodically removed by ploughing, which is a common farming practice in dehesas.

Fig. 1 shows the causal diagram of the model. It allows seeing the main processes represented, the way they interact and the time scales at which they work. Tables 1 and 2 show the lists of model parameters and equations, respectively.

3.1. Exogenous variables

The main exogenous variables of the model are placed outside the biggest rectangle in Fig. 1. Annual values of precipitation and average prices are assumed to follow stationary processes, i.e. no trend-like behaviour is considered, since the model is intended to make assessments of degradation under the current exogenous conditions. Annual precipitation and the average annual prices of meat and supplemental feed take random values at the first time step of each year and then remain constant for the rest of it (Eqs. (1)–(3)). The values are sampled from normal distributions in the three cases, whose means and coefficients of variation (CV) are model parameters (μ_{PP_y}, $\mu_{\mathrm{PR}_{mt}}$, $\mu_{\mathrm{PR}_{sf}}$, $\mathrm{cv}_{\mathrm{PP}_y}$, $\mathrm{cv}_{\mathrm{PR}_{mt}}$ and $\mathrm{cv}_{\mathrm{PR}_{sf}}$). Normal distributions showed smaller root-mean-square errors than gamma distributions when fitting the frequency distributions of historical data.

At each time step, the model assigns a random value to the runoff coefficient for the soil with the initial depth and at wilting point (Eq. (4)). This particular variable, hereafter referred to as RC_{wpi}, contributes to the calculation of the runoff coefficient for the soil with the depth and moisture existing at a given time step (Sections 3.2 and 3.3). The function of RC_{wpi} is to randomize runoff coefficients, thereby emulating the natural variability of rainfall intensity (Fig. 1). A random beta distribution, defined on the interval [0, 1], is used for RC_{wpi}. This distribution satisfactorily fitted the available data about the variable. The mean and CV of RC_{wpi} are model parameters ($\mu_{\mathrm{RC}_{wpi}}$ and $\mathrm{cv}_{\mathrm{RC}_{wpi}}$).

Any other exogenous variable of the model is non-random. Thus, subsidies and annual reference evapotranspiration ($\mathrm{ET_o}$) (Allen et al., 1998), both showed in Fig. 1, are model parameters (sb_h and $\mu_{\mathrm{ET}_{oy}}$, respectively).

3.2. Seasonal processes

The model derives seasonal values of precipitation and $\mathrm{ET_o}$ from their annual values (Eqs. (5) and (6)). To accomplish this, the time steps of every year are classified into two periods, the humid-fresh and the dry-hot ones. The lengths of both periods are determined by the parameter 'average number of months when precipitation exceeds ET_o' (mr). This number divided by 12 is the proportion of time steps of the humid-fresh period in every year.

The precipitation accumulated over the humid-fresh period in a given year is obtained by multiplying the total precipitation of the year (recall Section 3.1) by a constant proportion (parameter fp_{mr}). The remainder corresponds to the dry-hot period. Similarly, the ET_o accumulated over the humid-fresh period is obtained by multiplying annual ET_o (parameter $\mu_{\mathrm{ET}_{oy}}$) by a constant proportion (parameter fe_{mr}). The remainder corresponds to the dry-hot period.

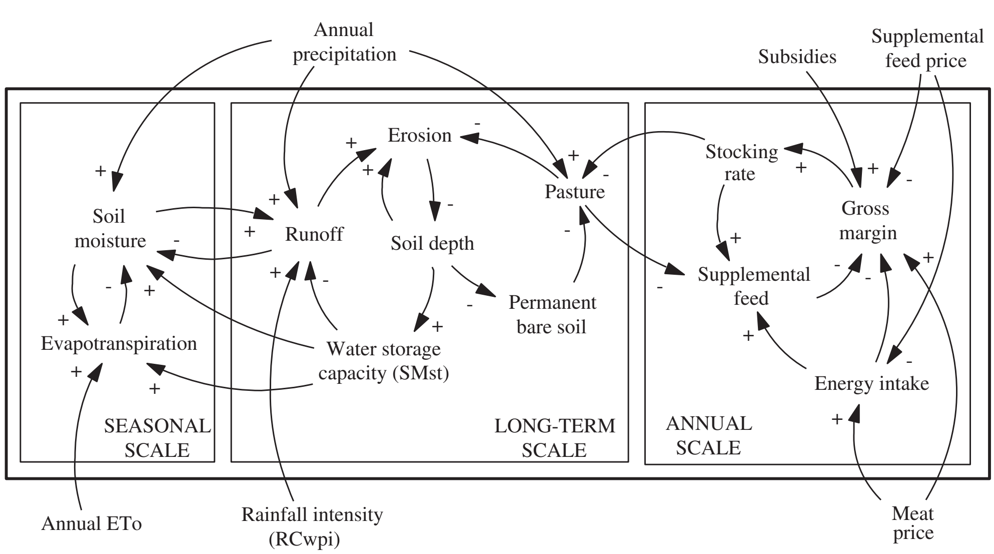

Fig. 1. Causal diagram of the model.

Table 2
Model equations.[a,b]

Annual precipitation [mm yr^{-1}]

$$PP_y = \text{if } t = \text{INTEGER}\{t\} \text{ then NORMAL}\left\{\mu_{PP_y}; cv_{PP_y}\right\} \text{ else } PP_y(t-\Delta) \quad (1)$$

Meat price [€ kg^{-1}]

$$PR_{mt} = \text{if } t = \text{INTEGER}\{t\} \text{ then NORMAL}\{\mu_{PR_{mt}}; cv_{PR_{mt}}\} \text{ else } PR_{mt}(t-\Delta) \quad (2)$$

Price of supplemental feed [€ kg^{-1}]

$$PR_{sf} = \text{if } t = \text{INTEGER}\{t\} \text{ then NORMAL}\left\{\mu_{PR_{sf}}; cv_{PR_{sf}}\right\} \text{ else } PR_{sf}(t-\Delta) \quad (3)$$

Initial runoff coefficient when soil is at wilting point [cm cm^{-1}]

$$RC_{wpi} = \text{BETA}\left\{\mu_{RC_{wpi}};\ cv_{RC_{wpi}}\right\} \quad (4)$$

Precipitation [cm yr^{-1}]

$$\begin{aligned} PP = {} & \text{if } t-\text{INTEGER}\{t\} \le mr/12 \text{ then } fp_{mr} \times PP_y \times 10^{-1} \times 12/mr \\ & \text{else}(1-fp_{mr}) \times PP_y \times 10^{-1} \times 12/(12-mr) \end{aligned} \quad (5)$$

Reference evapotranspiration [cm yr^{-1}]

$$ET_o = \text{if } t-\text{INTEGER}\{t\} \le mr/12 \text{ then } fe_{mr} \times \mu_{ET_{oy}} \times 12/mr \text{ else } (1-fe_{mr}) \times \mu_{ET_{oy}} \times 12/(12-mr) \quad (6)$$

Soil moisture [cm]

$$SM(t+\Delta) = SM + \Delta \times (IR-DR-ET) \quad (7)$$

$$SM(0) = sm_{wpi} \quad (8)$$

Infiltration rate [cm yr^{-1}]

$$IR = PP-SR \quad (9)$$

Surface runoff [cm yr^{-1}]

$$SR = PP \times RC \quad (10)$$

Soil drainage [cm·yr^{-1}]

$$DR = \text{MAX}\left\{0; IR-ET-(SM_{st}-SM)/\Delta + \text{MAX}\left\{0;\ SM-SM_{fc}\right\}\right\} \quad (11)$$

Maximum evapotranspiration [cm yr^{-1}]

$$MET = \text{MIN}\left\{ET_o; \left(SM_{fc}-SM_{wp}\right)\Big/\Delta\right\} \quad (12)$$

Evapotranspiration [cm yr^{-1}]

$$ET = \text{MIN}\{MET;\ IR + (SM-SM_{wp})/\Delta\} \quad (13)$$

Annual evapotranspiration [cm yr^{-1}]

$$ET_y(t+\Delta) = ET_y + \Delta \times (ET-\text{if } t = \text{INTEGER}\{t\} \text{ then } ET_y/\Delta \text{ else } 0) \quad (14)$$

$$ET_y(0) = ET_{yi} \quad (15)$$

$$ET_{yi} = ET_y \text{ in a simulation run under constant, average, initial conditions} \quad (16)$$

Runoff coefficient [cm cm^{-1}]

$$RC = {}^{(1)}RC_{wp} + (1-RC_{wp}) \times \left[(SM-SM_{wp})/(SM_{st}-SM_{wp})\right]^{\beta} \quad (17)$$

$$\beta = \left[\text{LN}\left\{\mu_{RC_{fci}}-\mu_{RC_{wpi}}\right\}-\text{LN}\left\{1-\mu_{RC_{wpi}}\right\}\right]\Big/\left[\text{LN}\left\{sm_{fci}-sm_{wpi}\right\}-\text{LN}\{sm_{sti}-sm_{wpi}\}\right] \quad (18)$$

Remaining soil depth [cm]

$$SD(t+\Delta) = SD + \Delta \times (wr-ER) \quad (19)$$

$$SD(0) = sd_i \quad (20)$$

(*continued on next page*)

Table 2 (*continued*)

Soil porosity multiplier [cm^2 cm^{-2}]

$$\mathrm{SP_{multi}} = {}^{(1)(2)}\mathrm{SD} \times (\alpha + \mathrm{sd}_i)/[\mathrm{sd}_i \times (\alpha + \mathrm{SD})] \tag{21}$$

$$\alpha = 0.5 \times \mathrm{sd}_i \times \mathrm{sd}_{0.5}/(0.5 \times \mathrm{sd}_i - \mathrm{sd}_{0.5}) \tag{22}$$

Topsoil bulk density and its initial value [g cm^{-3}]

$$\mathrm{BD} = {}^{(2)}2.65 \times (1 - \mathrm{sp}_i \times \mathrm{SP_{multi}}) \tag{23}$$

$$\mathrm{bd}_i = 2.65 \times (1 - \mathrm{sp}_i) \tag{24}$$

Soil moisture at saturation and its initial value [cm]

$$\mathrm{SM}_{st} = {}^{(3)}\mathrm{sp}_i \times (\alpha + \mathrm{sd}_i) \times [\mathrm{SD} + \alpha \times \mathrm{LN}\{\alpha/(\alpha + \mathrm{SD})\}]/\mathrm{sd}_i \tag{25}$$

$$\mathrm{sm}_{sti} = \mathrm{sp}_i \times (\alpha + \mathrm{sd}_i) \times [\mathrm{sd}_i + \alpha \times \mathrm{LN}\{\alpha/(\alpha + \mathrm{sd}_i)\}]/\mathrm{sd}_i \tag{26}$$

Soil moisture at field capacity and its initial value [cm]

$$\mathrm{SM_{fc}} = \mathrm{fc} \times \mathrm{SM}_{st}/\mathrm{sp}_i \tag{27}$$

$$\mathrm{sm_{fci}} = \mathrm{fc} \times \mathrm{sm}_{sti}/\mathrm{sp}_i \tag{28}$$

Soil moisture at wilting point and its initial value [cm]

$$\mathrm{SM_{wp}} = \mathrm{wp} \times \mathrm{SM}_{st}/\mathrm{sp}_i \tag{29}$$

$$\mathrm{sm}_{wpi} = wp \times \mathrm{sm}_{sti}/\mathrm{sp}_i \tag{30}$$

Runoff coefficient for the soil at wilting point [cm cm^{-1}]

$$\mathrm{RC}_{wp} = {}^{(2)(4)}1 - (1 - \mathrm{RC}_{wpi}) \times \mathrm{SP_{multi}} \tag{31}$$

Erosion rate [cm yr^{-1}]

$$\mathrm{ER} = {}^{(2)(4)}\mu_{\mathrm{ER}_{bsi}} \times (\mathrm{SR}/\mathrm{SR}_i)^2 \times \mathrm{EXP}\{-\varepsilon \times \mathrm{GV}\} \times \mathrm{SP_{multi}} \Big/ \left(\mathrm{BD} \times 10^2\right) \tag{32}$$

$$\mathrm{SR_i} = \text{SR in a simulation run under constant, average initial conditions} \tag{33}$$

$$\varepsilon = -\mathrm{LN}\{\mu_{\mathrm{ER}_i}/\mu_{\mathrm{ER}_{bsi}}\}/\mathrm{gv}_{sri} \tag{34}$$

Permanent bare soil area [ha ha^{-1}]

$$\mathrm{BS} = \mathrm{MIN}\{1; (\mathrm{BD} - \mathrm{bd}_i)/(\mathrm{bd_{pasture}} - \mathrm{bd}_i)\} \tag{35}$$

Area available for pasture growth [ha ha^{-1}]

$$\mathrm{GA} = 1 - \mathrm{BS} \tag{36}$$

Fractional pasture cover [ha ha^{-1}]

$$\mathrm{GV} = {}^{(1)(5)}\left[\mathrm{GA} \times \mathrm{MIN}\left\{1, \mathrm{GP}_h/\mathrm{gp}_{hgv1}\right\}\right] \times \mathrm{gv}_{sri} \times \mathrm{bf}_i/[\mathrm{gv}_{sri} \times \mathrm{bf}_i + (1 - \mathrm{gv}_{sri}) \times \mathrm{BF}] \tag{37}$$

Pasture production per hectare [kg ha^{-1} yr^{-1}]

$$\mathrm{GP}_h = \mu_{\mathrm{GP}_h} \times \mathrm{MAX}\left\{0; \mathrm{PP}_y - \mathrm{pp}_{y\ \min}\right\} \Big/ \left(\mu_{\mathrm{PP}_y} - \mathrm{pp}_{y\ \min}\right) \tag{38}$$

Pasture production [kg ha^{-1} yr^{-1}]

$$\mathrm{GP} = \mathrm{GA} \times \mathrm{GP}_h \tag{39}$$

Pasture consumption per breeding female [kg AU^{-1} yr^{-1}]

$$\mathrm{GC}_f = \mathrm{MIN}\left\{\mathrm{gc}_f; \mathrm{fs} \times \mathrm{GP}/\mathrm{BF}\right\} \tag{40}$$

Supplemental feed per breeding female [kg AU^{-1} yr^{-1}]

$$\mathrm{SF}_f = \left(\mu_{\mathrm{EI}_f} - \mathrm{GC}_f \times \mathrm{ec}_g\right) \Big/ \mathrm{ec}_{sf} \tag{41}$$

Table 2 (*continued*)

$\mu_{Elf} = ec_{sf} \times sf_{fi} + ec_g \times gc_f$	(42)
Gross margin per breding female and its initial mean value [€ AU^{-1} yr^{-1}]	
$GM_f = PR_{mt} \times \mu_{MP_f} + si_f + sb_h/BF - PR_{sf} \times SF_f - oc_f$	(43)
$\mu_{GM_{fi}} = \mu_{PR_{mt}} \times \mu_{MP_f} + si_f + sb_h/bf_i - \mu_{PR_{sf}} \times sf_{fi} - oc_f$	(44)
Expected gross margin per breading female and its initial value [€ AU^{-1} yr^{-1}]	
$GM_{f\ exp}(t+\Delta) = GM_{f\ exp} + \Delta \times \left[GM_f - GM_{f\ exp}\right] \Big/ dt_{GM_f}$	(45)
$GM_{f\ exp}(0) = \mu_{GM_{fi}}$	(46)
Breeding females [AU ha^{-1}]	
$BF = {}^{(1)}bf_i \times \left[\max\left\{0;\ GM_{f\ exp}\right\} \Big/ \mu_{GM_{fi}}\right]^{\rho}$	(47)
$\rho = {}^{(1)}LN\left\{1 + pi_{bf}/100\right\} \Big/ \ln\{1.1\}$	(48)

(1)The expression is formulated to ensure it passes through points defined by parameter values (Fig. 2).
(2)Porosity along the soil profile is given by $sp(SD) = sp_i \times SP_{multi}$ (Fig. 2B). Note that $SP_{multi} = 1$ when $SD = sd_i$.
(3)Definite integral between 0 and SD of sp(SD) (see note 2), with the boundary condition $sp(0) = 0$.
(4)Eqs. 23 and 24 imply that SP_{multi} equals the linear factor $(2.65-BD)/(2.65-bd_i)$.
(5)Here, the proportionality constant relating the stocking rate to BF (parameter fs) vanishes.
[a] $X(t)$ would denote the value of variable X at time t, but (t) is neglected as a simplification. Nevertheless, $X(t+\Delta)$, $X(t-\Delta)$ and $X(0)$ represent the values of X at times $t+\Delta$, $t-\Delta$ and zero, respectively.
[b] Strictly speaking, the term 'initial' refers to any t at which $SD = sd_i$. However, this will only happen at $t = 0$, normally, since erosion makes SD to be less than sd_i afterwards.

Within the humid-fresh period, precipitation is high and ET_o is low, so that soil moisture increases, on average; the opposite is true within the subsequent dry-hot period. In this way, the model makes a seasonal representation of soil moisture (Fig. 1). This influences the intra-annual distribution of runoff coefficients and erosion rates.

The following are details on the variables mainly involved in the seasonal processes: **Soil moisture** is the balance between infiltration, evapotranspiration and soil drainage (Eqs. (7) and (8)). **Infiltration rate** equals precipitation minus surface runoff (Eq. (9)). **Surface runoff** is precipitation times runoff coefficient (Eq. (10)). **Soil drainage** removes the water excess above field capacity, if any (Eq. (11)).

Maximum evapotranspiration is the lesser of two values: the current seasonal rate of ET_o and the available water content, i.e. field capacity less wilting point (Eq. (12)). **Evapotranspiration** is also the lesser of two values: maximum evapotranspiration and the current water content above wilting point (Eq. (13)). **Annual evapotranspiration** is obtained by adding up evapotranspiration over the time steps of one year (Eq. (14)–(16)).

Runoff coefficient. The model includes a function relating the runoff coefficient to soil moisture. This function is calibrated by fixing the initial mean runoff coefficients for the soil at field capacity and at wilting point (parameters $\mu_{RC_{fci}}$ and $\mu_{RC_{wpi}}$, respectively). The *Y*-intercept of this function is the runoff coefficient for the soil at wilting point, RC_{wp} (Eq. (17)–(18)). This variable is involved in a long-term process, so that it is fully explained in the following section. What is relevant here is that the function ensures that $RC_{wp} = RC_{wpi}$ (recall Section 3.1) when soil has the initial depth. Thus, if additionally rainfall intensity is average, then $RC_{wp} = RC_{wpi} = \mu_{RC_{wpi}}$ (Eq. (31)). When both conditions are met, the shape of the function is that illustrated by the thick curve in Fig. 2A. RC_{wpi} makes the *Y*-intercept RC_{wp} to be random, so that the whole curve shifts from time step to time step, thereby reflecting the effect of rainfall intensity, as already mentioned. This is represented in Fig. 2A by the dashed curves surrounding the thick one. As a result, the runoff coefficient for a given amount of soil moisture is also a random variable. The higher the soil water content, the higher is the mean of this random variable, and the lower is the variance. At the limit, the runoff coefficient for the saturated soil is always one with zero variance (Fig. 2A).

3.3. Long-term processes

Erosion drives the three long-term processes represented in the model. These processes constitute causal feedback loops with a pivotal role in degradation (Fig. 1). Two of them are positive, so making erosion to accelerate itself: i) as soil is lost, water storage capacity diminishes, thereby causing surface runoff, and thus the erosion rate, to increase, on average; and ii) as soil is lost, soil layers with a higher bulk density are exposed, thereby hampering seed germination and pasture rooting, causing the fraction of unprotected soil to increase and favouring erosion. The third feedback loop is negative: as soil is lost, topsoil bulk density increases, thereby hampering erosion. It is worth recalling that the rates at which these three processes evolve are ultimately determined by the specific values assigned to certain parameters.

The following are details on the variables mainly involved in the long-term processes:

Remaining soil depth. This state variable increases by the weathering of parent rock and decreases by erosion (Eq. (19) and (20)). The initial soil depth and the rate of weathering are model parameters (sd_i and wr, respectively). Soil organic matter is assumed to be in equilibrium, i.e. deposition rate equals mineralization rate on average. Hence, it does not cause variation in soil depth (and is not modelled).

Soil porosity decreases with soil depth from an initial topsoil value to zero, which is the porosity assumed for the parent rock (Eq. (21)–(22)). The functional form representing this distribution follows from the soil profile model provided by Kirkby (1985). As

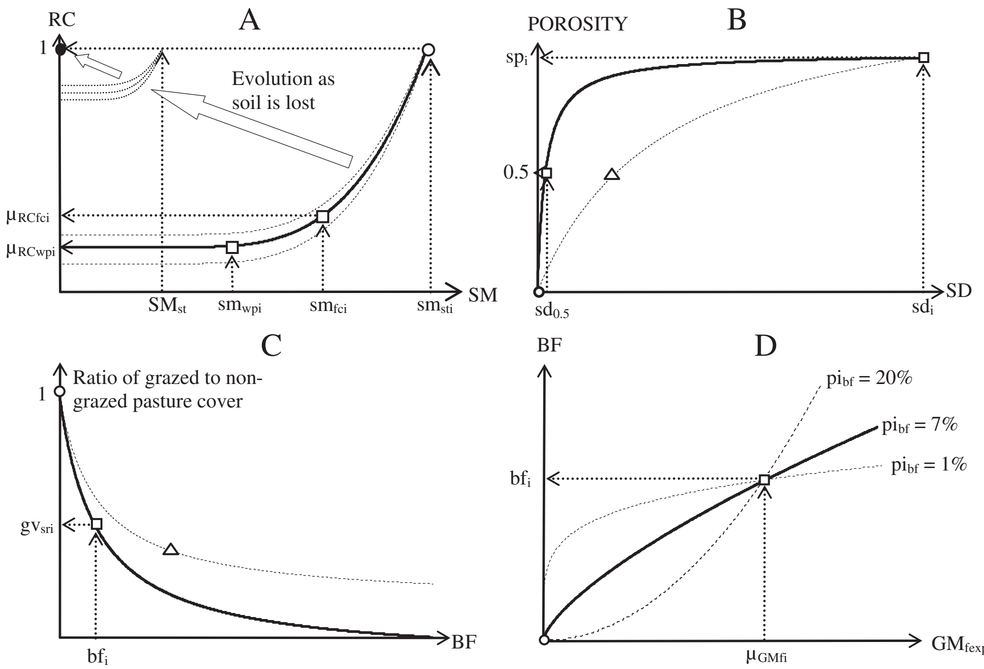

Fig. 2. Instances of the functional relationships in the model. They are calibrated by assigning the coordinates of the square points. Triangle points would lead to possible alternatives. Circle points are structural. (A) Relationship between runoff coefficient (RC) and soil moisture (SM). (B) Porosity distribution along the soil profile (SD is the remaining soil depth). (C) Relationship between the ratio of grazed to non-grazed pasture cover and the number of breeding females (BF). (D) Relationship between the number of breeding females (BF) and the expected gross margin per breeding female (GMfexp). See Table 1 for definitions of parameters.

already indicated, this function is calibrated by fixing two parameters: the initial topsoil porosity and the remaining soil depth at which topsoil porosity is reduced by a half (sp_i and $sd_{0.5}$, respectively). Two illustrative instances of this function, both referred to the same initial topsoil porosity, are shown in Fig. 2B. **Bulk density** distribution along the soil profile directly derives from that of soil porosity (Eq. (23)–(24)).

Soil moisture characteristics. Soil moisture at saturation equals the total pore volume existing in the remaining soil depth (Eq. (25)–(26); see also note 3 in Table 2). Therefore, it is equivalent to water storage capacity (Fig. 1). Field capacity and wilting point are two constant proportions of soil moisture at saturation (parameters fc and wp, respectively) (Eq. (27) and (30)).

Runoff coefficient for the soil at wilting point (RC_{wp}). Apart from what it was said about RC_{wp} in Section 3.2, this variable increases linearly with topsoil bulk density, i.e. as soil is lost. In the absence of data to check such a relationship, the assumption of linearity has the advantage of requiring no parameter (Eq. (31); see also notes 2 and 4 in Table 2). On the other hand, soil moisture at saturation (water storage capacity) diminishes as soil is eroded. As a result, the loss of soil makes the function relating the runoff coefficient to soil moisture to shrink and shift towards the upper left corner in Fig. 2A. Therefore, the mean of the runoff coefficient for a given amount of soil moisture increases and the variance decreases, as soil is lost. The runoff coefficient when the parent rock emerges is one with zero variance, i.e. if soil runs out, curves collapse into the black point in Fig. 1.

Erosion rate. The equation for this variable (Eq. (32)) is a reformulation of Thornes's erosion model (Thornes, 1985, 1989, 1990). This model is

$$\mathrm{ER} = k \times \mathrm{SR}^2 \times s^{1.6} \times \exp\{-\varepsilon \times \mathrm{GV}\}$$

where ER is erosion rate, SR is surface runoff, GV is pasture cover, s is the slope gradient and k and ε are parameters. It follows that

$$\mu_{\mathrm{ER}_{bsi}} = k \times \mathrm{SR}_i^2 \times s^{1.6}$$

$$\mathrm{ER} = \mu_{\mathrm{ER}_{bsi}} \times (\mathrm{SR}/\mathrm{SR}_i)^2 \times \exp\{-\varepsilon \times \mathrm{GV}\}$$

where $\mu_{\mathrm{ER}_{bsi}}$ is the initial mean bare soil erosion rate, which is a model parameter, and SR_i is surface runoff under average, initial conditions (Eq. (33)). Note that the value of $\mu_{\mathrm{ER}_{bsi}}$ condenses factors such as rainfall erosivity, soil erodibility, slope length and slope gradient. The expression given to ε (Eq. (34)) ensures that the erosion rate would equal its initial mean value (parameter μ_{ER_i}) under initial average conditions. Eq. (32) is completed by adding to Thornes's model a factor reflecting a linear inverse relationship between the erosion rate and topsoil bulk density. This factor ensures that the erosion rate falls to zero if parent rock is exposed. Linearity is again assumed to avoid increasing the number of model parameters. Note that pasture cover partly depends on livestock density, and thus on business profitability (Fig. 1). In this way, the model connects erosion rates to economic and behavioural variables.

Permanent bare soil is the area where pasture growth is no longer possible because topsoil bulk density is excessively high. This area is initially zero and is assumed to increase linearly with topsoil bulk density (Eq. (35)). Linearity has the advantage that only one parameter is needed to calibrate the relationship. This is the topsoil bulk density threshold above which pasture does not grow ($\mathrm{bd}_{\mathrm{pasture}}$). The **area available for pasture growth** is the rest of the total area (recall that shrubs are removed) (Eq. (36)).

3.4. Annual processes

The model represents the processes underlying the dynamics of the pasture-livestock subsystem in an annual scale. The core of this subsystem is a negative feedback loop. Indeed, the stocking rate is assumed to be positively related to the annual gross margin earned per breeding female (pbf). Thus, if the stocking rate increases, the amount of supplemental feed needed pbf on average over the years is greater, so that the average annual gross margin pbf becomes smaller and the stocking rate is eventually reduced (Fig. 1). This negative feedback tends to make the pasture-livestock subsystem self-regulating. However, the dynamics of such regulation depend on how fast farmers perceive changes in profit conditions and how reactive they are to them. Both behavioural factors have been included in the model because they may have implications for rangeland degradation (e.g. Higgins et al., 2007; Anderies et al., 2002).

The following are details on the variables mainly involved in the annual processes:

Pasture cover. Potential pasture cover, i.e. in the absence of livestock, is assumed to grow linearly from zero to one with pasture production. This relationship is calibrated by only one parameter, the 'minimum pasture production entailing full cover in the absence of livestock' (gp_{hgv1}) (potential pasture cover is within square brackets in Eq. (37)). Pasture cover (with livestock) is a variable fraction of the potential cover (the factor outside the square brackets in Eq. (37)). This fraction is one in the absence of livestock and approaches zero with increasing stocking rates (Fig. 2C). Its expression is calibrated by fixing two parameter values: the average initial value of pasture cover and the initial number of breeding females (gv_{sri} and bf_i). This function would synthesize the grazing and trampling effects of animals on pasture cover.

Pasture production per hectare linearly depends on annual precipitation as long as it exceeds a minimum threshold amount (Sullivan and Rohde, 2002) (Eq. (38)). This relationship is expressed in terms of two meaningful parameters: 'initial mean pasture production per hectare' and 'annual precipitation under which pasture does not grow' (μ_{DP_h} and $pp_{y_{min}}$, respectively). **Pasture production** is the product of pasture production per hectare and pasture area (Eq. (39)).

Pasture consumption pbf is the lesser of two values: a representative value, which is a model parameter (gc_f), and the maximum amount of available pasture pbf, which is pasture production divided by the stocking rate (Eq. (40)). The stocking rate is assumed to be proportional to the number of breeding females. The proportionality factor is a model parameter (fs).

Supplemental feed. It is assumed that breeding females are supplied with the amount of supplemental feed required for them to reach a fixed target energy intake (Eq. (41)). This energy intake, which would aim at weaned offspring to reach the intended weight, is calculated on the basis of typical consumptions of pasture and supplemental feed and the average energy contents of both feeds (parameters gc_f, sf_{fi}, ec_{sf} and ec_g) (Eq. (42)).

Annual gross margin pbf is the difference between total revenue and total cost pbf. The former is the result of multiplying the average annual price of meat by the average annual meat production pbf (parameter μ_{MP_f}) and then adding secondary incomes and subsidies pbf (parameters si_f and sb_h, respectively). Total cost pbf is the result of multiplying the average annual price of supplemental feed by the amount of supplemental feed consumed pbf and per year and then adding other cost pbf (parameter oc_f) (Eq. (43)–(44)). Recall that the average annual prices are exogenous variables (Section 3.1).

Expected annual gross margin pbf. Farmers' expectations about the annual gross margin pbf are obtained by applying exponential smoothing. This well-known method is formulated here as an information delay, that is, by means of a state variable (Sterman, 2000) (Eq. (45)–(46)). The average length of this delay (parameter dt_{GM_f}) is a measure of the time that farmers take to perceive and accept changes in rainfall (pasture production) and economic conditions.

Breeding females. The number of breeding females, and thus the stocking rate, depends on expectations about the annual gross margin. A Cobb–Douglas function, widely used in economics, is used to formalize this relationship (Eq. (47)–(48)). The function is expressed in terms of the meaningful parameter '% increase in breeding females if the current average gross margin pbf increased by 10%' (pi_{bf}). This is a measure of how reactive farmers are to changes in profit conditions. Fig. 2D illustrates the shape of the function for three different values of pi_{bf}: 20%, 7% and 1% (opportunistic, medium and conservative farmers, respectively).

4. Parameter values

As already mentioned, to run the model, all its parameters must take values that are representative of the case study. Those corresponding to this application are shown in Table 1. Most of the values were estimated on the basis of measurements taken at a set of 22 field working units (fenced areas) selected from 10 representative farms distributed throughout the Spanish region of Extremadura. This region is located in the centre of the area covered by dehesas in the Iberian Peninsula. Besides, semi-structured interviews with farm owners were conducted in 2011 to obtain information on economic aspects and livestock management in their farms.

Table 3 shows the main characteristics of the working units. Mean altitude of the farms range from 299 to 695 m a.s.l. Precambrian schist and greywacke are the prevailing rock types. Soils developed on schist are Cambisols and Leptosols and those found on sediments are classified as Luvisols (IUSS Working Group WRB, 2006). In general, soils are shallow, acid, with a sandy loam texture and a low content of organic matter and nutrients.

The working units show a variable size, ranging from 2.8 to 146.2 ha. Pastures are grazed by sheep, cattle, pigs and goats at highly variable livestock densities (0.19–15.79 AU ha^{-1}; AU = Animal unit). The most common animal species is sheep, present in 17 of the 22 working units, particularly in treeless pasturelands. Free-ranging pigs are typically in combination with either cattle or sheep. Cattle are common in the more humid areas

Table 3
Main characteristics of the working units where most of the model parameters were estimated. Id.F: Farm number, UN: Unit number, Area: Area (ha), Precip.: Mean total annual precipitation (mm) (Ninyerola et al., 2005), StD: Livestock stocking density (AU ha^{-1}), TD: Tree density (trees ha^{-1}).

Id.F	UN	Area	Rock type	Precip.	Livestock	StD	Tree	TD
1	1	46.2	Schist	731.8	Cattle and pigs	0.54	Cork oak	18.4
1	2	103.5	Schist	731.8	Cattle and pigs	0.54	Cork oak	26.5
2	3	37.6	Schist	504.8	Sheep	0.62	–	0.0
2	4	136.5	Schist	504.8	Sheep	0.19	–	0.0
3	5	33.2	Schist	591.8	Cattle, pigs and sheep	1.82	Holm oak	3.2
3	6	2.9	Schist	591.8	Cattle, pigs and sheep	15.76	–	0.0
4	7	146.2	Sediments	596.2	Sheep and pigs	1.08	Holm oak	24.5
4	8	30.3	Sediments	596.2	Sheep	1.19	Holm oak	10.4
4	9	74.1	Schist	596.2	Sheep and pigs	1.09	Holm oak	19.8
4	10	19.1	Sediments	596.2	Sheep and pigs	2.99	Holm oak	15.6
5	11	21.8	Schist	646.3	Sheep and goats	1.17	Holm oak	25.4
5	12	52.0	Schist	646.3	Sheep and goats	1.17	Holm oak	41.5
6	13	10.7	Schist	661.1	Sheep	0.59	Holm oak	9.1
6	14	12.8	Schist	661.1	Cattle	0.78	Holm oak	9.0
7	15	120.3	Schist	526.9	Sheep	0.25	–	0.0
7	16	120.3	Schist	526.9	Sheep	0.25	–	0.0
8	17	34.2	Schist	565.2	Sheep and pigs	0.54	Holm oak	88.4
8	18	24.1	Schist	565.2	Sheep and pigs	0.54	Holm oak	63.0
9	19	24.5	Schist	689.3	Cattle and pigs	0.59	Holm oak	64.9
9	20	6.2	Schist	689.3	Cattle and pigs	0.59	Holm oak	147.8
10	21	7.1	Schist	681.3	Sheep and pigs	0.43	Holm oak	48.2
10	22	19.7	Schist	681.3	Sheep and pigs	0.43	Holm oak	100.9

where pasture productivity is higher and goats are only found in one of the studied farms.

4.1. Parameter values coming from the field working units

Values of soil depth (sd_i), topsoil porosity (sp_i), field capacity (fc) and wilting point (wp) correspond to the averages of data obtained from 47 soil profiles located in the working units. Mean bare soil erosion rate ($\mu_{ER_{bsi}}$) was based on Universal Soil Loss Equation (Wischmeier and Smith, 1978; Renard et al., 1991) and estimated for a representative slope and soil in farm No. 4 (Table 3). Mean and CV of RC_{wpi} ($\mu_{RC_{wpi}}$ and $cv_{RC_{wpi}}$) were calculated from data on rainfall events measured from July, 1990 to September, 1997. Although rather scarce ($n = 10$), data showed a distribution which satisfactorily fitted to a beta distribution ($R^2 = 0.96$). The average runoff coefficient for the humid soil ($\mu_{RC_{fci}}$) was obtained from a sample of 37 rainfall events measured during the winter and spring seasons of the period 1990–1997. Pasture production per hectare (μ_{GP_h}) and the annual precipitation under which pasture does not grow ($pp_{y_{min}}$) were obtained by fitting Eq. (38) to a scatter plot of precipitation vs. pasture production ($n = 61$; $R^2 = 0.62$). Minimum pasture production for full cover in the absence of livestock (gp_{hgv1}) was estimated on the basis of 63 data points of pasture cover vs. annual precipitation measured between 2008 and 2010. In turn, the value of pasture cover corresponding to the average stocking rate (gv_{sri}) was obtained from 21 data points of pasture cover vs. stocking rate measured in May, 2010, a year when total precipitation was normal.

The values of most of the economic parameters ($\mu_{PR_{mt}}$, $\mu_{PR_{sf}}$, sb_h, si_f and oc_f), the average meat production pbf (μ_{MP_f}), the average consumption of supplemental feed pbf (sf_{fi}), the number of breeding females per hectare (bf_i) (a mixed herd made up of cows, sheep, sows and she-goats) and the ratio of breeding females to livestock numbers (fs) were averaged over data obtained from the interviews with farm owners.

4.2. Parameter values coming from other sources

The mean and CV of annual precipitation (μ_{PP_y} and cv_{PP_y}, respectively) were calculated on the basis of a single series of monthly rainfall amounts obtained by averaging those corresponding to the cities of Cáceres and Badajoz, period 1955–2011 (Spanish National Meteorological Agency; www.aemet.es/en/portada). Annual ET_o ($\mu_{ET_{oy}}$) and its monthly distribution were estimated from daily measurements of pan evaporation in the meteorological station of Cáceres, period 1996–2006 (Spanish National Meteorological Agency). The evaporation data were transformed into ET_o using a correction factor described in Allen et al. (1998). The comparison between monthly distributions of precipitation and ET_o allowed estimating the length of the humid-fresh period (mr) and the proportions of annual precipitation and ET_o corresponding to this period (fp_{mr} and fe_{mr}).

Official figures were used to estimate the means and CVs of the average annual prices of meat and supplemental feed ($\mu_{PR_{mt}}$, $\mu_{PR_{sf}}$, $cv_{PR_{mt}}$ and $cv_{PR_{sf}}$), period 1976–2010 (Spanish Ministry of Agriculture, Food and Environment; http://www.magrama.gob.es/es/).

The values for the rest of parameters were worked out on the basis of information collected from the literature. Mean erosion rate (μ_{ER_i}) corresponded to the central value of the modal interval of erosion rates in Cáceres (MAGRAMA, 2005), which includes 70.6% of the provincial area. The value of topsoil bulk density above which pasture does not grow ($bd_{pasture}$) was taken from the United States Department of Agriculture (USDA, 2001). It was referred to a soil with a sandy loam texture, which is common in the study area. The energy content of pasture (ec_g) came from Cambero Muñoz (1998) and that of the supplemental feed (ec_{sf}) from García Dory et al. (1985). The weathering rate of the parent rock (wr) was taken from Nahon (1991) and refers to metamorphic rocks under temperate climate. The number of months per year with grazing activity was obtained from the interviews, and the typical pasture consumption per animal and day was provided by Cambero Muñoz (1998) for cows, Pascual et al. (1996) for sheep, López Bote et al. (2000) for pigs and Rodríguez-Estévez et al. (2007) for goats; all of them allowed calculating the pasture consumption per breeding female (gc_f).

Since no time series about gross margins were available, data provided by Pulido et al. (2010) on agricultural income and livestock numbers in the Extremadura region, period 1977–2010, were used to estimate the average delay time for farmers to form economic expectations (dt_{GMf}) and the expected percentage increase in breeding females if agricultural income (not gross margin) increased by 10% (pi_{bf}). To accomplish this, Eq. (47) was repeatedly fitted to data, each time using a different value of dt_{GM_f} when applying exponential smoothing to agricultural income data. The parameter values eventually chosen were those leading to the smallest root-mean-square error.

No reference could be found on the remaining soil depth at which topsoil porosity is reduced by a half ($sd_{0.5}$). Therefore, its value was calibrated to make the annual evapotranspiration obtained by running the model under average initial conditions to fit the observed mean value. The latter was estimated from daily measurements of pan evaporation in the meteorological station of Cáceres, period 1996–2006 (Spanish National Meteorological Agency). The estimate of $sd_{0.5}$ turned out to be very low thereby reflecting that porosity would be uniformly distributed along the soil profile until a distance very close to the parent rock (Fig. 2B, thick line).

5. Assessment of land degradation by water erosion in dehesas

5.1. Analyses aimed at achieving the purposes of the model

In agreement with van der Leeuw (1998), degradation is conceived of as a loss of suitability for some land use. Therefore, generally speaking, we associate risk of degradation with the rate of loss of a resource (natural or human) which is crucial for the functioning of the land use. In the present case study, this resource is soil.

In this way, in order to achieve Purpose 1, that is assessing the risk of degradation that dehesa rangelands are running (Section 2.1), the following analysis was carried out. A thousand model simulations were run over a period of 1000 years under randomly-generated scenarios of annual precipitation, RC_{wpi} (as a surrogate of rainfall intensity) and the prices of meat and supplemental feed. The time span of the simulations was 1000 years to ensure that the entire soil was lost in all of them.

The resulting 1000 time trajectories of the variable 'remaining soil depth' (SD, Eq. (19)) were recorded. Based on these data, the cumulative frequency distributions of the time it took to lose different amounts of soil were calculated. This was done at intervals of 5 cm of soil, counting from the initial surface. Note that a value of one of the mentioned distributions indicates the percentage of the 1000 simulations where the corresponding amount of soil was lost before a given number of years. This is likened to the risk of such a thing to happen. Additionally, means, standard deviations and minimum and maximum values of the time to lose the different amounts of soil were calculated over the 1000 time trajectories of SD. Mean values are taken as complementary measures of the risk of degradation in the studied case.

In order to achieve Purpose 2, that is assessing the degree to which different factors would hasten degradation if they changed from their current typical values (Section 2.1), a sensitivity analysis of the model was carried out. This is an appropriate type of analysis in this case because all the model parameters have real world counterparts. Thus, the sensitivity analysis fulfils the function of answering 'what if' type questions and not that of helping to calibrate and validate the model. The factors whose impacts were evaluated are mainly climatic and economic.

Specifically, a Plackett–Burman sensitivity analysis (PBSA) was carried out. A detailed description of this procedure is given by Beres and Hawkins (2001). Briefly, the PBSA is a statistically sound method that measures the impacts of each parameter on target output variables in an efficient way in terms of the number of simulations needed. First, upper and lower values must be assigned to each parameter. In our case, these values were obtained by increasing and decreasing those presented in Table 1 by 5%. In this way, the PBSA yielded the impacts resulting from increasing each parameter by 10%. Second, the upper and lower values must be specifically selected to form a prescribed number of scenarios, which turned out to be 40 in our case. An important feature of the PBSA is that the impact of every parameter is averaged over all of the simulations, so that it does not rely on the all-other-things-being-equal assumption. However, the estimated impacts must be interpreted under this assumption.

In order to assess how changes in the variability of the exogenous variables affected the system, their CVs were included within the set of parameters to be analysed. Hence every simulation in the procedure was stochastic. Therefore, to achieve a robust analysis, 100 simulations were run under each scenario, each one using a different random seed. Overall, the analysis was carried out on the basis of 4000 model simulations. The target variable was the average number of years (over the 100 simulations per scenario) taken for the loss of 20 cm of soil (about 85% of the initial soil depth). The time horizon was fixed at 1000 years to ensure that this loss occurred in all of the simulations.

5.2. Results of the analyses

The cumulative frequency distributions of the times to lose 5, 10, 15 and 20 cm of soil, calculated over the 1000 time trajectories obtained for the variable soil depth, are plotted in Fig. 3. The following are two examples of how they must be interpreted: i) it is estimated that the risk of losing the upper 5 cm of soil within 150 years is approximately 80%, i.e. such amount of soil was lost before that number of years in 80% of the 1000 simulations; ii) it is estimated that there is a 100% risk of losing the upper 20 cm of soil (85% of the initial soil depth) within approximately 400 years. Means, standard deviations and minimum and maximum values of these distributions are drawn in Fig. 4. The mean values turned out to be 138, 245, 317 and 360 years, respectively.

Table 4 shows the results of the PBSA. Recall that every impact reflects the percentage variation of the time elapsed to lose 20 cm of soil when the corresponding parameter is increased by 10%. A positive impact means that the soil loss is delayed; a negative one means that it is brought forward. All of the impacts obtained showed the expected sign. Top positions were occupied by climatic factors, without exception. Mean annual precipitation (−36.9%) showed the greatest effect on soil erosion. The parameter 'initial mean runoff coefficient for the soil at wilting point', associated with rainfall intensity, showed a considerably smaller effect (−9.5%). When the CVs of both factors (rainfall amount and intensity) were increased by 10%, the time for the soil to be depleted was brought forward by −8.1% and −4.5%, respectively; these effects were less than those of mean values. Increasing the fraction of rainfall within the humid season, when soil moisture and runoff are higher on average, showed an important negative effect (−17%). Mean annual ET_0 and the fraction of annual evapotranspiration accumulated within the humid season also had important effects (12.5% and 12.2%, respectively). Here, positive signs were expected since increasing evapotranspiration implies reducing soil moisture and thus runoff, on average. Economic and behavioural variables were located in the lowest positions in Table 4. The highest impact of one of these variables, namely subsidies, was only −1.4%.

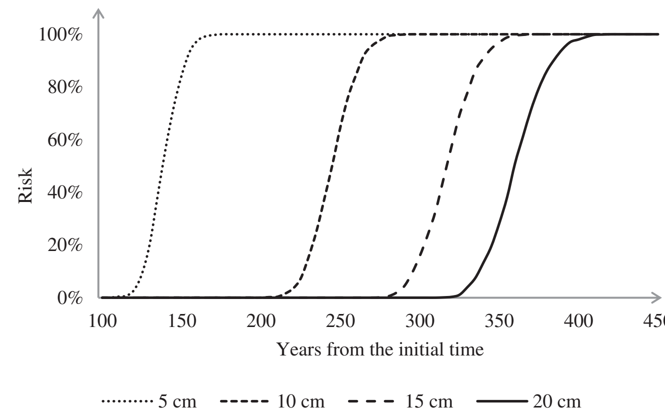

Fig. 3. Risk of losing different amounts of soil over time (see Section 5.2 for details).

5.3. Discussion

Erosion processes in real rangelands are highly variable in space and time, and places can be found where net gains and losses of soil significantly differ from those used in this work. Studying erosion as a natural process contributing to shape landscapes implies dealing with difficulties that lie beyond the scope of this research. Our aim is getting an overall view of the risk of degradation by water erosion that dehesa rangelands are currently running and of the main factors that drive this process. And we intend to make such an assessment by means of a model as transparent and user-friendly as possible, in case it is deemed convenient to convey the findings to stakeholders.

The results of our analyses show that the global risk of degradation by water erosion in dehesas would be moderate for the time being (Purpose 1, Section 2.1), so that it would not be a land use especially threatened by this type of degradation process in Spain

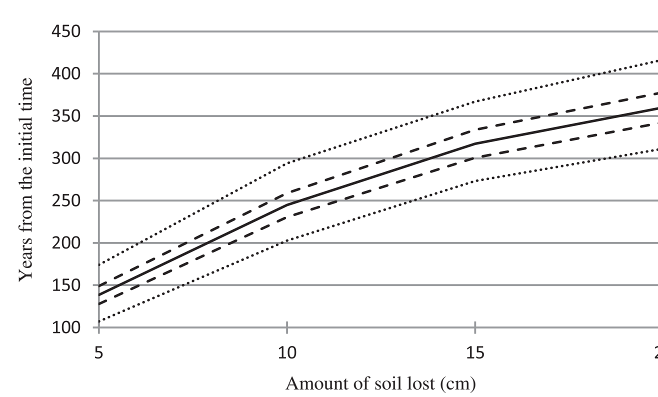

Fig. 4. Mean (thick line), mean plus and less one standard deviation (dashed lines) and minimum and maximum values (dotted lines) of the times to lose different amounts of soil in 1000 model simulations run under randomly-generated scenarios of rainfall amount and intensity and prices.

Table 4
Average impacts on the time to lose 20 cm of soil when a parameter is increased by 10%. A positive impact means that the loss is delayed; a negative one means that it is brought forward.

Parameter	Impact
Mean annual precipitation (μ_{PP_y})	−36.9%
Fraction of annual precipitation fallen in the humid season (fp_{mr})	−17.5%
Mean annual reference evapotranspiration ($\mu_{ET_{oy}}$)	12.5%
Fraction of annual evapotranspiration in the humid season (fe_{mr})	12.2%
Initial mean runoff coefficient soil at wilting point ($\mu_{RC_{wpi}}$)	−9.5%
Coefficient of variation annual precipitation (cv_{PP_y})	−8.1%
Coefficient of variation runoff coefficient soil at wilting point ($cv_{RC_{wpi}}$)	−4.5%
Months when precipitation > ET_o (length of the humid season) (mr)	2.4%
Total subsidies per hectare (sb_h)	−1.4%
Costs per female other than the cost of supplemental feed (oc_f)	1.3%
Mean meat price ($\mu_{PP_{mt}}$)	−1.2%
Weathering rate of the parent rock (wr)	1.1%
Average number of years to form gross margin expectations (dt_{GM_f})	−0.5%
Mean price of supplemental feed ($\mu_{PR_{sf}}$)	0.4%
Coefficient of variation supplemental feed ($cv_{PR_{sf}}$)	−0.2%
Coefficient of variation meat price ($cv_{PR_{mt}}$)	−0.2%
% Increase in breading females if gross margin increased by 10% (pi_{bf})	0.1%
Secondary income per breeding female (si_f)	0.0%

(Derived purpose 1, Section 2.1). This is not denying that some places affected by particular conditions are currently undergoing extreme degradation nor to say that more research on the matter is unnecessary, quite the contrary.

The question remains as to whether the described assessment should be conveyed to stakeholders or not, because there is a risk of reassuring them that no action is required. And this is unclear. For example, shrubs contribute to reduce soil water erosion and to preserve and ameliorate soil condition in dehesas (e.g. Smith and Wischmeier, 1962; Schnabel, 1997; Moreno and Pulido, 2009; Pulido et al., in press). As already mentioned, shrubs use to be removed from dehesas. However, managing the presence of dense patches of woody vegetation in time and space might help in further lowering the risk of land degradation, especially in areas of high sensitivity to erosion.

The risk of degradation by water erosion in dehesas could increase in the future mainly because of changes in climatic variables, particularly if average rainfall increased and evapotranspiration decreased (Purpose 2, Section 2.1). It must be said that projections of the effects of climate change for Extremadura show just the contrary, with precipitation decreasing and temperature increasing in the region, on average (AEMET, 2009). Nevertheless, rainfall intensity and the variability of both rainfall amount and intensity could increase in the future, and these changes hastened erosion in our analysis. It is unclear what the final balance of all these possible changes may be.

The modelled human activities would have a minor role in degradation both in absolute terms and relative to the climatic factors (Derived purpose 2, Section 2.1). A similar finding emerged from a study focused on a rangeland in Lagadas (Greece), which was based on a rather different model (Ibáñez et al., 2012). Although it is soon to generalize these results, they may contribute to the debate on the importance of natural and human factors in land degradation and desertification (e.g. Illius and O'Connor, 1999; Ellis and Swift, 1988).

As a final remark, we want to stress the major role played in our case study by the two positive feedback loops in which the erosion rate is involved (Fig. 1, Section 3.3). They are responsible for this variable to accelerate over time, thereby outweighing the effect of the negative feedback also involving the erosion rate. Such acceleration is evidenced in how the cumulative frequency distributions in Fig. 3 get closer over time and in how the curves in Fig. 4 bend downward. As a result, the average number of years needed to lose a 5-cm layer of soil decrease over time, namely 138 years for losing the upper layer, 107 for the subsequent layer, 72 for the third one and 43 for the deepest one. Besides, the average times to lose 5, 10, 15 and 20 cm of soil obtained with the model (138, 245, 317 and 360 years, respectively) fall far short of those obtained by naively considering a constant erosion rate, specifically the current value of the annual mean erosion rate (parameter μ_{ER_i}). Calculated in this way, the time to erode a 5-cm layer of soil is 372 years, so that the times to lose 5, 10, 15 and 20 cm of soil are 372, 744, 1116 and 1488 years, respectively. The differences between model results and these naive estimations evidence that the positive feedback loops may play a major role in the dynamics of erosion so that they should not be neglected in other related assessments.

6. Conclusions

The approach presented in this paper for carrying out an integrated assessment of land degradation has proved to be satisfactory. It is recommended under the following circumstances: i) the research is focused on representing breadth of the studied system and not depth on individual system components; ii) the goal is improving system understanding or social learning, and not prediction or forecasting; iii) simplicity and flexibility are required, e.g. to fit well in the ever-changing agendas of decision-makers (Oxley et al., 2004); iv) there is a relative scarcity of data; and v) the system's behaviour is affected by causal feedback loops.

However, even though the above conditions were met, the validity of the approach ultimately relies on the plausibility and coherence of the model used to carry out the assessment. In our case, every effort was made to satisfy this requirement. This allows us to be reasonably confident of the soundness of our assessment of degradation by water erosion in dehesas.

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