CASCADE
Catastrophic shifts in drylands:
How can we prevent ecosystem degradation?

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Report on integrated modelling strategy

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Executive summary
Widespread failure in ecosystem restoration and degradation prevention, even with massive investments, has underpinned the broad agreement that ecosystems can behave in complex, non-linear ways. Restoration of ecosystem performance and prevention of degradation can require considerably stronger efforts in non-linear than in gradually responding systems, but can also benefit from particular opportunities due to non-linear dynamics. Hence, knowledge on dynamic ecosystem regimes and threshold dynamics can provide crucial advances for sustainable land management enshrined in the Sustainable Development Goals.

This report presents a conceptualisation and modelling strategy to evaluate the socio-ecological effectiveness of land management considering non-linear ecosystem dynamics and windows of opportunities and risks. Socio-ecological effectiveness is defined here as the potential of a management strategy to help maintain or restore ecosystem services while ensuring land users meet their basic needs. It is evaluated based on insights into ecological effects and financial attractiveness focussing on drylands in southern Europe.

The conceptualisation presented here is the first to link ecological theory of non-linear ecosystem dynamics to Land Degradation Neutrality as a pre-requisite for meaningful operationalisation and monitoring of progress towards Land Degradation Neutrality. This conceptualisation provides the basis for a 5-step modelling approach. First, management scenarios are defined relating to land users' risk aversion, opportunistic and conservational strategies as well as windows of opportunities and risks arising in particularly dry and wet years. Second, a rangeland resilience model is used to simulate ecological impacts considering a set of management scenarios and windows of opportunities and risks. Third, economic impacts are investigated based on vegetation cover dynamics, investment costs and livestock income. Fourth, uncertainty analysis is performed to test the robustness of results. Fifth, socio-ecological effectiveness of management scenarios is evaluated.

The findings of this conceptualisation and modelling strategy demonstrate the utility of considering non-linear ecosystem dynamics to provide essential insights into appropriate timings, climate-induced windows of opportunities and risks and the financial viability of land management investments. They can directly inform cost-effective and efficient progress towards achieving sustainable land management for which Sustainable Development Goal 15 presents a strong demand.

Contents

1 Introduction

There is increasing international momentum to support more sustainable land management, driven by strong global acknowledgement that land degradation can have negative impacts for both climate change and biodiversity (Reed and Stringer 2015). As a result, the concept of Land Degradation Neutrality (LDN) has been formally introduced in global sustainability planning enshrined in the Sustainable Development Goals (SDG target 15.3). LDN refers to a state of zero net land degradation, where 'the amount and quality of land resources necessary to support ecosystem functions and services and enhance food security remain stable or increase within specified temporal and spatial scales and ecosystems' (UNCCD 2016). LDN therefore balances degradation with maintenance and improvement of the land's condition through restoration and sustainable land management (SLM) practices, on- or off-site (Barkemeyer et al. 2015). Restoration implies an ecosystem's return from a degraded to a functional state, while SLM practices aim to prevent the loss of ecosystem functioning and even further improve an ecosystem's functionality. SLM increases an ecosystem's resilience defined as the degree of disturbance it can withstand while remaining within critical thresholds, thus maintaining its core structure and functioning (Holling 1973).

How LDN can be operationalised is currently considered in the work programme of the United Nations Convention to Combat Desertification (UNCCD)'s Science-Policy Interface (SPI) (Orr et al. 2017). The SPI recognises that while LDN is an international policy target, aggregate efforts at smaller scales enable progress. Indeed, countries at the 2015 UNCCD Conference of the Parties agreed to set voluntary LDN targets, acknowledging that 'striving to achieve SDG target 15.3 is a strong vehicle for driving the implementation of the UNCCD' (UNCCD 2015; Decision 3). National level target-setting means that decisions will be needed on where and when best to invest in sustainable land management (SLM) and restoration, depending on the types and status of land degradation in each country. This presents a need for cost-effective decision making and a deeper understanding of the costs of inaction as well as the costs of different types of action.

The recent Economics of Land Degradation (ELD) Initiative report 'The Value of Land' provided a new evidence base that partly addresses this need (ELD 2015). The ELD report has helped policymakers to better appreciate that globally, misuse of vegetation, soils and water has undermined the land's capacity to maintain healthy ecosystems and to provide important ecosystem services, and that this bears a significant cost (ELD 2015). However, land degradation cannot be easily decreased everywhere at acceptable cost: location-specific factors determine costs and success. It requires local socio-ecological causal factors and their interlinkages with broader contextual conditions to be well-understood for interventions to be effective (Suding 2011, Wilson et al. 2011, Diffenbaugh and Field 2013). Moreover, land degradation and climate change are closely linked phenomena. Widespread land degradation is both a driver and consequence of climate change (Reed and Stringer 2016). Degradation can cause stored carbon to be released while also reducing adaptation options and biodiversity. Higher atmospheric greenhouse gas concentrations will increase future climate variability, including more extreme droughts and peak rainfall, potentially driving even more severe degradation and limiting adaptation even further.

While existing scientific knowledge and practical implementation skills can clearly support sustainable land management decisions (Chasek et al. 2015, Stavi and Lal 2015), decision makers lack evidence that can guide them on where and when best to invest in restoration and SLM. In particular, decision-making requires an understanding of key non-linear ecosystem dynamics including critical thresholds, which ecosystems often, but not always, exhibit (Suding and Hobbs 2009). The CASCADE project has as a core objective to improve our understanding and thereby our ability to predict and prevent dryland degradation, and in particular catastrophic shifts, i.e. abrupt, unexpected and often irreversible degradation of dryland ecosystems. Through multi-faceted research activities including fieldwork, ecological modelling and economic appraisal of available management options, data and tools are produced to support management. By applying principles from ecological theory of non-linear ecosystem dynamics, it is possible to inform appropriate investments in recovering and sustaining ecosystems. It is therefore vital that approaches are identified that bring together these concerns to inform sustainable land management decisions and long-term cost-effective and efficient progress towards LDN.

In this report, we demonstrate the utility of considering non-linear ecosystem dynamics to provide essential insights into appropriate timings, climate-induced windows of opportunities and risks and the financial viability of land management investments. Using this conceptualisation, we outline a modelling strategy to evaluate the socio-ecological effectiveness of land management defined here as the potential of a management strategy to help maintain or restore ecosystem services while ensuring land users meet their basic needs. In linking non-linear ecosystem behaviour to an economic evaluation of land management options, we identify opportunities and challenges for cost-efficiently moving towards the LDN target.

2 Conceptualising socio-ecological effectiveness of land management

2.1 Guiding land management through a perspective on non-linear ecosystem dynamics

Widespread failure in ecosystem restoration and degradation prevention, even with massive investments, has underpinned the broad agreement that ecosystems can behave in complex, non-linear ways (Westoby et al. 1989, Scheffer et al. 2001). In contrast to gradual responses, several studies demonstrate that a range of terrestrial and aquatic ecosystems exhibit alternative dynamic regimes and threshold dynamics (Scheffer and Carpenter 2003, Folke et al. 2004, Hirota et al. 2011, Suding 2011). Restoration of ecosystem performance after a decline and prevention of degradation can require considerably stronger efforts in non-linear than in gradually responding systems, but can also benefit from particular opportunities due to non-linear dynamics. Hence, recognition of dynamic ecosystem regimes and threshold dynamics can provide crucial advances to operationalising LDN.

A dynamic ecosystem regime is a region in a state space – also called a basin of attraction – in which an ecosystem develops towards a stable equilibrium (Scheffer et al. 2001). Small disturbances or management impacts can change an ecosystem's state, but the system remains within a given regime and ultimately tends towards the stable equilibrium due to positive internal feedbacks. Dynamic regimes are separated by thresholds defined as boundaries in time and space. At a threshold, a small change in environmental conditions, such as precipitation variability, herbivore pressure, fire frequency or soil fertility, triggers a large change in ecosystem state implying abrupt shifts from one dynamic regime to another. Existence of two alternative dynamic regimes under the same environmental conditions implies hysteresis (Fig. 1a) such that a system's degradation path can strongly differ from its restoration path. Severe disturbances or large management impacts can shift the system over the border of a basin of attraction to an alternative basin of attraction. Changes in environmental conditions exceeding a threshold (T1 and T2 in Fig. 1a) can also trigger a regime shift. Responses manifest as alterations in the productivity and cover of grasses, shrubs or trees and species composition as well as other ecosystem state variables. Such alterations can demand minor or major investments in order that they may be avoided, reduced and/or reversed.

A grass- and a shrub-dominated landscape can be considered as two alternative regimes, which are useful to illustrate shifts in internal feedbacks. Intense livestock grazing can drive degradation shifts from grassland (healthy state) to shrubland (degraded state), leading to decreased fuel connectivity and lack of fire disturbance (Friedel 1991). Without fire, germinating shrubs which are not grazed can survive and outcompete grasses. Under significantly changed feedback mechanisms governed by grass-shrub competition, shrubs can persist even after grazing pressure reductions. Land management needs to reduce grazing intensity in order to improve environmental conditions well beyond the pre-degradation threshold at which the ecosystem shifted to the alternative regime (T2 in Fig. 1a) for the grass-dominated regime to recover. This demonstrates that under hysteresis, ecosystem restoration may require greater efforts and investments compared with a non-hysteretic ecosystem. Changes in environmental conditions may alter regime boundaries and hence the size of a basin of attraction affecting its resilience to disturbance. An increase in basin size can reduce the probability of a regime shift, as the system is less easily driven over a threshold into an alternative regime, implying greater resilience. Likewise, preventive actions such as livestock rotation to reduce grazing pressure are crucial when a healthy grassland approaches a threshold (T2 in Fig. 1a). By increasing the distance to a threshold, this can reduce the likelihood of a shift to the degraded shrub-dominated regime.

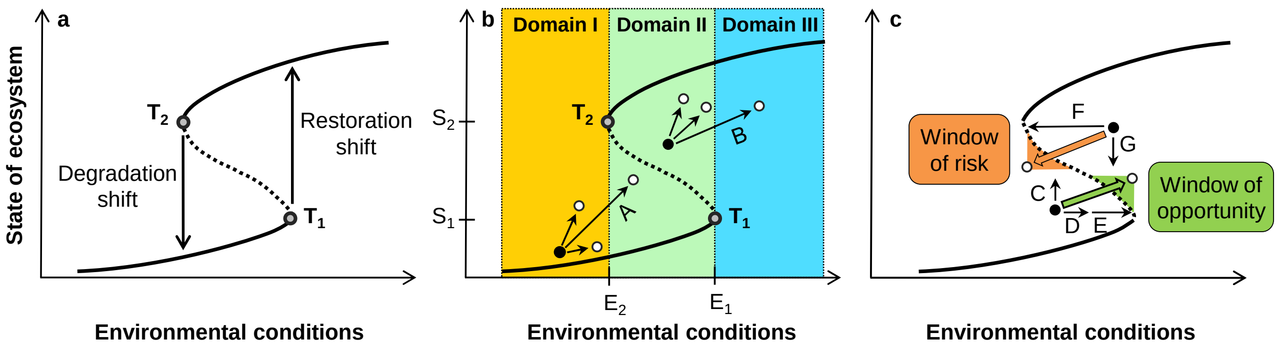

Figure 1 Non-linear dynamics: Dynamic ecosystem regimes and priority situations for LDN interventions (Fig. 1a adapted from Scheffer et al. 2001). (Note: Environmental conditions capture e.g. increase in precipitation or reduction in herbivory and fire frequency. Ecosystem state variables encompass e.g. vegetation cover, density and diversity. Bold lines represent stable equilibria; dotted lines unstable equilibria (borders between basins of attraction). Black dots indicate an ecosystem's current state; white dots show possible management- and climate-induced changes. Fig. 1a shows hysteresis including critical thresholds T_1 and T_2 that distinguish degradation and restoration pathways. Fig. 1b depicts stability domains. The bi-stable Domain II represents priority situations for restorative and preventive actions. Rightward pointing arrows show possible trajectories of land management effects. Movement along arrow A = ecosystem enters bi-stable domain; movement along arrow B = ecosystem leaves bi-stable domain. Fig. 1c illustrates windows of opportunities and risks. Arrows exemplify effects of different types of management practices and external climate drivers: C = seeding, D = reduced grazing pressure, E = extremely wet episode, F = drought and G = deforestation.)

As ecosystems are complex systems displaying high variability in constituting processes and states, there is no single one-dimensional threshold that determines restoration or degradation outcomes. Underlying processes must therefore be adequately captured in threshold models to avoid misinterpretation of conditions under which ecosystems may not be restorable because a historical reference cannot be re-established (Bestelmeyer 2006). Recent work on 'novel ecosystems' highlights the necessity of distinguishing situations in which original states cannot be restored, for example due to constraining interactions between climate change and land use (Hobbs et al. 2013). Land management considering diverse ecosystem functions and multi-dimensional thresholds is a pre-requisite to achieve LDN.

An ecosystem's state relative to critical thresholds can provide key insights into appropriate timings and urgency of restorative and preventive interventions. Ecosystems in a bi-stable situation (Domain II in Fig. 1b) must be prioritised. Experimental evidence shows that arid grasslands in the southwestern United States that degraded to shrub-dominated ecosystems due to intensive grazing can be restored when livestock are excluded (Valone et al. 2002). In the dynamic regime perspective, livestock exclosure induced improved environmental conditions, up to or beyond E1 (see Fig. 1b), enabling a restoration shift. However, shrub-dominated systems may respond slowly to livestock removal as a single management strategy, requiring >20 years before natural grasslands regenerate (Valone et al. 2002). These time lags create delays before management effects materialise highlighting that restoration efforts often require a long-term vision and commitment to be successful.

In a domain with a single degraded regime, Domain I in Fig. 1b, land management principally cannot induce a shift to the healthy (e.g. vegetated) regime due to the absence of an alternative regime. Yet, management such as reduction in grazing pressure and erosion control (especially in regions with erodible soils, highly variable and intensive rainfall and strong winds) is required to avoid a further deterioration of ecosystem state, which would make restoration more difficult. For example, bush encroachment and repeated wildfires affecting abandoned landscapes are known to lead to long-term loss of productivity (Roques et al. 2001, Hill et al. 2008) and the high cost of reversing such degradation is prohibitive (Reed et al. 2015). Similarly, an ecosystem in Domain III cannot shift to an alternative regime, even with a severe disturbance. Here, land management would ideally maintain environmental conditions beyond E1 (Fig. 1b), avoiding the possibility of a regime shift.

2.2 Identifying climate-dependent windows of opportunities and risks

Environmental conditions can strongly vary, opening windows of opportunities and risks for restoration and degradation prevention. Opportunities include exceptionally wet episodes, such as those associated with the El Niño Southern Oscillation (ENSO; Holmgren and Scheffer 2001). Field monitoring and remotely-sensed estimates of tree cover demonstrate that seeding (arrow C in Fig. 1c) and protecting seedlings from herbivores (arrow D in Fig. 1c) at the onset of a rainy El Niño episode (arrow E in Fig. 1c) facilitated tree recruitment and regeneration of extensive dry forests in coastal Peru (Sitters et al. 2012). This fine-tuned dual management strategy was particularly successful in wetter low-lying areas and sandy soils. In contrast to seeding as a single restoration strategy, which was insufficient to induce forest restoration (Sitters et al. 2012), this combination can trigger the passage of thresholds, inducing sudden, long-lasting restoration shifts towards a high vegetation cover regime (green arrow in Fig. 1c). These dual management strategies together with more frequent extreme precipitation events associated with future climate change may generate important windows of opportunities for the recovery of dry forests in some coastal regions in western South America (Holmgren et al. 2013) upon which people's livelihoods rely. Benefitting from such opportunities however requires efficient flood and erosion control measures to avoid land degradation.

Land management to prevent degradation shifts must consider windows of risks when typical degradation drivers, such as drought and deforestation, interactively affect an ecosystem's state. For example, dynamic modelling suggests that combined drought and deforestation can result in more widespread shifts from rainforest to savanna regimes in the south-eastern Amazon basin than those triggered by either drought or deforestation (orange arrow in Fig. 1c; Staal et al. 2015). Here, both drought and deforestation favour grass invasion which increases flammability, decreasing the rainforest's fire resilience and therefore increasing the probability of a degradation shift to a savanna regime. As the combined effects of drought and deforestation can move a forest out of Domain III into the bi-stable Domain II (see Fig. 1b), land management is required to stabilise internal feedbacks (e.g. preventing fragmentation of forest canopy and grass invasion) in order to reduce the probability of a degradation shift. This underlines the importance of policies and mechanisms to prevent deforestation, particularly when future climate change is associated with more frequent and intense droughts (Malhi et al. 2008) and coupled degradation drivers limit the boundaries within which forests can be sustainably managed (Scheffer et al. 2015).

2.3 Deciding when to invest

For financial viability of investments, stability domains (Fig. 1b) matter greatly, as does the opening of a climate-dependent window of opportunity or risk (Fig. 1c). Cost-benefit analysis is traditionally applied to assess expected financial impacts of land management interventions (Qadir et al. 2014, Giger et al. 2015, Baptista et al. 2016). While the feasibility of interventions may depend on a variety of criteria, a major assumption is that a land manager would invest only in those measures whose expected returns are positive. It is however often difficult to anticipate the effects of land management with certainty (Suding 2011, Wilson et al. 2011, Nilsson et al. 2015).

A global meta-analysis of ecosystem restoration depicts large variations in benefit-cost ratios across a range of biomes including grasslands, forests and wetlands (De Groot et al. 2013). Similarly, a global analysis of successful SLM cases reveals great differences in the costs and benefits that stakeholders perceived in establishing and maintaining SLM measures depending on management type, region and area size (Giger et al. 2015). Further differentiation of costs and benefits according to varying degradation levels, environmental conditions and climate risks and opportunities is essential to inform investment decisions. Clearly, a better understanding of dynamic ecosystem regimes can advance decision making on investment in land management, particularly concerning large-scale restoration and SLM programmes. Here, timing is a key factor: investment costs are required immediately and maintenance costs may pose an additional strain on resources in the initial years following an investment, whereas the later the benefits are anticipated to occur, the less they are valued at the time of establishment of SLM programmes. In cost-benefit analysis this is captured through discounting of future costs and benefits. In the following paragraphs, we discuss the effects and cost-effectiveness of seeding as a key restoration measure to illustrate major differences in the costs and benefits arising from action across the stability domains. Seeding makes for a good illustrative case as it directly affects an ecosystem's state and its success may vary with environmental conditions. Other restoration measures such as fencing off degraded land can be cheaper and equally effective but do not affect an ecosystem's state directly.

Considering a degraded ecosystem in a bi-stable domain (Domain II in Fig. 1b), a priority situation for restoration, investments coinciding with a window of opportunity have greater chances of succeeding and generating higher gross benefits (green line and area in Fig. 2b) than those outside such a window of opportunity. This also raises chances of a positive return on investment. Insights from germination biology can support the evaluation of soil moisture and weather conditions, especially in regions with a highly variable and changing climate (Broadhurst et al. 2016). When seeding and improved environmental conditions are insufficient for the system to cross a threshold, recurrent costs to maintain the achieved improvement and prevent a degradation tendency are incurred while waiting for a new window of opportunity (see plateau in green line and repeated sharp decline in grey line during early years in Fig. 2b). Once an ecosystem has passed a critical threshold during a new window of opportunity, vegetation cover increases naturally without any further maintenance costs (increase in green and grey lines and areas in Fig. 2b).

In contrast, improving a severely degraded ecosystem under adverse environmental conditions (Domain I in Fig. 1b) is expensive and takes longer to materialise (grey line and area in Fig. 2a). Here, we illustrate a case in which site preparation did not immediately result in vegetation improvement but disturbed the existing vegetation and led to an initial decline in vegetation cover. This decline implies a lack of benefits in the first years even with additional maintenance (see early negative values of grey line and area in Fig. 2a). As ecosystems tend to return to the lower stable equilibrium (i.e. degrade) if situated above the lower branch of the hysteresis curve in Domain I, recurrent maintenance costs arise (resulting in repeated sharp decline in grey line in Fig. 2a), as in Domain II. In the case depicted in Figure 2a, maintenance costs are exemplified to occur every other year (repeated sharp decline in grey line in Fig. 2a) reflecting variability in rainfall and vegetation establishment. However, such investments to sustainably improve a degraded ecosystem may not be economical as shown by both total negative present and future net benefits (grey line and area in Fig. 2a).

Investment in a healthy ecosystem that tends to improve naturally (located below the upper branch of the hysteresis curve in Domain III, Fig. 1b) can increase the speed of improvement (pronounced slope in light blue line and area in Fig. 2c), usually at modest investment cost. Net benefits only arise at an early stage and vanish once the ecosystem would have reached the healthy stable equilibrium without the intervention (grey line and area in Fig. 2c). The healthy stable equilibrium that is reached will be the same with and without investment. Here, the acceleration of restoration as the ecosystem develops towards the higher stable equilibrium (healthy regime) needs to be high enough to render investment attractive.

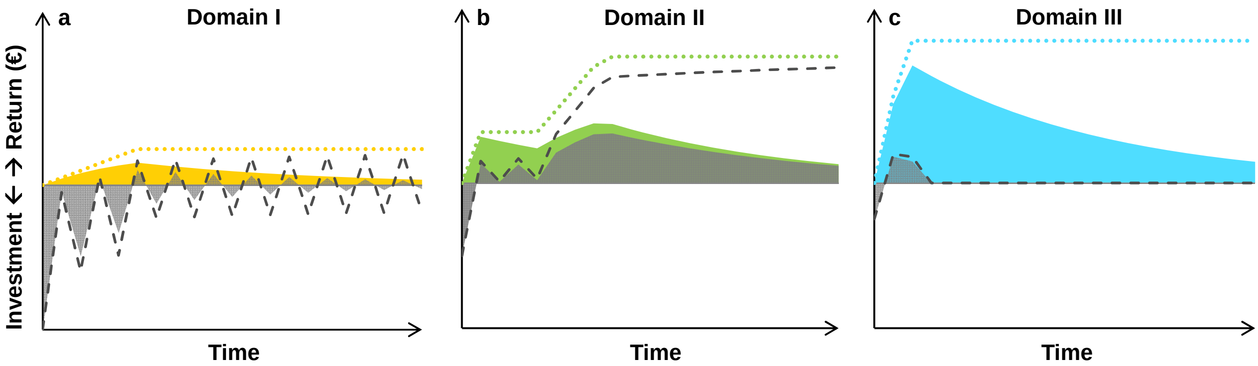

Figure 2 Cost-efficiency of management interventions dependent on stability domains (Fig. 1b) and window of opportunity (Fig. 1c). Areas represent discounted (present) value of investment costs and benefits, while lines represent future values. Coloured areas refer to gross present benefits. Grey areas refer to net present benefits (i.e. subtracting from gross benefits the intervention costs and any benefits that would have been obtained without the intervention). Coloured lines refer to gross future benefits and grey lines to net future benefits. Gross future benefits depend on productivity levels which vary between stability domains. Fig. 2b indicates management effects concurring with a window of opportunity. Note the declining level of initial investment costs and recurrent maintenance costs going from Domain I (Fig. 2a) to Domain III (Fig. 2c).

SLM as a preventive measure has in the long run frequently been found to be cheaper than ecosystem restoration (ELD 2015, Nkonya et al. 2016). However, investment costs need to be considered in conjunction with expected benefits, risk of failure and the passage of thresholds, meaning that higher upfront costs might in the long run be offset by restoration benefits (Zahawi et al. 2014, Gilardelli et al. 2016). Long-term field experiments with controlled management and environmental conditions are crucial to test and refine important ecosystem properties and feedbacks captured in models to advance existing and build new theories and inform decision making (Foster et al. 2016). They are key for improving our often incomplete knowledge about the socio-ecological dynamics that facilitate or constrain the implementation of specific land use strategies (Sietz and Van Dijk 2015) and evaluating threshold behaviour (Suding and Hobbs 2009). This is a pre-requisite for land-based management decisions that are well-suited to address heterogeneity in global sustainability challenges such as loss of biosphere integrity, livelihood insecurity and socio-ecological vulnerability (Sietz 2014, Steffen et al. 2015, Kok et al. 2016).

In the face of ever-present uncertainty, learning through monitoring of key processes and feedbacks, scenario analysis and adaptive management is central for decision making and inherently linked to resilience thinking. Efforts aimed at increasing response diversity may be particularly beneficial to address uncertainty in future disturbances and environmental conditions (Suding and Hobbs 2009). Response diversity describes the variety and heterogeneity of species, ecological communities and feedbacks but also managerial processes, allowing ecosystems and human flexibility to respond in various ways and prepare for anticipated effects of disturbances and ongoing change. High response diversity enables some system components or functions to persist, recuperate or transform when disturbed, while others may experience damage or vanish. Further, as costs and benefits associated with alternative ecosystem regimes can differ significantly depending on land users' perceptions, demands and expectations (James et al. 2015, Tarrason et al. 2016), stakeholder involvement is paramount in decision making.

3 Modelling socio-ecological effectiveness of land management

Costs and effectiveness are the most important considerations in land management. Their evaluation helps to select the best out of several management options. To assess the socio-ecological effectiveness of land management, we evaluate insights into ecological effects and financial attractiveness. This approach merges major aspects of two economic methods commonly used to inform decision-making on land management: cost-effectiveness and cost-benefit analysis.

In cost-effectiveness analysis, relative costs and outcomes (e.g. ecological effects) of various management strategies are compared. This implies the definition of a specific target, such as a critical threshold in vegetation cover, as a criterion against which each strategy's performance is evaluated. However, cost-effectiveness analysis is sensitive to subjective decisions in target setting necessitating robust empirically grounded choices that are relevant in a particular management context. For example, 40% vegetation cover was determined a critical threshold to prevent erosion and maintain important ecosystem services in CASCADE study sites (see D7.3).

In contrast, cost-benefit analysis requires all effects to be expressed in monetary values considering the time value of an investment. It compares management options according to cash flows, that is to say time series of monetary costs and benefits, taking into account a discount factor. In CASCADE grazing sites, pasture and livestock productivity are of major interest to land users and can be monetarised in a straightforward way. Yet, cost-benefit analysis has often been criticised regarding the desirability and ways to attribute monetary value to ecosystem services, such as aesthetic, cultural and recreational services and its exclusively forward-looking nature (Anderson et al. 2016).

Despite these concerns, a key question however remains: how can we inform decisions on sustainable land management? Given the complexity of management impacts and potentially long time scales before effects materialise, this question needs to be addressed through scenario studies that establish relevant assumptions and simulate the socio-ecological impacts of various management options. Although land management is a multifaceted process, it can be assumed that land users would consider to adopt only those strategies that yield positive expected returns expressed as costs and benefits and that policy makers' decisions would depend on cost-effectiveness considerations. A combination of cost-effectiveness and cost-benefit analysis is therefore used in this modelling strategy to evaluate ecological and economic impacts of selected management scenarios (Fig. 3).

Ecological impacts are simulated applying an advanced rangeland resilience model that captures the effects of livestock grazing on vegetation dynamics (Kefi et al. 2007, Schneider and Kefi 2016). It allows to assess the impacts of interacting grazing pressure and environmental conditions (e.g. aridity) on ecosystem stability considering time frames that are relevant for land users (e.g. annual time steps over a period of 10-30 years). In particular, this model simulates spatially heterogeneous grazing impacts caused by local facilitation (e.g. shading and water retention when plants grow in patches), associational resistance (i.e. joining physical defences such as spines and thorns) and competition for scarce resources (e.g. water and nutrients). The model development was informed by ecological conditions observed in Randi, Cyprus but can be adjusted to conditions found in other grazing sites. Management strategies such as controlled grazing and supplementary feeding that may imply long-lasting effects and regime shifts receive particular attention. We assume that this type of management directly affects vegetation cover (state of ecosystem) and livestock density captured in the model which in turn influence the chance of restoration in a degraded ecosystem and the risk of degradation in a healthy ecosystem. The ecological assessment is complemented by insights into investment costs (e.g. costs to purchase additional fodder) and income through livestock production (e.g. meat and milk). In contrast, the models investigating fire and drought effects on forests reveal dynamics in the distribution of major functional types including pines and oaks (D6.1). They demonstrate long-term vegetation dynamics including species succession and recurrence after stress exposure. Due to the specific parameters used, these models do not sufficiently link to the management strategies applied to reduce fire risk and damage in forest stands (e.g. conservational and traditional logging). In addition, the models' focus on long-term impacts (e.g. >200 years) goes far beyond land users' planning and decision-making horizons. Therefore, the modelling strategy presented here refers to livestock grazing in the Mediterranean drylands in southern Europe.

Overall, a 5-step modelling approach is presented here to analyse the socio-ecological effectiveness of land management. First, management scenarios are defined relating to land users' risk aversion, opportunistic and conservational strategies as well as windows of opportunities and risks arising in particularly dry and wet years. Second, the rangeland resilience model is parametrised in such a way that it represents observed ecological conditions and used to simulate ecological impacts (vegetation cover dynamics) considering the management scenarios and windows of opportunities and risks. Third, economic impacts are investigated based on vegetation cover dynamics, investment costs and livestock income yielding cash flow series. Fourth, sensitivity analysis is performed to test the robustness of results. Fifth, socio-ecological effectiveness of management scenarios is evaluated and discussed with stakeholders to validate the findings. This approach is outlined in more detail in the following sections.

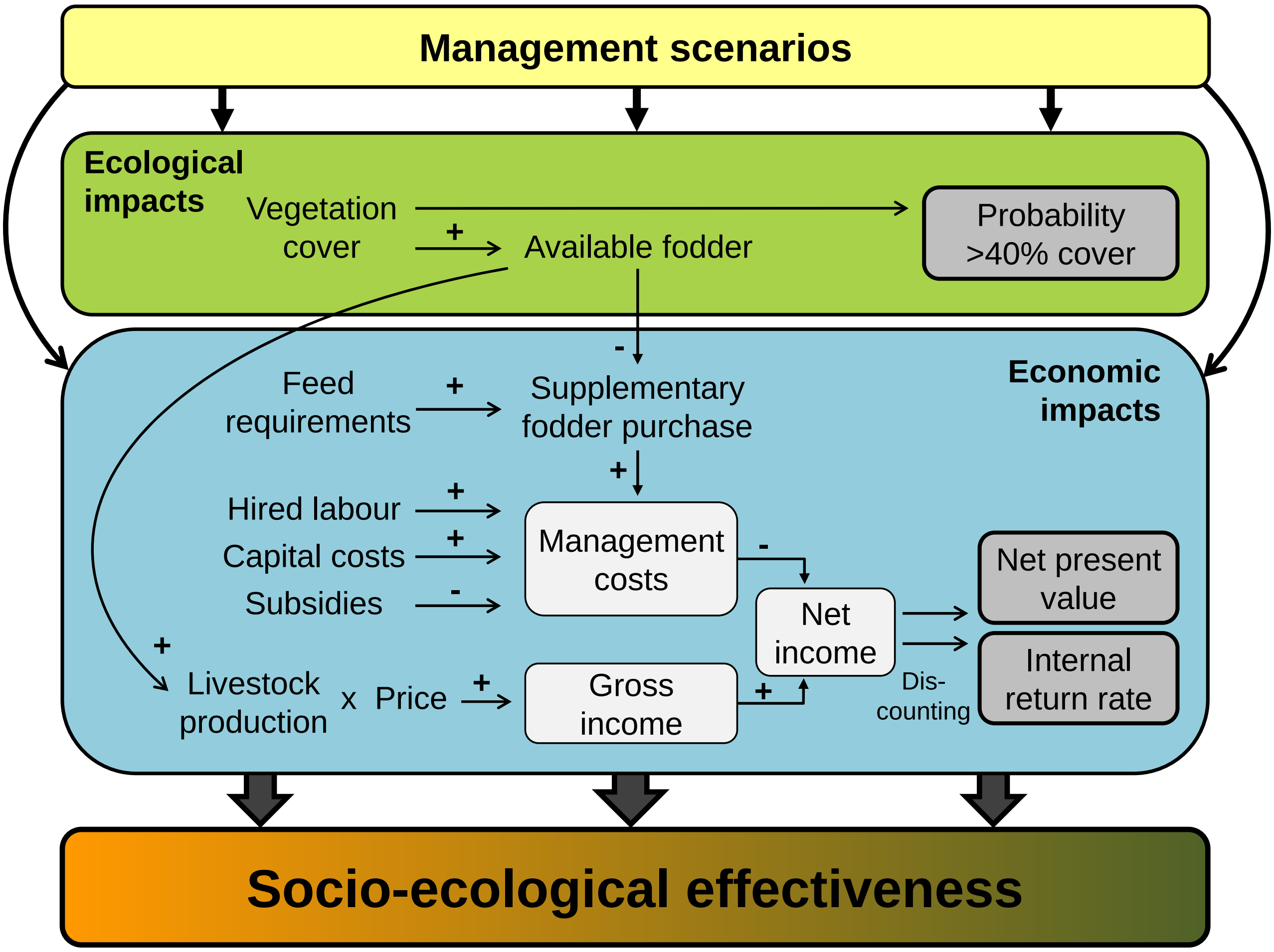

Figure 3 Overview of the socio-ecological modelling approach presented in this report. The five steps of this modelling approach encompass Step 1) Defining management scenarios, Step 2) Assessing ecological impacts of land management, Step 3) Estimating economic impacts of land management, Step 4) Uncertainty analysis considering ecologic and economic dimensions and Step 5) Evaluating socio-ecological effectiveness. (Note: +/- indicated next to arrows symbolise positive and negative effects).

3.1 Step 1: Defining management scenarios

Drylands are marginal regions where low and variable rainfall, infertile soils and land degradation often constrain agricultural productivity (Safriel et al. 2005, Hein and De Ridder 2006, Zika and Erb 2009). Livestock grazing plays a major role in land use and income generation in drylands but is also one of the major desertification drivers globally (Asner et al. 2004, Safriel et al. 2005). Hence an acute sustainability challenge in land management is to adjust livestock pressure to the marginal dryland conditions. Adaptive grazing strategies have been developed in drylands including destocking and restocking in response to low and variable rainfall and associated vegetation dynamics. The term 'adaptive' indicates that land users base their decisions on ecological and economic considerations such as vegetation cover, pasture productivity, precipitation at the start of the growing season or capital available to purchase supplementary fodder. In adjusting stocking rates, land users take into account a land's grazing capacity. The stocking rate depicts the ratio of livestock and available fodder while grazing capacity refers to the number of livestock that the vegetation can sustain. Hence, sophisticated balancing of stocking rate and grazing capacity is key to sustainable dryland management. In this modelling strategy, we consider common adaptive management strategies including opportunistic and conservational grazing management in combination with varying degree of risk aversion.

When using opportunistic strategies, pastoralists adjust livestock density to pasture productivity in each year (Westoby et al. 1989, Behnke et al. 1993). This means that the stocking rate is adjusted to the land's grazing capacity. This allows pastoralists to directly benefit from annual productivity changes yet without considering the time span necessary for the vegetation to recover sufficiently. Other land users may follow conservational strategies excluding livestock from some part of their land in wet years while fully grazing the land in other years (Frank et al. 2007, Müller et al. 2007, Quaas et al. 2007). This 'resting' implies that stocking rates remain below the grazing capacity in wet years facilitating vegetation recovery and potentially higher stocking rates in a relatively short time. In addition, management depends on the degree of risk aversion. An important aspect is the perceived likelihood that productive land would degrade or degraded land would recover. Observed vegetation cover serves in this modelling approach as a proxy for degradation risk with lower cover indicating higher risk of potentially irreversible degradation. For example, land users who are not risk averse may continue to graze a given livestock number on a degraded, sparsely vegetated pasture (e.g. 30%) assuming that this vegetation cover could be maintained. Late destocking may however cause long-lasting or irreversible degradation. In contrast, more risk averse land users may reduce stocking rates earlier (e.g. 40% vegetation cover) possibly allowing the pasture to recover. The management scenarios considered in this modelling approach are summarised in Table 1.

Table 1 Scenarios of adaptive land management

Management scenario		**Description**	**Start conditions**	
			Degraded sites	Restored sites
Baseline scenario	Least risk aversion	If vegetation cover smaller 30% → reduce number of animals grazed on pasture to half	X	X
Scenario 1 (S1)	Higher risk aversion	If vegetation cover smaller 40% → reduce number of animals grazed on pasture to half	X	X
Scenario 2 (S2)	Resting in wet years and extreme risk aversion	In wet years or if vegetation cover smaller 60% → reduce number of animals grazed on pasture to half	---	X

Taken together, the management scenarios are designed to test the relevance of land users' risk aversion and resting periods. Outcomes are evaluated over a 10-year period assuming that land users would expect improvements to materialise within this range. In a land users' perspective, a major question is: how effective is a management strategy in safeguarding sufficient pasture in the following year while ensuring the restoration of a degraded pasture in the next 10 years? Or: Is there a risk that a healthy pasture may degrade under a certain management strategy in the next 10 years? Extreme environmental conditions can significantly alter ecological conditions and the potential for restoration and degradation including regime shifts (see Sections 2.2 and 2.3). Hence, management scenarios include windows of opportunities and risks, which consist of wet and dry years, both of which on average occur twice in 10 years in the Mediterranean. This frequency reflects observed climate variability in the Mediterranean region (Sousa et al. 2011, Vicente-Serrano et al. 2014).

All management scenarios imply annual decisions on livestock destocking or restocking. In the case of destocking, part or all of the livestock is kept in stables, rather than being sold, and entirely fed with supplementary fodder, whereas restocking implies that those animals are brought back to graze on the pastures. Ecological impacts may manifest themselves only gradually and longer adjustment periods may be required to yield significant changes. For example, low livestock pressure in a single year may not be sufficient for a severely degraded vegetation to recuperate. Longer reduction of livestock pressure may be necessary to trigger the desired vegetation recovery demanding longer-term expenditures, e.g. to purchase supplementary fodder when keeping livestock in stables.

3.2 Step 2: Assessing ecological impacts of land management[1]

After defining the management scenarios and windows of opportunities and risks in Step 1, the rangeland resilience model (Kefi et al. 2007, Schneider and Kefi 2016) needs to be parametrised. This is done in relative terms according to ecological conditions observed at the study sites in Randi/Cyprus, Crete/Greece, Castelsaraceno/Italy and Albatera/Spain reflecting CASCADE's potential for comparative analysis. The ecosystems at the CASCADE study sites exemplify particular environmental conditions (such as aridity and soil characteristics) and related feedback mechanisms that can be differentiated in the model simulations in a hypothetical way. Representative theoretical ecosystem properties are used to define three types of ecosystems depicting contrasting environmental conditions and vegetation states. These include (i) extremely marginal environment and sparse vegetation cover, (ii) marginal environment and medium vegetation cover and (iii) better environment and higher vegetation cover. Covering broad variations in environmental conditions and ecological feedbacks enables systematic analysis of the range and limitations of a specific management scenario to restore or maintain healthy pastures. This can improve our principle understanding of the coupled nature of pasture conditions, strategic management choices, impacts on ecosystem functioning and sustainability outcomes. The types of ecosystems represented are outlined below and summarised with the respective model parameters in Table 2.

Extremely marginal and sparse vegetation cover:

- This ecosystem type is assumed to display very marginal environmental conditions such as extreme water scarcity and shallow soils. In these harsh dryland conditions, local facilitation, associational resistance and strong local competition are considered important factors affecting vegetation dynamics (Kefi et al. 2007). Local facilitation means that plants accumulate organic matter, provide shade and retain water, improving growing conditions and support other plants' recruitment in their vicinity. Along a gradient of low to high rainfall in semi-arid ecosystems, field studies have shown shifts from competition to facilitation and back to competition (Maestre and Cortina 2004, Maestre et al. 2005). Therefore, high local competition and low facilitation are assumed for this ecosystem type. Moreover, associational resistance refers to the joint protection when plants grow in patches decreasing the chance of being grazed by livestock (Schneider and Kefi 2016). It is particularly important in sparsely vegetated landscapes. The strongly degraded study sites in Randi, Cyprus and Albatera, Spain depict high aridity (Trabucco and Zomer 2008), shallow soils and very low vegetation cover due to overgrazing (D5.2) providing empirical support for these conditions.

Marginal and medium vegetation cover:

- In this type of ecosystem, environmental conditions are still considered to be marginal, including low precipitation and eroded soils, but more favourable than in the previous ecosystem type. The slightly more favourable environment is supposed to support intermediate vegetation coverage. Hence, ecological feedbacks related to local facilitation are assumed more important, while local competition and associational resistance are considered less important for vegetation dynamics than in the previous ecosystem type (Maestre and

[1] The procedure to assess ecological management impacts described in this section has been developed together with Florian Schneider and Sonia Kefi (WP6).

Cortina 2004, Maestre et al. 2005). These circumstances represent core ecosystem properties and feedbacks observed at the study site in Messara, Greece where aridity, water overuse and partial overgrazing have led to land degradation including reduced vegetation cover (Trabucco and Zomer 2008, D5.2).

Better environment and higher vegetation cover:

- This type of ecosystem is considered to represent better environmental conditions including higher humidity and more favourable soil properties that support higher vegetation cover in the absence of livestock grazing. Yet in more densely vegetated landscapes, plant recruitment is assumed to depend to a greater extent on facilitation and less on competition for limited resources (Maestre and Cortina 2004, Maestre et al. 2005, Kefi et al. 2007) and associational resistance than in the previous ecosystem types. The more humid and more densely vegetated study site in Castelsaraceno, Italy (Trabucco and Zomer 2008, D5.2) reflects major aspects of these circumstances.

Table 2 Types of ecosystems captured in the model simulations. (Note: All parameters are scaled to 0-1 range reflecting minimum and maximum values.)

Type of ecosystem	**Ecosystem's properties**	**Model parameters**				**Representative study site**
		Environ-mental quality (b)	Local facili-tation (f)	Local compe-tition (c)	Associa-tional resistance (p)	
Extremely marginal environment and sparse vegetation cover	Extremely dry and very shallow/infertile soils, low facilitation and strong local competition	Very low (0.1)	Low (0.2)	High (0.8)	Very high (0.9)	Randi/Cyprus (degraded site) (Albatera, Spain*)
Marginal environment and medium vegetation cover	Dry and shallow/infertile soils, low-medium facilitation and medium-high competition	Low (0.3)	Low-Medium (0.4)	Medium -High (0.6)	High (0.7)	Crete/Greece Randi/Cyprus (reference site)
Better environment and high vegetation cover	More humid and deeper/more fertile soils, medium-high facilitation and low-medium competition	Medium (0.5)	Medium -High (0.6)	Low-Medium (0.4)	Medium (0.5)	Castelsaraceno, Italy

* The study site in Albatera, Spain is hardly grazed anymore due to severe degradation of vegetation and soils.

The model is parametrised using a 0-1 parameter scale enabling comparability across study sites. This allows us to qualitatively discuss the effects of parameter changes in relation to minimum and maximum values which may useful to frame policy recommendations or explore the impacts of future development scenarios. For example, we may explore in which situation a certain EU policy might provoke most change in relative terms. However, a given parameter value can have very different real-world values across the study sites meaning that a large relative change may mean only a small (absolute) change in real world processes and outcomes. This needs to be taken into account when evaluating the model results.

To finalise the model parametrisation, the livestock density parameter is chosen in such a way that the modelled vegetation cover corresponds to the observed perennial vegetation cover (data collected in regional survey) reflecting distinct degradation levels in each study site (e.g. reference and degraded sites). The actual livestock density and share of palatable vegetation are then used to estimate site-specific conversion factors for observed and modelled livestock density assuming that 1 livestock unit (LU) corresponds to 1 cattle, 6 sheep and 4 goats.

Based on this parametrisation, the rangeland resilience model is used to simulate vegetation cover dynamics for the various management scenarios including two stochastic dry and wet years to simulate the effects of extreme environmental conditions. Simulations are performed separately for each management strategy using the same environmental conditions (e.g. precipitation and soil depth) and all results need to be compared with the baseline situation. These model simulations provide time series of ecological management impacts without yet capturing any socio-economic impact. The timing of ecological effects is important in the economic evaluation conducted in Step 3 (Section 3.3). Step 2 generates vegetation cover dynamics with annual resolution for time series of 10 years (see example output in Fig. 4). Evaluation of ecological management impacts include critical levels of vegetation cover (probability ≥40%; see example output in Fig. 5) as defined in WP7 (D7.3), reversibility of degradation and potential restoration/degradation shifts. All example outputs presented in Figs. 4-8 provide a coherent overview of findings based on a consistent set of management scenarios used to simulate vegetation cover and economic dynamics.

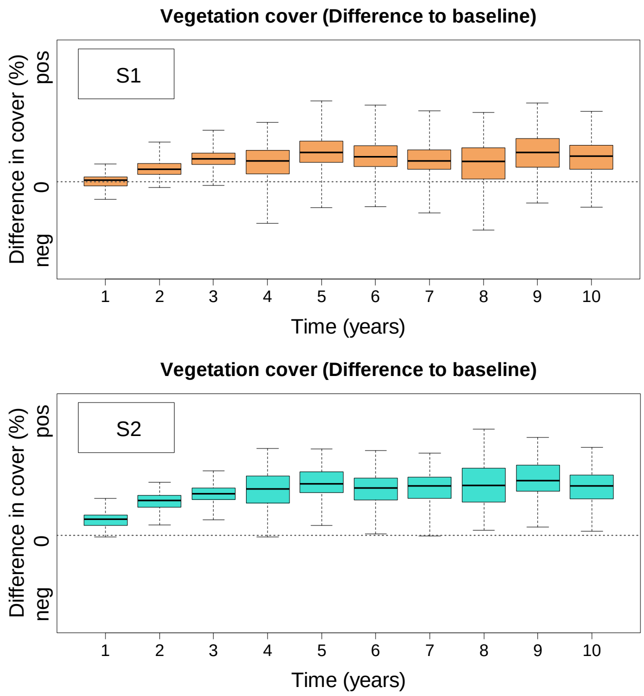

Figure 4 Example output: Vegetation cover time series for two management scenarios S1 and S2 (e.g. higher risk aversion and resting in wet years and extreme risk aversion). (Note: Positive and negative deviations from baseline scenario are shown. Zero indicates the vegetation cover in the baseline scenario. Colour coding refers to Tab. 1. Box boundaries denote the 25th and 75th percentiles of data while whiskers indicate minima and maxima. The line in a box depicts the median value.)

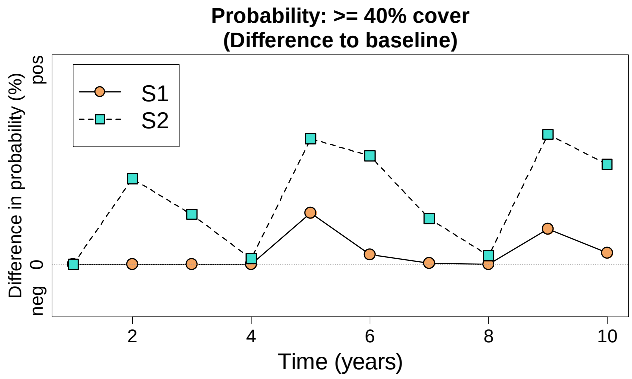

Figure 5 Example output: Differences in probability of reaching ≥40% vegetation cover over time for the same two management scenarios S1 and S2 used to generate all example outputs. (Note: Positive and negative deviations from baseline scenario are shown. Zero indicates the probability in the baseline scenario.)

3.3 Step 3: Estimating economic impacts of land management

In this step, the ecological model outputs derived in Step 2 are subject to an economic analysis. For this, the modelled vegetation cover is translated to livestock productivity. The rangeland resilience model provides time series of perennial vegetation cover as an output variable describing dynamics in an ecosystem's state. Similar to the ecological impacts, costs and benefits are assessed for yearly time steps over a 10-year period resulting in annual cash flow series, and management scenarios are compared to the baseline scenario. As a basis for the economic assessment, regional input data have been collected in an expert survey (Appendix 1) conducted with regional land users and scientists within WP8 covering all CASCADE grazing sites to support the modelling process. These include empirical factors to convert vegetation cover to biomass and data on economic costs and benefits associated with livestock management.

In linking model outputs and economic considerations, perennial vegetation cover is converted to available biomass, i.e. pasture productivity, using site-specific empirical conversion factors. For example in Castelsaraceno, Italy, the biomass conversion factor is 55kg dry matter/ha per 1% perennial vegetation and 80-98% of vegetation is palatable. This is comparable with sub-desertic steppes in North Africa where 1% perennial vegetation cover correlated with 43 ± 3.6 kg dry matter perennial phytomass/ha and 80-100 ± 25kg for alfa grass steppes only (Le Houérou 1987). The original rangeland resilience model simulated the effects of grazing on vegetation cover without considering a pasture's grazing capacity and feedbacks on livestock productivity. In this work package, the model has been extended to deliver information on grazing capacity considering the empirical biomass conversion factors. This provides a necessary link to assess dynamics in livestock productivity.

Livestock is usually maintained by pasture production (available biomass) and supplementary fodder purchased on the market e.g. in winter season. The lower the pasture productivity, e.g. due to degradation, the more supplementary fodder needs to be purchased throughout the year. The sum of pasture production and supplementary fodder needs to meet the total feed requirements considering animal-specific requirements and the number of livestock grazed on a plot. Feed requirements vary with animal weight, age and activity level. These aspects remain beyond the scope of this modelling strategy to keep an appropriate balance with the level of detail captured in the management scenarios and fundamental ecological processes.

For livestock grazed on pasture land, the difference between total feed requirements and fodder available through pasture production is assumed to be purchased contributing to management costs. For livestock kept in stables, e.g. when reducing livestock pressure in normal years and resting in wet years, we assume that these animals are entirely fed on purchased fodder.

The costs for fodder purchase are summed with the costs for hiring labour (e.g. livestock drover and cheese production) and other expenditures (e.g. land rent, veterinary consultation and stable construction) minus received subsidies to determine total management costs. Moreover, gross income is derived from total livestock produce (e.g. meat and wool) and the respective prices received for sale. From this gross income, management costs are subtracted to determine the net income from implementing a management strategy compared with the baseline situation. Net income dynamics are presented as annual cash flow series over 10 years (see Fig. 6 for example output).

Finally, all cash flow series are discounted to reflect the cost of capital. This allows comparing those management strategies that require similar expenditures and are equally effective. The strategy that generates the effects earlier is clearly more attractive. The discount factor (e.g. 10%) is estimated based on the costs that are incurred when land users borrow money. This is motivated by the consideration that borrowing money to invest in land restoration should yield at least the amount borrowed plus interests and transaction costs. The sum of discounted cash flows yields net present values (NPV; see Fig. 7 for example output). NPV is an appropriate economic indicator since management costs reach similar magnitudes in the scenarios considered in this modelling approach. If a management scenario results in negative NPV, it would be important to identify whether there is a discount factor at which the management becomes attractive. Corresponding to a situation in which NPV is zero, this information is expressed as internal rate of return (IRR) useful to inform financial policies on credit conditions. An additional key aspect to inform policies relates to the price of fodder that leads to a positive NPV. This is determined using a gradient of fodder price.

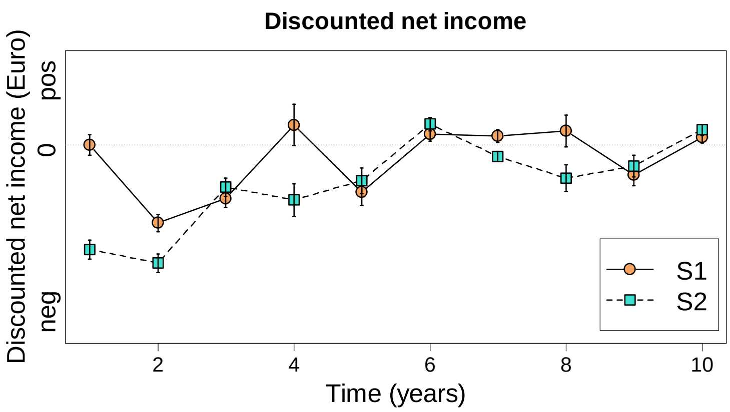

Figure 6 Example output: Discounted net income over time for the same two management scenarios S1 and S2 used to generate all example outputs. (Note: Positive and negative deviations from baseline scenario are shown. Zero indicates the discounted net income in the baseline scenario. Error bars show standard deviation.)

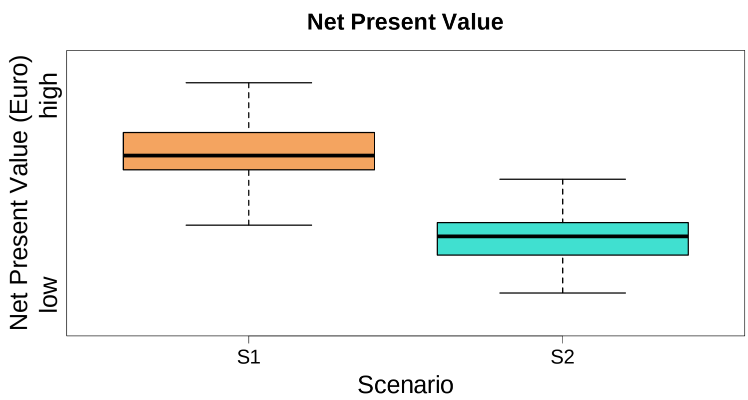

Figure 7 Example output: Net present value for the same two management scenarios S1 and S2 used to generate all example outputs. (Note: Deviations from baseline scenario are shown. Box boundaries denote the 25th and 75th percentiles of data while whiskers indicate minima and maxima. The line in a box depicts the median value.)

3.4 Step 4: Uncertainty analysis

There are three major dimensions within the model that require uncertainty analysis of future outcomes of land management. First, the spatial vegetation structure can vary according to the initial distribution of vegetation patch size. Potential variations are captured by repeating model simulations using 200 landscapes with random initial vegetation structure. Second, environmental variability can trigger long-lasting changes in vegetation cover, particularly when an ecosystem is approaching a critical threshold. Variability in environmental conditions is implemented stochastically over time. Third, the stochasticity in vegetation structure and environmental variability implies uncertainty in discounted net income and net present value.

The resulting ecological and economic variations are quantified in three ways: a) by calculating the probability of reaching ≥40% vegetation cover (see Fig. 5), b) by indicating the degree of data dispersion – graphically displayed in boxplots of vegetation cover and net present value – without making an assumption about the statistical data distribution (see Figs. 4 and 7) and c) by determining the standard deviation – displayed as error bars of discounted net income (see Fig. 6). Taken together, the boxplots and error bars provide estimates of how future vegetation cover and income vary according to variability in vegetation structure and environmental conditions. Moreover, results may be associated with uncertainty depending on site-specific fluctuations, land users' perceptions and memory bias in recalling expenditures. Since this modelling strategy focuses on typical land management situations to investigate principle future development trajectories, these variations remain unconsidered and may be included in an extended future assessment.

3.5 Step 5: Evaluating socio-ecological effectiveness

Steps 2 and 3 served to determine ecological and economic impacts for particular types of ecosystems and a set of management scenarios (see Step 1). In this step, ecological and economic insights are integrated to evaluate the socio-ecological effectiveness of management. This evaluation is based in the following criteria:

- If a management scenario is effective in preventing or reversing degradation (i.e. high probability ≥40% cover) and its NPV is high, it is effective in socio-ecological terms and likely to be attractive to land users and policy makers.
- A management scenario that effectively prevents or reverses degradation (i.e. high probability ≥40% cover) but has a negative or low NPV (e.g. S2 in Fig. 8) indicates

that policy incentives such as subsidies would be useful to increase land users' motivation to implement this type of management.

- If a management scenario yields high NPV but cannot effectively prevent degradation (i.e. low probability ≥40% cover; e.g. S1 in Fig. 8), it would seem acceptable only from a profit-maximisation perspective that focusses on short-term goals. However, such a scenario does not contribute to sustainable land management.
- If both NPV and degradation prevention effectiveness are low, a management scenario is socio-ecologically ineffective and should also be disregarded in efforts aimed at achieving sustainable land management.

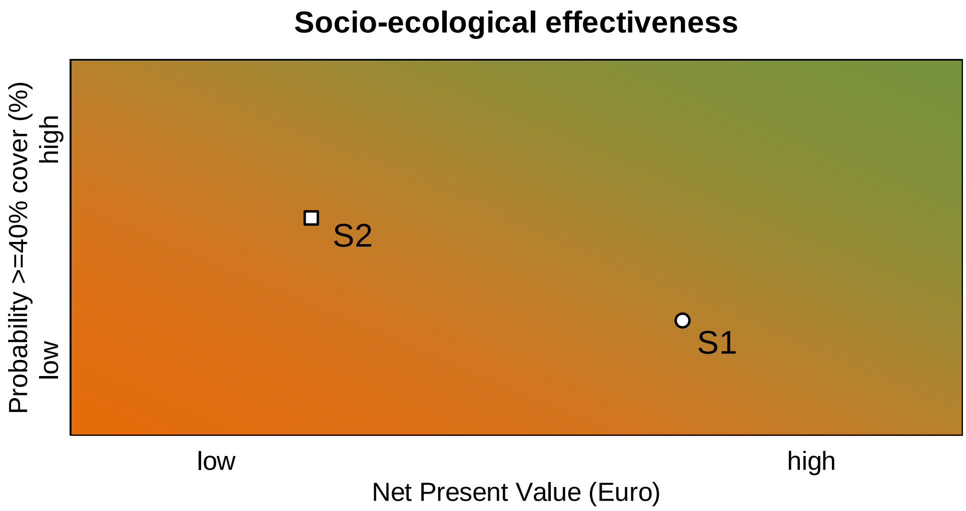

Figure 8 Example output: Socio-ecological effectiveness for the same two management scenarios S1 and S2 used to generate all example outputs. (Note: Graduation from orange to green background colour indicates gradient from lower to higher socio-ecological effectiveness.)

This socio-ecological evaluation together with the modelled vegetation dynamics and economic findings was discussed with stakeholders at regional workshops to test the robustness of findings in particular regional contexts. Potential disagreement between the findings of this modelling strategy, field observations and land users' as well as decision-makers' experience were explored and can lead to refinements in the ecological and economic assessments. The results can be found in D8.3.

Finally, the evaluation of socio-ecological effectiveness of land management can be up-scaled to a regional level representing the Mediterranean drylands in southern Europe. This requires capturing the full range of environmental conditions and livestock density observed in Mediterranean drylands. Suitable spatially-explicit indicators include aridity index (Trabucco and Zomer 2008) and livestock density data (Robinson et al. 2014) available at high-resolution for the Mediterranean region (Figs. 9 and 10). Other indicators may deliver complementary information on environmental conditions such as soil quality data such as organic carbon content (ESDB 2004, Jonas et al. 2005, Panagos et al. 2012) and land cover (e.g. CORINE land cover).

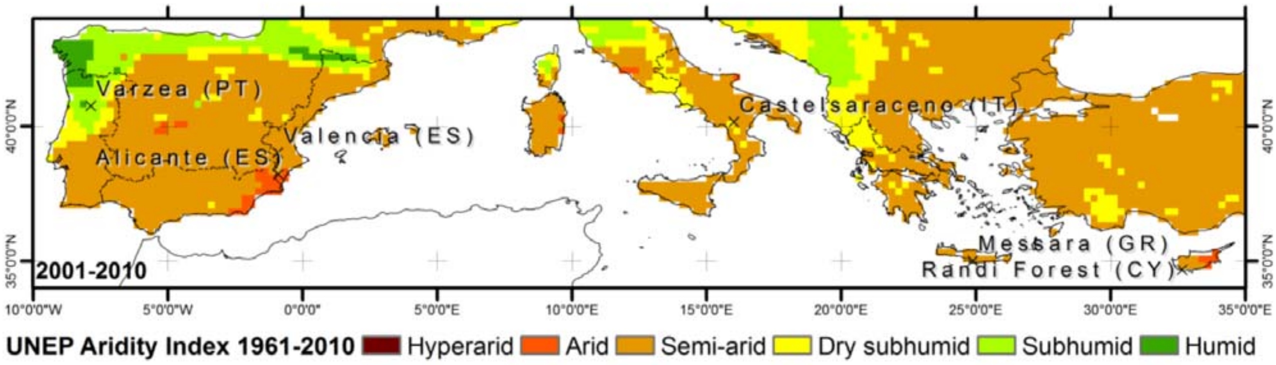

Figure 9 Aridity index in the Mediterranean (Tsanis and Daliakopoulos, 2014)

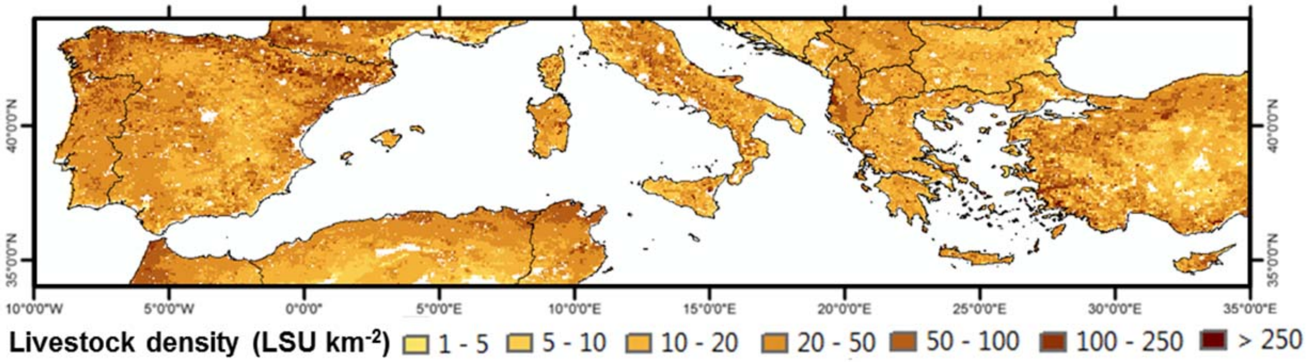

Figure 10 Livestock density in the Mediterranean (Data source: Robinson et al., 2014)

The gradients of observed environmental conditions are assumed to be reflected by the 0-1 parameter scale (i.e. min-max values) used to determine the respective parameter (i.e. b) in the rangeland resilience model. In future up-scaling, the observed livestock density will be converted to the livestock density parameter used in the model based on reference conditions (i.e. vegetation cover) without livestock grazing. All potential combinations of environmental conditions (i.e. based on aridity index and soil organic carbon content) and grazing pressure (i.e. livestock density) will be used as starting conditions to model ecological management impacts (i.e. vegetation cover). Model parameters such as local facilitation and competition will be adjusted reflecting particular ecological processes and feedbacks (see Tab. 2). According to the specifications of management scenarios, vegetation cover will be modelled over 10 years.

Differences in vegetation cover dynamics directly translate into differences in the costs that arise to purchase supplementary fodder depending on the price of fodder. The fodder price is therefore the most important economic aspect differentiating the financial viability (i.e. net present value) among the management scenarios (see Fig. 3). Hence, a gradient of fodder price is considered to capture variations in economic management impacts. This gradient is informed by fodder prices reported for Mediterranean countries (Eurostat, 2017). Taken together, the combined gradients of environmental conditions, livestock density and fodder price constitute the basis for evaluating socio-ecological effectiveness at a regional scale considering the evaluation criteria outlined above. An example output is given displaying the vegetation cover that results from two management scenarios applied along gradients of environmental conditions and livestock density (Fig. 11).

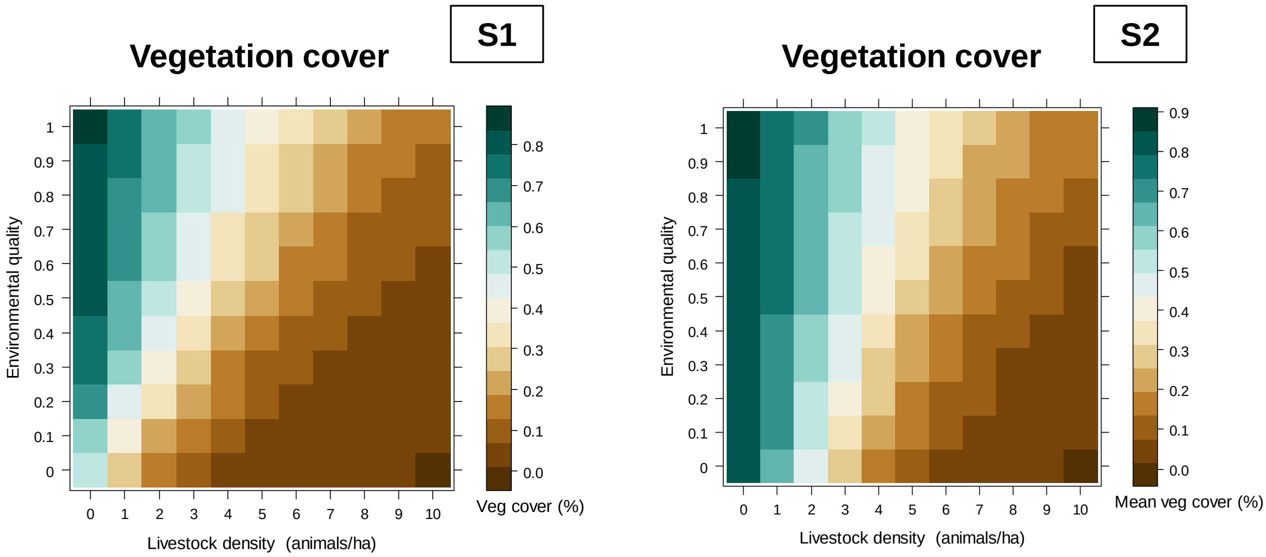

Figure 11 Example output: Vegetation cover resulting from gradients of environmental conditions and livestock density after 10 years considering two management scenarios S1 and S2 (see Tab. 1).

4 Outlook

We have in this report set out an integrated modelling strategy that uses scenario analysis and dynamic ecosystems understanding to generate recommendations for decision makers. This perspective on dynamic ecosystem regimes appraises actions that both foster restoration of degraded ecosystems and prevent degradation of functioning ecosystems. There is a high level of conceptual thinking embedded in ecological models, with model parameters that are not easily calibrated based on field data. In particular, there is an issue of scaling in going from conceptual 0-1 parameter values to indicator values that are observed in reality. In order to provide useful scenario output, an important step is to parametrise the model in such a way that it suitably represents the ecological conditions in a given study site. After successful parametrisation, the modelling strategy can be used to look for the spatial dimensions of best practices across gradients of environmental conditions (e.g. as a function of aridity index), livestock density and fodder price in the Mediterranean region.

The strategy offers three key lessons in operationalising LDN. First, long-term field experiments are essential to strengthen advances in identifying dynamic ecosystem regimes including a variety of relevant ecosystem properties and developing reliable predictions of site-specific degradation and restoration drivers and outcomes. In particular, we call for probabilistic assessments of current ecosystem states in relation to stability domains and systematic use of early warning signals for predicting regime shifts to advance the spatial balancing of land degradation and recovery for achieving LDN. Second, prediction of windows of opportunities and risks is essential to identify critical land management timings that realise ecological benefits at minimum risk and cost. Improved seasonal weather forecasts and ENSO early warnings can provide key information for such predictions, especially if packaged with restoration and SLM advice tailored to land users' needs. Third, successful multi-level LDN planning requires managerial flexibility that allows to continuously adapt investment decisions, including timing, to existing environmental conditions and ecosystem trajectories in relation to critical thresholds. This is a pre-requisite to rapidly take action once opportunities or risks emerge. These insights into non-linear ecosystem dynamics help to better evaluate the effectiveness of land management options for achieving policy goals setting a positive trajectory for achievement of the Sustainable Development Goals and LDN.

5 References

Anderson MW, Teisl MF, Noblet CL. 2016. Whose values count: is a theory of social choice for sustainability science possible? *Sustain Sci* **11**: 373–383. DOI 10.1007/s11625-015-0345-1

Asner GP, Elmore AJ, Olander LP, Martin RE, Harris AT. 2004. Grazing systems, ecosystem responses, and global change. *Annu Rev Environ Resour*. **29:** 261-299.

Baptista I, Irvine B, Fleskens L, Geissen V, Ritsema C. 2016. Modelling the biophysical impact and financial viability of soil management technologies under variable climate in Cabo Verde drylands: the PESERA-DESMICE approach. *Land Degrad. Dev.* **27**: 1679–1690. doi: 10.1002/ldr.2552.

Barkemeyer R, Stringer LC, Hollins JA, Josephi F. 2015. Corporate reporting on solutions to wicked problems: sustainable land management in the mining sector. *Environm. Sci. Policy* **48**: 196-209.

Behnke RH, Scoones I, Kerven C. 1993. Range Ecology at Disequilibrium: New Models of Natural Variability and Pastoral Adaption in African Savannas, Overseas Development Institute, London.

Bestelmeyer BT. 2006. Threshold concepts and their use in rangeland management and restoration: the good, the bad, and the insidious. *Restor. Ecol.* **14(3)**: 325-329.

Broadhurst LM, Jones TA, Smith FS, North T, Guja L. 2016. Maximizing seed resources for restoration in an uncertain future. *Biosci.* **66**: 73-79.

Chasek P, Safriel U, Shikongo S, Fuhrman VF. 2015. Operationalizing zero net land degradation: the next stage in international efforts to combat desertification? *J. Arid Environm.* **112**: 5-13.

De Groot RA, Blignaut J, Van Der Ploeg S, Aronson J, Elmqvist T, Farley J. 2013. Benefits of investing in ecosystem restoration. *Conservat. Biol.* **27(6)**: 1286-1293.

Diffenbaugh N, Field, CB. 2013. Changes in ecologically critical terrestrial climate conditions. *Science* **341**: 486-492.

ELD. 2015. The value of land: prosperous lands and positive rewards through sustainable land management. The Economics of Land Degradation Initiative, Bonn, Germany.

ESDB. 2004. The European Soil Database distribution version 2.0, European Commission and the European Soil Bureau Network, CD-ROM, EUR 19945 EN, 2004. http://eusoils.jrc.ec.europa.eu/content/european-soil-database-v20-vector-and-attribute-data

EUROSTAT. 2017. http://ec.europa.eu/eurostat/web/agriculture/data/database.

Folke C, Carpenter S, Walker B, Scheffer M, Elmqvist T, Gunderson L, Holling CS. 2004. Regime shifts, resilience, and biodiversity in ecosystem management. *Ann. Rev. Ecol. Evol. Systemat.* **35**: 557-581.

Foster CN, Sato CF, Lindenmayer DB, Barton PS. 2016. Integrating theory into disturbance interaction experiments to better inform ecosystem management. *Glob. Chang. Biol.* **22(4)**: 1325-1335.

Frank K, Baumgärtner S, Becker C, Müller B, Quaas M. 2006. Ecological-economic models for sustainable grazing in semi-arid regions: between concepts and case studies. In: Leser, H. (Ed.) The changing culture and nature of Namibia. Case studies. Basler Afrika-Bibliographien, Basel, pp. 69 – 89.

Friedel MH. 1991. Range condition assessment and the concept of thresholds: A viewpoint. *J. Range Manag.* **44(5):** 422-426.

Giger M, Liniger H, Sauter C, Schwilch G. 2015. Economic Benefits and Costs of Sustainable Land Management Technologies: An Analysis of WOCAT's Global Data. *Land Degrad. Dev.* doi: 10.1002/ldr.2429.

Gilardelli F, Sgorbati S, Citterio S, Gentili R. 2016. Restoring Limestone Quarries: Hayseed, Commercial Seed Mixture or Spontaneous Succession? *Land Degrad. Dev.* **27***:* 316–324. *doi:* 10.1002/ldr.2244.

Hein L, De Ridder N. 2006. Desertification in the Sahel: a reinterpretation. *Glob. Chang. Biol.* **12**: 751–758.

Hill J, Stellmes M, Udelhoven T, Röder A, Sommer S. 2008. Mediterranean desertification and land degradation: Mapping related land use change syndromes based on satellite observations. *Glob. Planet. Chang.* **64**: 146–157. DOI: 10.1016/j.gloplacha.2008.10.005

Hirota M, Holmgren M, Van Nes EH, Scheffer M. 2011. Global resilience of tropical forest and savanna to critical transitions. *Science* **334**: 232-235.

Hobbs RJ, Higgs ES, Hall C. 2013. *Novel Ecosystems: Intervening in the New Ecological World Order*. John Wiley, Oxford, UK.

Holling CS. 1973. Resilience and stability of ecological systems. *Ann. Rev. Ecol. Evol. Systemat.* **4**: 1–23. DOI: 10.1146/annurev.es.04.110173.000245

Holmgren M, Hirota M, Van Nes EH, Scheffer M. 2013. Effects of interannual climate variability on tropical tree cover. *Nature Clim. Chang.* **3**: 755-758.

Holmgren M, Scheffer M. 2001. El Niño as a window of opportunity for the restoration of degraded arid ecosystems. *Ecosyst.* **4**: 151-159.

James JJ et al. 2015. Managing Medusahead (Taeniatherumcaput-medusae) on rangeland: a meta-analysis of control effects and assessment of stakeholder needs. *Rangel. Ecol. Manag.* **68**: 215-223.

Jones RJA, Hiederer R, Rusco E, Loveland PJ, Montanarella L. 2005. Estimating organic carbon in the soils of Europe for policy support. *Europ. J. Soil Sci.* **56**: 655-671.

Kefi S, Rietkerk M, van Baalen M, Loreau M. 2007. Local facilitation, bistability and transitions in arid ecosystems. *Theor. Popul. Biol.* **71**: 367–379. doi:10.1016/j.tpb.2006.09.003

Kok M, Lüdeke MKB, Lucas P, Sterzel T, Walther C, Janssen P, Sietz D, De Soysa I. 2016. A new method for analysing socio-ecological patterns of vulnerability. *Reg. Environm. Chang.* **16(1)**: 229–243. DOI: 10.1007/s10113-014-0746-1

Le Houérou HN. 1987. Aspects de la croissance et du météorologiques développement végétal dans les déserts et les zones menacées de désertisation. PNUE, Nairobi, Kenya, and OMM, Genéve, Switzerland. 59 pp.

Maestre FT et al., 2005. Is the change of plant-plant interactions with abiotic stress predictable? A meta-analysis of field results in arid environments. *J. Ecol.* **93**: 748e757

Maestre FT, Cortina J. 2004. Do positive interactions increase with abiotic stress? A test from a semi-arid steppe. *Proc. R. Soc. B Biol. Sci.* **271**: S331eS333.

Malhi Y et al. 2008. Climate Change, Deforestation, and the Fate of the Amazon. *Science* **319**: 169-172.

Müller B, Frank K, Wissel C. 2007. Relevance of rest periods in non-equilibrium rangeland systems - a modeling analysis. *Agric. Syst.* **92**: 295–317.

Nilsson C, Polvi LE, Gardeström J, Hasselquist EM, Lind L, Sarneel JM. 2015. Riparian and in-stream restoration of boreal streams and rivers: success or failure? *Ecohydrol* **8(5)**: 753-764.

Nkonya E et al. 2016. Global cost of land degradation. In: Nkonya et al. (Eds.) Economics of Land Degradation and Improvement – A Global Assessment for Sustainable Development. International Food Policy Research Institute, Washington, DC, USA, Center for Development Research, University of Bonn, Germany, Ch. 6, 117-165 pp.

Orr BJ, Cowie AL, Castillo Sanchez VM, Chasek P, Crossman ND, Erlewein A, Louwagie G, Maron M, Metternicht GI, Minelli L, Tengberg AE, Walter S, Welton S. 2017. *Scientific Conceptual Framework for Land Degradation Neutrality*. A Report of the Science-Policy Interface. United Nations Convention to Combat Desertification (UNCCD), Bonn, Germany.

Panagos P, Van Liedekerke M, Jones A, Montanarella L. 2012. European Soil Data Centre: Response to European policy support and public data requirements. *Land Use Pol.* **29(2)**: 329-338. doi:10.1016/j.landusepol.2011.07.003

Qadir M, Quillérou E, Nangia V, Murtaza G, Singh M, Thomas RJ, Drechsel P, Noble AD. 2014. Economics of salt-induced land degradation and restoration. *Nat. Resour. Forum* ***38***: 282–295. doi:10.1111/1477-8947.12054

Quaas MF, Baumgärtner S, Becker C, Frank K, Müller B. 2007. Uncertainty and Sustainability in the Management of Rangelands. *Ecol. Econ.* **62**: 213–234.

Reed MS, Stringer LC, Dougill AJ, Perkins JS, Atlhopheng JR, Mulale K, Favretto N. 2015. Reorienting land degradation towards sustainable land management: Linking sustainable livelihoods with ecosystem services in rangeland systems. *J. Environm. Manag.* **151**: 472–485. DOI: 10.1016/j.jenvman.2014.11.010

Reed MS, Stringer LC. 2016. Land Degradation, Desertification and Climate Change: Anticipating, Assessing and Adapting to Future Change. Routledge, UK.

Robinson TP et al. 2014. Mapping the Global Distribution of Livestock. *PLoS ONE* **9(5)**: e96084. doi:10.1371/journal.pone.0096084

Roques KG, O'Connor TG, Watkinson AR. 2001. Dynamics of shrub encroachment in an African savanna: relative influences of fire, herbivory, rainfall and density dependence. *Journal of Appl. Ecol.* **38**: 268–280. DOI: 10.1016/S0095-0696(03)00069-X

Safriel U, Adeel Z, Niemeijer D, Puigdefabres J, White R, Lal R, Winslow M, Ziedler J, Prince S, Archer E, King C. 2005. Dryland systems. In: Hassan R, Scholes R, Ash N (Eds.) Ecosystems and human well-being: current state and trends. Vol 1., Island Press, Washington, pp 623–662.

Scheffer M et al. 2015. Creating a safe operating space for iconic ecosystems. *Science* **347**: 1317-1319.

Scheffer M, Carpenter S, Foley JA, Folke C, Walker B. 2001. Catastrophic shifts in ecosystems. *Nature* **413**: 591-596.

Scheffer M, Carpenter S. 2003. Catastrophic regime shifts in ecosystems: linking theory to observation. *Trends Ecol. Evol.* **18(12)**: 648-656.

Schneider F, Kefi S. 2016. Spatially heterogenous pressure raises risk of catastrophic shifts. *Theor. Ecol.* **9**: 207-217.

Sietz D, Van Dijk H. 2015. Land-based adaptation to global change: What drives soil and water conservation in western Africa? *Glob. Environm. Chang.* **33**: 131-141.

Sietz D. 2014. Regionalisation of global insights into dryland vulnerability: better reflecting smallholders' vulnerability in Northeast Brazil. *Glob. Environm. Chang.* **25**: 173–185. DOI: 10.1016/j.gloenvcha.2014.01.010

Sitters J, Holmgren M, Stoorvogel JJ, López BC. 2012. Rainfall-tuned management facilitates dry forest recovery. *Restor. Ecol.* **20**: 33-42.

Sousa PM, Trigo RM, Aizpurua P, Nieto R, Gimeno L, Garcia-Herrera R. 2011. Trends and extremes of drought indices throughout the 20th century in the Mediterranean. *Nat. Hazards Earth Syst. Sci.* **11**: 33-51, doi:10.5194/nhess-11-33-2011. doi:10.5194/nhess-11-33-2011

Staal A, Dekker SC, Hirota M, Van Nes EH. 2015. Synergistic effects of drought and deforestation on the resilience of the south-eastern Amazon rainforest. *Ecol. Complexity* **22**: 65-75.

Stavi I, Lal R. 2015. Achieving zero net land degradation: challenges and opportunities. *J. Arid Environm.* **112**: 44-51.

Steffen W, Richardson K, Rockström J, Cornell SE, Fetzer I, Bennett EM, Biggs R, Carpenter SR, de Vries W, de Wit CA, Folke C, Gerten D, Heinke J, Mace GM, Persson LM, Ramanathan V, Reyers B, Sörlin S. 2015. Planetary boundaries: Guiding human development on a changing planet. *Science* **347**: 6223. DOI: 10.1126/science.1259855

Suding KN, Hobbs RJ. 2009. Threshold models in restoration and conservation: a developing framework. *Trends Ecol. Evol.* **24(5)**: 271-279.

Suding KN. 2011. Toward an era of restoration in ecology: successes, failures, and opportunities ahead. *Ann. Rev. Ecolog. Evolut. Systematics* **42**: 465-487.

Tarrason D, Ravera F, Reed MS, Dougill AJ, Gonzalez L. 2016. Land degradation assessment through an ecosystem services lens: Integrating knowledge and methods in pastoral semi-arid systems. *J. Arid Environm.* **124**: 205-213.

Trabucco A, Zomer RJ. 2009. Global Aridity Index (Global-Aridity) and Global Potential Evapo-Transpiration (Global-PET) Geospatial Database. CGIAR Consortium for Spatial Information. Published online, available from the CGIAR-CSI GeoPortal at: http://www.csi.cgiar.org.

Tsanis IK, Daliakopoulos IN. 2014. Drivers of change in the study sites. CASCADE Deliverable 2.2. Technical University of Crete, Chania, Greece.

UNCCD. 2015. Report of the Conference of the Parties on its Twelfth Session. Addendum (ICCD/COP(12)/20/Add.1). Available from: http://www.unccd.int/Lists/OfficialDocuments/cop12/20add1eng.pdf. Accessed February 2016.

UNCCD. 2016. Land in balance. The scientific conceptual framework for land degradation neutrality (LDN). Science-Policy Brief 02. September 2016, United Nations Convention to Combat Desertification (UNCCD), Science-Policy Interface, Bonn, Germany.

Valone TJ, Meyer M, Brown JH, Chew RM. 2002. Timescale of perennial grass recovery in desertified arid grasslands following livestock removal. *Conserv. Biolog.* **16(4)**: 995-1002.

Vicente-Serrano et al. 2014. Evidence of increasing drought severity caused by temperature rise in southern Europe. *Environ. Res. Lett.* **9(4)**: 044001. doi:10.1088/1748-9326/9/4/044001

Westoby M, Walker B, Noy-Meir I. 1989. Opportunistic management for rangelands not at equilibrium. J. *Range Manage.* **42(4)**: 266-274.

Wilson KA et al. 2011. Optimal restoration: accounting for space, time and uncertainty. *J. Appl. Ecolog.* **48**: 715-725.

Zahawi RA, Reid RL, Holl KD. 2014. Hidden costs of passive restoration. *Restor. Ecol.* **22(3)**: 284-287.

Zika M, Erb K-H. 2009. The global loss of net primary production resulting from human-induced soil degradation in drylands. *Ecol. Econ.* **69**: 310–318.

Appendix 1 D8.2

Expert survey on economics of land management
Evaluating socio-ecological effectiveness of rangeland management

Diana Sietz and Luuk Fleskens

1. Socio-ecological costs and benefits of land management

One of the major aims in WP8 is to assess the socio-ecological effectiveness of sustainable land management (SLM) strategies for simultaneously enhancing land users' wellbeing and ecosystem functioning (Tasks 8.2 and 8.3.). We ask you as regional experts for support in evaluating the following aspects:

- Rangeland productivity
- Converting vegetation cover to livestock productivity
- Economic costs and benefits at household level
- Ranking of grazing sites
- Critical thresholds for degradation and restoration shifts

We consider the socio-ecological effectiveness of a particular SLM strategy to be determined by financial attractiveness and ecological effects. Better understanding the socio-ecological effects of SLM strategies can provide essential new knowledge suitable to guide land management decisions and policies in specific ecological conditions. As an overview, we will evaluate socio-economic costs and benefits and the ecological effects of SLM strategies:

- **Socio-economic costs**: Investment costs (e.g. costs to fence an area or purchase additional fodder)
- **Socio-economic benefits:** Changes in livestock productivity (e.g. meat and wool production).
- **Ecological effects**: SLM directly generates changes in the pressure on and state of an ecosystem and may affect the chance of restoration and risk of degradation.

We will use a rangeland resilience model (Schneider and Kefi 2016) to evaluate the ecological effects of land management. We discussed the opportunity to parametrise the rangeland resilience model in relative terms according to the study sites in Randi/Cyprus, Crete/Greece, Castelsaraceno/Italy and Albatera/Spain[2] to take advantage of CASCADE's potential for comparative analysis. These study sites differ in the levels of aridity, soil characteristics, livestock density etc. – ideally they can be ordered along a gradient that can be captured in the model simulations.

The evaluation of ecosystem services, degradation levels and restorability performed in WP 5 is a useful starting point to distinguish a range of parameter values which best represents the respective sites. The model uses a 0-1 parameter scale, for example to depict poorest to optimum environmental conditions, enabling comparability across study sites (spatial) and within sites (temporal). Depending on the sites, a large relative change in a parameter may however mean only a small absolute change in real world processes and outcomes. In discussing the effects of land

[2] The Albatera site is a heavily degraded shrubland which may have undergone a degradation shift (D5.2). Although current management strategies at Albatera relate to tree plantation, this site may serve to discuss potential management scenarios related to ecosystem restoration.

management, the qualitative insights derived from the model need to be translated back to real-world values and implications to explore development scenarios and inform policy recommendations.

We will focus on SLM strategies with immediate effects, such as controlled grazing, supplementary feeding or fencing, potentially implying long-lasting effects and regime shifts. We assume that SLM directly affects the livestock pressure and vegetation cover (state of ecosystem) captured in the model.

2. Rangeland productivity

First, the rangelands' carrying capacity is an important characteristic to be considered. It will help to relate parameter values in the model (e.g. perennial vegetation cover and livestock density) to real world values.

Table 1 Carrying capacity of rangelands

	Carrying capacity					
Study site: Reference (ungrazed)	**PERENNIAL vegetation**			**ANNUAL vegetation**		
	PERENNIAL vegetation cover (%) [a] (average)	**Dry matter above-ground biomass of PERENNIAL vegetation** [b] (kg/ha) (average)	**Share of palatable PERENNIAL vegetation** (%) (average)	**ANNUAL vegetation cover** (%) [a] (average)	**Dry matter above-ground biomass of ANNUAL vegetation** [b] (kg/ha) (average)	**Share of palatable ANNUAL vegetation** (%) (average)
Randi, Cyprus						
Messara, Greece						
Castelsaraceno, Italy						
Albatera, Spain						

[a] maximum potential vegetation cover without grazing, *Note:* Figures in Sect. 5 in D5.2 show total aboveground biomass at reference sites. Are there also data available for *perennial* above-ground biomass?

[b] maximum potential biomass without grazing. *Note:* Fig. 2 in D5.2 shows total aboveground biomass in reference state. Are there also data available for *perennial* above-ground biomass?

[c] maximum potential livestock density that can be sustained indefinitely given optimal vegetation cover and biomass defined in previous columns

Second, the actual rangeland productivity provides insights into the ecological effects of degradation and restoration efforts useful to evaluate the model results. The plant cover and biomass data given in D5.2 (Tab. 6, p. 81) relate to TOTAL cover and biomass going beyond perennial plants (*Remark: I noted differences between the data in Tab. 6 and the data given on page 41*). Are there also data available for perennial vegetation cover and above-ground biomass at degraded, restored and managed sites? Can you estimate the average share of palatable plants in perennial vegetation? Can you already oversee when the missing data for Messara will become available? This would be helpful to fill the following tables.

Table 2 Actual rangeland productivity

Study site	**PERENNIAL vegetation**			**ANNUAL vegetation**			**Livestock density (number/ ha)** (average)		
	Vegetation cover (%) (average)	**Dry matter above-ground biomass (kg/ha) (SD)** (average)	**Share of palatable vegetation (%)** (average)	**Vegetation cover (%)** (average)	**Dry matter above-ground biomass (kg/ha) (SD)** (average)	**Share of palatable vegetation (%)** (average)	Sheep	Goat	Cattle
Randi, Cyprus									
Degraded									
Restored									
Messara, Greece									
Degraded									
Restored (Odigitri)									
Castelsaraceno, Italy									
Overgrazed									
Fenced									
Under-grazed									
Cleared									
Albatera, Spain									
Degraded							NA	NA	NA

Third, information on meat, wool and milk production is important to calculate financial attractiveness, i.e. economic benefits and costs. Is leather production also relevant in some sites?

Table 3 Production of meat, wool and milk

Livestock type	**Share of total herd used for produce (average)** [a)]			**Annual production (average)**		
	Meat (%)	Wool (%)	Milk (%)	Meat (kg/head)	Wool (kg/head/year)	Milk (l/head/year)
Randi, Cyprus						
Sheep						
Goat		NA			NA	
Cattle		NA			NA	
Messara, Greece						
Sheep						
Goat		NA			NA	
Cattle		NA			NA	
Castelsaraceno, Italy						
Sheep						
Goat		NA			NA	
Cattle		NA			NA	
Albatera, Spain --- in absence of grazing on study site, an estimate at regional level would be useful.						
Sheep						
Goat		NA			NA	
Cattle (used?)		NA			NA	

[a)] E.g. only a certain share of animals may be sold/slaughtered and calves are not milked.

3. Converting vegetation cover to livestock productivity

The rangeland resilience model provides data on perennial vegetation cover as an output variable describing an ecosystem's state. To derive financial attractiveness, this needs to be translated to livestock productivity. Therefore, an essential question is how can we best convert perennial vegetation cover to perennial above-ground biomass and livestock productivity in our study sites? While some initial insights may be derived from Tables 1 and 2, it would be great if you could identify conversion factors reflecting our sites. For example in Andalusian alfa grass steppes, 1% of cover by alfa grass corresponded to 380.4 kg dry matter/ha (Gauquelin et al. 1996). These very high values resulted from absent or limited human impacts and relatively favourable annual rainfall (370 mm). In contrast in sub-desertic steppes in North Africa characterised by different precipitation and temperature patterns, 1% perennial vegetation cover correlated with 43 ± 3.6 kg dry matter perennial phytomass/ha and 80-100 ± 25kg for alfa grass steppes only (Le Houérou 1987).

Table 4 Conversion of perennial vegetation cover to above-ground biomass

Study site	**Cover-to-biomass (above-ground) conversion factor** (kg dry matter/ha per 1% perennial vegetation cover)	**Reference if available** (otherwise expert knowledge assumed)
Randi, Cyprus		
Messara, Greece		
Castelsaraceno, Italy		
Albatera, Spain		

To further convert available above-ground biomass in livestock productivity, it is important to identify feed requirements. Productivity determinants include animal-specific requirements, animal weight, growth rate and activity level. Moreover, we would need information on the average daily feed requirement per livestock unit (LU) and the contribution of complementary fodder bought on the market (and tree pruning?) to total feed requirements in order to determine livestock productivity and potential rangeland degradation. Information on the price of complementary fodder will inform the evaluation of costs and benefits.

Another aspect related to livestock productivity is the conversion of livestock units (LU). Can you suggest suitable conversion factors, e.g. cattle = 1, sheep = 0.1 and goat = 0.1. Again, a reference would be valuable.

Table 5 Livestock feed requirements

Livestock type	**Daily livestock feed requirements** (kg dry matter/Livestock Unit*day) (average)	**Reference if available** (otherwise expert knowledge assumed)	**Contribution of complementary fodder to total livestock feed requirements** (%) (average)	**Farm gate price of complementary fodder** (€/kg) (average)
Randi, Cyprus				
Sheep				
Goat				
Cattle				

Messara, Greece				
Sheep				
Goat				
Cattle				
Castelsaraceno, Italy				
Sheep				
Goat				
Cattle				
Albatera, Spain --- in absence of grazing on study site, an estimate at regional level would be useful.				
Sheep				
Goat				
Cattle (used?)				

4. Economic costs and benefits at household level

An important aspect in evaluating financial attractiveness relates to economic costs and benefits at a household level. Here we assume a typical land user representing a given case study region and ask you for information about prices of produce at the farm gate, labour requirements, size of livestock herds and costs for hired labour as well as other investments in livestock (e.g. veterinary).

Please indicate the conversion factor from milk to cheese here: …

Table 6 Farm gate price of produce, subsidies and net income/animal

Livestock type	**Farm gate price (average)**				**Subsidies received per head or hectare of land** (€/year) [b]	**Total net income per animal** (€/year)
	Meat or meat products (€/kg)	Wool (€/kg)	Milk (€/l)	Cheese (€/kg) [a]		
Randi, Cyprus						
Sheep						
Goat		NA				
Cattle		NA				
Land area	NA	NA	NA	NA		NA
Messara, Greece						
Sheep						
Goat		NA				
Cattle		NA				
Castelsaraceno, Italy						
Sheep						
Goat		NA				
Cattle		NA				
Land area	NA	NA	NA	NA		NA
Albatera, Spain --- in absence of grazing on study site, an estimate at regional level would be useful.						
Sheep						
Goat		NA				
Cattle (used?)		NA				
Land area	NA	NA	NA	NA		NA

[a] See conversion factor from milk to cheese

[b] Please use specific rows to indicate if subsidies are received per animal head or hectare of land

Table 7 Labour requirements and costs for livestock keeping

Livestock type	**Daily labour requirements** (hours/head*day) (average)	**Number of animals kept per household** (number/household) (average) [a]	**Number of hours dedicated to livestock keeping** (hours/day) (average)	**Total costs for hired labour per year** (€/year) (average)	**Wage rate for hired labour per hour** (€/hour) (average)	**Labour requirements for processing cheese, meat and other products** (days/year) (average)	**Capital costs per year**, e.g. land rent, veterinary, stable maintenance, equipment, milk tank, membership farmers' association (€/year) (average)
Randi, Cyprus							
Sheep							
Goat							
Cattle							
Messara, Greece							
Sheep							
Goat							
Cattle							
Castelsaraceno, Italy							
Sheep							
Goat							
Cattle							
Albatera, Spain --- in absence of grazing on study site, an estimate at regional level would be useful.							
Sheep							
Goat							
Cattle (used?)							

[a] If animals are usually kept in combination, please indicate typical herd composition below.

If animals are usually kept in combination, please indicate the typical composition of herds:

- ... sheep,
- ... goats and
- .... cattle

Are there any capital costs that arise only occasionally, e.g. construct a stable? If so, please indicate purpose, lifetime and amount of capital costs:

- Purpose:
- Lifetime (years):
- Amount (€):

5. Ranking of grazing sites

Information on climate, soils and other environmental conditions useful to rank the grazing sites is available across the grazing sites (see below excerpts of D5.2) which serves to rank the sites along an aridity gradient, for example. This ranking can be directly related to loss of ecosystem services, with a higher loss in more arid areas (see D5.2, p. 9). However, the aridity index is very similar in Messara and Randi. Therefore, it may be useful to rank the sites according to aggregated environmental conditions, for example combining climate and soil properties. Soil properties may further differentiate the sites in Messara and Randi. Would you have a suggestion of how to rank the grazing sites depending on combined environmental conditions?

Table 8 Ranking of grazing sites according to environmental conditions (Note: 1 = max, 4 = min)

Study site	**Aridity**	**Soil**	**Climate**	**Others**
Randi, Cyprus				
Messara, Greece				
Castelsaraceno, Italy				
Albatera, Spain				

Excerpt from D5.2 → Table 1. Climatic characteristics of the six CASCADE field sites (extracted from D2.1, Daliakopoulos and Tsanis 2013).

	Albatera	**Castel-saraceno**	**Messara**	**Randi**
Climate	Semi-arid	Humid	Dry sub-humid	Dry sub-humid
Average annual rainfall (mm)	267	1289	503	489
Average mean temperature (°C)	18.0	9.1	17.9	19.5
Aridity Index (mm/mm)	0.16	1.05	0.31	0.29
PET (monthly)	136.0	102.5	136.0	141.5

NOTE: This is not consistent with aridity definition by UNEP (1992)

Excerpt from D5.2 → Table 2. Summary of main characteristics of the six CASCADE field sites (extracted from D2.1, Daliakopoulos and Tsanis 2013).

	Albatera	**Castelsaraceno**	**Messara**	**Randi**
Elevation	225-310 m	972-1284 m	100-230 m	90-230 m
Bedrock	Dolomites, conglomerates and sandstones	Limestones and dolomites	Limestones and marls	Marls
Soils	Calcisols, Cambisols and Fluvisols	Regosols	Cambisols and Luvisols	Calcaric regosols
Land use	Agriculture (52%) and shrublands (24%)	Cropland, pasturelands and forests	Croplands and shrublands	Croplands and shrublands
History	Abandonment of rainfed croplands, alpha grass harvesting and wood gathering. Afforestations	Land abandonment (especially after 1990s)	Overgrazing and overexploitation of water resources	Agriculture and grazing

References

Bestelmeyer BT, Duniway MC, James DK, Burkett LM, Havstad KM. (2013) A test of critical thresholds and their indicators in a desertification-prone ecosystem: more resilience than we thought. Ecology Letters 16: 339-345.

Gao Y, Zhong B, Yue H, Wu B, Cao S. (2011) A degradation threshold for irreversible loss of soil productivity: a long-term case study in China. Journal of Applied Ecology 48: 1145-1154.
Gauquelin T., Jalut G., Iglesias M., Valle F., Fromard F. and Dedoubat, J.J. (1996) Phytomass and carbon storage in the Stipa tenacissima steppes of the Baza basin, Andalusia, Spain. Journal of Arid Environments 34: 277–286.
Le Houérou, H. N. 1987. Aspects de la croissance et du météorologiques développement végétal dans les déserts et les zones menacées de désertisation. PNUE, Nairobi, Kenya, and OMM, Genéve, Switzerland. 59 pp.
Robinson et al. (2014) Mapping the Global Distribution of Livestock. PLoS ONE 9(5) e96084
Schneider and Kefi (2016) Spatially heterogenous pressure raises risk of catastrophic shifts. Theor. Ecol. 9: 207-217.
UNEP (1992) World Atlas of Desertification, Edward Arnold, London, UK.