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Model-based analysis of the environmental impacts of grazing management on Eastern Mediterranean ecosystems in Jordan

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ABSTRACT

Eastern Mediterranean ecosystems are prone to desertification when under grazing pressure. Therefore, management of grazing intensity plays a crucial role to avoid or to diminish land degradation and to sustain both livelihoods and ecosystem functioning. The dynamic land-use model LandSHIFT was applied to a case study on the country level for Jordan. The impacts of different stocking densities on the environment were assessed through a set of simulation experiments for various combinations of climate input and assumptions about the development of livestock numbers. Indicators used for the analysis include a set of landscape metrics to account for habitat fragmentation and the "Human Appropriation of Net Primary Production" (*HANPP*), i.e., the difference between the amount of net primary production (NPP) that would be available in a natural ecosystem and the amount of NPP that remains under human management. Additionally, the potential of the economic valuation of ecosystem services, including landscape and grazing services, as an analysis concept was explored. We found that lower management intensities had a positive effect on *HANPP* but at the same time resulted in a strong increase of grazing area. This effect was even more pronounced under climate change due to a predominantly negative effect on the biomass productivity of grazing land. Also Landscape metrics tend to indicate decreasing habitat fragmentation as a consequence of lower grazing pressure. The valuation of ecosystem services revealed that low grazing intensity can lead to a comparatively higher economic value on the country level average. The results from our study underline the importance of considering grazing management as an important factor to manage dry-land ecosystems in a sustainable manner.

1. Introduction

The Eastern Mediterranean ecosystems in Jordan are classified as dry-land systems which are potentially prone to desertification. The UN Convention to Combat Desertification defines the term "desertification" as "land degradation in arid, semi-arid and dry sub-humid areas resulting from various factors, including climatic variations and human activities" (UNEP, 1994). Potential proximate causes are identified by Geist and Lambin (2004) and include land-use change processes such as cropland expansion, overgrazing, and the expansion of road infrastructure and urban area. In Jordan, the main reasons for land-use change are the growing demands for settlement area and food by an increasing human population, aiming at a higher standard of living. Traditionally livestock grazing plays an important role in the agricultural sector. Here, the overarching problem is the overuse of the dry-land ecosystems caused by inadequately high stocking densities of grazing animals (overgrazing). Potential environmental impacts are changes in the vegetation cover/composition and soil degradation (Gillson and Hoffman, 2007; Ibanez et al., 2007) which reduce the productivity of forage grasses (van de Koppel and Rietkerk, 2000). In consequence, these processes can threaten the livelihoods of farmers and regional food security as well as biodiversity (Alados et al., 2004; Alhamad, 2006). An additional pressure on both ecosystems and livestock grazing will be climate change. The 4th IPCC Assessment Report points out that the Mediterranean region will face increasing mean annual temperatures and decreasing precipitation accompanied by a likely increase in length and frequency of dry spells in the coming decades (Christensen et al., 2007).

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In order to meet these challenges, there is a need for a more sustainable management of grazing land in Jordan (Millennium Ecosystem Assessment, 2005). According to World Bank (2006), the objective of sustainable land management is to fulfill the growing food and fiber demands (provisioning services) while sustaining other ecosystem services such as soil fertility, erosion control, or landscape esthetics. de Groot et al. (2010) illustrate that intensive management favors the provisioning services on cost of the portfolio of other ecosystem services. Regarding the management of grazing land this implies an adjustment of stocking densities at a level where environmental impacts are considerably reduced (Köchy et al., 2008) and in consequence a balance between provisioning services and other services is achieved.

An essential prerequisite for the development of regional strategies for a sustainable land management is to improve the scientific understanding of the functioning of the land-use system under consideration. Therefore, the aim of our study is to explore the effects of different driving factors on the future development of grazing land and provisioning of ecosystem services in Jordan as well as to quantify potential environmental impacts. The considered drivers include changing livestock numbers, climate change, and different options of grazing management, expressed as allowable maximum stocking densities. We applied the land-use model LandSHIFT (Koch et al., 2008, 2012; Schaldach et al., 2011) in combination with the vegetation model WADISCAPE (Köchy et al., 2008) to simulate the spatial distribution of grazing land and stocking densities of grazing animals under different scenarios, which are defined as combinations of these drivers. Based on the model output, which comprises information on the change in landcover and land-use intensity, we assessed the resulting environmental impacts as well as the consequences for the provisioning of ecosystem services. Indicators for the environmental impacts were the Human Appropriation of Net Primary Productivity (*HANPP*) (Haberl et al., 2007) and a set of landscape metrics, including "Number of Patches", "Largest Patch Index", "Proximity Index", and "Total Core Area" (Alhamad et al., 2011). In our case, the *HANPP* indicator defined the impact of grazing on the available biomass of ecosystems and served as a local metric for the human influence on ecosystem structure. In contrast, the landscape metrics were used to quantify ecosystem fragmentation as an important factor for the loss of biodiversity (e.g. Gustafson and Parker, 1994; Fahrig, 2003). Additionally, beneficial and adverse effects of the applied management schemes were evaluated by estimating the economic ecosystem service value, which integrates services from intact landscapes and savings in feed costs due to livestock grazing (Fleischer and Sternberg, 2006).

2. Material and methods

2.1. Study region

Study region is the Hashemite Kingdom of Jordan which is bordered by Syria in the north, by Iraq and Saudi Arabia in the east, and by Israel as well as the West Bank in the west (Fig. 1). The country has a land area of about 90 000 km^2. The climate is characterized by hot, dry summers and cool, wet winters. Mean annual precipitation ranges from less than 50 mm in the southeast to 660 mm in the northwest. In 2000, the population of Jordan was approximately 5 million people; about one fifth of which lived in Amman, the administrative capital and largest city in the country (United Nations, 2009). With about 2.2 million goats and sheep (FAO, 2011), the production of small ruminants is an important factor of Jordan's agricultural sector. The landscape can be classified as an eastern-Mediterranean ecosystem, which has been modified by human activity for several thousand years. Besides limited natural freshwater resources, current environmental problems include overgrazing and a high risk of desertification (Abahussain et al., 2002).

2.2. Modeling framework

The modeling framework (Fig. 2) that we have applied for our study includes a regional version of the land-use model LandSHIFT (Koch et al., 2008, 2012; Schaldach et al., 2011) and the WADISCAPE model to determine biomass productivity of semi-natural vegetation under grazing pressure (Köchy, 2007; Köchy et al., 2008). The models were used to calculate a series of grid maps showing the spatial distribution of grazing land and the respective stocking density of sheep and goats on each cell between 2005 and 2050. Based on these maps the environmental impacts of different types of grazing management were analyzed with a local level indicator

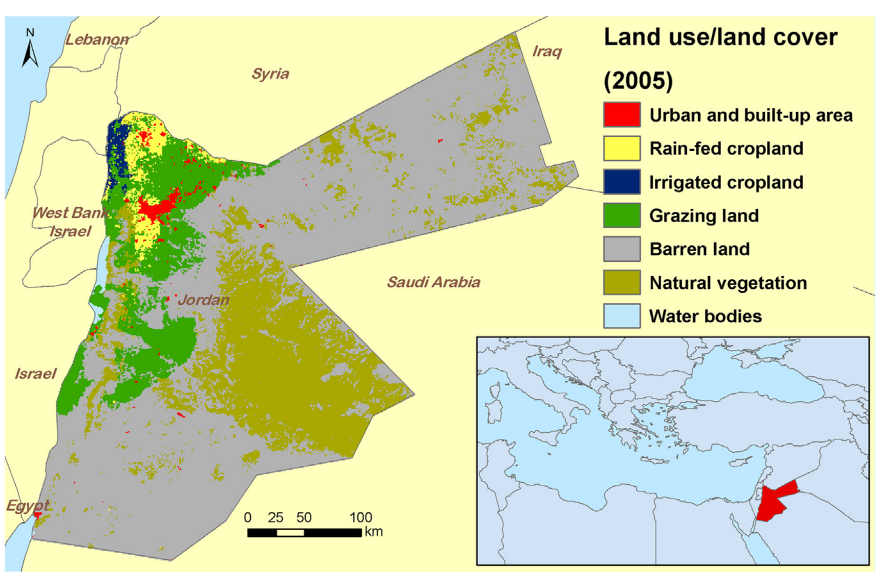

Fig. 1. Map of Jordan, the study area of this analysis, showing the land-use/land-cover distribution for the year 2005 as simulated with LandSHIFT.

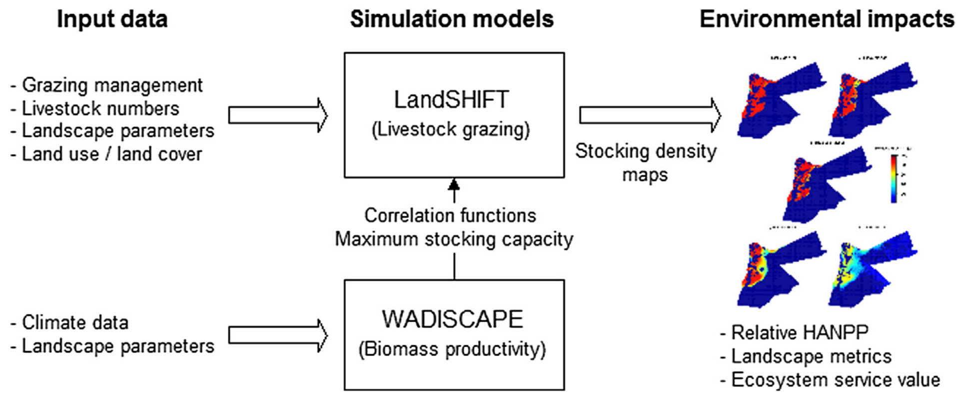

Fig. 2. Block diagram of the modeling framework. Changes in grazing land and stocking density are calculated with the land-use model LandSHIFT. Maximum stocking capacity and the relationship between biomass productivity and stocking density (correlation functions) are provided by the WADISCAPE model. The model input includes information on climate change, changes in livestock numbers as well as spatial land-cover/land-use and landscape parameters. Based on the model output from LandSHIFT, environmental impacts were calculated on local and landscape level.

(*HANPP*) and a set landscape level indicators (landscape metrics) as well as by an economic valuation of ecosystem services.

The LandSHIFT model is based on the concept of land systems (Turner et al., 2007) and couples components that represent the respective anthropogenic and environmental sub-systems. It includes a sub-module (Livestock grazing) to simulate grazing management utilizing information on biomass productivity and maximum stocking capacity provided by the WADISCAPE model. LandSHIFT operates on two spatial scale levels. Driving variables are specified on the country level (macro level) and comprise the numbers of sheep and goats held in Jordan expressed in Livestock Units (LU), the daily feed demand per LU as well as assumptions on management and policy measures (e.g. grazing intensity and nature conservation policy). Land-use changes are determined on a regular grid (micro-level) with a spatial resolution of 30 arc-seconds ($\sim$1 km $\times$ 1 km) in 5-year time-steps. Each grid cell has information about the dominant land-use/land-cover type, the stocking density of grazing animals, human population density, different landscape parameters (terrain slope, river network density, biomass productivity), and nature conservation area. The land-use/land-cover types are based on the IGBP classification (Loveland et al., 2000) which differentiates between cropland, urban land, and a variety of natural land-cover types such as forests, grassland, and shrublands.

As grazing land cannot be identified as a separate land-cover type in available remote sensing datasets (see Section 3.1), information on its spatial extent, location, and respective stocking densities is calculated by the sub-module Livestock grazing which follows two process steps.

The first process step, the suitability assessment, is carried out on the micro level. The suitability of a single grid cell for grazing is assessed with a multi-criteria analysis (Eastman et al., 1995):

$$\Psi_k = \underbrace{\sum_{i=1}^{n} w_i p_{i,k}}_{\text{suitability}} \times \underbrace{\prod_{j=1}^{m} c_{j,k}}_{\text{constraints}}, \quad \text{with} \sum_i w_i = 1, \text{ and } p_{i,k}, c_{j,k} \in [0,1] \tag{1}$$

The factor-weights w_i define the importance of each suitability factor p_i at grid cell k; c_j define constraints for changing the land-use type at that given cell. In this study, p_i includes the three above mentioned landscape parameters and information on human population density (see Koch et al., 2008; Wint et al., 2003). Both p_i and c_j were normalized by value functions (based on a logistic regression analysis) transforming the factor values to a co-domain from 0 to 1. The determination of the factor weights is described in Section 3.2. Existing nature conservation area, cropland, and urban land are excluded from being converted to grazing land.

The second process step, resource allocation, is also carried out on the micro level. Within this process, the macro level numbers for goats and sheep are distributed to the micro level grid cells with the highest suitability by changing their land-use type to "grazing land". In addition, various allocation modes representing different options of grazing intensity are implemented. The allocation of grazing land in the starting year 2005 of the simulation is identical for all these modes (model initialization). Using the WADISCAPE model, the local biomass productivity without grazing is assessed (see below). The local SD in livestock units per km^2 is then calculated from this productivity via the feed demand per livestock unit and subsequently assigned to the grid cell. In the following time steps, the allocation modes differ in their grazing intensity. The highest intensity level is given by the maximum stocking capacity from WADISCAPE, while moderate grazing strategies are characterized by reduced intensity levels represented as a certain fraction, e.g. 50%, of the maximum stocking capacity. In order to allow for a gradual transition of the intensity level, the maximum SD is reduced stepwise between 2005 and 2020. The allocation procedure itself uses the above mentioned SD to derive the new biomass productivity value in the next time step from the cell specific correlation function provided by WADISCAPE. This productivity again serves as basis for the calculation of the new local SD, as described for the first simulation step. This procedure is repeated for each simulation time step (Koch et al., 2008).

The WADISCAPE model determines biomass productivity by simulating the growth and dispersal of herbs and dwarf shrubs in artificial wadi landscapes. Vegetation dynamics are controlled by water availability, which varies with topographic conditions, and by grazing animals. This dynamic behavior forms the basis of the Livestock grazing sub-module of the LandSHIFT model (see above).

First, the output from WADISCAPE is used to determine the maximum stocking capacity (MSC) of a habitat, defined as the number of small ruminants (sheep and goats) per hectare, for which the vegetation provides sufficient food in 9 out of 10 years of year-round grazing. Without grazing, biomass productivity and MSC increase as a function of mean annual precipitation. The effects of climate change on biomass productivity and maximum stocking capacity are calculated for the years 2005 and 2050. As the LandSHIFT model needs this input information for each simulation step (i.e. in 5-year intervals), the values for intermediate time steps are calculated by linear interpolation.

Second, WADISCAPE output provides sigmoidal correlation functions between stocking density (SD) and green biomass productivity for each factorial combination of five precipitation classes ("Arid": 80 mm–<200 mm, "Semiarid": 200 mm–<400 mm, "Dry Mediterranean": 400 mm–<500 mm, "Typical Mediterranean": 500 mm–<700 mm, "Mesic Mediterranean": 700 mm–<960 mm) and five classes of terrain slope (0°–<5°, 5°–<12.5°, 12.5°–<17.5°, 17.5°–<25°, ≥25°) (Köchy et al., 2008), which are mapped to individual grid cells, accordingly. Foraging by sheep and goats reduces the average productivity. In relative terms, this effect is more pronounced, the dryer the landscape. When SD tends to MSC, productivity declines as the grazed herbs produce fewer seeds than required to maintain the un-grazed plant density. Climate change effects on the relationship between SD and green biomass productivity are covered by two sets of correlation functions, one for current climate conditions (applied in the simulation period 2005–2020) and one for projected future climate conditions (applied in the period 2025–2050).

2.3. Quantification of environmental impacts of grazing

Environmental impacts of grazing were assessed on different spatial scale levels. On the cell level (local level), we used the concept of *Human Appropriation of Net Primary Production* (*HANPP*). On the landscape level, different pattern metrics were applied. In addition a monetary valuation of the ecosystem services was conducted.

2.3.1. Cell-level analysis – HANPP

According to Haberl et al. (2007) *HANPP* defines the aggregate impact of land use on biomass available each year in ecosystems. It is calculated as follows:

$$HANPP = \Delta NPP_{LC} + NPP_h \tag{2}$$

In this equation ΔNPP_{LC} defines changes in net primary production (*NPP*) induced by soil degradation, soil sealing, and ecosystem change, while NPP_h is the part of *NPP* which is harvested or destroyed during harvest. As the focus of our study is on grazing land, we assume that NPP_h equals the forage demand of sheep and goats. Based on these prerequisites the relative *HANPP* ($HANPP_{rel}$) is calculated according to Equation (3) with NPP_0 being the *NPP* of the potential natural vegetation.

$$HANPP_{rel} = \left(\frac{HANPP}{NPP_0}\right) \cdot 100 \; [\%] \tag{3}$$

In our analysis, the variables of these equations were determined by the WADISCAPE model (see Section 2.2). NPP_0 was derived from the non-linear correlation functions using stocking density (*SD*) and productivity of green biomass at $SD = 0$. The *NPP* of the actual vegetation (NPP_{act}), which was determined via the applied *SD* in the previous time step, was subtracted from the NPP_0 in order to calculate ΔNPP_{LC}.

2.3.2. Landscape-level analysis – habitat fragmentation

According to Cushman (2006), landscape composition describes the variety and abundance of land-cover types, while landscape configuration is the spatial arrangement of a specific land-cover class within a landscape – the landscape pattern. Landscape patterns and ecological processes are closely related to each other, i.e. a strongly fragmented landscape may lead to isolation of habitats and a higher extinction risk for plant and animal species (e.g. Kruess and Tscharntke, 1994). In this context, Fahrig (2003) describes habitat fragmentation as a landscape-scale process involving the loss of habitat area and the breaking apart of that habitat area into fragments.

In order to quantify the impact of different grazing management strategies on habitat fragmentation, we analyzed the results of our simulation runs with a set of 6 landscape metrics that were identified as suitable for Mediterranean landscapes in Jordan (Alhamad et al., 2011). For analysis at the class level, the land-use maps generated by the LandSHIFT model for the years 2005 and 2050 were combined with the corresponding maps of $HANPP_{rel}$ in order to separate the two classes "strongly human-influenced vegetation cover" (Class 1) and "semi-natural vegetation cover" (Class 2). It is assumed that Class 2 cells provide a higher habitat quality for flora and fauna than Class 1 cells. We tested three threshold levels of $HANPP_{rel}$ (30%, 40% and 50%) as criterion to distinguish between the two classes. Grazing cells with a $HANPP_{rel}$ greater than the respective threshold and cells of land-use types "urban area" and "cropland" were grouped in Class 1, while Class 2 comprised the remaining grazing cells and cells with semi-natural vegetation cover. In the next step, Class 2 was evaluated with our set of metrics using the software package FRAGSTATS Version 3.3 (McGarigal et al., 2002).

The "Number of Patches" (NP) metric simply counts the patches of a specific habitat (in our case cells grouped in Class 2) within a landscape. According to McGarigal et al. (2002) it is only a general measure for spatial subdivision with limited meaningfulness, but it forms the basis to calculate more advanced metrics. In contrast, the "Largest Patch Index" (LPI) and the "Proximity Index" (PROX) are regarded as measures for describing the "breaking apart" of landscapes as part of a fragmentation process. The LPI describes the percentage of the largest patch of the habitat compared to the total habitat area, while PROX distinguishes sparse distributions of small habitat patches from clusters of large patches. According to Gustafson and Parker (1994), the PROX value "becomes large when a patch is surrounded by larger and/or closer patches and decreases as patches become smaller and/or more sparse". Based on Alhamad et al. (2011), the mean value (PROX_MN) and the coefficient of variation of this index (PROX_CV) were calculated. The fifth metric is the total core area (TCA), which sums up the core area of all habitat patches with a specified minimum distance to the edge of each patch (here we selected 1 km). It can be interpreted as a measure for the loss (or gain) of habitat area. Lastly, the "Patch Cohesion Index" (COHESION) measures the connectedness of the corresponding habitat type and increases with a stronger aggregated spatial configuration of patches.

2.3.3. Ecosystem service value

The ecosystem service value (*ESSV*) represents the economic value per unit area of an ecosystem because of its provision of different types of goods and services to the human society. According to Fleischer and Sternberg (2006), we assumed that a semi-natural Mediterranean ecosystem provides a landscape service (LS) that includes the regulating, habitat, and recreation services listed in de Groot et al. (2002). When such an ecosystem is used for grazing, it additionally provides a grazing service (GS), i.e., the provision of feed for sheep and goats. Hence, the total *ESSV* of a managed semi-natural ecosystem is the sum of the economic value of both services: the landscape service value (*LSV*) and the grazing service value (*GSV*).

The landscape service value *LSV* was derived from a basic estimate of \$969 ha^{-1} for forest (Costanza et al., 1997), which we assumed to be adequate for natural vegetation under mesic Mediterranean climate (MM). In order to adjust the *LSV* to the physiographic conditions in the remaining climate regions of the study area, we scaled this value by the proportion of (average) total biomass relative to the total biomass for MM. Fleischer and

Sternberg (2006) provide total biomass values of 19.1 t ha^{-1} for MM, 11.3 t ha^{-1} for Mediterranean climate (MT), 6.1 t ha^{-1} for semi-arid climate (SA), and 2.8 t ha^{-1} for arid climate (AR). Hence, the maximum landscape value LSV_m for natural vegetation was set to \$969 ha^{-1} for MM, \$574 ha^{-1} (=59.2%) for MT, \$309 ha^{-1} (=31.9%) for SA, and \$142 ha^{-1} (=14.7%) for AR.

The actual *LSV* of an ecosystem deviates from LSV_m if it is used for grazing since total biomass decreases depending on the stocking density. This decrease is represented by the $HANPP_{rel}$ as calculated from the WADISCAPE model output (Section 2.3.1). Accordingly, we calculate the actual landscape value LSV as

$$LSV = LSV_m \cdot \frac{100 - HANPP_{rel}}{100}. \quad (4)$$

The grazing service value (*GSV*) was derived from the savings in feed costs for sheep and goats due to grazing. It is calculated from the stocking density (expressed in livestock units) multiplied with the costs for feed per livestock unit. In their case study for Israel, Fleischer and Sternberg (2006) report savings in feed costs of \$116.5 ha^{-1} given a green biomass production of 0.832 t ha^{-1} (under MM climate). In LandSHIFT, the proportion of green biomass that can actually be grazed is 70%, which reduces the green biomass consumed by small ruminants to 0.582 t ha^{-1}. From these figures and the feed demand, the annual feed costs per sheep or goat results in \$49.

In order to compare scenarios assuming different livestock numbers and management options, the *ESSV* is expressed as the average value per unit area (\$ ha^{-1}) of Jordan's total country area.

3. Modeling procedure

3.1. Input data

Spatial land-cover data for LandSHIFT in the year 2001 is provided by the gridded land-cover map from the MODIS global remote sensing dataset (Friedl et al., 2002). The map has a spatial resolution of 30 arc seconds and distinguishes land-cover types according to the IGBP classification (Loveland et al., 2000). During the initialization step, LandSHIFT combines the land-cover data with information on grazing land and stocking densities of grazing animals (see 2.2). Result is a land-cover/land-use map for the year 2005 as starting point for the scenario simulations. For the assessment of grazing suitability, the LandSHIFT model uses grid-level information on land-use, landscape parameters, and nature conservation area: Human population density was derived from the Global Rural-Urban Mapping Project (CIESIN, 2004), while terrain slope is based on the HYDRO1k dataset (USGS, 1998). River network density was calculated from the line density of rivers per grid cell, using a 10 arc minutes search radius, based on the RWDB2 River-Surface Water Body Network dataset (FIMA, 2011). Data on national and international nature conservation area was taken from the world database on protected areas (WDPA, 2004).

Main model drivers include livestock numbers to calculate the spatial pattern of grazing land with LandSHIFT and climate data to calculate biomass productivity with the WADISCAPE model. Livestock numbers were derived from statistical data for the period 1961–2006 (FAO, 2011). For the base year 2005, the model was initialized with the three-year average of livestock numbers from 2004 to 2006, which are 625 511 goats and 2 051 683 sheep. For the conversion of goat and sheep numbers to Livestock Units (LU), needed as input format by LandSHIFT, we assumed that one sheep or goat accounts for 0.1 LU in developed countries (Seré and Steinfeld, 1995). For Israel (which belongs to that category), Perevolotsky et al. (1998) estimate a daily feed demand of 1350 g dry matter. According to Seré and Steinfeld (1995), the daily feed demand of goats or sheep in Jordan is approximately 50% compared to Israel, which results in a conversion factor of 0.05 LU per sheep or goat for Jordan. Further, we assumed that 30% of the feed demand is covered by grazing (Al-Jaloudy, 2001).

Climate data was taken from the regional climate modeling exercise for the Jordan region described in Smiatek et al. (2011). Assuming that the political target for limiting global warming to 2 °C is reached (Meinshausen et al., 2009), we chose the simulation results from the A1B scenario calculated with the MM5 model (Grell et al., 1995), driven with boundary data from the global circulation model ECHAM5. The spatial resolution is 18 km. For the period 2031–2060, which represents the year 2050, temperature increases by about 2 °C compared to the climate normal (1961–1990), while annual mean precipitation decreases by 10%, accompanied by more frequent dry spells.

3.2. Parameterization and validation of the LandSHIFT model

3.2.1. Model parameterization

The factor weights for the suitability assessment within the LandSHIFT model were determined according to the CRITIC method (Diakoulaki et al., 1995). We calculated "objective weights" based on the contrast intensity of the evaluation criteria, i.e., the standard deviation of normalized criteria values, and the inter-criteria correlation. Resulting factor weights are 0.43 for river network density, 0.28 for terrain slope, 0.21 for biomass productivity, and 0.08 for population density.

3.2.2. Model validation

Ideally, spatially explicit land-use models should be validated against changes in at least two land-use maps over time, as has been done for example by Pontius (2000) and Pontius et al. (2004). As grazing land cannot be identified as a separate land-cover/land-use type in available remote sensing datasets (see 3.1), this approach is not suitable for our study. As an alternative, our strategy is (1) to compare the grid maps of calculated suitability for grazing land with maps of observed change of livestock density and (2) to compare the calculated extent of grazing land in the starting year 2005 with the observed area of grazing land.

We defined the observed change of grazing land as the change of small ruminant density between 2000 and 2005 derived from the Global Small Ruminant Density Map (FAO, 2011). A change from non-grazing to grazing was assumed if the small ruminant density increases by 25% and by a minimum of 25 animals per km^2. The calculated suitability map was tested by means of a relative operating characteristic (ROC) analysis, which relates proportions of correctly and incorrectly classified spatial predictions, in our case changes of the land-use type from non-grazing to grazing (e.g. Pontius and Schneider, 2001). The ROC performance measure is calculated as the trapezoidal approximation of the area under the curve (AUC). The resulting AUC value is 0.84 (Fig. 3), which is significantly higher than the value for randomly distributed suitability values (AUC = 0.5) and hence indicates that changes in land-use can be found predominantly at locations where LandSHIFT calculates high suitability values.

The simulated extent of grazing land in 2005 of 10 206 km^2 (see Section 4) was compared to available data and literature sources. We apparently overestimate the extent of permanent grazing land given by FAO (2011), which is 7410 km^2, by almost 42%. However, Al-Jaloudy (2001) estimates the total grazing area of Jordan is 80 710 km^2, with 69 077 km^2 of which receiving less than 100 mm annual rainfall. In these regions grazing is very extensive and management can be classified as a traditional nomadic system. The remainder of 11 633 km^2 can be managed more intensively with

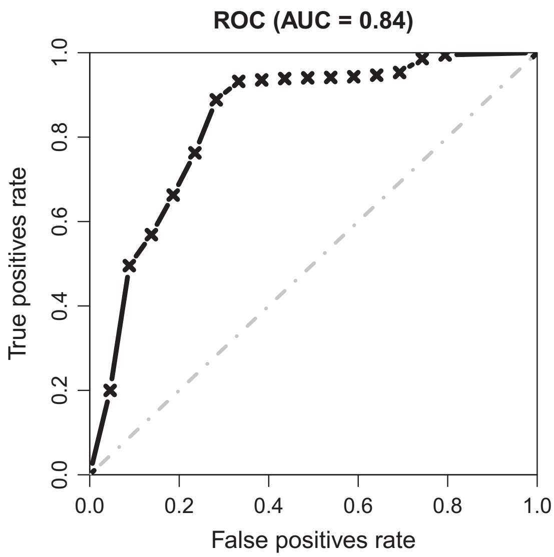

Fig. 3. ROC-curve for validation of the calculated grazing suitability map. The resulting area under curve (AUC = 0.84) indicates that changes of grazing land between 2000 and 2005 can predominantly be found on well suited grid cells.

semi-nomadic, settled, or intensive systems. These data and the location of regions with more than 100 mm rainfall show a good agreement with the model output for the base year 2005.

3.3. Scenario analysis

The scenario analysis consists of 24 simulation runs from 2005 to 2050, in the following referred to as scenarios (Table 1). The spatial extents of urban area and cropland remain constant at the year 2005 level. Each scenario represents a management option together with a specific combination of the two model drivers “livestock number” and “climate change”. For our analysis we have defined four management options representing different grazing intensities: maximum stocking densities of 34% (“MSC34”), 50% (“MSC50”), 66% (“MSC66”) and 100% (“MSC100”) of the maximum stocking capacity (MSC) of each cell as calculated by the WADISCAPE model. Regarding the driver livestock number, we distinguish three alternatives. The first alternative is constant 2005 livestock numbers (“ConstLS”). The second alternative is the linear extrapolation of the historical trend, based on data from 1961 through 2009 (FAO, 2011), until 2050 (“TrendLS”). Under these assumptions, livestock numbers increase to 0.9 million goats (+44% compared to 2005) and 3.6 million sheep (+80%). The third alternative extrapolates the historical trend with the annual growth rate reduced by 30% (“RTrendLS”). Additionally, the climate change driver has two options. Either MM5 simulation results for the period 1971–2000 were used for the whole simulation period (“NCC”) or the MM5 simulations results for the A1B emission scenario were applied (see Section 3.2), representing future changes of temperature and precipitation patterns (“CC”).

Subsequently, we refer to the various combinations of the development of livestock number, climate input, and management option with unique identifiers. These are composed from the abbreviations (given in parentheses above) in three groups according to the scheme ”livestock development”–“climate input”–“management option”. For example, ConstLS-CC-MSC34 identifies the scenario with constant livestock number considering climate change and a management scheme with maximum $SD = 0.34^{*}MSC$. If one or more of the groups are absent, the identifier refers to the subset of scenarios matching the specified groups

Table 1
Model drivers and results from the 24 simulation runs for the year 2050. The left three columns describe the combination of drivers for each simulation run with respect to livestock number, climate change and management. “ConstLS” assumes a constant livestock number whereas “TrendLS” is the extrapolation of the historical trend and “RTrendLS” reduces this trend by 30%. The climate driver describes either the current climate (NCC) or a climate change scenario (CC). Management distinguishes between the maximum stocking density (MSC100) and reduced stocking densities with 34% (MSC34), 50% (MSC50) and 66% (MSC66) of the maximum stocking density.

Livestock number	Climate	Management	Extent [km^2]	Stocking density [LU/km^2]	$HANPP_{rel}$ [%]	Grazing suitability [0…1]
ConstLS	NCC	MSC100	10 508	17.15	83	0.536
ConstLS	CC	MSC100	10 578	17.04	82	0.514
RTrendLS	NCC	MSC100	14 205	17.34	83	0.512
RTrendLS	CC	MSC100	14 497	17.00	81	0.483
TrendLS	NCC	MSC100	16 125	17.06	83	0.499
TrendLS	CC	MSC100	16 597	16.57	81	0.468
ConstLS	NCC	MSC66	10 681	16.87	82	0.535
ConstLS	CC	MSC66	11 298	15.95	76	0.508
RTrendLS	NCC	MSC66	14 743	16.71	80	0.508
RTrendLS	CC	MSC66	17 153	14.35	70	0.464
TrendLS	NCC	MSC66	16 840	16.33	80	0.495
TrendLS	CC	MSC66	24 480	11.23	57	0.426
ConstLS	NCC	MSC50	12 881	13.94	67	0.520
ConstLS	CC	MSC50	14 565	12.29	57	0.483
RTrendLS	NCC	MSC50	18 675	13.19	65	0.482
RTrendLS	CC	MSC50	35 289	6.98	35	0.398
TrendLS	NCC	MSC50	21 695	12.68	64	0.463
TrendLS	CC	MSC50	47 466	5.80	30	0.380
ConstLS	NCC	MSC34	20 319	8.81	41	0.472
ConstLS	CC	MSC34	42 554	4.21	21	0.386
RTrendLS	NCC	MSC34	33 532	7.35	37	0.418
RTrendLS	CC	MSC34	65 833	3.63	18	0.353
TrendLS	NCC	MSC34	39 380	6.99	36	0.406
TrendLS	CC	MSC34	65 833	3.63	19	0.353

(e.g., NCC-MSC66 refers to the scenarios assuming constant climate and management option MSC66 in combination with all three options regarding livestock numbers).

$HANPP_{rel}$ was calculated for the start and the end year of each simulation run. In order to explore the effect of grazing intensity on habitat fragmentation, landscape metrics were analyzed for the four management options and a scenario that considers climate change together with constant livestock numbers (ConstLS-CC). To determine beneficial and adverse economic effects of the management schemes applied, the economic ecosystem service value (*ESSV*) in the start and end-year was assessed for each simulation run.

4. Results

4.1. Model initialization

For the model initialization, we calculated a grazing area of 10 206 km^2 in the year 2005. The average $HANPP_{rel}$ of grazing land accounted for 82.3% of the aboveground green biomass while the average stocking density was 17.66 LU/km^2, which is equal to 3.53 sheep and goats per ha. We assumed that allocation was done with the highest management intensity (MSC100). Medium suitability of the grazing cells is 0.534. Fig. 4a shows the spatial distribution of $HANPP_{rel}$ in the initial time step.

4.2. Expansion of grazing land and resulting impacts on $HANPP_{rel}$

Table 1 depicts the results from the 24 simulation runs for the year 2050. In the following, each simulation run has been named with its identifier, composed as described in Section 3.3.

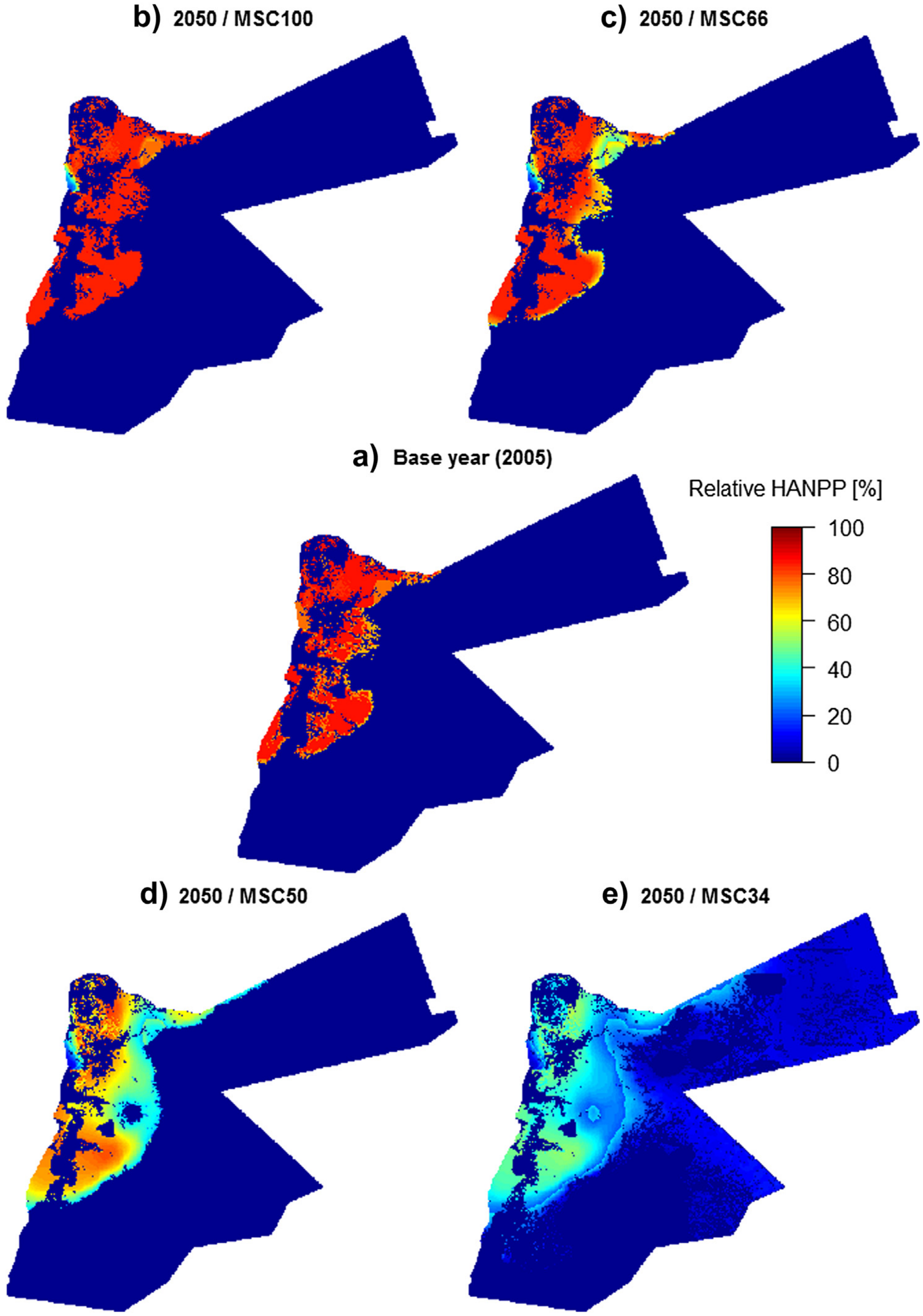

Fig. 4. Spatial distribution of grazing land and $HANPP_{rel}$ across Jordan for ConstLS-CC in a) 2005 and 2050 applying management options b) MSC100, c) MSC66, d) MSC50, and e) MSC34. Values of $HANPP_{rel}$ greater than zero indicate the allocation of livestock grazing.

For the ConstLS-NCC-MSC100 scenario, a minor expansion of grazing area was observed (10 508 km^2). The respective livestock trend scenario (TrendLS-NCC-MSC100) showed an increase up to 16 125 km^2, while the scenario assuming a reduced trend of livestock numbers (RTrendLS-NCC-MSC100) occupied a grazing area of 14 205 km^2. In both scenarios with growing livestock numbers, additional grazing land was needed to fulfill the higher feed demand. The slightly lower medium values for cell suitability (Table 1) already indicated that less productive land had to be used in the allocation procedure of the model.

Under climate change conditions all MSC100 scenarios showed a stronger expansion of grazing land together with a lower medium SD and suitability values. This could be explained by the negative effect of increasing temperatures and decreasing precipitation on net primary productivity under grazing pressure in the majority of the grazing area. This effect was most pronounced for TrendLS-CC (e.g. under the strongest increase of livestock numbers). *Relative HANPP* values showed only minor differences between the simulation runs with management option MSC100 (around 82%).

The reduction of the management intensity in MSC66, in combination with current climate (NCC-MSC66), only lead to small increases in the extent of grazing land compared to MSC100. The other output variables also remained relatively stable. Under climate change, a further expansion of grazing area to more than

24 000 km^2 in TrendLS-CC-MSC66 was observed. At the same time, livestock density and $HANPP_{rel}$ decreased further. Interestingly, the difference between the extent of grazing land under current climate and under climate change was larger than under the MSC100 management, e.g., in the trend scenario this difference was more than 45% compared to only 3% under MSC100.

Under the management option MSC50, the extent of grazing land showed a higher increase than under the more intensive management options, while the values for SD and $HANPP_{rel}$ decreased. The effect of climate change was even stronger with grazing land encompassing 47 466 km^2 in TrendLS-CC-MSC50. This means that under climate change conditions more than twice the area was needed to fulfill the livestock feed demand than under current climate (TrendLS-NCC-MSC50). This negative impact of climate change was also obvious due to the decline in suitability values, meaning that more unproductive land had to be used. Grazing land was managed in a very extensive manner indicated by a mean $HANPP_{rel}$ value of 30%.

The lowest management intensity (MSC34) showed the most dramatic impact. Under current climate, the increase of grazing land was between 93% (ConstLS-NCC-MSC34) and 144% (TrendLS-NCC-MSC34) higher than in the respective scenarios with MSC100 management. Mean livestock density was 8.81 LU/km^2 and 6.99 LU/km^2, while mean $HANPP_{rel}$ values were 41% and 36%, for ConstLS-NCC-MSC34 and TrendLS-NCC-MSC34, respectively. Under this management option, climate change had a strong impact even for the ConstLS-CC-MSC34 scenario. Here, the extent of grazing area almost tripled compared to MSC50 (from 14 565 km^2 to 42 554 km^2). For RTrendLS-CC-MSC34 and TrendLS-CC-MSC34, the demand for grazing area could not be met anymore by the available land resources. Mean stocking density dropped to 3.63 LU/km^2.

We could identify two factors that were responsible for decreasing *SD* and $HANPP_{rel}$. First were the management options with lower intensities that were aiming at a more extensive use of grazing land with lower environmental impacts, expressed by the $HANPP_{rel}$ indicator. This was accompanied by an increase in the extent of grazing land. Depending on the strength of the livestock driver, from a certain point the area increase showed a non-linear behavior. For current climate this dynamics became visible only for the lowest management option (NCC-MSC34). At this point, the second factor came into play. Very unproductive grasslands were used for grazing (expressed by their low grazing suitability), which could be managed only with very low stocking density. This effect positively fed back by further increasing the area demand. Climate change seemed to strengthen these dynamics through its negative impact on biomass productivity. Particularly for TrendLS-CC, this positive feedback effect could already be observed under the MSC66 management option.

The effect of the different management options on the extent of grazing land and the spatial distribution of $HANPP_{rel}$ for ConstLS-CC can be observed in Fig. 4. In general, the extent of grazing land grew with decreasing maximum stocking density, while $HANPP_{rel}$ decreased. Compared to the situation in the initial time step in 2005 (Fig. 4a), the management options MSC100 (Fig. 4b) and MSC66 (Fig. 4c) caused relatively small changes in the extent of grazing land alongside a moderate reduction in $HANPP_{rel}$. However, limiting maximum *SD* to 50% of MCS caused the allocation of livestock grazing to large areas of lower biomass productivity (Fig. 4d). This effect was highly non-linear, which led to a total extent of grazing area, with much lower *rel.* $HANPP_{rel}$, covering almost half of Jordan for ConstLS-CC-MSC34 (Fig. 4e).

4.3. Impacts of grazing on landscape metrics

In order to better understand the effect of the management intensity on habitat fragmentation we analyzed four scenarios, each characterized by constant livestock number, climate change (ConstLS-CC), and one of the four different management intensities. This set of scenarios was combined with three different threshold levels of $HANPP_{rel}$ used to differentiate between semi-natural vegetation and land cover strongly influenced by grazing (see Section 2.3.2). As the COHESION metric remained constant for all simulation runs it was excluded from the analysis.

The results (Table 2) showed large differences between the management options. All options, except the most intensive management (MSC100), were sensitive to the threshold for $HANPP_{rel}$. In contrast to 2050, applying the different thresholds did not result in changes of any metric in 2005 since $HANPP_{rel}$ was greater than the maximum threshold applied throughout.

In the case of the strictest threshold ($HANPP_{rel} \leq 30\%$), we found the largest areas of land cover influenced by grazing, while semi-natural vegetation was at its smallest extent. This effect was best illustrated by the total core area metric (TCA), which was lower for all management options compared to the base year, indicating a loss of habitat area. Although the total grazing land was expanding, the resulting decrease in the lower intensity management options for semi-natural land was partly compensated for by low *SD* and small $HANPP_{rel}$ values, which led to a classification of grazing land as semi-natural vegetation. Patch numbers were lower than in 2005 due to (i) the growing extent of land cover influenced

Table 2
Landscape metrics and $HANPP_{rel}$ calculated for the 4 management options in 2050 using a scenario with constant livestock numbers and climate change (ConstLS-CC). Three different thresholds of $HANPP_{rel}$ were analyzed to differentiate between "strongly human-influenced vegetation cover" and "semi-natural vegetation cover".

	NP [#]	LPI [%]	PROX_MN [−]	PROX_CV [−]	TCA [ha]	COHESION [−]	$HANPP_{rel}$ [%]
2005	99	83.54	9807	98	7 285 440	99.92	82
2050, 30% $HANPP_{rel}$ + threshold							
MSC100	92	82.83	11 196	96	7 272 441	99.89	82
MSC66	85	82.15	10 343	95	7 226 317	99.90	76
MSC50	57	77.51	6771	134	6 919 905	99.84	57
MSC34	66	81.02	5160	170	7 128 929	99.90	21
2050, 40% $HANPP_{rel}$ threshold							
MSC100	91	82.83	11 039	97	7 275 378	99.89	82
MSC66	85	82.22	10 391	94	7 238 142	99.89	76
MSC50	39	80.13	5068	164	7 121 658	99.88	57
MSC34	135	87.11	3776	220	7 691 003	99.86	21
2050, 50% $HANPP_{rel}$ threshold							
MSC100	91	82.83	11 040	97	7 277 949	99.89	82
MSC66	89	82.49	9620	102	7 256 283	99.89	76
MSC50	38	83.02	5876	146	7 404 935	99.89	57
MSC34	100	94.10	10 145	116	8 245 512	99.92	21

by grazing (conversion of semi-natural patches) and (ii) the closing of the gaps between semi-natural patches by lowering $HANPP_{rel}$. At the same time, the LPI was lower, which indicated that the largest patch of semi-natural vegetation became smaller compared to the total area of semi-natural vegetation. This development could be explained by an increase of grazing land with $HANPP_{rel}$ being larger than 30% that sprawls into existing semi-natural land. Looking at the spatial patch distribution, the proximity index PROX_MN was slightly higher than in 2005 for MSC100 and MSC66, meaning that this aspect of habitat fragmentation decreased (Table 2). In contrast, this value was dropping for MSC50 and MSC34, showing that the remaining patches were more scattered than in 2005. This finding was supported by the coefficient of variation which increased in these two cases.

Setting the threshold to 40% of $HANPP_{rel}$, the TCA metric grew in particular for ConstLS-CC-MSC34 indicating an increase of habitat area. Under the most extensive management, area classified as semi-natural vegetation exceeded the 2005 extent despite the land used for grazing being almost 4 times larger than in the base year. The 40% threshold pronounced the stronger positive effect of the lower *SD* and $HANPP_{rel}$ values typical for that management option. This effect could also be seen in the LPI which surpassed and the PROX_MN which is lower than the respective 2005 values indicating a lower habitat fragmentation compared to the base year.

A relaxation of the threshold to 50% $HANPP_{rel}$ led to a further increase of total core area (TCA), LPI, and PROX_MN, in particular under the ConstLS-CC-MSC34 scenario.

4.4. Impacts of grazing on the economic ecosystem service value

Due to identical assumptions in the initial time step, the *ESSV* in 2005 was the same for all simulation runs and had a value of \$199.05 ha^{-1}. Fig. 5 shows the change in *ESSV* until 2050 ($ESSV_{2050}$ minus $ESSV_{2005}$), which varied in the range between \$ + 4.27 ha^{-1} (CONSTLS-CC-MSC34) and \$ − 6.42 ha^{-1} (TrendLS-CC-MSC100). In general, the *ESSV* decreased with a growing absolute number of goats and sheep. The only exception, observed for the scenarios assuming constant climate, was management option MSC34 (NCC-MSC34), which showed a slight increase in average *ESSV* with growing livestock numbers. This indicated that additional livestock grazing was allocated to areas where the additional savings in feed costs over-compensated for the reduction of *LSV* as a function of $HANPP_{rel}$ Another finding was that the *ESSV* in 2050 was always higher for management option MSC34 as compared to scenarios with identical assumptions on climate and livestock number but different management options.

A comparison of the CC- and NCC-scenarios showed that the management options had varying potential to alleviate adverse impacts of climate change on the *ESSV* (Fig. 6). Under MSC100, climate change led to an additional small loss in *ESSV* until 2050 regardless of the livestock number, while climate change impacts were negligible or positive if MSC66 was applied. Interestingly, climate change resulted in an increasing *ESSV* throughout for management option MSC50. Thus, the change in *ESSV* was between \$ + 0.45 ha^{-1} (MSC50-ConstLS) and \$ + 1.62 ha^{-1} (MSC50-TrendLS). In the case of management option MSC34, such a positive effect could not be observed for increasing livestock numbers because the reduced average biomass productivity under climate change was insufficient to feed the projected number of sheep and goats in RTrendLS-CC-MSC34 (failure to feed 49 726 sheep or goats) and TrendLS-CC-MSC34 (failure to feed 211 658 sheep or goats). Therefore, out of the four management options, MSC50 can be considered as the best alternative to adapt to the climate change impacts projected in this study.

5. Discussion and conclusion

5.1. Environmental impacts of grazing

Our analysis reveals major differences between the management options regarding their environmental impacts on the local level. Management options with lower stocking densities are characterized by a less intensive use of local resources (lower $HANPP_{rel}$) while the area required in order to fulfill the feed demand becomes successively larger. A secondary effect, which cannot be observed directly in the output maps, is the further reduction of productivity of (semi-)natural vegetation under high stocking densities. We account for this land degradation by using the correlation functions between stocking density and landscape productivity provided by the WADISCAPE model for the calculation of the $HANPP_{rel}$ indicator (ΔNPP_{LC}).

Using $HANPP_{rel}$ to discriminate "semi-natural vegetation" against "strongly human influenced land cover" was the key to linking grazing intensities with the analysis of landscape patterns (see Section 2.3). This approach goes beyond relating the natural vegetation class to specific land-cover types derived from remote sensing data as for example done by Geri et al. (2010). We could observe that lower management intensity also has benefits on this level of analysis. This is expressed by a lower level of habitat fragmentation characterized by greater proximity between the patches (PROX_MN and PROX_CV) and by stable or increasing area classified as semi-natural vegetation (habitat area) described by the TCA metric, even if the extent of grazing land is strongly increasing. A major difficulty in evaluating the habitat fragmentation with landscape metrics is the definition of an adequate threshold value to distinguish the two land-use classes. We could show that the

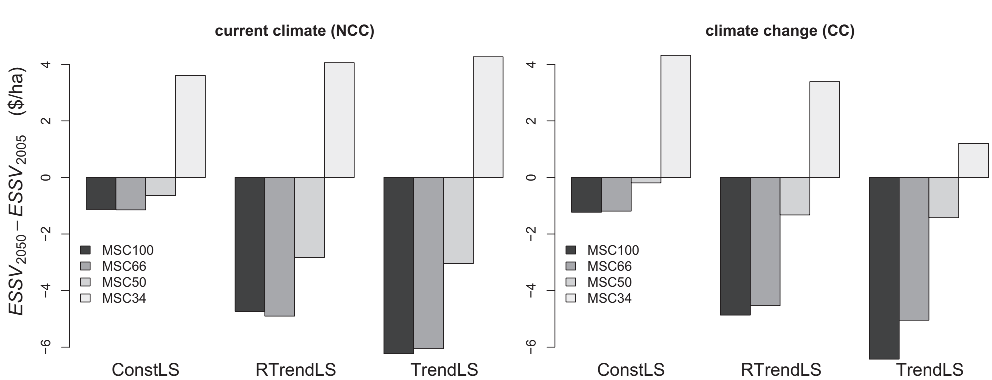

Fig. 5. Change in ecosystem service values (*ESSV*) between 2005 and 2050 for all simulation runs grouped by variants with current climate (NCC) and climate change (CC).

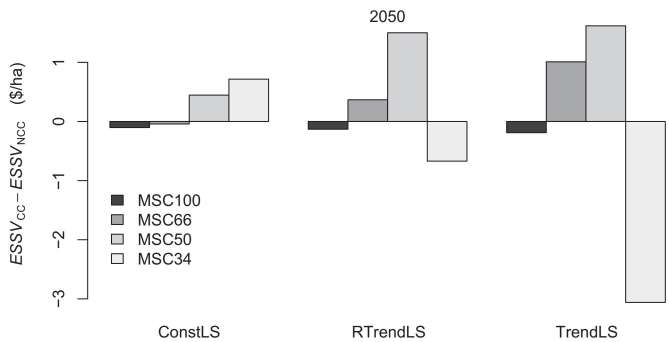

Fig. 6. Differences in ecosystem service values (*ESSV*) between climate change (CC) and non-climate change (NCC) simulation runs in 2050.

results are sensitive to the threshold value. The threshold value for $HANPP_{rel}$ should be in a plausible range, e.g., values far beyond 50% or much lower than 30% are not useful.

In summary, the combination of local and landscape level indicators provide a suitable tool to analyze the effects of different types of grazing management on biomass productivity and habitat fragmentation as important environmental impacts. Nevertheless, it would be desirable to establish a closer link to ecological field studies in order to interpret the results in respect to the disturbance of ecosystem processes and biodiversity (Gustafson, 1998). In this context, it also has to be noted that the chosen landscape metrics were originally applied to quantify the effect of cropland and urban expansion on fragmentation (Alhamad et al., 2011). Hence, further studies should systematically investigate if other metrics might be more suitable to account for effects of intensively managed grasslands on habitat fragmentation.

Our simulation results support the findings of Abahussain et al. (2002) that climate change will play an important role for the future expansion of grazing land in Jordan. An important finding is that under changing climate conditions and low-intensity management options the livestock numbers of the trend (TrendLS) and reduced trend (RTrendLS) scenarios cannot be allocated within Jordan anymore. Hence, in these scenarios the limits of growth for management with low environmental impacts are reached.

5.2. Valuation of ecosystem services

By integrating the landscape service value (*LSV*) and savings in feed costs (*GSV*), the ecosystem service value allows us to account for trade-offs between the preservation of open semi-natural vegetation patterns and the economic benefits of grazing. However, the *ESSV* is sensitive to the economic valuation of these two services. Regarding the determination of the landscape value, Fleischer and Sternberg (2006) discuss possible approaches as well as their problems and uncertainties to valuate services that are not traded at markets (non-market-values). As a first approximation, we linked *LSV* to total biomass productivity as proposed by Costanza et al. (1997) allowing us to directly assess the effect of grazing intensity on *LSV* by combining the concepts of *HANPP* and ecosystem services. The monetary *LSV* has been derived by a benefit-transfer approach that does not consider regional characteristics (e.g. social attitudes or personal income), which is clearly a drawback of our current study design. However, we could illustrate that the *LSV* can be interpreted as an indicator to integrate two major effects of lowering the grazing intensity, which may counteract each other. Reducing the stocking density leads to less degradation of the landscape service, i.e. less reduction of *LSV*, on grazing land, while at the same time the conversion of additional semi-natural vegetation into grazing land is required to fulfill the feed demand. This in turn reduces the *LSV* on these newly allocated cells (see Equation (3)). Savings in feed costs (*GSV*) refer to the market value of fodder and therefore can be determined relatively accurately. As long as the feed demand is met, these savings directly correlate to the livestock number. However, it is an important simplification that additional costs on farm level involved in the expansion of rangeland at a fixed number of livestock, e.g., for transportation, provision of drinking water, or sun shades, are not considered in our approach. Taking into account these additional costs might reduce the economic benefit of extensive grazing considerably.

Due to the strict linkage of *LSV* to biomass productivity, the interpretation of the *ESSV* should be done with care as other environmental aspects such as biodiversity, soil functions, or structural diversity of the vegetation cover were not considered directly. For example, the reduction of the *ESSV* decreases when area with a lower *LSV* is used for grazing. We can see that the ecosystem service value increases monotonically as a function of *SD* in the climate region AR (LSV_m = \$142 ha^{-1}), while it decrease monotonically with SD in the climate region MM (LSV_m = \$969 ha^{-1}). This might mislead to the conclusion that it was economically reasonable to shift grazing livestock production to less productive ecosystems, although this may not be advisable from a broader ecological point of view.

Despite the aforementioned shortcomings, we found that *ESSV* is a suitable instrument to compare and to rank the different management options regarding their ability of avoiding negative impacts on the landscape and regarding their performance as a climate adaptation measure. It helps to address trade-offs in an intuitive way that can be effectively communicated to stakeholders and decision makes involved in regional planning processes or in natural resource management activities.

5.3. Uncertainties and limitations

For our analysis we have soft-coupled a biophysical model (WADISCAPE) to a spatially explicit land-use model (LandSHIFT). The coupled model could be validated against spatial data of changing livestock densities and statistical data on the extent of grazing land. A limitation of our study design is the focus on a single land-use activity. Competition for suitable land between grazing, farming, and urban development is not yet considered but plays an important role for land-use change in Jordan (Alhamad et al., 2011). Hence, our model approach does not account for potential effects of these processes on the spatial extent and patterns of grazing land.

Regarding the input data we can identify three major sources of uncertainty. (1) There is a lack of detailed historical spatial data describing the extent of grazing land. In contrast to cropland, this land-use type cannot be observed directly even with very high resolution remote sensing techniques, which is a major problem when building relevant datasets. This causes large uncertainties for the initialization of our land-use model as assumptions about grazing land area in Jordan range from 7500 km^2 (FAO, 2011) to more than 70 000 km^2 (Al-Jaloudy, 2001). Furthermore, this situation hinders a more rigorous validation of the model. (2) The *MSC* used in this study is calculated from green biomass production assuming a constant feed demand per sheep or goat of 0.675 kg d^{-1} (dry matter). This value corresponds to the demand of a goat with body weight of 30 kg when fed in stables (low muscular activity) (NRC, 1981). However, Lachica et al. (1999) report that, in semi-arid Mediterranean rangelands, goats require an extra 31–47% of metabolic energy over maintenance for locomotion. Thus, a reduction of *MSC*, and hence the maximum *SD* used in our scenarios, would increase the simulated area of rangeland by about the same amount. Consequently, livestock demand would no longer be met in LSTrend-CC-MSC50, LSRTrend-CC-MSC34, and LSTrend-CC-MSC34, while grazing area in LSConst-CC-MSC34 would reach the limits of the system by 2050. (3) We applied only one regional climate projection. As we have seen that climate change is an important driver for the expansion of grazing area, a more detailed analysis should incorporate at least two realizations of regional climate scenarios calculated with different models, or with an even larger model ensemble, in order to portray the uncertainties involved in the climate simulations (Smiatek et al., 2011; Stainforth et al., 2004).

6. Conclusion: implications for natural resource management

The results of our study are a contribution to the discussion about maintaining intact open landscapes (Köchy et al., 2008; Tielbörger et al., 2010) as a means of sustainable land management of Mediterranean ecosystems. We could illustrate that it is possible to reduce the environmental impacts of grazing both on local and landscape level by lowering the management intensity. When the monetary value of landscape services is taken into account, also the economic benefits of preserving an intact vegetation cover become obvious. These effects are even more pronounced if we consider the potential negative impacts of climate change on biomass productivity. In addition, climate change is likely to limit the suitable area for grazing in Jordan and low-intensity grazing may be constrained considerably. Consequently, policies for the protection of natural resources should also promote a reduction of total livestock numbers. At the same time, such policies must aim at providing alternative sources of income to pastoralists and farmers, e.g., by alternative uses of open landscapes for recreation and eco-tourism (e.g. Tielbörger et al., 2010). In this context, especially the method of ecosystem service valuation can become a suitable tool to support the development of climate adaptation strategies integrating both ecological and socio-economic aspects.

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