Session 3:

Integrated environmental modelling

Regional integrated assessment modelling of desertification for policy support

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ABSTRACT

This paper reviews the role of integrated assessment modelling in the better understanding of land degradation and desertification and specifically for improved policy formulation to prevent, mitigate or adapt to the consequences of these processes. Integrated assessment modelling (IAM) attempts to couple the socio-economic drivers (of desertification) with the biophysical processes and their outcomes. This is best achieved as a bi-directional feedback in which: (a) the socio-economic drivers force the biophysical desertification processes (alongside external biophysical drivers such as aridification), (b) the desertification processes lead to some development of land or environmental condition which may be either stasis, aggradation or degradation and (c) the socio-economic system responds to those changes which then further force the socio-economic drivers, and so on, in a feedback loop.

Integrated assessment modelling is a relatively new approach with most models to date having been confined to either the biophysical or the socio-economic domains, rarely both. Indeed the majority of models used in desertification cover only part of the biophysical system (for example. soil erosion models) or part of the socio-economic system (for example land use change models) whereas desertification is actually a suite of multiple, interacting processes and factors (for example aridification, soil erosion, biological decline, tourism, agricultural change). Integrated assessment models are – necessarily – spatial in nature and thus take account of spatially varying biophysical and socio-economic states, patterns and processes as well as the important lateral flows which connect natural and economic landscapes. These models are also focused on policy relevant scales - which tend to be larger than the data-driven, experimental scales of most process models. This spatiality and scale necessitates the availability of significant spatial data for the parameterisation of climate, soils, vegetation, water resources, landscape and land use, population and infrastructure. Thus, integrated assessment modelling highly dependent (for model parameterisation and validation) upon inputs from remote sensing and geoinformation processing. This is especially true for desertification because desertification processes operate at fine grains yet the policies that drive (or prevent) them are usually implemented regionally. A desertification IAM must thus represent both fine grains and large extents, in both space and time.

I briefly review the science behind the assessment of land degradation indicating the role of measurement, modelling and remote sensing in the assessment process. I indicate the value of modelling, the process of building IAMs and the tools available to do so. I discuss the significance of model integration and feedbacks in policy-relevant modelling. I then discuss the nature and requirements of research and policy models and of decision support systems. Finally, I then present a case study of one of the most advanced Integrated Environmental Models for the assessment of Desertification: the MedAction Policy Support System (PSS). The model is described briefly and the role of remote sensing in providing data for model application and validation are discussed. Finally, a case study on the application of the MedAction PSS for better understanding the impact of climate and policy changes on desertification in the Guadalentin Basin, SE Spain is discussed with particular emphasis on the feedbacks between socio-economic and biophysical processes, negative feedback and unforeseen consequences.

1 INTRODUCTION AND AIMS

Land degradation takes many forms and occurs in many different environments. Where land degradation occurs in dry sub-humid, semi-arid and arid zones and is characterised by a diminution of the biological potential of the land caused by climatic aridification and/or the impact of human land or water use, then it is often called ‘desertification’. Desertification is not a process but rather is the outcome of a series of processes that act together to transform environmental conditions from productive to less productive (and thus more desert-like). Desertification is not often a simple transformation to desert as might occur at an existing desert margin, for example. Rather, desertification most often occurs in the context of a spatially complex mosaic of patches of aggradation, stasis and degradation *but* where the patches of degradation have the greater spatial extent or impact on society. The balance between the state of these patches and the location of them relative to human need from the land will determine the perception (and impact) of the state of desertification. Patches may be degrading because of their physical properties (for example south facing with high evaporation losses; low hydraulic conductivity or high upslope area leading to high runoff and erosion), because of human use of the patch (unsuitable land use and land management practices) or because of impacts imported from other (connected) patches (for example fire, flood, reduced runon, wind-eroded sediment deposition). This latter, lateral connection is another important feature of desertification, and adds another level of complexity in terms of understanding on-site and off-site controls (usually upstream[1]) and effects (usually downstream) of desertification. Desertification is the result of processes that may be slow and gradual (such as aridification, soil erosion and exhaustion, productivity decline) or much more rapid but episodic, ‘extreme events’ (for example flooding, erosion and mass movements, wildfire) and thus rates of desertification may be highly spatially and temporally variable [1]. Human drivers may also be gradual (e.g. long term overgrazing) or episodic (e.g. seasonal burning).

Desertification engages with human society because it affects the reliability of natural resources on which large, complex and demanding societies have come to depend. Since it is unpleasant (inconvenient, costly and at its worst impossible) for those societies to adapt farming and living conditions to cope with resource change, desertification can be a threat to human health and wellbeing. Desertification also engages with society because human impacts in the landscape are often a driver of desertification. So, humans can be part of the cause and are certainly prone to the effects of desertification. The purpose of this paper is to examine desertification with a particular emphasis on the European Mediterranean and a particular focus on the spatial complexity of the processes leading to it and the extent of coupling and feedback between the biophysical and human components. My aims here are to:

(a) describe more of the (process, spatial and temporal) complexity of the processes which produce desertification,

(b) review some key IAMs for understanding European desertification in an integrated way (including human and biophysical components, their linkages and feedbacks) over space and time,

(c) indicate areas in which further conceptual and technical development are necessary to improve the science, the data and the models.

(d) provide examples of the use of these models in better understanding desertification dynamics.

I will first discuss the science and rationale for desertification assessment before presenting a short review of the variety of processes of desertification that should be taken into account in any particular assessment. The bulk of this paper examines some key IAMs for desertification and the data requirements for the successful application of these models, indicating the role of remote sensing in the spatial provision of these data. I conclude with a case study of a state-of-the-art integrated desertification model (the MedAction policy support system) and its application. The model is described briefly and a series of case studies on the application of the model for better understanding the impact of climate and policy changes on desertification in the Guadalentin Basin, SE Spain are discussed. This leads to a concluding discussion on necessary developments for the improvement of integrated assessment models generally and also specifically in desertification policy support including the potential role of remote sensing and geoprocessing.

[1] ‘Stream’ here is considered analogous to ‘network‘ so the connections may be upstream in the case of fluvial processes, upwind in the case of Aeolian processes or up or downstream along some other relevant network, which may even be socio-economic.

2 LAND DEGRADATION AND DESERTIFICATION

In this text I will use the terms land degradation and desertification interchangeably since desertification is a subset of land degradation applied according to the definition given in section 1.0. My use of the term land degradation (which I prefer to desertification because of its more accurate and less alarmist description of the issue) should be understood to refer only to the many types of *dryland* land degradation. In other words when I say land degradation here I mean the same as when I say desertification : a reduction in the biological potential of the land and consequent effects in dry environments as a result of combined climate and human impact.

2.1 Defining land condition and degradation

2.1.1 Measures

In order for one to assess land degradation one must first have some non-destructive measures of land condition that can be repeated over time and compared in order to first define and then measure degradation, stasis or aggradation. The measures of land condition that one chooses will depend upon the impacts of land degradation that one is interested in. An ecologist would probably use measures such as species richness, natural vegetation cover, change in extinction rates. A farmer would be more interested in crop yields, artificial (water energy and nutrient) input requirements and the outcome of these two –agricultural profit. A water resource manager would focus more on water resource reliability, quality and quantity in the main subsurface, surface and fluvial 'reservoirs'. Thus all of these 'policy makers' will have different views on the most important measures of land condition and also different views of the direction in which a change in land condition is degradation or aggradation.

2.1.2 My degradation, your aggradation

By way of example, reduction in natural forest cover over time may be an important measure of land condition to the ecologist and the water resource manager. The reduction is a degradation to the ecologist but an aggradation to the water resource manager (because of reduced evaporation from shrublands compared with closed forests[1]). Degradation means different things to different 'policy makers'. Moreover, as is clear from the case of the consequences of forest loss for the water resource manager, a simple change in condition may have many opposing effects (see footnote) some of which are beneficial to the water resource manager and others of which are not and thus some judgement as to whether this results in an overall aggradation, stasis or degradation has to be made by the policy maker.

2.1.3 Scientific uncertainty

Our scientific understanding of the processes is such that even determining whether a particular change in land condition represents a degradation or not for a particular policy maker, like the water resource manager, may not always be possible. Of course some measures of land condition are much more clearly either degradation or aggradation: reduced agricultural productivity is a degradation for the agriculturalist because it leads to reduced incomes and profits –clearly. However, reduced agricultural productivity may be an aggradation to the ecologist, especially where reduced productivity leads to agricultural abandonment and natural regeneration. So, even in this very first stage of desertification assessment there can be difficulties in defining (a) appropriate measures of land condition and (b) which changes in those measures constitute an overall degradation. This is partly because of the scientific uncertainty in understanding the controls on desertification and partly because of the variety of stakeholders operating in any one landscape and their very different views of what constitutes an adverse change.

2.1.4 Part of the problem

In providing science-based policy or decision support one sometimes has a single stakeholder in mind (a water resource manager, tourism enterprise, agricultural stakeholder or environmental group). The danger of building systems for a limited set of stakeholders are that the policy actions that may seem to be appropriate in preventing, adapting to or mitigating the impacts of desertification on the single or few stakeholders may have negative consequences for other stakeholders. Thus - as much as possible - science needs to move away from partial descriptions of the environmental and human interactions in desertification, towards a more holistic treatment of multiple processes and multiple stakeholders. Since the same human-environment interactions can have very different outcomes in different spatial locations (because of their different properties) and at different times (because of the impact of cyclical changes in drivers such as climate and global markets) spatial and temporal

[1] though this ignores the potential water resource benefits of forest over shrubland in terms of potentially increased baseflows, better flow regulation, reduced erosion and thus increased water quality. Thus the water resource manager must determine whether the balance of these effects makes reduced forest cover a hydrological aggradation or degradation.

variation must also be accounted for in these systems. Such an approach requires a multidisciplinary effort that brings together data and process descriptions from a variety of disciplines describing the relevant human-environment interactions and integrating them with appropriate feedback loops in a scientifically meaningful and technically plausible way. As we will see, such an approach is very difficult to achieve outside a modelling framework.

2.2 The Assessment of Land Degradation

2.2.1 The purpose of assessment

Avoiding the deleterious impacts of desertification involves prevention, adaptation or mitigation. The primary purpose of desertification assessment is to sectorally and spatially target efforts at prevention, adaptation and mitigation. Assessment assists by providing spatially targeted information on desertification hazard that can be used to assist land use and land management planning. Thus the ideal desertification assessment must provide the appropriate information for the relevant strategy. No system exists to date that will do all of this especially because different strategies for overcoming desertification (prevention, adaptation or mitigation) require different types of information.

2.2.2 Assessment for prevention

The prevention of desertification involves monitoring changes in land condition, identifying the cause of any degradation and adapting land and water use practices and intensities in order to put less pressure on the more degrading or more sensitive areas of the landscape. In other words preventing desertification by removing its drivers. This may mean changing land use to require less irrigation, having lower cropping densities, using more native or drought-tolerant varieties and accepting lower yields. It may also mean changing land management practices to encourage lower water use and protection against soil erosion by, for example, contour ploughing or low tillage, retention of rock fragments [33], or low input farming. Thus, an assessment system that is focused on prevention, needs to be able to forecast the cumulative impact of current practices occurring in particular landscape positions, examine the cost of the resulting desertification and be capable of simulating the impact of land use and land management scenarios designed to reduce desertification. The cost of implementing these scenarios should be less than the benefit of reducing desertification that results from their implementation. It is then down to the regional and national financial bureaucracy to find mechanisms for redistributing costs and benefits between the different stakeholders such that the costs are incurred in an equitable manner relative to the benefit winners.

2.2.3 Assessment for adaptation

Adaptation is quite different to prevention. It does not focus on halting desertification but rather on adapting the use of the land to the changing land condition. Adaptation is particularly important where desertification is forced by external drivers (for example global climate change) and is thus unpreventable by the local or regional stakeholders or where prevention is otherwise not possible or the degradation not so serious as to warrant preventative measures. The requirements of an assessment system for adaptation are less focused on scenario testing and assessment of long term cumulative impacts and more focused on comparative analysis of different land use and land management practices in order to know which is possible (sustainable) under a degrading environment. Thus, the focus is not on preventing desertification through land use and land management change but on finding the use-management combination that provides the greatest benefits (without necessarily accelerating desertification) and continuing to adapt those practices as the land condition changes. It is thus an optimization exercise rather than a comparative cost:benefit exercise. The key difference between prevention and adaptation is that prevention attempts to find a stable use-management combination which is sustainable in the long term and avoids land degradation whereas adaptation recognises that degradation cannot be prevented and provides an adaptive scenario which gets the most from the land as degradation proceeds, always in the hope that stasis or aggrading conditions will return.

2.2.4 Assessment for mitigation

The focus of mitigation is in the development of land use and management practices that lessen the actual or potential negative effects of desertification so that, although degradation is allowed to take place, mitigation focuses, not on adapting to the degrading potential of the land but rather, on developing land management strategies that help recover land condition and reduce the signs of desertification. Mitigation is the application of 'technical fixes' for desertification. Such strategies might include the development of irrigation schemes, the construction of reservoirs and erosion dams, the transfer of water from wetter regions to drier ones and the development of high input agricultural techniques that import production resources from outside the degrading region (e.g. greenhouse crops). Assessment for mitigation focuses on understanding the dominant processes of desertification in a region and simulating the success of different mitigation techniques in reducing the deleterious impacts. Assessment systems may examine user interventions such as terracing and the construction of check dams and compare the cost of construction and maintenance of these features with the benefits gained from them and even taking into account any negative impacts of these solutions in other areas or process domains.

2.2.5 The science of assessment

When we talk of desertification assessment we must distinguish between the assessment of desertification intensity and extent and the assessment of desertification causality. Most of the scientific effort to date has focused on the former with various assessments of desertification extent both globally [9],[3] and regionally, especially in the Mediterranean [10],[6]. The global assessments have often taken the form of maps from the World Desertification Map (1977) prepared by UNEP, FAO, UNESCO and WMO for the United Nations Conference on Desertification (UNCOD), through the UNEP (1998) Plan of action to combat desertification (PACD) map to the assessment of UNEP/ISRIC[1] (1990) and the World Desertification Atlas (UNEP, 1992), now in it's second edition [16]. Most of these efforts combined national expert opinions and field-collected data with mapping and later GIS techniques and some simple multi-criteria analysis. These efforts continue in the various desertification early warning systems (EWS) such as the Degradation Early Warning System and (DEWS)[2] and The European Land Degradation Monitoring System[3]. Some of these latter efforts make extensive use of satellite remote sensing as well as ground-collected data. These efforts also continue in the various indicator approaches to desertification assessment summarized by Geeson [5].

Any indicator of change should be quantitative and objective, sensitive to the change of interest, few in number and readily measured at the necessary scale [3]. Most indicators are focused on desertification assessment rather than causality and most are indirect since not all symptoms of desertification (e.g. reduction in the biological potential of the land) are always the consequence of desertification. Moreover, indicators, like most environmental variables are rarely constant in time so that a sufficiently long record of observation is necessary to separate an important trend in such an indicator from natural background variability. Different indicators will be representative of different processes of desertification and will usually be most representative at a well-defined scale. Moving from the complex 'landscape' of desertification to the few indicators of desertification necessary for operational use is a rather subjective and challenging process.

In order to select and apply appropriate actions to counter desertification we must also assess desertification causality since maps of desertification extent alone cannot be used by policy makers in the implementation of control measures. This is not to say that maps are not the correct tool: the spatially variable nature of desertification processes and thus of the controls necessary to counter them means maps are appropriate. However rather than focusing on *where* desertification occurs, they need to focus on *why* it occurs there. This transition from studying *where* to studying *why* has occurred as desertification has progressed from a poorly known issue in need of attention into one of the best understood of the global environmental changes in which the focus is now on remedy rather than diagnosis (policy rather than science). Understanding the controls on desertification in a particular area requires assessment to focus on the causal processes of desertification. The direct cause of desertification is usually (mis) management of the land in a way which is inappropriate for its climate but this is, in turn, a consequence of various indirect causes which may include: population pressure, land tenure (or lack thereof), international markets for crops, inappropriate land management or agricultural policies, drought, poverty, inadequate agricultural extension services, and lack of research [3]. Thus desertification assessment tools that aim to inform policy by tackling the question '*why does desertification occur here?'* must continue the use of ground based and remote sensing data and continue to provide spatial assessments but also incorporate an understanding of physical processes as well as these direct and indirect causes. This is best achieved by integrating environmental and socio-economic modelling with spatial information systems driven by remote sensing data and focused on policy relevant scales. Such an approach provides a formal integration of process, data and context and may be used to develop policy-focused indicators of desertification in a more robust manner than can be achieved by direct human interpretation of the data and the science.

[1] International Soil Reference and Information Centre
[2] http://www.geog.umd.edu/LGRSS/Projects/degradation.html
[3] ftp://ftp.fao.org/agl/agll/lada/montana.pdf

2.3 Geospatial modelling of land degradation

Before examining in detail the role of modeling in desertification, let us look more generally at the purpose o modeling as a research activity in the environmental sciences. This is necessary in order to understand the flavours of modelling that are most appropriate in tackling desertification.

2.3.1 Definition of modelling

To produce a model is to produce a simplification of reality. A model can be used to formalise understanding gained through data collection or theoretical advance and to explore the properties of that understanding. Models can take the form of conceptual models which formalise theories in a pictorial, illustrative, rule-based or tabular, non-computational form, physical models - which are hardware representations of processes, or mathematical models which use the formal language of mathematics to represent processes and the relationships between them. Within the context of scientific research, models are tools for simplifying, formalising and testing theories as well as for implementing projections of particular scenarios for future change. Models are often used to integrate different research activities, in particular theoretical cogitation, delphic knowledge, field and laboratory-based experimentation and monitoring as well as earth observation. Within the context of desertification research there appear to be two basic types of modelling strategy. The first strategy sees models as single equations to represent a specific process whilst the second sees models as software tools for integrating the results of interdisciplinary, multi-institution research efforts. As well as integrating the research activity itself, models are also used to explore the consequences of theoretical developments that are made as a result of the research and are therefore used as exploratory tools for developing and presenting research results.

The development of models can be a primary research objective, a post-research integrative tool or both since modelling can be used as a means of:

i. Understanding the system. Particularly where research includes a number of simple processes connected in complex ways (such as desertification) or where the research deals with understanding complex processes, models are a useful way to simplify the systems and explore their sensitivity to different parameters or manipulations,

ii. Testing of hypotheses. Models can be used to formally test hypotheses under particular environmental conditions that are representative of the same or different locations to that of interest and in the past, present or future.

iii. Prediction and scenario development. Models can be used to develop predictions based on current knowledge of the impact of present environmental changes or scenarios of future environmental changes.

2.3.2 Simple versus complex models

Environmental models come in all levels of complexity. If we follow the principal of parsimony or Ockham's razor, the best of two equally predictive models is always the simplest of them. This is better expressed as the best model is that which achieves its objective most simply. This is not the same as saying the simplest model is always the best…. sometimes a particular objective requires a complex model. Indeed, different objectives require vastly different levels of model complexity. The objective of modeling is sometimes prediction (of empirical events or states), sometimes explanation (of processes, events or states) but increasingly exploration of data, concepts and processes and their interaction. For prediction, the simplest model can often be as good a predictor of a variables trajectory as a complex model and so simplicity (parsimony) is paramount. For explanation, it is important that the process descriptions are as accurate and complete as possible and this may require an added level of complexity. For exploration, the model needs to be as complete a representation of processes and their interactions as necessary and this requires the most complex parsimonious model of the three cases since the model is trying to replicate the significant complexity of the real system. If the measure of a model’s worth is its predictive capacity then Ockam’s razor (“Given two equally predictive theories, choose the simpler”) clearly applies but when the measure is the accuracy of representation of the internal workings of the system, since the real system will always be more complex than the model, the best model will be the more complex one if it is also a better representation of the system. But it should only be as complex as necessary, and no more. Though, according to Leonardo da Vinci “simplicity is the ultimate sophistication“, nature is inherently complex and thus even simplified representations of nature tend towards complexity.

Complexity in nature arises in many ways, some of which are inherently simple. Complexity can arise from the (a) existence of complex processes, (b) the existence of simple processes with complex outcomes, (c) the accumulation of repeated simple processes over spatial or temporal variation, or (d) from the interaction of multiple simple processes. (a) and (b) are truly complex whereas (c) and (d) are better termed ‘complicated‘ since they are not in any way simple. So simplicity and complexity are not necessarily y at opposite ends of a spectrum but simple and complicated are. A more thorough representation of environmental complexity would have to include:

(a) Process complexity – the sophistication and detail of the description of processes,

(b) Spatial complexity – the spatial extent and grain of variation (and any lateral flows) represented,

(c) Temporal complexity – the temporal horizon and resolution of system dynamics,

(d) Inclusivity (complication) – the number of processes included,

(e) Integration – the extent to which the important feedback loops between processes are closed.

Environmental researchers have tended to concentrate on (a) whereas (b)-(e) are probably more important in natural systems, especially at spatial and temporal scales outside of those used in experimentation. There is an enormous difference between the complexity that one works with in an experimental or laboratory setting and complexity in natural systems at policy-relevant scales.

2.4 Model integration

For a long time the focus in science has been on reductionism, splitting nature into pieces in order to understand how the parts work. Reductionism has led to significant progress in many areas but it does not allow us to understand whole systems well though we need to be able to do exactly that in managing desertification and the environment generally. Nowadays modelling and computers are enabling us, for the first time, to put these pieces together and understand their interactions and emergent properties as a (process, spatial and temporal) whole and this capability is critical to understanding response to environmental change. IAM or integrated environmental modelling attempts integration across: issues, scales, stakeholders, disciplines (biophysical and socio-economic), and models. IAM helps in exploring the behaviour of bi-directionally coupled human-environment systems and helps to highlight the previously-unforeseen consequences of scenarios of change and policy or other interventions. Integration is important because : (a) the links and feedbacks between processes can be as important for system behaviour as the processes themselves, (b) integration facilitates the production of decisions and policies that are well tested with respect to their long term sustainability and that do not have deleterious unforeseen and unintended consequences in other policy domains and (c) nature is complex (especially with humans around) and much of this complexity results from the links between processes rather than the processes themselves.

We have seen that desertification is a bewildering series of processes connected by a complex web of feedbacks loops. The human brain is not well adapted to understanding or tracking the outcomes of these processes and feedbacks (even at only one point in space and time). Miller [17] showed that a person can store 7 ± 2 items (numbers, faces, words, processes) in their short term memory under optimal conditions. In the presence of distractions even three items can become difficult and may disappear in 2-18 seconds [27],[11]. Thus the outcome of a complex set of processes can be more robustly examined computationally than mentally. The real mental effort is in the understanding of system components and linkages that are required in the development of the model rather than the routine execution of that model. Such models are thus ‘thinking tools’ that can be used to shed light on problems that otherwise cannot be managed by the human brain alone and thus, a model based on the best available knowledge and assembled in a careful and well-tested manner can be a useful aid in the decision making process.

2.4.1 Challenges in Integrated Assessment Modelling

In this paper integrated modelling is used to refer to (spatial) modelling systems which couple models from a variety of disciplines in the science and social science domains with a view to understanding the outcomes of their interaction and the impact of scenarios for change (e.g. climate change) and policy interventions (e.g. land use planning) upon them. This integrated modelling promises a great deal in terms of better understanding complex environmental problems but also provides a series of important challenges that must be solved before its full potential can be realized. Model integration itself is a challenge. Engelen [4] points out that there are very few operational definitions or procedures for model integration available from the scientific literature. Appropriate model integration requires attention to scientific integration early on (*which models should be included?, which parameters and variables should be passed between them?, how should differences in model philosophy, time and space scale be dealt with?*). In addition, one has to tackle the technical integration aspects (*how should the integrated model be assembled and run?*). Other challenges posed by integrated modelling include:

(a) *Parameterisation* – integrated models by their very nature tend to have a large number of parameters. If the integrated model is to be used for decision support (as is often the case) then model parameterisation at policy relevant scales becomes a significant challenge and the need for integration with satellite remote sensing and geo-information processing becomes all the more important.

(b) *Calibration* – even if individual models operate well in isolation, they may need re-calibration (or redesign) when coupled with other models. Calibration of an integrated model is complex and time-consuming because model integration focuses on tight coupling and thus parameter interdependence is common.

(c) *Uncertainty and Error propagation* – though there is some evidence that errors in model components become more evident when the models are integrated rather than run in isolation [2], this may not always be the case and understanding error propagation in a complex, highly connected model is a significant intellectual and computational challenge, especially for DSS where input data are lean and end users are require some estimates of the uncertainty of model outcomes in order to trust and use them.

(d) *Validation* – validation is always difficult, never more so than in multi-component and projective models. The validation problem stems in part from lack of available data but also from the need to ensure that validation is a true measure of model performance against some 'reality' rather than a measure of differences in scale, method and time of collected validation data compared with the model representation of the same data, as is often the case. The focus of validation in integrated models has to be primarily on verification of model behaviour against expectations and secondly in the validation of critical subcomponents rather than attempts at validating the whole integrated model, which are very likely to be unfeasible. Validation at the latter level takes the form of critical appraisal of outcomes rather than numerical validation *per se*.

(e) *Understanding the outcomes* – the more complex the model the more difficult it is to ascertain the series of process interactions that led to a particular outcome. As models become more inclusive they gain exploratory power but lose interpretability. There are - to date - few tools to assist in the 'mining' of model results and the interpretation of complex model outputs. In the end, the few simple indicators of desertification causality required in the policy domain will need to be generated from model output in the same way that indicators of desertification extent will need to be generated from remote sensing output. Well-developed protocols and methodologies for this do not exist and much work is necessary in that area.

(f) *Documentation* – the larger and more inclusive the model, the greater the task of fully documenting its routines, assumptions and codes and the more difficult the task of making this documentation accessible and interpretable to the models user community in both the scientific and the policy domain.

There are many advances that need to be made in terms of computational power, data availability and integration, multiparameter calibration, error analysis, verification and validation, interpretive tools and indicators and documentation technologies before IAMs will be on an equal footing with simpler or single process approaches to modeling. We will later highlight how these specific problems have been tackled in the MedAction PSS but there remains a long way to go. Consequently, IAMs and any encompassing decision support capacity should be used for explorative rather than predictive purposes. Indeed simple (empirical) models are usually better for prediction, complex or integrated and physically based models are better for understanding and exploring the system and that is the *only* way that these types of model should be used. If they are used a prediction machines then they are used incorrectly.

The basic process for development of an IAM is as follows:

(a) Manageable slices of science are produced tested and wrapped into modules.

(b) The modules are linked appropriately with parameters, state variables and other linked process modules.

(c) A scientific integration ensures the modules are scientifically compatible (conceptually and philosophically).

(d) A technical integration ensures the modules connect appropriately in terms of data being passed at the right times and spaces (and in the correct physical units!).

(e) A spatial database is added.

(f) Appropriate temporal scenarios (outside control of policymaker) and policy interventions (controllable by the policy maker) are added.

(g) The model is tested and run. The computer deals with the (short term) memory and logic leaving the scientists to deal with the science.

2.4.2 Tools for building Integrated Assessment Models

Of course any low or high level computing language can be used to build IAMs but most IAM developers are domain specialist scientists (geographers, ecologists, economists, land use planners) and not computer programmers and thus require high level modelling tools that allow one to build IAMs without investing a lot of time and effort into computer programming or deep mathematics. IAMs are usually built and tested using these systems even if they are later 'hard-coded' in a computer programming language for improved execution efficiency (such as the MedAction system). An IAM development system requires two main capabilities (a) the capacity to deal with spatial and temporal data and (b) the ability to easily modularize and integrate a range of submodels. We can separate the existing IAM tools into those that have evolved as modelling extensions of GIS software (e.g. ESRI MODELBUILDER and AML, ERDAS SPATIAL MODELER, those that have evolved as spatial extensions of modelling software (Stella , Modelmaker ,Vensim , Extend and SIMILE) and custom built spatial modelling tools (e.g. PCRASTER[31]; SME[13]). Of course, systems like Stella, Modelmaker, Vensim, Extend and SIMILE can and have been used to develop IAMs but they are usually not so suited to environmental IAMs where spatial variability is important, thus custom built spatial modeling tools like PCRASTER and SME are much better suited to this task.

In general, Geographical Information Systems (GIS) have been rather slow to take on simulation modelling. Whilst these systems can have a very considerable arsenal of spatial functions which can be used to represent processes on raster or vector datasets, very few provide an easy to use modelling environment which facilitates spatio-temporal modelling. In the early ESRI products the most effective means of developing spatial models was through programming in arc macro language (AML) for ARCGIS workstation, though this is a rather low level scripted language and requires a significant investment of time to build - even simple - spatial models. On the other hand AML does provide access to the full array of ESRIs spatial modelling functionality. More recently in ARCGIS Desktop 9 the so-called 'model builder' provides a drag and drop functionality for building spatial models, though spatio-temporal models are constructed with great effort this way. Scripting in ARCGIS Desktop 9 is provided by the open source Python programming language but this is again, beyond the un-trained capabilities of most domain specialists.

Modelling systems have, again, been rather slow to take on board the spatial modeling that is so critical to Environmental IAMs (E-IAMs) . Many now incorporate routines that allow for replication of calculations through a cellular grid and thus the development of spatial or individual based models but very few offer significant spatial functionality in addition to their in-built mathematical, trigonometric, logical and other functions and are thus rather limited for most spatial modelling efforts. Thus both standard GIS and modelling software are deficient for the kind of spatial modeling required in E-IAM. For GIS this is because, though the spatial functionality is good, the modelling capability is either limited or based on rather technical programming languages (and thus user-unfriendly). Modelling software are much easier to work with for most domain specialists. They are based on systems thinking approaches and usually with drag and drop model building functionality and sophisticated modelling-support tools for development, testing, visualisation and sensitivity analysis. However, modelling software usually have very limited spatial analysis capability and focus only on replication of calculations for different individuals or objects that may have no spatial relationship (e.g. neighbourhood, upstream, downwind).

The third type of system - purpose built spatial modelling tools - overcome many of these deficiencies. There are very few of these in comparison with the number of GISs and modelling software. Though a number of blueprints for such a system exist, the main operational examples are PCRASTER and SME. PCRASTER is a script based spatio-temporal modelling language with over 115 functions more than half of which are truly spatial functions. It supports only raster grids, not vectors or objects. The software is developed by the University of Utretcht and the PCRASTER Environmental Software company and is currently freely available as windows or linux executable (though not open-source) and has been used widely. The shell-based script language is relatively simple and the system has useful visualisation tools but a domain specialist must have some experience of script based languages to get by and the various utilities are rather loosely knit compared with standard GUI based modelling packages such as STELLA. Nevertheless PCRASTER has an extraordinarily powerful array of spatial and non-spatial functions and its development is ongoing.

SME is developed by a team at the International Institute for Ecological Economics of the University of Maryland. It takes simulationsmodels developed in STELLA-like systems and allows the user to integrate them within the context of a gridded space. SME also supports vector and object data models. STELLA, Vensim, Extend or Simulab models can be easily built using drag and drop but integrating them spatially in SME is an altogether more technically involved process involving the definition of the project, the creation and linkage of Simulation Module Markup Language (SMML) modules from the STELLA models, the assembly of data in the appropriate format, the generation of (C++) code from the SMML objects, and the configuration of the code prior to model run and visualisation. Though 'generators' are available to do much of this, it appears rather technically challenging for the average domain specialist. SME is Unix software which also somewhat limits its application to most non-specialists but this does enable it to support parallel processing, important for much large scale, long term or fine resolution simulation.

Both PCRASTER and SME remain in development though PCRASTER in particular has a considerable user base. They represent that state-of-the-art in true spatial modelling software which is thus E-IAM capable.

Both of these software have tackled some of the major developmental needs in IAM development:

(a) the need for simple high level functions for spatial processes (diffusion, water flow in landscapes, neighbourhood operations) – especially PCRASTER,

(b) the need for rapid model development and testing environments – PCRASTER and SME,

(c) the need to optimise use of computing resources (PCRASTER because its database is held in memory for rapid execution and SME because it permits parallel processing),

(d) the need for simple visualisation tools for spatio-temporal model outputs (especially PCRASTER),

(e) the need for error propagation and uncertainty analysis tools (especially PCRASTER).

However, some serious limitations for the domain specialist interested in building IAMs remain:

(f) the use of text based (scripting) languages in at least part of the modelling process,

(g) the need to convert input data from their original format to a software specific format for use by the systems,

(h) requirements of the user to manage the spatial extent, spatial resolution timestep and order of command processing independently (something that most non-spatial commercial modelling software 'looks after' without user intervention),

(i) Weak capabilities for 'on-the-fly' visualisation and interpretation of results and for results-mining,

(j) The lack of a sophisticated GUI and the integrated feel of many commercial modelling packages,

(k) In the case of PCRASTER weak modularisation and thus code reusability,

(l) In the case of SME specialist hardware requirements.

Many of these limitations are the result of obvious constraints on the model sophistication achievable with graphical model building tools compared with model scripts and on the user-simplicity of scripts compared with graphical 'drag and drop' systems. Those systems that adopt graphical model building will have limited functionality and those that adopt scripting will have greater functionality but be less marketable to non-computer scientists. The ideal IAM development system would have both of these means of model development and a sophisticated set of spatial functions and data mining tools, good modularity of models enabling distribution, use and re-use by the user community as is common with some GIS software (eg. ARCVIEW extensions). The tool would have an easy to use GUI that incorporate all of the systems functionality for parsing data, building, testing, running and analysing models. In addition the software would support very large spatial databases for the large extent, fine scale work which will characterise this field in future decades and would also support parallel or grid processing. This is, if you like, a modeller's wishlist and would enable significant advances in modelling by domain specialists who would, with these needs met, be less constrained by the (largely software-based) technical barriers that exist today.

2.5 'Research' versus 'policy' models

Models are usually built by scientists and social scientists not by policy makers. In many cases their purpose is to advance understanding through research though they are often useful for, and applied to, policy or decision support, particularly where research funding agencies require this as a condition of funding and this has certainly been the focus of EC funded desertification research since 2000. The characteristics of a model designed for research may be somewhat different to those required for a model developed specifically for policy application. But models which begin as research models often evolve into what might be called policy models - models used in decision or policy support. Of course there are no clear boundaries that separate research versus policy models and most models lie somewhere on a continuum from research to policy use. **Table 1** identifies some of the key differences in emphasis between the endpoints of this continuum. The major difference between a research and policy model are in their objectives. Research problems are usually extremely well defined as a testable hypothesis that the model addresses whereas policy problems are rather ill-defined and may be of the type: *what is the optimum land use pattern in the area for sustainable production?; how can we improve nature conservation in the area?; what should we do to adapt to desertification in the area?; how might we improve the livelihoods of the communities in the area*?. These kinds of questions require a quite different approach to modelling than a hypothesis does. The approach required by ill-defined questions is an exploratory and thus integrative approach. The model is thus designed to be more of a general description of the workings of the system ("an instrument for exploration representing part of a complex reality with some degree of certainty", [4]) rather than a specific description of the processes required to confirm or refute a particular hypothesis.

Research models are used primarily to understand how a system functions and thus the description of processes should be as accurate as possible for the particular objective. Policy models, on the other hand, require only the accuracy of process description which is necessary to achieve the policy objective, and no more. Similarly model complexity and resolution for research models often reflect the complexity and resolution of the processes whereas for policy models they reflect the complexity and resolution of available data. Research models are usually scientifically innovative (much of the value of the model is its description of new science) whereas policy models need to be scientifically proven such that they can be trusted in application. Research models usually raise more questions than answers whereas policy-focused models are required to give simple, clear (and true!) answers and the value of a policy model is solely within its output whereas research models have value in their own right (as formal descriptions of the system of interest). Thus, policy models tends to be input/output centred and research models, process centred. All models should be the simplest possible description of the process required for the objective but this is a particularly stringent requirement for policy models that are both data and runtime limited.

Table 1. The characteristics of research and policy models.

Research models	**Policy models**
Research problem well defined as hypothesis which model addresses	Policy problem ill-defined, model more generalized
accurate representation of processes	adequate representation of processes
complexity and (time and space) resolution reflect processes	complexity and (time and space) resolution reflect data
accurate representation of spatial variability	adequate representation (existing data)
Sectoral and detailed	Less detailed but multi-sectoral (integrated or holistic)
scientifically innovative	scientifically proven
raises more questions than answers	provides simple(?), definitive(?) answers
interesting and worthwhile in its own right	interesting and worthwhile only through its output
process centred	input/output centred
numbers validatable	outcomes validatable
as complex as necessary	as simple as possible
as fast as possible	faster (no more than a few minutes running)
data hungry, if necessary.	data lean.

2.6 Models as decision and policy support systems

Policy questions have reached a level of complexity that can no longer be dealt with by politicians alone. High-level technicians play an ever-increasing role, and, the significant advances in hardware and software technologies have equipped them with "very powerful multi-media calculators" [4]. In the last decade there has been considerable growth in the number of governmental organizations who develop these rather sophisticated model-based information systems to support the policy making process. This growth is fuelled by the growing awareness that policy-making should be based on an integrated approach [4] in order to develop policies that are sustainable and that are not prone to negative, unintended consequences in other sectors than the sector of interest. The systems that policy-makers deal with rarely exist in isolation, rather they are parts of a larger entity and interact in feedback loops with other local and global systems and with other spaces and times. Even weak connectivity between system components can have important consequences for the behaviour of the system as a whole (see [12]). Policy makers have to manage these fragile systems that exhibit a very rich behaviour, not least because of the presence of intelligent (human) actors which steer the system in the direction of their interest [4],[34]. They require technical support systems which summarise knowledge (data and understanding) of their system and help them to better anticipate the effects of their interventions within the system as fully as possible [4] in order to do this. The development of such support systems has been fuelled by the requirement for integrated natural resources management [1] and integrated assessment[7], by the availability of research funding to provide the support tools that it requires, by the massive advances in computing and analytical power of the last two decades and the increasing availability of high resolution remote sensing and Geographical Information Systems (GIS) data that make spatial modelling for policy support a reality.

Policy-focused models are often also called decision support systems (DSS) or policy support systems (PSS). DSS refers to a wide range of support tools from maps through spreadsheets to the kind of spatial IAMs which are the focus of this paper. PSS are DSS applied specifically to supporting the policy domain (as opposed to other forms of decision). A D(P)SS has aims to aid a decision maker in addressing unstructured or semi-structured decisions through realizing one or more of the following D(P)SS objectives :

(a) To assist with problem recognition (clarifying or better specifying the problem). This is an important function of DSS and often their most important contribution.

(b) To supplement one or more of a decision maker's abilities e.g. knowledge collection, knowledge derivation, analysis (what if?), and formulation of potential plans for analysis or action.

(c) To facilitate one or more of the decision-making phases e.g. intelligence (providing relevant information), design (identifying or analysing alternatives), choice (which alternative to choose).

In addition, within a desertification policy context a DSS should:

(a) SIMPLIFY- distill complex but good data and science into usable models or simple rules.

(b) INTEGRATE - integrate research results from very different disciplines in a common and formal language (mathematics).

(c) COMMUNICATE - hide complex science from the end user and link scientists with policy advisors.

(e) BE FLEXIBLE - be flexible in the analysis of *scenarios* for change and *policy options*.

(f) BE INTERACTIVE - be interactive, fast, easy to understand and cater for a short attention span.

(g) PROVIDE - to provide the end user with the information they want at the scale they like when they like

The value of a DSS depends upon the nature of the DSS itself, the nature of the decision maker and the decision context (for example the data availability and range of stakeholders). Where these conditions are suitable the deployment of a DSS can augment a decision maker's innate knowledge handling abilities and can solve problems that the decision maker alone would not even attempt or that would consume significant decision-maker time and resources due to their complexity. Even for relatively simple problems, a DSS may be able to reach solutions faster and/or more reliably than a decision maker. If a problem is too complex to be solved using a DSS, the DSS could be used to stimulate the decision maker's thoughts about the problems through exploratory retrieval, analysis, by solving a "similar" problem which triggers insight about the present problem or by providing more compelling evidence to justify or support the decision maker's position. Thus a DSS can also be a learning (and teaching) tool. Finally, one of the most valuable aspects of DSS is the activity of constructing it (with end user or decision maker involvement) which may advance the scientific understanding of the system, reveal new ways of thinking about the decision domain and even partially formalise aspects of the decision making.

DSS are no panacea. They have a number of limitations, especially in tackling problems such as desertification. Two of these are outside of the scientific and technical area of DSS: (a) they only provide information to support decisions, those decisions are then made within a social, cultural and political context which may or may not allow the decisions to mature into policies and (b) such policies then need to be acted upon in a sustained manner if they are to succeed and politics does not always favour this kind of long-termism. The remaining limitations are scientific and technical:

(a) DSS are unable to replicate some innate human knowledge management skills/talents such as scientific reasoning,

(b) DSS may be too specific, that is, many models and DSSs may be needed for a single decision so a means of coordinating them is required,

(c) A DSS may not match the decision makers mode of expression or perception, making communication difficult,

(d) DSS cannot overcome a faulty decision maker or one who is not making science-based decision/policy,

(e) a DSS is constrained by the knowledge it possesses (the data, the models and the analysis and visualization tools) ,

(f) No amount of glossy presentation can overcome scientific or technical flaws in a DSS, one has to remain always critical of the output of DSS (as with any information system) and not become over-dependent upon them for making decisions,

(g) Models are never finished, they should continuously evolve as the knowledge and experience of the modeler and the computational capacity evolves. A stagnant model is not a good one. The same also applies to DSS.

2.7 Building a DSS

Engelen [4] defines the basic structure of a DSS as containing four major components, (a) the database that includes model parameters, GIS and remote sensing based spatial data and timeseries data, (b) the model base (process descriptions and their connections), (c) the tool base (analytical techniques and functions), and (d) the user interface which hides the complexity of the first three components and presents the results to the decision maker in a way that they can understand. There are two very different ways in which DSS can be constructed. In some cases existing research models are tweaked and have user interfaces added or are integrated loosely with other models within a DSS shell (this so-called soft-integration was the approach taken by many of us in the MODULUS DSS [4],[25]. In other cases scientific models are redesigned from scratch as policy models, taking note of the requirements outlined in Table 1 (this so-called hard integration was the approach adopted in the MedAction PSS [32]). The former is often necessary where there already exists a bank of research models representing a substantial investment which, with a little further funding, can interface better with the policy domain. However, the latter is usually a much more successful entrance to the policy domain.

2.7.1 Soft integration

Soft-integration is a matter of choosing appropriate models which share a common modeling philosophy and which together simulate the processes and spatio-temporal domains of interest. These models are analysed in order to know where their points of connection are, how one might link them temporally and spatially with the model database and with other models and which variables they should obtain from the database and which from other models, where, when and in what units. Integration is then a technical matter of either recoding or software 'wrapping' the existing models and having them run together with the appropriate database, toolbase and user interface and then verifying that the system works. Such a process often produces a rather imprecisely aimed, buggy, slow and cumbersome system since the models are developed with different objectives in mind and may not follow the requirements identified in Table 1 very well, if at all.

2.7.2 Hard integration

Hard integration is altogether more challenging and rewarding. It involves the re-development of models as policy models through changing their: spatial and temporal domains, data requirements, output focus, level of connectivity and integration and general philosophy and then tightly integrating them within a time and spatial domain, with a database, toolbase and appropriate user interface. The models would simulate the relevant processes, allow user exploration of the system and its dynamics and display outcomes as readily interpretable indicators. This approach produces a much more policy-relevant model but is a significant investment of time and effort.

2.8 Integrated desertification modelling : recent developments

I will now cover in more detail the most recent (and current) developments in IAM of desertification that took place in three EU projects in which the author was/is a partner, starting with the MODULUS[1] project, then MedAction[2]. In MODULUS the research team essentially took a wide variety of models written by different

[1] http://www.riks.nl/projects/MODULUS
[2] http://www.riks.nl/projects/MedAction

research groups in different previous EU projects (ERMES[1], ARCHEOMEDES[2], EFEDA[3]) and coded in different languages (Pascal, C, LISP...), chose the most suitable from them, making the necessary modifications for better integration and wrapped them in ACTIVE-X wrappers specifying the connections between models and passing data between them at runtime within the Research Institute for Knowledge Systems (RIKS) GEONAMICA© DSS system. This approach allowed existing models to be used but : (a) does not make those models into policy models, they tend to remain rather heavy research models, (b) produces a rather slow, clunky product that is not well streamlined for policy application, (c) requires a lot of work to ensure that the model philosophies are similar/compatible and that there is a good scientific rationale for their wrapping and integration in this way. The MODULUS system proved that this kind of scientific and technical integration was indeed possible but produced no more than a prototype DSS which is useful as a scientific test-bed IAM but is probably not very attractive as a DSS. Nevertheless MODULUS proved that it was technically feasible to produce a working system using this soft integration and the system was applied to desertification processes in the Marina Baixa of SE Spain and in the Argolid of Greece [23],[24],[25].

In MedAction we took a quite different approach and redeveloped all of the models from scratch in hard code. The models were based on knowledge developed from previous desertification research activities and models but were re-developed to be less like ‘scientific models’ and more like ‘policy models’ though there is, of course, a continuum between them. The models were tightly coupled, fine tuned, made to work with routinely available data (particularly from remote sensing) , tightly integrated with feedback loops, closed and coupled with the appropriate database, toolbase and GUI. The PSS development was driven by policy makers in the region and used model-derived indicators as a tool for better communication of model outcomes to policy makers. The advantages of this approach over that of MODULUS are that the end product is more likely to be useful to a policy analyst, it is likely to be more streamlined, simpler, better targeted at the specific policy issue and better rounded in terms of the robustness of the underlying model philosophy and the quality of the process integration (especially across the biophysical/socio-economic divide). The model was applied to desertification issues in the Guadalentin Basin of SE Spain [32].

2.9 The Medaction PSS

Here I describe the basics of the MedAction PSS before presenting an example application of the tool. The purpose of the MedAction PSS was to make use of the best models of Mediterranean desertification to assist technical policy analysts in:

(a) understanding the processes shaping the region and their linkages,

(b) identifying and anticipating current and future problems in the region,

(c) designing policy measures to mitigate the problems and assessing their effectiveness,

(d) evaluating different alternatives and selecting a candidate for implementation.

This was achieved with tightly coupled process models with a regional extent, 100m grain, timescales from sub-minute to annual and a 30 year time horizon, from 2000. The model includes six core modules (see **Figure 1** : climate and weather, hydrology and soil, vegetation, water management, land use and farmer’s decisions) within which a series of submodules exist, for example hydrology and soils includes submodules for hydrology, sedimentation and salinisation. Within a submodule are a series of submodels for example the hydrology submodule contains models for the soil moisture balance, runoff generation, streamflow and groundwater. Each sub-model is made up of a series of processes. For the water balance sub-model these are evapotranspiration, infiltration, soil drainage, throughflow and continuity. The process is defined by a number of parameters and state variables that are either generated by another process in the same or different submodel, submodule or even module or derived from the spatial or temporal databases. The process is run for each relevant raster cell of the spatial database, conditioned by interactions with other processes and with inputs from the temporal database at the appropriate temporal resolution (for example rainfall inputs). The running model provides a variety of spatial (map-based), temporal (graph based) and statistical (spreadsheet based) output of scientific interest and a series of moving indicators of environmental sensitivity of policy relevance. The policy analyst interacts with the system via these indicators but also via a series of policy options and scenarios which they may apply to the system.

The MedAction system couples external and internal biophysical processes with external and internal human processes. External biophysical processes include climate and weather which must be supplied as either historic

[1] http://www.maths.leeds.ac.uk/Applied/news.dir/issue6/arts/ermes.html
[2] http://www.ucl.ac.uk/archaeology/research/profiles/mcglade/glde9.htm
[3] http://medias.obs-mip.fr/ricamare/interface/projet/efeda.html

data files or are downscaled from GCM-derived grid box projections of climate change for three GCMs (HADCM2, ECHAM and GFDL, IS92 scenarios). The downscaling makes use of local station data in order to increase the spatial and temporal resolution of supplied historic daily data or projected monthly data. Climate and weather thus containing a series of weather generators designed to improve the spatial and temporal realism of routinely available meteorological data in line with the needs of process-based hydrological models in environments where desertification can be an event-based process [21] and thus it is important to properly represent storm dynamics. Internal biophysical process are all of those within the spatial realm of the model and which do not have to be supplied as a boundary condition. These include all of the hydrological processes (surface and subsurface), soil wash erosion and sedimentation, vegetation growth, development and succession for crops and natural vegetation and soil salinisation. The external socio-economic components include external markets for crops, agricultural incentives, water 'imports' and the various policy options (water pricing, terracing, crop planning and irrigation). These must all be supplied to the model as data. The internal socio-economic components include water demands and usage, water resources allocation, land use (change), profit and crop choice and dynamic land suitability.

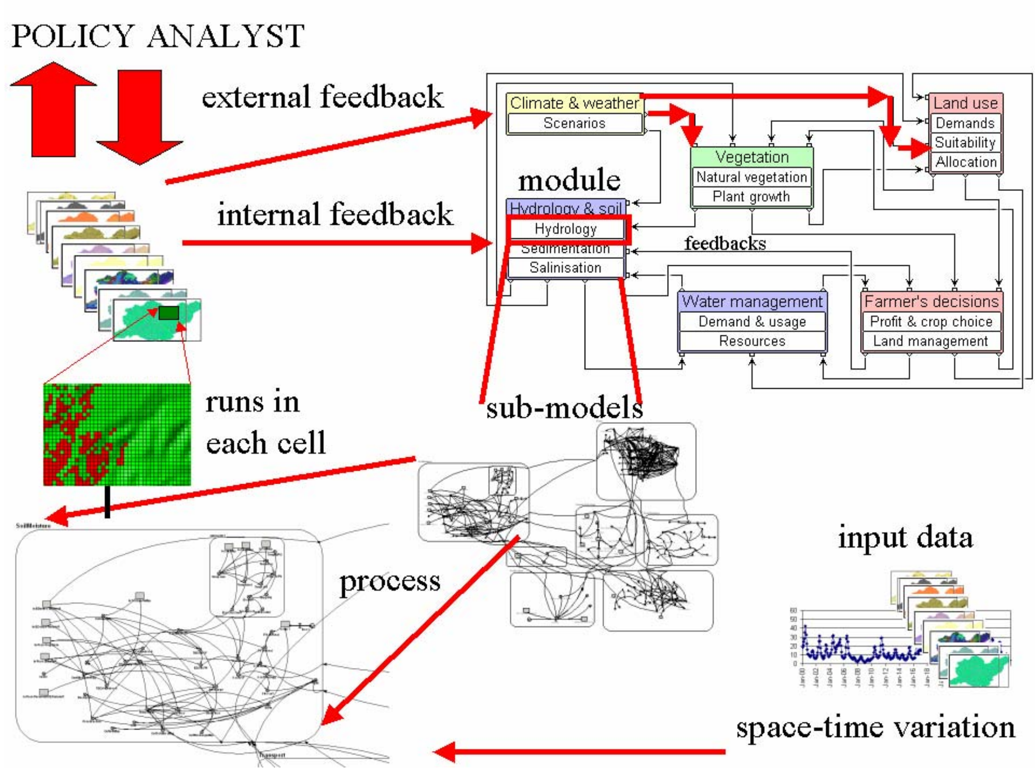

Figure 1. A Basic outline of the structure of the MedAction PSS.

The detail for the individual biophysical models is described fully in Mulligan [22] and for the socio-economic models in van Delden et al. [32]. The biophysical models were developed and tested first in the PCRASTER GIS[1] before being ported to the Geonamica system. A short summary of the models is given here.

2.9.1 Climate and weather

The climate and weather module simulates spatially distributed, sub-hourly rainfall using a timestep in inverse proportion to the rainfall intensity (down to a few seconds) which recognizes the importance of high intensity events for runoff generation. All other inputs are integrated twice-daily, at sunset and sunrise, to give a mean value during the daytime and during the night-time, which are considered appropriate timesteps and also different process domains. These inputs include solar, net and phosynthetically active radiation and air temperature. The data are based on GCM scenario inputs or historic data from a single base station and are spatialised and downscaled in temporal resolution using data from a nearby automatic weather station (AWS) for a single year. Data are

[1] www.pcraster.nl

spatialised using inputs from additional stations around the catchment along with appropriate climate covariates such as altitude and distance to coast. Further details can be found in Mulligan [22].

2.9.2 Hydrology

The hydrology module integrates at sunrise and sunset during periods of no rain but at the resolution determined by the rainfall intensity (the 'bucket-tip' timestep) during rainfall periods. It has sub-models for interception, evapo-transpiration, soil sealing, infiltration and runoff, throughflow, and drainage in the soil compartment, groundwater recharge, groundwater flow and return flow in the groundwater compartment and runoff accumulation, transmission loss and river evaporation in the fluvial compartment. Water is routed downriver using a kinematic wave or a simpler cascading scheme depending on user requirements. Further details can be found in Mulligan [22].

2.9.3 Soil erosion and deposition

Since one of the major desertification issues in the Guadalentin basin is soil erosion, the erosion and sedimentation module includes a wash erosion model which simulates erosion based on the soil properties, slope gradient, land cover and stream power [30] along with a sediment transport and deposition model after Kirkby [8]. This model both reduces soil thicknesses and breaks surface seals in areas where erosion occurs but also deposits soil on slopes, in streams, rivers, checkdams and reservoirs - which has implications for water storage and associated policy options for dredging and check dam construction and clearing. Further details can be found in Mulligan [22].

2.9.4 Salinisation

The salinisation module uses salt concentrations from the various water sources in the region (rainfall, river, groundwater, desalinised sea water and water from the Tajo-Segura transfer as well as salt concentrations in the soil, to calculate the accumulation of salinity in soils by evaporation and the washout of this salinity with infiltration water (eventually to the aquifer and back) and with runoff water (to the rivers and also, eventually to the aquifer). Further details can be found in Mulligan [22].

2.9.5 Water resource allocation, demands and use

The water resources module calculates the water balance of the aquifer, any reservoirs and stocks of desalinated seawater and allocates this water spatially based on its availability and any current policy restrictions. The water demands module calculates the requirements for water by each agricultural sector on the basis of the unit demands and the spatial extent of the sector. Actual water use is calculated taking into account the policy restrictions and water prices from the different sources. Further details can be found in van Delden et al. [32].

2.9.6 Land use and dynamic suitability

The land use model is based on a constrained cellular automata model and allocated land according to the demand by different sectors, neighbourhood attraction and repulsion by particular land uses and the land use specific suitability of the land, any policy zoning and accessibility requirements ([35]). Land use suitability is dynamic, based on the changing conditions of soil salinity, soil moisture and available soil thickness, all of which respond to desertification processes as well as air temperature, which responds to climate change and slope gradient, which is rather static in the absence of a terracing policy. In this way land use determines the trajectory of environmental processes which in turn set the state of the land suitability for any further land use change in a tightly coupled dynamic feedback. Further details can be found in van Delden et al. [32].

2.9.7 Plant growth and natural vegetation

The plant growth model uses a production efficiency approach [18],[28],[29] to calculate net biomass growth on the basis of environmental conditions. This growth is then partitioned to the relevant plant parts including yield using a resource dependent partitioning algorithm. On this basis plant structural properties (biomass, leaf area index, vegetation cover fraction) can be determined by and also, in part, determine the productivity of the plant in the next timesteps. At the biophysical level natural vegetation and crops are represented as a series of functional types: nonirrigated tree (e.g. Pine, Olive), irrigated tree (e.g. Citrus), dwarf shrub (e.g. Quercus), nonirrigated grass (e.g. Stipa) and irrigated grass (e.g. wheat). These classes are determined from the EEA CORINE land cover data that is used to initialize land use in the model. Further details can be found in Mulligan [22]. The natural vegetation and land management modules use a much more detailed specification of natural vegetation and crop types which are mapped onto these for biophysical purposes. The natural vegetation model represents the processes of succession at the community level on the basis of the vegetation maturity (height) and the various grazing and firing conditions. The module also permits vegetation dispersal through seed diffusion [14],[15].

2.9.8 Land management

The land management module is one of the main interfaces with policy analysts and incorporates procedures on crop planning, terracing and irrigation. The planning model allows a user to define the timing of sowing, ploughing and harvesting for each crop type. The irrigation model allows the user to define a minimum soil moisture which, if reached under a crop, will start crop irrigation and determines from which source the water will come and how much will be available for use, based on the approach also adopted in MODULUS ([26]). The terracing model allows users to construct terraces - which has profound implications for aspects of the hydrology. Further details can be found in van Delden et al.[32].

3.0 EXPLORING DESERTIFICATION WITH THE MEDACTION PSS

The MedAction PSS has been extensively tested and verified [22],[32]. Though some of the individual models have been validated in isolation [19],[20] outputs from the whole IAM have not been validated against historic data as yet – though, as discussed earlier, this is less relevant for explorative models. Here we present some case studies which explore the dynamics of desertification in response to climate change using the MedAction PSS. The experiment examines the implications of projected climate change on the integrated environmental system for the Guadalentin. The experiment is an equilibrium experiment in which the model is run twice, once as a baseline and then again with the perturbation. The model results (averaged over the catchment and the 30 year simulation period) are compared between the two sets of equilibrium output.

3.1 Exploring Climate Change in the Guadalentin

In this 30 yr simulation we run a baseline scenario based on current climate and another based on the downscaled IS92 climate change scenario of the ECHAM GCM (+0.48ºC over the period, rainfall change is +1.87 mm/month or 22.5 mm/year (around 7.5% in the lowlands) so an overall slight wetting). After running the two simulations, all of the 392 output variables were analysed and those changing by more than ±5% between the simulations on average across the catchment and period were set aside. 254 out of 392 (65%) of variables changed by more than 5% in response to this scenario. This process allows us to examine the model outcomes in a more holistic way than would be possible by analyzing specific processes or variables in isolation. By looking at the suite of variables that have changed the greatest we can generate a narrative of change from the model outcomes. The variables increasing by more than 5% included: dryland vineyard productivity and cover (150%), matorral productivity and cover (300%), dryland olives and fruit productivity and cover (100%), dryland almonds productivity and cover (60%), profit for dryland fruit (86%), greenhouse vegetables (39%), vineyards (34%) and olives (21%), evaporative interception losses (50%), aquifer demands for rural residential (14%), reservoir demands for industry and commerce (7.5%), urban residential (7.5%) and tourism (9.5%), soil moisture (6%), runoff (13%) and river discharge (13%) . Variables decreasing by more than 5% included: desalinated sea water demands in tourism, rural residential and expats (all 1-00%), water shortage in the tourism sector (-70 to -90%), sea water irrigation for pretty much all crops, aquifer salinity (-43%) and soil salinity (-9%), biomass and productivity for most irrigated crops (Olives, Almonds and Fruit, -34 to -42%), aquifer recharge (-30%), reservoir (-19%) and checkdam (-15%) sedimentation, drip irrigation (-13.6%) and aquifer demands for agriculture (-14%), soil erosion (12.2%) and productivity and cover for irrigated Almonds, Fruit, Olives, Cirus and Cereal (-14 to -10%). The net area under natural vegetation fell by 28 Ha. over the period to be replaced by agriculture.

The mild climate change thus precipitated a reduction in the productivity of irrigated crops (which tend to be based on the warmer, drier lowland plains) and an increase in the productivity of dryland crops (which tend to occur on the cooler, wetter hillslopes) with resulting profit reductions for irrigated crops and thus a shift to non-irrigated or reduced irrigation crops. There is also a general 'blooming' of agricultural and natural biomass with the extra water availability. Although soil moisture and runoff slightly increase, soil erosion and sedimentation decrease, probably as a result of the increases in vegetation cover. Moreover salinisation also decreases because of the extra water inputs. The reductions in erosion are consistent with the train of feedbacks that the climate change precipitates: increased water availability►reduced dependence on irrigation►growth of the vegetation cover ►reduced erosion. None of the other profits fell or rose sufficiently to be counted in this analysis so that although large changes in the landscape occurred, the economic consequences were not serious - in a positive or negative way - regionally and in the long term.

The three GCMs used (HADCM2, ECHAM and GFDL) all show marginal changes in rainfall, in fact often slight increases for the SE Spanish grid box so the previous analysis is the most likely outcome. We can, however, produce an aridification scenario to test the opposite scenario. This was done with a temperature increase of +1ºC over the 30 year period and a rainfall decrease of 4mm/month (47 mm/year or 12.4% of the rainfall). As before we compared this aridification scenario with the baseline scenario for all 392 model output variables and 249 variables

(64%) showed a change of more than ±5%. Variables that increased by more than 5% as a result of this scenario included : desalinated sea water demands for expatriots (+390%), desalinated sea water irrigation for irrigated greenhouse vegetables (+149%), olives (+181%) and almonds (+183%), vegetables (+61%), fruit (+72%), citrus (+76%) and vineyards (+77%), water shortage for the tourism sector (+134%) and thus desalinated sea water demands for tourism (+110%), rural residential (+128%) and agriculture (+47%). Reservoir demands for expats (+91%), rural residential (+14%) and agriculture (+15%) also increase as do extraction from desalination sea water (+30%) and reservoirs (+15%). The shortage of water for urban residential increases by 11% and for industry and commerce by 5%. Aquifer salinity increases by 10%. Variables that decreased by more than 5% as a consequence of the scenario included: biomass and profit for dryland, greenhouse and irrigated fruit and vegetables. Profit for greenhouse vegetables fell by -116%, by -70% for irrigated fruit, by -50% for irrigated vegetables. Biomass and profit also fell for dryland olives (-44%) and vineyards (-40%) as well as matorral and forest communities. Interception losses fell by -36% as a result of reduced rainfall and reduced vegetation cover. Profit for olives fell by -30%, for irrigated citrus by -20%, irrigated almonds (-16%), dryland vineyards (-16%), irrigated vineyards (-13%) and this leads to reduced aquifer irrigation for these crops (around -15%) and thus lower demands on the aquifer. Even with less rainfall, the fall in biomass production and profitability and consequent land use changes led to a reduction in the shortage of water for agriculture of -27%! The lower rainfall also causes lower runoff and river discharge (-16%) and thus -10% less check-dam and slope sedimentation, -19% less river sedimentation and transmission loss to the aquifer (-12%) and -7% less water in the reservoirs. There is also lower transpiration (-15.8%) in part also because of the reduced plant covers. This leads to -22.5% less recharge, -19% less aquifer replenishment and 8% less infiltration and soil moisture. The net area under agriculture fell by 1518 ha over the period to be replaced by natural vegetation.

So, the aridification scenario showed an overall decrease in biomass and productivity for both natural vegetation and crops (especially irrigated ones but also for dryland crops), leading to much-reduced profitability, greater dependence on desalinized sea water and reservoirs (also for the tourism sector). Reduced recharge led to higher levels of aquifer salinity. The resulting biomass decline, reduced transpiration losses reduced agricultural hectareage and increased natural vegetation hectareage leads to a reduction in the agricultural water deficit in the region! Less water leads to less runoff and erosion and reduced soil moisture and groundwater recharge but higher salinity. This is more like the typical desertification scenario and indicates the significant dependence of the agricultural and tourism economies in this region to the available rainfall. Since rainfall in the Mediterranean is extremely temporally variable at all timescales [20] it is a rather shaky foundation upon which to build enormous rainfall-dependent agricultural and touristic economies.

3.3 Leave it to the feedbacks

It is clear from the analyses above that models with **feedback loops** closed tend to behave less dramatically than partial and unclosed models. If we would run a soil erosion model in isolation on any of the scenarios above we would get a quite different picture of the outcomes of that scenario than we get from this analysis. Where feedback loops are closed, systems react to change and the reaction usually dampens the change. After all, most of the feedbacks that exist in natural systems are negative feedbacks. Where positive feedbacks exist they usually become negative or else they tend to destroy the system in which they exist (for example hurricanes, earthquakes) and are thus short lived. Positive feedbacks that are sustained in the longer term tend to be artificially sustained by human activity and artificial economic constructs without such natural feedbacks (for example subsidy-sustained agriculture on eroding soils or in drying climates). Because of the prevalence of natural negative feedbacks, in many cases the expected consequence of the input scenario is not realized…but other, counter-intuitive consequences are, for example *reduced* water deficits as a result of a drying climate.

In addition to many negative feedbacks, the MedAction model (as the real world) incorporates a high degree of **connectivity** and, as a result, the consequences of the scenarios can be seen in very different domains from those in which the scenario directly operates. So, both a wetter and a drier climate leads to less erosion leads to less erosion. The more of the processes we include the more complicated it becomes to interpret the outputs of models and the causal chains that have led to those outcomes but also the greater the opportunity for identifying unexpected or counter-productive outcomes of particular policies in the domain in which they are aimed or in other domains. The complication of the MedAction PSS is not necessary for identifying desertification but is for understanding precisely why it is occurring and building successful policies that will manage it without causing (too many) other (possibly worse) problems. I hope that I have been able to convince the reader that complex models also have their place in dealing with issues as multi-factoral, spatially variable and interactive as desertification. It is not the modeller's aim to make the models complex. They are complex because the reality of desertification is complex and, looking at one or a few processes in isolation will give only a partial view of the potential trajectories of desertification is a region. Worse still, this partial view will tend to be more dramatic than the reality because the important negative feedbacks are ignored.

3.4 ...and the policy makers?

As support systems these tools do not provide all the answers. Any model (even these) are partial and there may also be processes, feedbacks or actors that are not included but that may very well reverse the success of a particular policy to adapt to, prevent or mitigate desertification. These models can assist in highlighting 'dangerous' or negative interaction, can help to identify unforeseen consequences of apparently benign interactions, but are most useful as learning tools to help those working in science and policy to view the connectedness and behavioural dynamics and interactions of landscapes and their occupants. As we all know, change can be more to do with poli-tics than poli-cies. Whilst the kind of tool outlined here can help to compare the long-term and multi-domain impact of policies, policies will only be implemented if leaving things as they are is not a feasible alternative, if the stakeholders and thus their politicians will accept it and if the (short-term) benefits of tackling desertification are greater than the long term losses of not doing so – and are accounted for as such.

3.5 Developmental Needs for improved IAM

If these tools are to be used operationally at least the following developments are required some of which will require significant development in the analysis but also the dissemination of the remote sensing products discussed elsewhere in this volume. Remote sensing has a lot to offer these regional scale integrated models but to-date there has been relatively little interaction between the remote sensing community and the modeling community at this scale (much more has occurred at the global scale driven by various of the global change initiatives). The following basic data need to be improved, particularly in terms of the representation of spatial variability:

(a) Spatio-temporal climate fields (solar radiation, temperature, precipitation),

(b) Better (and higher spatial resolution) LUCC (and land management) products,

(c) More spatial data on soil properties (thickness, bulk density, texture, stoniness),

(d) Validation products (soil erosion, vegetation change timeseries, moisture dynamics timeseries).

In addition more and deeper exploration of the dynamics of complex spatial models is necessary so as to better understand their behaviour and that of the processes that they represent. In order for this to happen the systems themselves require significant theoretical and software developments in order to facilitate more efficient mining through the model results. These tools include:

(a) Error and uncertainty tracing tools capable of representing model results alongside a measure of cumulative error,

(b) Analytical tools focused on interpreting detailed model output and converting it to narratives which represent the behaviour of the system much more simply than by multitude graphs, maps and data tables.

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Combination of LDI and topographic attributes for land degradation mapping using a decision tree method

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ABSTRACT

This study investigates the utility of the combination of topographic attributes and the Land Degradation Index (LDI) approach to express land degradation in a small Mediterranean watershed located in the Moroccan western Rif. The LDI is based on the concept of the soil lines and was elaborated from ASTER sensor data and field spectra measurements. The implementation of the LDI index uses all the bands of the spectral domain in the visible, NIR and SWIR. Field measurements were carried out in the Saboun experimental basin located in the marl soil region of the Moroccan western Rif for which soil, hydrological and erosion databases are readily available. The topographic attributes were derived from a digital elevation model. Decision Tree (DT) was considered as mapping method to assess the classification performance in mapping land degradation.

LDI based analysis provided the allowed to analyse and characterize the state of soil degradation with a KHAT statistic of 0.79. Classification accuracy was increased significantly with the combination of LDI and topographic attributes with a KHAT of 0.84 or a global accuracy of 88%. The global results are in good agreement with ground truth. The approach should help decision makers in their soil conservation planning process.

Keywords: Land degradation, LDI, Decision tree, ASTER, Spectroradiometry, Topographic attributes, Morocco

1 INTRODUCTION

Land degradation, defined as the loss or reduction of the potential utility or productivity of the land [1], is a major environmental problem in the Mediterranean area. Changes introduced in the agricultural production systems, such as mechanization and the modification of agricultural practices, have led to soil degradation in the north of Morocco. To analyze the state of land degradation and to evaluate the risks of expansion or aggravation of the phenomenon, remote sensing is an appropriate tool. One of the approaches used for mapping land degradation is the spectral Land Degradation Index LDI [2]. Haboudane et al. [3] showed that combining geomorphometric indices with Spectral Mixture Analysis (SMA) improves the accuracy of land degradation mapping. Florinsky [4] presents a review of the combined analysis of Digital Elevation Model (DEM) and remotely sensed data in landscape investigations.

In this paper, we examine if the combination of topographic attributes and LDI under a Decision Tree (DT) approach can improve the results of LDI based mapping. In most cases, studies using this approach focus on land cover classification [5] or vegetation mapping [6; 7]. To date, only a few researches have investigated the use of DT to produce maps of land degradation. DT is a non-parametric classification method and accepts a wide variety of input data [8]. Also, DT has been shown to provide more accuracy compared to other conventional classifiers [9]. The assessment of land degradation is based on the Global Assessment of Soil Deterioration (GLASOD) classes as defined by Oldeman [10]. Hoosbeek et al. [11] recommended this qualitative method to classify soil degradation using remote sensing data.

The principal objective of this study is to evaluate how combining LDI index with topographic attributes will enhance the performance of land degradation mapping. Some guidelines and the effects of the training set size are also investigated.

Data collection is described in the next section, which also presents the remote sensing methods considered for land degradation mapping. Results and the validation of retrieved soil degradation maps are reported in section three. A general conclusion is outlined in section four.

2 MATERIAL AND METHODS

2.1 Study area

The study was carried out in the Saboun (720 ha) experimental catchment located in the western Rif mountains of Morocco (figure 1) for which soil, hydrological and erosion databases are available. The Saboun watershed is composed mainly of the Tangiers geological unit, characterized by predominantly altered argillaceous, gray and yellow shale facies dating from the Senonian. The soils of the basin have a predominance of swelling clays (montmorillonite and smectite). Oligo-Miocene deposits with shaley facies and quaternary deposits can also be found [12].

The soil map covering the study area was established by the *Direction provinciale de l'agriculture de Tétouan* [13] and shows the presence of 4 soil classes: vertisols (Typic Chromoxerert), paravertisols (Vertic Palexeroll), calcimagnesic (Typic Calcixeroll) and poorly developed (Vertic Xerothent).

Figure 1. Study area: the Saboun basin in Morocco.

The Saboun watershed is subject to intensive agricultural activity. The climate in the region is Mediterranean. Mean annual rainfall amounts to 743 mm with an inter-annual coefficient of variation of 23% [14]. Temperatures vary between 7°C (monthly minimum average) and 26°C (monthly maximum average).

2.2 Data acquisition and processing

2.2.1 Field spectroradiometric data and ASTER image

The measurement campaign was conducted between 18 and 26 October 2000. The choice of the measurement sites is based on the soil, geological and topographic maps and on our knowledge of the study area so as to include most of the surface conditions or the different levels of soil degradation.

The spectroradiometer used on the ground is a high spectral resolution ASD (Analytical Spectral Device). It operates in the visible, the near infrared and the shortwave infrared (350-2500 nm). A Spectralon board served as reference before and after each measurement. This permits the calculation of the target reflectance factor according to the method described by Jackson et al. [15]. The objective of this procedure is to minimize errors due to variations in atmospheric conditions and sun inclination. The bidirectional effects of the target reflectance were accounted for by carrying out measurements over very short and close time intervals and by keeping the viewing angle constant and in vertical position.

The characterization of the spectral properties of the different soil types was carried out based on the technique of principal components analysis (PCA) using the ground spectra [16]. A K-means analysis was subsequently applied. This permitted the discrimination of the different classes corresponding to the level of degradation. To obtain these results, the relative coordinates at the orthogonal level (PC_1 and PC_2) of the 22 spectra were adopted. The chain of treatment adopted for ASTER data was published in more detail by Chikhaoui et al. [2].

2.2.2 Topographic attributes

A topographic map (scale of 1/50,000) was used to generate the DEM, which served to create others attributes: Slope, Aspect and Topographic Index. These variables are believed to control the land degradation.

Slope (in percentage): represents a key parameter in the study and the modelling of the land degradation. Previous studies demonstrated that the susceptibility to soil erosion is related to slope. Also, the runoff factor responsible for erosion is more sensitive to gradient slope.

Aspect (in degrees from north): indicates the principal direction of the streams. As regards the Mediterranean environment, this variable is often used in soil studies [17]. Furthermore, it's strongly correlated with the vegetation cover [18], the north-exposing slopes having a denser cover. The effect of vegetation cover on reducing runoff and erosion has been demonstrated in many researches. The values of the aspect are included between 1 and 360 °, a value of 0 indicating a North orientation.

Topographic Index TI: represents the extent of flow accumulation [19] and is calculated by the equation below:

$$TI = \frac{A}{\tan(\beta)} \tag{1}$$

where A is the upslope drainage area and β, the local slope.

TI permits the determination of the potential soil moisture and the distribution of saturation zones. Also, TI is correlated with the thickness of the soil horizon, organic matter, pH and soil texture [20]. In investigating soil degradation risks, Haboudane et al. [3] integrated the TI in defining homogeneous hydrologic response units. The use of this parameter in modelling and predicting soil properties improves the results [21]. All parameters were generated with the GIS Arc View software.

2.3 Land degradation mapping methods

2.3.1 LDI (Land Degradation Index)

The LDI index is an alternate approach for expressing land degradation, which does not limit itself to the choice of specific bands [2]. The implementation of this method relies both on a slightly degraded bare soil line and a highly degraded bare soil line. The definition of these lines requires a collection of points from the pixels of the ASTER image itself or derived from ground spectral measurements. The use of ground data necessitates the calibration of the image data in order to produce a reflectance image corrected for atmospheric and instrumental effects. Any pixel representing bare soil is located between the highly degraded bare soil line and that of the slightly degraded soil line. The ratio between the distance of the highly degraded soil line over the distance between the highly and slightly degraded soil lines allows the attribution of any point to a given class [2].

2.3.2 Decision Tree

Decision tree classifiers are supervised methods in which the selection of training data is based on the map provided by the LDI approach. This step allows examining if the combination of topographic attributes and LDI under a DT approach can improve results in the LDI based mapping. The DT approach is based on a recursive partitioning of the data into mutually exclusive homogeneous subsets which best explain the variation in the dependent variable under observation [22]. The decision tree software used in this study is the product of the Estritel team (CARTEL-Université de Sherbrooke), and is available at http://estritel.geo.usherb.ca/~voirin/. Furthermore, a graphical user interface of the software operates in the Microsoft Windows environment.

2.4 Validation procedure

To reach our objective, the literature presented a number of methods for assessing classification results accuracy. The most widely used method is the error matrix, which serves to calculate the global accuracy and KHAT statistic (an estimate of the kappa coefficient) (i.e. $\hat{K}$). KHAT statistic is a method to measure of the accuracy of classification results [23]. To test the significance of the differences between the two approaches, the Z test is used. It was calculated by Cohen [24] as:

$$Z = \frac{\hat{K}_1 - \hat{K}_2}{\sqrt{\hat{\sigma}_1^2 + \hat{\sigma}_2^2}} \tag{2}$$

However, other studies use a qualitative approach based on thematic maps of factors responsible for spatial differences in land degradation such as slope gradient, lithology and soil proprieties [2, 3, 16]. The thematic maps were resampled to the same pixel size (15 m). A total of 120 points selected randomly over the surface conditions map were used with the thematic map to determine the accuracy assessment.

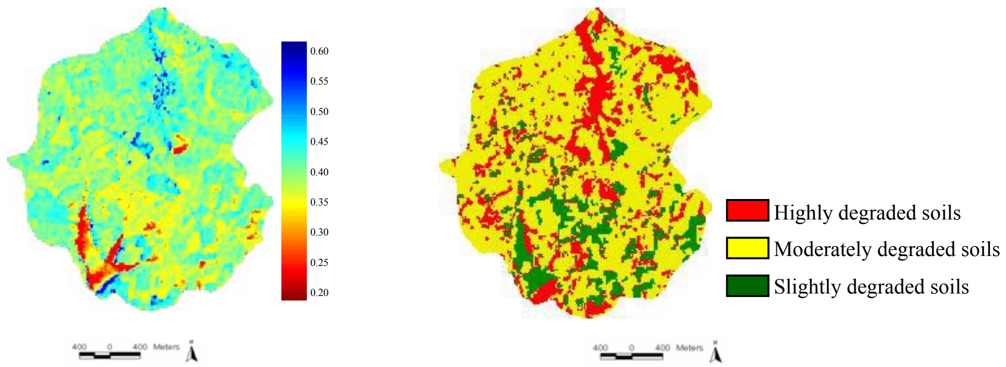
Finally, based on field observations, three land degradation classes were identified in our basin, based on the GLASOD method:

- **Slightly degraded soils:** part of the surface soil is removed and affected by overland and sheet runoff. This class is occupied by olive orchards and agricultural land is cultivated with cereals.
- **Moderately degraded soils:** large part of topsoil is removed and also affected by rills, overland and sheet runoff with sparse vegetation. This class is dominated by pasture land.
- **Highly degraded soils:** all topsoil and part of subsoil or substratum is removed. The surface is affected by rills, gullies and bank undermining forms of erosion. We noted a physical deterioration caused by domestic animals (compaction). This class is characterized by steep hillsides and mostly ploughing in the slope direction.

3 RESULTS AND DISCUSSION

3.1 Land degradation mapping using the LDI index

Figure 2.1 shows the results obtained with the LDI index. We observe that the highly degraded soils have a high LDI index value with 0.60 and that the slightly degraded soils have a quite smaller LDI value around 0.20. The application of a thresholding to the histogram of the LDI index provided well defined classes. Subsequently, we applied a median filter with a 3x3 window to obtain homogeneous classes and to reduce the presence of isolated pixels. Figure 2.2 shows the results of this post-processing.



2.1 **2.2**

Figure 2. Surface conditions map determined by the LDI.

Classification accuracy results for the LDI approach are shown in table 1. According to the error matrix, the highly degraded soils were mapped with high user's and producer's accuracy (> 0.92). Moreover, we can observe that the thematic map produced by the application of the LDI index shows a dominance of poorly developed soils of the order of 39%.

Table 1. Accuracy assessment result of the LDI approach

Class	Producer's accuracy	User's accuracy
Slightly degraded soils	0.83	0.79
Moderately degraded soils	0.83	0.85
Highly degraded soils	0.92	0.95
Overall accuracy	0.85	
KHAT statistic and variance	$\hat{K}$ = 0.79 and σ^2 = 0.0015	

3.2 Land degradation mapping using the DT approach

Classification accuracy results from the DT method are shown in table 2 with an overall accuracy of 0.88 and include the error matrix. According to this data, the highly degraded soils were mapped with high producer's accuracy (> 0.96). Figure 3 shows the results obtained with DT. This method improves the user's and producer's accuracy of slightly degraded soils class. The classification accuracy increased significantly with DT by 5%.

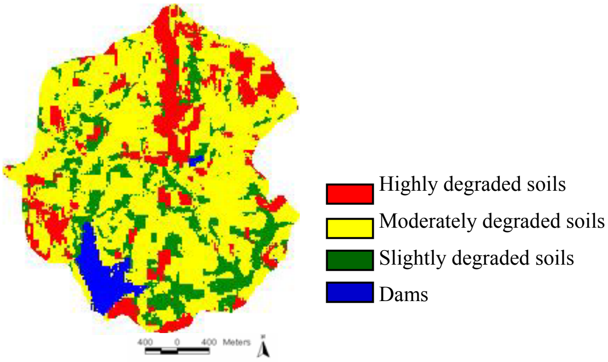


Figure 3. Surface conditions map determined by the LDI

Also, DT identified the water class (dam reservoir). This result can be explained by the fact that the sample of pixels included in the training of the DT method is adequate. To evaluate the effects of training set size on classification accuracy using a DT classifier, figure 4 shows the relationship between accuracy (Kappa) and training set size. Analysis of this figure indicates that accuracy increases with the size of the training set. This parameter must be considered in processes of classification with DT method.

Table 2. Accuracy assessment result of the DT approach

Class	Producer's accuracy	User's accuracy
Slightly degraded soils	0.84	0.87
Moderately degraded soils	0.85	0.84
Highly degraded soils	0.96	0.95
Overall accuracy	0.88	
KHAT statistic and variance	$\hat{K} = 0.84$ and $\sigma^2 = 0.0012$	

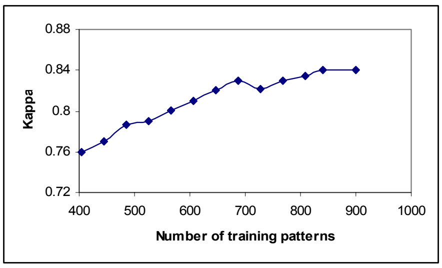


Figure 4. Variation of classification accuracy with increasing number of training patterns using DT

3.3. Validation and discussion of results

Comparison of the results obtained by the two approaches reveals that DT produced a much higher accuracy than LDI with KHAT statistic=0.84. According to Montserud and Leamans [25] this value indicates that the method shows very good to excellent classification performance. We observe that the results obtained with the two approaches are similar overall and clearly show the degraded soil class. Moreover, we note that the use of DT enabled us to get more details and more accuracy. In addition, the combination of topographic attributes and LDI under a DT approach can improve LDI based mapping results. The high accuracy obtained with DT shows that this method is a useful means of extracting the relationships between input data and land degradation. Tables 1 and 2 show the KHAT statistic used for the accuracy evaluation. The difference between the two approaches shows the interest and the contribution of LDI in the study of land degradation. We found that Z is equal to 1.02 (table 3). This value does not exceed Zt = 1.96 with a 95% confidence level. The difference between the LDI and DT methods is not significant. This result shows that the LDI is a useful tool for land degradation mapping using remotely sensed data only.

Table 3. Comparison of classification accuracies from the two methods with kappa statistics and its variance

Methods	Overall accuracy	KHAT statistics: K	Variance σ^2
LDI	0.85	0.79	0.0015
DT	0.88	0.84	0.0009

The use of the photo-interpretation results, DEM, slope gradient and vegetation cover served as an additional tool for the validation of our results. The analysis of the photo-interpretation result shows that our study area is entirely affected by water erosion. The following erosion forms were found: overland flows, gullies, rills, and bank undermining (table 4). Overlay of the map of surface conditions with the soil map, DEM and slope gradient showed that highly degraded soils are associated with the Typic Calcixeroll soil class. It is characterized by a soil erodibility factor k=0.44 measured in laboratory [14] indicating a high susceptibility to erosion [26]. An examination of table 4 shows good correlation between higher elevation, slope gradient and the highly degraded soil class. Also, the analysis of the table 3 shows that the highly degraded soils are characterized by serious vegetation degradation, with a vegetation cover under 5%.

Based on these results, the DT appears interesting for evaluating soil surface conditions and opens new perspectives for future studies in similar landscapes.

Table 4. Severity of the land degradation in our study area

Severity	Erosion Form	Erodibility factor k	Elevation m	Slope gradient %
Slight	Overland flow Sheet	< 0.3	20-40	Flat (<5)
Moderate	Overland flow - Rills	0.3<K<0.4	50-80	10-20
High	Rills-Gullies- Bank undermining	> 0.44	80-120	>25

4 CONCLUSION

This study shows that it was possible to map the state of soil degradation using the LDI with a KHAT statistic of 0.79. However, it is possible to increase the classification accuracy by 5% with the combination of LDI and topographic attributes (DEM, slope, aspect and topographic index) by using a DT classifier for mapping soil degradation. The performance of the DT approach is affected by the training data set size. The approach allowed to recognize three out of the four GLASOD land degradation classes and shows the interest of the use of the LDI with ASTER data to study or map land degradation in the Mediterranean basin (north of Morocco). The validation and evaluation of the results are based on ground data and photo-interpretation and show that globally, results represent the ground reality with sufficient accuracy to help decision makers in their soil conservation planning process.

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Use of a modified GIS-based Environmentally Sensitive Areas index (ESAI) to evaluate desertification risk in rangelands of northern Greece

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ABSTRACT

The aim of this study was to evaluate the desertification risk of rangelands in three village communities of Lagadas county, northern Greece, with the use of the Geographic Information Systems (GIS). Geoinformation data were used in order to combine several layers of information involved in desertification for the formation of a GIS model. The latter is a modified approach of the methodology used for mapping Environmentally Sensitive Areas (ESAs). It is found that the majority of the study area is in a critical stage in terms of desertification. The majority of rangelands are under intensive grazing activities. Overgrazing however, contributes to the desertification risk in combination with physical parameters such as soil, vegetation and climate.

Keywords: Desertification, Greece, Lagadas, Indicators, GIS.

1 INTRODUCTION

Desertification is a complex issue, closely related with land degradation and is caused by both human and physical forcing functions [1, 2, 3]. Among other definitions, the most recent and generally accepted is the one proposed by the Convention to Combat Desertification as "*land degradation in arid, semi-arid and dry sub-humid areas resulting from climatic variations and human activities*" [4].

In the Mediterranean region, desertification is not a new phenomenon. Historically, this region has had widespread problems because of intense land degradation [3]. In the northern part in particular, dryland regions are suffering desertification on a wide scale due to human activities in combination with harsh climatic conditions [4]. Several areas in Greece are threatened by desertification with most notable examples the islands of Lesvos [5] and Crete [6].

The Greek National Action Plan for Combating Desertification [7] lists as main factors of desertification the following: climate, physiography, geology, soil, hydrology and the human effects. Separately, the first five physical factors do not cause desertification, unless they are combined with irrational human practices. Furthermore, desertification is attributed to socio-economic factors, which can be expressed by several key indicators [8, 9, 10, 11].

Remote Sensing and GIS can be used either alone or in combination for the estimation of desertification risk [12, 13, 14] by employing special assessment models [14, 15, 16, 17]. The objective of this research was to estimate the environmental sensitivity of rangelands to desertification in three representative village communities of northern Greece as well as to identify which factors are critical for each of these communities.

2 MATERIAL AND METHODS

2.1 Study Area

The research was carried out in Lagadas county located about 30 km NE of the city of Thessaloniki, in Macedonia, northern Greece. Livestock husbandry is an important economic activity in the area and livestock have slightly increased in numbers over the last years [18].

Three village communities, namely Kolchiko, Lofiskos and Kryoneri (Fig. 1), were chosen as study area. Each village is located at different altitude, i.e. at the low (0-200 m.), middle (200-600 m.) and high (>600 m.) elevation zones respectively. All communities are located above the Koronia lake, and cover a total area of 14619.78 ha (3537.54 ha, 5237.37 ha, 5844.87 ha for each one respectively without urban areas). They have been studied by the European research projects GeoRange (Contract no. EVK2 -2000 - 22089) and VISTA (Contract no.EVK2 - 2001 -

000356) for the years 2001-2005 and the national research project Pythagoras (Contract no. 21893) for the years 2004- 2006.

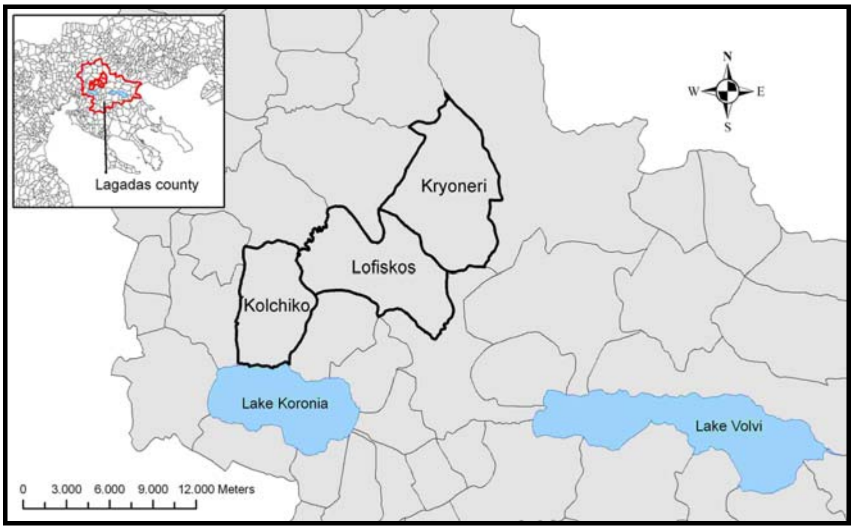

Figure 1. Map of the study area.

2.2 Methodology

Geoinformation data were used in order to combine several layers of information (physical, ecological and economic) involved in desertification for the formation of a GIS model. This model is a modified approach to the methodology used for mapping Environmentally Sensitive Areas (ESAs) in the MEDALUS project [14]. The modification can be found in the use of a reduced number of information layers as some layers were excluded from the model and some others were differently calculated e.g. stocking rate or added e.g. distance from sheds.

To determine the quality maps, data that were prior digitized and processed for the needs of the European project GeoRange were used. They included indices for climate quality, soil quality, vegetation quality and management quality. These indices were given weighted scores to emphasize or not the relative importance of information layers according to MEDALUS methodology (Table 1).

Soil quality: The parent material categories of the geology map [19] were regrouped into larger ones and the scores were given according to the MEDALUS methodology, as well as to the characteristics of the soil. The soil depth categories were derived from the soil map of Greece [20] and expressed in cm (Table 1). The slope was derived from the digitized contours of the topographic map of Greece (Hellenic Military Geographical Service, scale 1:50000).

Climate quality: The mean annual precipitation was estimated from the linear relation between precipitation and elevation [21]. This was done due to the fact that no meteorological stations existed in study area, and five nearby but located at different altitudes were used instead. The equation (1) between precipitation (Y) and altitude (X) produced was

$$Y=0.3272\,X+434.78 \qquad R^2=0.573, \quad p<0.05. \tag{1}$$

Slope aspect was the second layer used for climate quality and derived from the digitized contours of the topographic map of Greece (Hellenic Military Geographical Service, scale 1:50000).

Vegetation quality: Vegetation quality was estimated from the land cover/use categories of the orthophotomaps of the Greek Forest Service (scale 1:20000). In this classification, all kinds of agricultural land are grouped as one unit. For this reason, the CORINE land cover database [22] was used in addition, in order to determine the type of agricultural use. According to the GeoRange [18], at least 80% of the arable lands in Lagadas county is grown with cereals crops (annual). Also the majority of grasslands of the area are perennial. Finally, woody cover data were used for denoting plant cover.

Table 1. Quality indices with their relative information layers and scores.

Quality	Information layers	Classes		Scores
Soil	Parent material	Deposits, Ultra basic, Molasses of Lagadas		1
		Gneiss, Granite, Quartzite		1.7
	Soil depth (Description coded in soil map of Greece and in MEDALUS project).	Deep	Deep (>75 cm)	1
		Deep and shallow, Deep and bare	Moderate (75-30 cm)	2
		Shallow and deep, Shallow, Shallow and bare, Bare and deep	Shallow (15-30 cm)	3
		Bare and shallow, Bare	Very shallow (<15 cm)	4
	Slope (%)	<6		1
		6-18		1.2
		18-35		1.5
		>35		2
Climate	Rainfall (mm/year)	> 650		1
		280-650		2
		< 280		4
	Slope aspect	North, NW, NE, plain		1
		South, SW, SE		2
Vegetation	Fire risk	Barren land, Olives		1
		Abandoned agricultural land, Annual agricultural crops, Grasslands, Deciduous forests (oak, mixed)		1.3
		Shrublands		1.6
		Conifers		2
	Erosion protection	Conifers, Shrublands, Grasslands, Olives		1.3
		Deciduous forests (oak, mixed)		1.6
		Barren land, Annual agricultural crops, Abandoned agricultural land		2
	Drought resistance	Barren land		1
		Conifers, Deciduous forests, Olives		1.2
		Shrublands, Grasslands		1.7
		Annual agricultural crops, Abandoned agricultural land		2
	Woody plant cover (%)	> 40		1
		10 -40		1.8
		<10		2
Management	Stocking rate (AUM/ha)	<0.5		1
		0.5 – 2.5		1.5
		> 2.5		2
	Distance from sheds (m)	0-150		2
		150-400		1.8
		>400		1

Management quality: The management quality map was created from the point view of land use intensity, considering only stocking rate and the piospheric impact of animal sheds. Stocking rate was expressed in animal units per unit area, i.e. by dividing the number of free grazing animal units of each village community separately with the area of rangelands, derived from the forest orthophotomaps (scale 1:20000) that were corrected in 2002 [23]. For the calculation of stocking rate, sheep and goats raised in flocks, as well as the free grazing cows which are totally depending on rangelands were considered as well as. The grazing units were expressed in sheep equivalents (1 sheep or 1 goat or 5 sheep for each cow) [18].

Animal sheds have a piospheric impact on woody cover and plant height within a zone smaller than 800 m [24]. In Ref. [25] also was found that different plant heights of shrubs in the study area correlate with different biomass production and woody cover. Taking into account these two findings and the results from linear regression analysis of woody plant cover and plant height with the distance from the sheds [24], a strong reduction of the woody cover in the first 150 m was found and then a moderate one up to 400 m.

The computing algorithms, expressed with Eqs. (2) and (3) were calculated to produce the quality and desertification risk maps for each village community as described in Ref. 13 and 14.

$$\text{Quality\_x}_{ij} = (\text{layer\_1}_{ij} * \text{layer\_2}_{ij} * \text{layer\_3}_{ij} * \ldots\ldots * \text{layer\_n}_{ij})^{(1/n)}, \tag{2}$$

where: i,j = rows and columns of a single elementary pixel (30 x 30 m) of each layer; n = number of layers used and

$$\text{ES}_{ij} = (\text{Quality\_1}_{ij} * \text{Quality \_2}_{ij} * \text{Quality \_3}_{ij} * \text{Quality \_4}_{ij})^{(1/4)}, \tag{3}$$

where: i,j = rows and columns of a single elementary pixel (30 x 30 m) of each quality; Quality_n_{ij} = computed values.

Subsequently the available digital data from the GeoRange project were introduced into the GIS softwares, ArcGIS Desktop 9.0 and ArcInfo Workstation 9. In order to depict the physiographic characteristics of the area, the Digital Elevation Model (DEM) was created and the slope and exposure maps were derived. The 20 m spacing contour line used for the creation of the DEM was corrected with the "topogrid'' command of the ArcInfo Workstation 9 in order to make a hydrologically correct model. The topogrid command is an interpolation method specifically designed for the creation of hydrologically correct digital elevation models (DEMs) from comparatively small, but well selected elevation and stream coverages [26]. The animal shed impact layer was processed with the Spatial Analyst tool (ArcGIS 9). The remaining digital vector data sets (shapefiles and coverage) were also processed with the Spatial Analyst tool (ArcGIS 9) to convert them into a grid format. This grid format had a minimum pixel size of 30 m, which is considered to be a satisfactory size for the land surface research [27]. Finally, the computing algorithm of all information layers in grid format was combined with the "raster calculator'' command, resulting in the quality and desertification maps.

3 RESULTS AND DISCUSSION

Four quality maps were produced (Fig. 2). The majority of the study area (78.75%) has low vegetation quality with respect to desertification risk followed by moderate quality (21.25%). The absence of high quality class is probably due to the dominance of annual crops in terms of cover and to the presence of very low cover of woodlands. The large annual crop cover results in low erosional protection and very low drought resistance (annual crops in Mediterranean vegetation are classified in the less drought resistance category). Almost 52.55% of the area is covered by rangelands, from which 30.39% are shrublands and open forests with moderate cover (10-40%), while 11.28% is covered by perennial grasslands. The vegetation quality improves (from low to moderate) where dense deciduous forests (oaks) and shrublands (23.78% in total) occur.

Soil quality is predominantly low (67.09%) in the area. The greatest part has a shallow soil depth (54%), steep slopes (21.36%) and gneiss rocks up to 72%. The remaining part has high (17.28%) to moderate (15.63%) soil quality which is attributed to the gentle slopes (over 45%) and tertiary deposits (18%).

The climate quality is low (63.76%) due to the extended cover (over 70%) of the S, SE and SW slopes aspect of the area. The mean annual precipitation is over 556 mm in 83% of the area, while the remaining part, mostly located in higher elevation zone, has a mean annual precipitation of over 650 mm.

Finally, high management quality was found in the 68.86% of the area, and moderate in 25.65%. However, over 55.24% of rangelands have moderate management quality because of overgrazing and only 33.13% have high

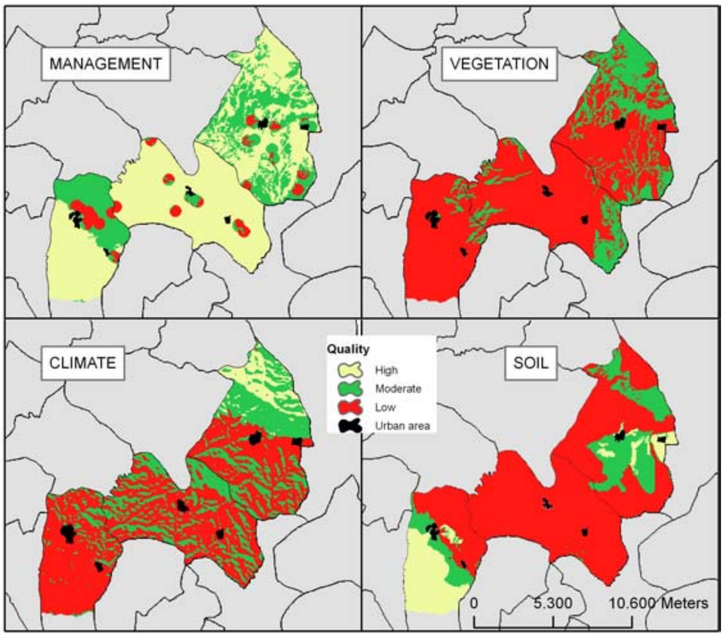

Figure 2. Quality maps of the study area.

quality due to moderate grazing. Low management quality is restricted to 11.63% of rangelands which is attributed to the piospheric impact of animal sheds.

The final map of ESAs (Fig. 3) was derived according to indices ranges, as described by MEDALUS project. The types of ESAs are defined as critical (C), fragile (F), potential (P) and non affected (N). Furthermore, the critical and fragile areas are defined on a three-point scale ranging from 3 (high sensitivity) to 1 (lower sensitivity). The map of the ESAs indicated as critical the major part of the three village communities (88.03%), followed by fragile (8.31%) and potential (1.51%). Only 2.15% of the area is not affected (N) by desertification because of deep soils and dense deciduous forests. The description of ESAs of the study area is:

Critical ESAs: Areas with gentle to very steep slopes with shallow to moderate soil depth on gneiss and granite parent material and tertiary deposits. The rainfall is generally <650 mm. Areas of this subtype are found in south-facing and in cases north facing slopes. The dominant vegetation is shrublands with low cover (10-40%), grasslands and cereals (<10%). These areas sustain husbandry, and stocking rate varies from high (Kolchiko and Kryoneri) to moderate (Lofiskos). In addition, areas around animal sheds are threatened from the piospheric impact. The use of woody cover indicator instead of plant cover probably increased the percentage of critical ESAs, especially in areas with woody cover lower than 10%, e.g. grasslands and cereals.

Fragile ESAs: Areas located mainly at Kryoneri village with steep to gentle slopes and in cases very gentle to flat, with shallow to deep soils. They formed mainly gneiss parent materials. Precipitation height is higher than 650 mm in the northern areas of Kryoneri, while it turns to moderate (less than 650 mm) in the remaining area. These areas are mainly found in south-facing slopes and in some cases in north-facing slopes. The dominant vegetation is deciduous forests with moderate cover (40-70%) to high (70-100%) and cereals (<10%). These areas are experienced mild grazing activities and are not threatened from the piospheric impact of animal sheds.

Potential ESAs: These areas are located only in Kryoneri with moderate soil depth and gneiss rocks with gentle to steep slopes. They are found on north-facing slopes or they are flat, with high rainfall >650 mm. The dominant vegetation is dense (>70%) deciduous oak forests with moderate fire risk, erosion protection and resistance to drought. Grazing activities are restricted and animal sheds are far away.

Non threatened areas: These areas are located only in Kryoneri village community with the same characteristics of potential ESAs but with deep soils.

Over 90% of Kolchiko (Fig. 4) is characterized by critical environmental sensitivity to desertification because of low vegetation quality (94.54%), attributed to the high percentage of annual crops and the moderate woody cover of shrublands in the north side of the village. Areas with fragile sensitivity are restricted (2.44%) to where dense shrublands occurs. Also rangelands have low woody cover and moderate management quality (up to 80%) because of overgrazing (stocking rate 3.39).

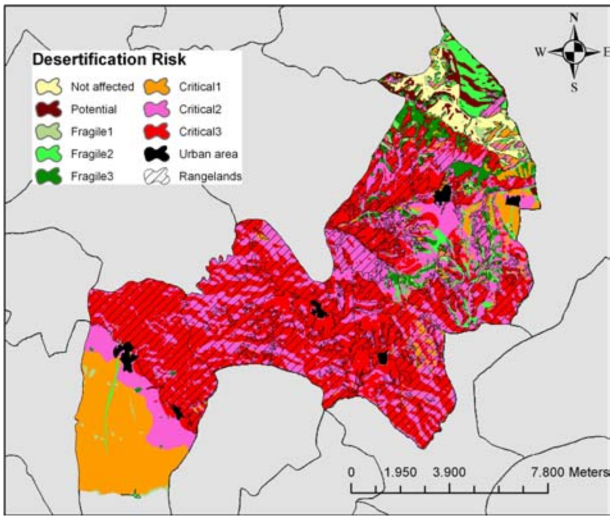

Figure 3. Environmental Sensitive Areas of the three village communities.

The village also has very low climate quality, because of the dominance of S, SW and SE slope aspects (86.07%). The mean annual precipitation is over 500 mm (<650mm). Tertiary deposits occupy more than 60% indicating a high soil quality (45.86%); they are mostly located above the Koronia lake. Also, there is deep and very gentle to flat soil. However, in the north side of the village where gneiss occurs (over 30%), the depth is shallow and the slope steep, the soil quality is low. The soil quality becomes moderate in the middle of the village community because the soil has moderate depth and gentle slope.

The whole Lofiskos indicates critical environmental sensitivity to desertification. There is very low soil quality (over 99%) because of very shallow soils. Gneiss cover more than 87% and granite only 11.33%. Slope is gentle with a percentage over 60%. In cases where there are ultra basic rocks and deposits, the soil quality becomes moderate. The vegetation quality is low (over 80%) and becomes moderate (over 16%) in the east part of the village and in cases in the west part. Low vegetation quality is the result of moderate woody cover of shrublands (over 40%) and the low woody cover of the grasslands. There is high management quality (over 90%) due to moderate stocking rate (1.36) in rangelands. The village has low (over 60%) to moderate climate quality (over 37%), because of the N, NW and NE aspects and the mean annual precipitation is over 550 mm (<650mm).

Kryoneri showed over 65% of its area with critical environmental sensitivity to desertification (south part) and the north part (over 20% of its area) with fragile environmental sensitivity. About 6% of the village is not affected and over 4% is potential. Low soil quality has the greatest impact on desertification (63.79%). Gneiss covers more than 80% and granite only 12.28%. In places where the soil depth is moderate to shallow the soil quality is low. Slope is very gentle to flat in the center of the village and rises from gentle to steep as we move to the margins. The soil quality becomes high in parts of the area where deep soil is found and the slope is gentle with tertiary deposits (5.97%). The vegetation quality is low (58.57%) in the south part of the village and moderate in the north. Low vegetation quality is the result of the low woody cover (over 10% of the shrublands and 10% of the forests). The village has moderate to low climate quality in the south because of S, SW and SE slopes and precipitation up to 600 mm. In the north, where N, NW and NE aspects occurs and the mean annual precipitation is up to 700 mm, the quality is high. In the lower part of the village and especially in the center of it the climate quality becomes low and in some places moderate, according to the exposure. Kryoneri has high management quality (55.10%). However, rangelands which represent the 40% of the area are overgrazed (stocking rate 4.87), resulting to moderate management quality.

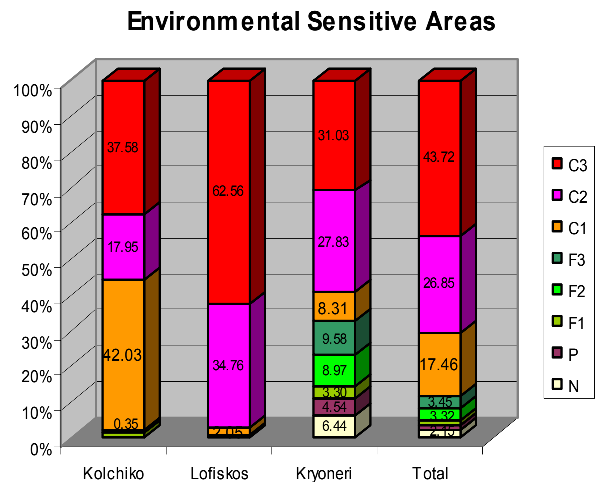

Figure 4. Percentages of ESAs of the three village communities, separately and as total.

It is obvious that the three village communities have areas with critical environmental sensitivity to desertification. Desertification risk in the area is the result of the interaction of all parameters studied. Despite the intensive grazing activities, management quality is low only around animal sheds, because of the piospheric impact of animal sheds. The most sensitive village communities are Lofiskos and Kolchiko because of low soil quality and low vegetation quality respectively. Kryoneri has high desertification risk because of low soil quality.

4 CONCLUSIONS

It is crucial to estimate the environmental sensitivity of areas at a local scale, so that we have a real estimation of desertification risk. The methodology used in the MEDALUS project provides the ability to add or remove information layers, according to available data, and making ESAI an important index in desertification risk assessment. Unambiguously, the proposed simplified GIS model can manage easily obtained data, but can be significantly improved if more information layers related to desertification are added.

Rangelands, especially in Kolchiko and Kryoneri village communities are in critical desertification stage mainly due to intensive grazing activities. The use of woody cover indicator instead of plant cover probably increased the percentage of desertification risk, especially in areas with woody cover lower than 10%. Finally overgrazing results in desertification in combination with other parameters related to desertification. However, piospheric impact of animal sheds had a critical effect in management quality, although restricted around the sheds.

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Indicating desertification in the Sahel of Burkina Faso based on Remote Sensing derived vegetation and soil indices

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ABSTRACT

Desertification, drought and poverty create a scenario of misery across the dry lands of Africa. There are many factors that trigger desertification, including the unpredictable effects of drought, fragile soils and geological erosion, livestock pressures, nutrient mining, growing populations, landlessness and an inequitable distribution of assets, poor infrastructure and market assess, neglect by policy makers and agricultural and environmental research systems. Given this complexity of causal factors, an integrated approach is essential to evaluate and indicate desertification. The north of Burkina Faso is part of the central Sahel of Westafrica and features all the above mentioned problems. The object of this study was to develop a **d**esertification **e**valuation **s**ystem (*DES*) for the district Oudalan in northern Burkina Faso. The evaluation system should include a set of indicators that manifest degradation of soils and vegetation cover. In order to provide a starting point, we have selected two indicators that are of large importance in all aspects of desertification: percentage of vegetation cover and soil loss. The other two indicators are hematite content in soils (*Hm*) and partial relationship between hematite and goethite in soils (Hm / (Hm + Gt). These two indicators present morphodynamic conditions of the landscape and provide valuable insights into the status of desertification. All indicators are tested over the study area by calibration of remote sensing derived information with the field sampling data obtained during field trips [1, 2]. Percentage vegetation cover was calculated from Normalized Difference Vegetation Index (*NDVI*). The annual soil loss was modeled for each pixel by applying Universal Soil Loss Equation (*USLE*). The Hm and Hm / (Hm + Gt) ratio for the whole study area were calculated by calibration of satellite derived indices developed by Madeira et al. [3] against values from test sites derived by laboratory work and Munsell color notations. Two linear regression models predicting *Hm* and Hm / (Hm + Gt) values from terrain conditions were developed and residuals from the corresponding mean regression lines were calculated for each pixel. Based on residual values, the desertification status was calculated for each pixel. The four indicators were then combined to a resulting desertification map. The developed *DES* allows determining spatial trends in land degradation and desertification in the study area.

Keywords: Desertification Evaluation System (*DES*), Sahel (Westafrica), Burkina Faso, Vegetation and Soil indices.

1 INTRODUCTION

The complexity of the desertification processes demands the use of a wide set of desertification indicators by an attempt to assess and monitor desertification. Indicators usually describe one or more aspects of desertification and provide data on threshold levels, status and evolution of relevant processes. Remote sensing-derived indicators provide an attractive way to map and monitor the desertification processes at all scales. Remote sensing data are indispensable especially for desertification assessments at broad scale, when the approaches based on field sampling and observation are not possible or cost-ineffective. Since the late 1970th many studies developed methods for deriving individual indicators of desertification from remotely sensed data. The majority of these studies used only one vegetative indicator or its few varieties as the main evaluation aspect [4]. The object of this study was to develop a desertification evaluation system (*DES*) for the district Oudalan in northern Burkina Faso. The evaluation system should include a set of indicators that manifest degradation of soils and vegetation cover in the study area. In order to provide a starting point, we have selected two indicators that are of large importance in all aspects of desertification: percentage of vegetation cover and soil loss. Changes in percentage of vegetation cover and soil erosion are important indicators of desertification as they both result from all degradation processes. Further indicators are hematite content in soils (*Hm*) and partial relationship between hematite and goethite in soils (Hm / (Hm + Gt). These indicators provide information about morphodynamic conditions and provide valuable insights into the status of desertification. All indicators are tested over the study area by calibration of remote sensing derived information with the field sampling data obtained during the field trips in 1994, 1995, 1996 and 2004. All four indicators -are then combined in order to produce a integrated map of desertification.

2 BACKGROUND

Burkina Faso and its Northern Province Oudalan are considered to be already overpopulated in the early 1970th. Since the 1950^{th}, the population of the province Oudalan had doubled and reached 70,000. The population density was counted 5.3 people / km². Before the severe drought (1970-1976) the peoples in Oudalan owned 225,000 heads of cattle and 450,000 heads of sheep. During the drought years the carrying capacity of the natural environment rapid decreased, what leads to overgrazing the pastures overall in the region and their degradation. The decrease of biological potential of the pasture land had forced a large part of nomadic peoples to turn to settled life and crop areas rapid increased. The increase of crop land areas in a region that lies beyond the border of non-irrigated crop production increases a risk of land degradation and desertification. The trend of rapid population growth in the study area will last during the next 20 years. According to assessments of the FAO, the rural population in Burkina Faso grew from 6.2 million in 1980 to about 10 million peoples in year 2000 [5]. The rapid population growth multifaceted with the below normal annual rainfall amounts during the last 30 years are reasons for sustainable degradation of nature resources in the study area. More preceding studies also in the 1990^{th} showed a high degree of land degradation and desertification in the province Oudalan [1, 2].

3 STUDY AREA

The study area is located in the northern part of Burkina Faso, in the province Oudalan, and is spatial limited to the watershed of the river Gorouol (figure 1). Although limited in extend, the area displays the characteristics of the Sahelian environments. As known, Sahel is a transitional zone where a biologically enriched Sahara and an impoverished Sudanian zone meet. The landscapes of the study area are dominated by longitudinal dune system fixed by vegetation and flat planes. The climate is characterised by a great inter-annual and intra-annual variation of precipitation. The dry season dues from October to May and a short rainy season last from June to September. During the 20th century, the mean annual value of precipitation was 438 mm. The soils are characterized by low humus content, weak development of horizons and the presence of crusts. The dune areas are covered by thick sandy soils that show low humus content – Luvic and Ferrallic Arenosols. These soils are the most important areas of millet cultivation. Luvisols and cambisols cover the most inter-dune area. They have 10-15 cm thick, loamy-sandy A-horizon, with superimposes a sandy-loamed or clayed B-horizon. Under 0.5 m lateritic crust is located which is cemented in patches. Another soil type that is to found on the plains is Vertisol. The inter-dune areas are generally used as rangelands. Fluvisols are associated with the areas of influence of the river Gorouol and other perennial water streams. They impose a 40-60 cm thick, yellow-brown very sandy A-horizon, and are covered by gallery forests. The natural vegetation outside from the river flood-plains is *Acacia*-steppe, but this vegetation type was degraded by overgrazing and now the most commonly vegetation in the study area is grass-steppe with discontinuous bush cover. On many places the bush-steppe is interrupted by areas of bare ground. These areas are heavily influenced by wind erosion.

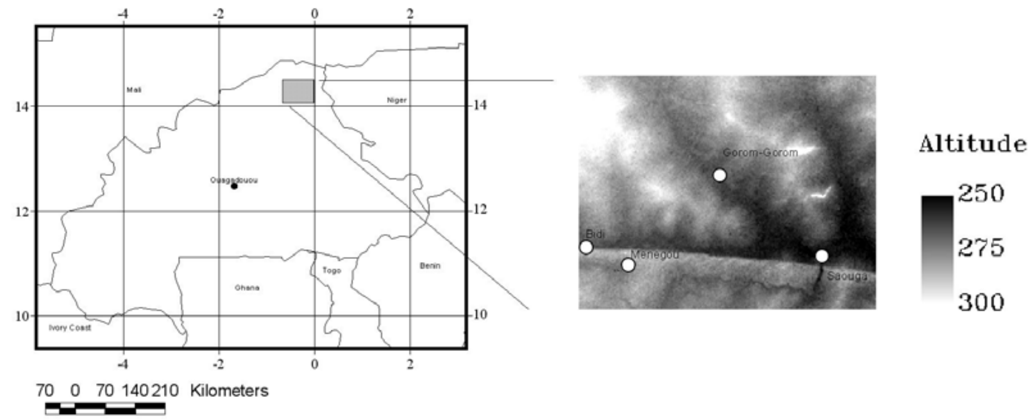

Figure 1. Study area and DTM.

4 IMPLEMENTATION AND EVALUATION OF INDICATORS

The following sections introduce the potential of each indicator, provide an outline how they were derived and assess their utility for land degradation evaluation. The most detailed descriptions are devoted to soil degradation indicators.

4.1 Estimation of vegetation cover

A number of techniques have been development in the literature to determine vegetation cover from NDVI [6]. In this study we calculated a linear regression relationship between vegetation cover (V_c) and NDVI by the use of vegetation cover data from the field sample sites described by Kappas & Wandelt [1] and the calibration of NDVI measurements. The regression equation calculated using the Landsat TM derived NDVI and the vegetation cover information from 24 test sites is the following:

$$V_c = 31.56 + 130.63 * NDVI \tag{1}$$

With fit statistics $R^2 = 0.87$ and $p = 2.13 * 10^{-17}$.

After the rain season in October and November we find good vegetation conditions and high NDVI values. The highest percentage of V_c show the areas of gallery forests at perennial water streams. There V_c reaches 60-80 %, but some places of gallery forests are degraded by cutting wood for fuel and overgrazing the grass cover. The lowest vegetation cover show the inter-dune areas covered by *Acacia*-steppe. On many places the vegetation cover is fool distracted (0 - 5 %) and the soil is bare. These places are to note as the areas with high degree of land degradation. The desertification status of each pixel was evaluated accordingly to the evaluation criteria by Babaev & Kharin [7]: $V_c > 50$ %, slight or no degradation; 20 % < V_c< 50 %, moderate degradation; 5 % < V_c < 20 %, high degradation; V_c < 5 %, severe degradation. Although a useful indicator of land degradation, the estimation of vegetation cover should be treated carefully. Because not only the desertification processes control the percentage of vegetation cover, but it is strongly influenced by other predictors such as inter-annual and intra-annual variations in rainfall, phenology and land use practices, unrelated to land degradation.

4.2 Estimation of soil erosion

The estimation of soil erosion was done by the Universal soil Loss Equation (*USLE*). The *USLE* represents how climate, soil, topography, and land use affect sheet and rill soil erosion caused by raindrop impact and surface runoff. It has been used to estimate soil erosion loss, to assess soil erosion risk, and to guide development and conservation plans in order to control erosion under different land-cover conditions, such as croplands, rangelands, and disturbed forest lands. During the last 20 years various modernized types of the *USLE* such as *RUSLE*, etc. were developed. In our research we used the standard form of USLE [8] that is expressed as:

$$\text{Annual Soil Loss } (t/ha) = 0.224 * R * K * LS * C \tag{2}$$

Where *R* is the rainfall-runoff erosivity factor; *K* is the soil erodibility factor; *L* is the slope length factor; *S* is the slop steepness factor; and *C* is the cover management factor.

The factor *R* was calculated by using the simplified equation described by Zachar [9]: the mean annual rainfall amounts in the study area, 438 mm, was multiplied with the coefficient 0.068. We did not distinguished spatial rainfall variations in the study area, because of the local scale of the research (25*25 km) they are not significant. The soil erodibility factor *K* is related to the integrated effects of rainfall, runoff, and infiltration on soil loss. It is controlled primarily by soil texture and soil organic matter content. The *K* factor varies from one soil type to another. The integrated soil map for the region developed by Kappas and Wandelt [1] and the table of values of *K* for different soil textural classes and percentages of soil organic matter content from Mitchell and Bubenzer [10] was used to determinate the soil erodibility factor. The *LS* factor accounts for the effect of topography on erosion in USLE. The slope length factor *L* represents the effect of slope length on erosion, and the slope steepness factor *S* reflects the influence of slope gradient on erosion. The standard form of the *LS* factor provided by the *USDA Agricultural Handbook* [8] requires to know the slope length in meters which was not possible to compute with the used software. Therefore, for calculation of the *LS* factor we used the modified form developed by Mitasova et al. [11]:

$$LS = (m+1) * \left(\frac{U}{22.13}\right)^m * \left(\frac{\sin\beta}{0.09}\right)^n \tag{3}$$

where U is the upslope area per unit width (measure of water flow) in meters (m²/m), β is the slope angle in degree, 22.13 is the length of the standard *USLE* plot in meters and 0.09 = 9% = 5.15° is the slope of the standard *USLE* plot. The values of exponents were m = 0.4 and n = 1.3. These values are usually used when is nothing known about the type of flow or when both types (sheet flow and rill flow) are combined.

The factor *C* reflects the effects of vegetation canopy. The methodology for the estimation of the percentage of vegetation cover V_c have been described in 3.2. The calculated values for the *C* factor in the study area ranged from 0.05 (V_c = 80%) to 0.95 (V_c < 10%). The majority of the study area has *C* values between 0.3 and 0.6. It appears that soil erosion is most serious in the inter-dune areas covered by degraded *Acacia*-savannah. Human-induced reduction of vegetation cover here made well conditions for sheet and rill linear erosion. The erosion rate is relatively low in the areas of gallery forest on flood-plains of perennial water streams. Four levels of soil loss related to general land degradation evaluation were identified and mapped: low (<0.5 t/ha), moderate (0.5-1.0 t/ha), intensive (1.0-5.0 t/ha) and severe (>0.5 t/ha).

4.3 Estimation of iron oxides content in soils

4.3.1 Hematite and goethite content in soils as indicators for land degradation

Iron oxides are building up from the iron-containing silicate rocks such as biotite, pyroxene, amphibole and olivine. After the build up the most part of iron oxides is not transferred to another place but crystallizes at the place of weathering. Therefore, the relation between whole iron content and content of iron oxides in soil characterizes a degree of weathering of the soil. The most frequent iron oxides in soils are hematite (*Hm*) and goethite (*Gt*). The relative proportions of these both minerals are related to factors determining the thermodynamic stability of soil. A hematite-goethite relationship in soil sequences of these regions is mostly determined by the following factors:

- moisture variability in the toposequence: higher moisture means little hematite content. The soils that lay higher in the toposequence have more reddish colour (higher hematite content) than the soils placed lower.
- organic matter content in soil plays an important role in hematite-goethite transformations. Lower temperature and more moisture in soil cause a slower decomposition rate of organic matter. In these conditions, increased amounts of organic compounds migrate into the soil and dissolve the hematite. The new formed oxide is goethite [12].
- content of other iron forms and other minerals.

The relative proportions of both iron oxides reflect the pedogenese conditions and are expressed by the ratio Hm / (Hm +Gt). An increase of Hm / (Hm + Gt) predicts a decrease of organic matter content in soils. Transformation from goethite into hematite is only possible with a change of the morphodynamic conditions. On this way, they influence natural dynamic and distribution of hematite and goethite. Therefore, one can use hematite and goethite content in soil and their relative proportions as an important biophysical/pedogenic indicator for land degradation evaluation. Each deviation from this rule can be explained as results of influence of other predictors. We supposed anthropogenic impact to be a main predictor: loss of organic matter, reduction in infiltration rates and drying up of upper soil horizons, change of soil water regime due to human-induced reduction in vegetation cover, overgrazing and stamping by stocks, all these factors lead to changing the ratio Hm / (Hm + Gt) on degrading places. The ratio Hm / (Hm + Gt) should increase with the increase of degradation degree.

4.3.2 Colorimetric and spectral indexes for estimation of Fe oxides content

Soils with a high content of goethite have yellowish brown colour, on the contrary, soils containing only, or almost exclusively, hematite show a red colour. The red colour of hematite is known to be very effective in masking the yellow colour of goethite. Colour measurements have already been used to quantify hematite and goethite contents in soils. Numerical indexes for calculations of hematite content and Hm / (Hm + Gt) ratio based on notations of Munsell colour system, such as hue (*H*), value (*V*) and chroma (*C*), were developed and are very suitable for field work. One commonly used equation quantifying the relationships between soil colour and hematite content was developed by Torrent et al. [13] and is expressed as:

$$RR = \frac{(10 - H) * C}{V} \tag{4}$$

where *RR* is the redness ratio, *H* is hue, *V* is value and *C* is chroma in Munsell colour system.

Madeira et al. [3] derived indexes based on the Landsat TM visible spectral bands to quantify the relationships between the iron oxides and their contents in soils. By development of the indexes a simulation model of spectral responses for the TM visible channels was compared to the colorimetric parameters derived from CIE chromatic coordinates. The spectral soil indexes are expressed as:

- the hematite index I_{Hm} for estimation of hematite content Hm ,

$$I_{Hm} = 10000 \frac{TM3^2}{TM1 * TM2^3} \tag{5}$$

- the ferric index I_{Fe} to estimate the Hm / (Hm + Gt),

$$I_{Fe} = \frac{TM3 - TM2}{TM3 + TM2} \tag{6}$$

where *TM1, TM2* and *TM3* are the channels of Landsat TM in visible spectrum.

We used in our study both spectral indexes to indicate the spatial variations in hematite content and Hm / (Hm + Gt) ratio in the study area. As was described above, these variables can significantly present a degree of land degradation and desertification in a landscape. The validation of the spectral hematite indexes was performed by a comparison with the hematite content calculated by use of colorimetric indexes (redness rating: *RR,* derived from Munsell colour notations).

4.3.3 Estimation of hematite content from soil colour

Hematite is the most pigmenting iron oxide in soils. Even a little presence of it in soil radical changes the soil colour. Torrent et al. [13, 14] stated that only 1.7 % hematite could give a soil a red colour; the Munsell hue was altered from 10YR to 5YR, when 1 % fine hematite was added; the addition of only 1 % hematite and 3 % goethite cased a shift in Munsell hue from 2Y to 5YR. Therefore we converted Munsell colour to redness rating (*RR*). Redness rating was also calculated for each of the 24 soil probes from the field sites. The *RR* is proved to be strongly correlated with the soils hematite content. The equation of linear regression developed by Torrent et al. [13] for Luvisols from Southern Europe was used to estimate the hematite content of the soils from the field sites in the study area. The equation is expressed as:

$$RR = 2.6 * Hm - 0.1 \tag{7}$$

where *RR* is the redness rating and *Hm* is hematite content in soils (weight, %).

For this study it was important to prove that a higher hematite presence in soils can be interpreted as an indicator of its degradation.

4.3.4 Computing Hm and Hm / (Hm + Gt) ratio from Landsat TM scene

An extrapolation of results of *Hm* calculations for the 24 sample test sites was made by validation of Landsat TM computed I_{Hm} to hematite contents derived from numerical index. First, the hematite index I_{Hm} was calculated for each pixel of the study area by applying equation (5). Second, a regression equation was calculated between the obtained I_{Hm} values and *Hm* values calculated by applying colorimetric index for the sample sites. Third, the hematite content was calculated for each pixel of the image with the help of the regression equation. We found that a quadratic regression describes the relationship better (R^2 =0.81). The obtained regression equation is expressed as:

$$Hm = 0.0281 * I_{hm}{}^2 - 1.086 * I_{hm} + 10.833 \tag{8}$$

The hematite content for each pixel was computed by this equation. The following characteristics of spatial distribution of hematite content can be discriminated: the most inter-dune areas are dominated by reddish-brown to dark brown coloured *Luvisol, Cambisol* and *Vertisol* soils with high hematite content (> 3.0 %); the dune areas in the southern part of the study region are covered by hematite-rich *Arenosol* soils with contents of 3-4.5 %, the colour of the soils is orange and reddish; the hematite-poor areas (< 1.5 %) are associated with the Gorouol river bed and other perennial watercourses. The calculation of Hm / (Hm + Gt) for the whole study area was possible by computing the TM based ferric index for each pixel of the image. The ferric index I_{Fe} was calculated from the TM2 and TM3 channels according equation (6). Madeira et al. [3] proved a good correlation between the ferric index I_{Fe} and Hm / (Hm + Gt) ratio. The linear regression equation for these both variables is given as:

$$I_{Fe} = 0.09432 + 0.00288 * Hm/(Hm + Gt) \tag{9}$$

The values of Hm / (Hm + Gt) vary from zero to more than 0.7 in dependence on soil type and relief. The spatial pattern of Hm / (Hm + Gt) is similar to that of the hematite content *Hm*, but there are notable differences: the values seem to be more diagnostic for the soil types and they show more homogeneity within each soil type.

4.3.5 Modelling Hm and Hm/(Hm+Gt) from toposequence analysis

To define the relationships between relief and hematite-goethite content in soils, correlation analysis was performed for 10 toposequences of various length and various frequencies for hematite and Hm / (Hm + Gt) ratio (R^2 varies from 0.54 to 0.88). In all causes, the linear regression models were fitted with a confidence level higher than 99 percent. After that, based on the equations calculated for the separate toposequences, we performed two linear regression equations, one for hematite content *Hm* and one for Hm / (Hm + Gt) ratio, that explain spatial variations of these two variables over the whole study area by variations of relief factor. The regression model equations are expressed as:

$$Hm = 0.0322 * A - 6.3386 \qquad (10)$$

$$Hm/(Hm + Gt) = 0.0134 * A - 3.2012 \qquad (11)$$

where *A* is altitude in meter.

Then, we used these equations to compute a regression predicted value for both variables of each pixel. A digital elevation model (*DEM)* for the whole study area was applied as the predicting variable *A*. The *DEM* has a pixel size of 30 arc seconds (~90 meter, SRTM). Next the *DTM* was resized by cubic convolution method to a pixel size 28.5 meter. As results, we derived two images representing a spatial distribution of hematite content and Hm / (Hm + Gt) ratio over the study area in that all pixel values exactly lay in the regression line.

4.3.6 Discriminating areas under ongoing desertification processes by applying the Hm and Hm/(Hm+Gt) models

In 4.3.1 we discussed why areas with same environmental conditions can show various hematite content or Hm / (Hm + Gt) ratio. We supposed that a human induced degradation process would be the main factor to produce a higher value of the both variables. Therefore, a high deviation of *Hm* or Hm / (Hm + Gt) from the expected value predicted by the environmental factors would indicate that this place is under ongoing process of degradation. In a region with high climatic variability and low percentage vegetation cover the relief would be the main predicting factor.

To discriminate the areas with high deviation from expected *Hm* and Hm / (Hm + Gt) values, we calculated residuals from the spatial regression lines for each pixel. It was done by subtracting the regression images from the corresponding images presenting *Hm* and Hm (Hm + Gt) distribution.

4.3.7 Modelling the classes of land degradation from regression residuals

The derived residuals varied from –3.5 to 3.5 for hematite content *Hm* and from –0.3 to >0.3 for the Hm / (Hm + Gt) ratio. The residuals images contained the information about the spatial distribution of the residuals over the study area. We considered that a higher value deviation from the main regression line would indicate areas under ongoing land degradation. But a scale (measure) for a quantitative evaluation of degradation status for each pixel is needed. This scale of degradation status was developed by the validation of the residuals from mean regression lines calculated for the test sites in comparison to their degradation degree measured in the field. We plotted the residuals against the degradation state of the test sites and computed a linear regression accordingly for the both variables, *Hm* and Hm / (Hm + Gt). The regression equations are the following:

$$DD = 8.1751 * Hm/(Hm + Gt)_{res} + 3.053 \qquad (12)$$

$$DD = 0.6954 * Hm_{res} + 2.895 \qquad (13)$$

where *DD* is the degradation degree; Hm_{res} and Hm / $(Hm + Gt)_{res}$ are residual values, in accordance with *Hm* and Hm / (Hm + Gt).

The regression equations (12) and (13) were then used to calculate two maps of degradation conditions for the study area derived from the corresponding residuals images: one based on hematite content deviation, and one based on Hm / (Hm + Gt) deviation for each pixel. The resulting images we reclassified into 4 classes of desertification:

- 1 class: “no degradation”, $0 < DD < 1.5$ for the Hm_{res} and $0 < DD < 0.06$ for the Hm / (Hm + Gt)res;

- 2 class: “slight degradation”, 1.5 < DD < 2.5 and 0.05 < DD < 0.15;
- 3 class: “moderate degradation”, 2.5 < DD < 4 and 0.15 < DD < 0.22;
- 4 class: “high degradation”, 4 < *DD* and 0.22 < *DD*.

4.4 Combining all indicators and producing land degradation map

The results of all indicators – vegetation cover, soil erosion risk, *Hm* and Hm / (Hm + Gt) content in soils – were combined to finally build up a map of desertification (figure 2). The classes 1 and 2 covered the most part of the study area (> 80 %). About 35 % of the whole area is none or only slight degraded (1 class), more than 45 % of the area is moderate degraded (2 class) and about 18 % are high degraded (3 and 4 class). The degradation processes mostly comprise the inter-dune areas used for pasture and the sand dunes used for crops production. The gallery forests in valleys of perennial water streams being slightly influenced by human impact. We compared the derived degradation map with the “Synthetic soil map for the district Oudalan” developed by Kappas and Wandelt [1] in order to evaluate the accuracy of our work. A confusion matrix was calculated for 100 sampling points. Overall accuracy derived from the confusion matrix was 76 %.

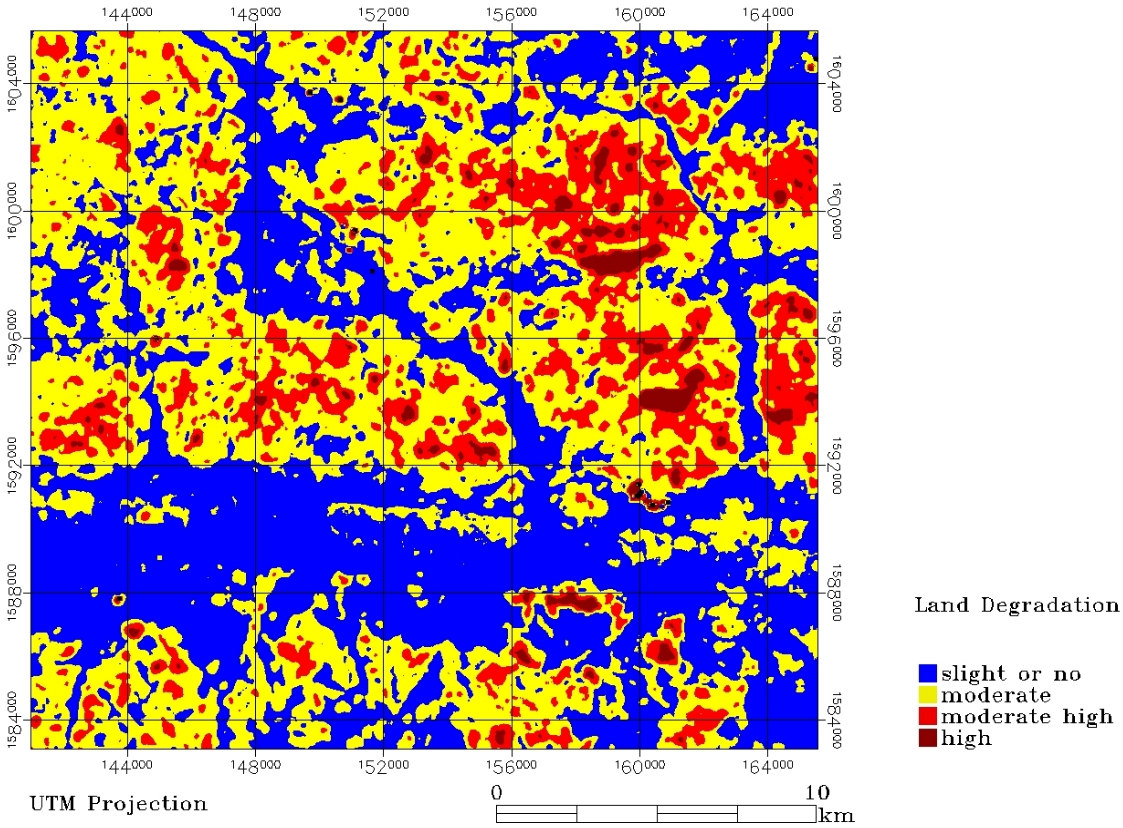

Figure 2. Integrated Map of Desertification based on DES.

5 CONCLUSION

Many indicators based on remotely sensing data were developed in order to improve the assessment of land degradation at local, regional and coarse scales. But the majority of this indicator evaluates vegetation conditions and excludes all other biophysical variables of ecosystem. It is unlikely that the most desertification or land degradation assessments were based on one single index or variable which should represent the complex process of desertification and land degradation. Environmental management strategies for these regions can be improved by use of more biophysical variables for indicating the desertification phenomenon. Combining GIS and remote sensing methods can help to solve this task. We have developed a method of implementing four important indicators of desertification and land degradation at a local scale. We used one vegetative and three biophysical soil indicators for our evaluation of land degradation. Two indicators presenting Fe oxides in soils used in this work (hematite content and partial relationship between hematite and goethite content) have been considered for an evaluation of land degradation. The *Hm* and the Hm / (Hm + Gt) ratio indicate stability of morphodynamic conditions and sensible react on its change caused by degradation process. Two other indicators estimate degradation of vegetation cover and risk of soil erosion. By combining all four indicators we made a map of land degradation for the region Gorom. The results generated provide a solid visual and quantitative basis for decision making in regard to implementing environmental protection, rehabilitation and resource allocation programs in the study area.

ACKNOWLEDGMENTS

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Satellite Image Processing and Geo-statistical Methods for Assessing Land Degradation around Watering Points in the Central Asian Deserts

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ABSTRACT

Desertification around watering points has been well observed by satellite images in many drylands around the world. It can be recognized as radial brightness belts fading as a function of the distance from the wells. The primary goal of the study was to characterize the spatial and temporal land degradation/rehabilitation in the Central Asia drylands, in term of vegetation and soil patterns, in different time periods, with respect to the socio-economic changes before and after the collapse of the Soviet Union. More specific objectives of the study were: (1) to develop a geo-statistical model, based on the kriging technique and using high-resolution satellite image processing in order to assess spatial and temporal land cover patterns in three key different time periods (mid-late 1970's, late 1980's, and 2000); (2) to conduct a change detection analysis based on the geo-statistical products in order to assess the direction and intensity of changes between the study periods; and (3) to link the previous findings to the socio-economic situations before and after the collapse of the Soviet Union that influenced the grazing gradients and hence the landuse/ land-cover state of the study site.

The Tassel-Cup's Brightness Index was found as the best spectral transformation for enhancing the contrast between the bright degraded areas close to the water wells and the darker surrounding areas far and in-between these wells. Empirical variograms were computed for each of the images and the exponential models were fitted. The Kriging geo-statistical technique utilized the variograms for creating brightness maps. The maps demonstrate the grazing gradient as levels of degrading belts around the wells. Change detection analysis, based on the Kriging maps, reveals some land rehabilitation between the 1975 and the 1987 images. However, mixed results, degradation and rehabilitation, were observed between the 1987 and the 2000 images.

Degradation of the area occurs due to recent exploration and exploitation of the gas and oil reserves in the region. Consequently, large areas went through intensive 'technological desertification' that means utilizing large amount of heavy-duty equipments, large-scale plants, and vehicles that damage the soil surface. The rehabilitation of the rangelands can be explained by the historical events of the last decades. Following independence of the former Soviet states in 1991 and the imposition of difficult economic conditions with transition reforms, several major socio-economic changes occurred that caused drastic declines in livestock populations, with the major drop in the number of sheep and goats, and hence vegetation recovery and land rehabilitation.

Keywords: Grazing Gradient, Watering Point, Brightness Index, Landsat, Geo-statistics, Kriging, Change Detection, Ust-Urt Plateau, Kazakhstan.

1 INTRODUCTION

According to the United Nations Environment Program (UNEP) the term *overgrazing* refers to a practice of allowing a much larger number of animals to graze at a location than it can actually support. As a result, overgrazing by different types of livestock is perhaps the most significant anthropogenic activity that degrades rangelands and causes desertification in terms of plant density, plant chemical content, community structure, and soil erosion [1]. In arid and semi-arid environments, land (soil and vegetation) degradation is particular related to area surrounding point-sources of water, either natural or artificial, such as wells or boreholes [2]. Pickup and Chewings [3] were the first who defined the term '*grazing gradient*' as "spatial patterns in soil or vegetation characteristics resulting from grazing activities and which are symptomatic of land degradation".

Domestic animals (sheep, goats, cattle, camels, yaks, and horses) prefer to graze in vicinity to a watering point. When food is depleted in this area they move away from the source of water but return regularly for drinking. Consequently, higher number of individuals is frequently concentrated around the watering points; density that decreases gradually with increasing distance from water [4, 5]. The livestock grazing distance is limited, depends on the water demand of different animals, season and weather conditions, and quality of the forage. Typical distance is 4-6 km that can increase to 10 or even 20 km under extreme conditions [6].

Since the radial pattern around watering points is well observed from space, most of the recent studies are based on the interpretation and modeling of remotely sensed data, which can be analyzed in a semi-automated and repeatable way over vast and remote areas. Various remote sensing based models have been developed to estimate the spatial distribution of the different variables around the watering points, these are the PD54 [4], Normalized Difference Vegetation Index (NDVI) calculated from Advance Very High resolution Radiometer (AVHRR) [7], or Probabilistic linear spectral mixture model, called AutoMCU, based on Monte Carlo analysis [8].

The above-described ground and spaceborne observations have demonstrated not only that the grazing gradients are characterized by concentric circles around the watering points, but also the spatial nature of the measured biotic, abiotic, and environmental variables that are distributed in a common fashion. Each variable (such as vegetation cover, grass and annuals production, bush encouragement, soil pH, organic content, phosphate, and nitrate, soil nutrient concentrations, particularly potassium and phosphorus, and track density) has low (or high) value near the center and changes exponentially as the distance increase. Moreover, most of these observations show that the rate of improvement (or decline) of each variable does not change after several kilometers (usually 5 as mentioned before) from the watering point.

The current paper presents another approach for assessing and mapping the grazing impacts around watering points. The primary goal of the study was to characterize the spatial and temporal land degradation/rehabilitation in the Central Asia drylands, in term of vegetation and soil patterns, in different time periods, with respect to the socio-economic changes before and after the collapse of the Soviet Union. More specific objectives of the study were: (1) to develop a geo-statistical model, based on the kriging technique and using high-resolution satellite image processing in order to assess spatial and temporal land cover patterns in three key different time periods (mid-late 1970's, late 1980's, and 2000); (2) to conduct a change detection analysis based on the geo-statistical products in order to assess the direction and intensity of changes between the study periods; and (3) to link the previous findings to the socio-economic situations before and after the collapse of the Soviet Union that influenced the grazing gradients and hence the land-use/ land-cover state of the study sites.

2 METHODOLOGY

2.1 Study site

The study site is located in the Ust-Urt desert plateau, ca. 160,600 km^2, between the Caspian and Aral seas in Central Asia and occupies the southern part of Kazakhstan, the northern part of the Karakalpak Republic, and Turkmenistan (Figure 1). It rises to between ca. 150 and 300 m AMSL. Its semi-nomadic population raises sheep, goats, and camels. It consists primarily of stony desert with grey-brown soils. The area has a semiarid continental climate with hot summers and cold windy winters. Vegetation consists on *Artemisia terrae-albae*, *Anabasis*, and *salsae*. The North Caspian basin is a petroleum-rich region with large oil and gas reserves but hardly explored yet.

2.2 Image processing

Three Landsat images of the study sites were used, acquired by different sensors (MSS, TM, and ETM+) and three different time periods 1975, 1987, and 2000. From the original images, subsets of the study fields were made, bounding an area of 1,184 km^{-2}

Reflectances were used for calculating several vegetation indices for each image subset. Performance analysis, in terms of standard deviation and stretch of the index values within the dynamic range, was applied to all indices. These analyses revealed that the Tasseled Cap-derived Brightness Index (BI) had produced the best contrast and consequently had been selected for further analysis. Besides the statistical significant results, and that this index was originally designated to examine soil properties, its advantage is in the ability to compare between the different sensors that have different spectral bands, as it reduces their different spectral bands to one normalized layer of BI values. Also it was found that the BI is uniquely independent from the spatial dimension and thus all image frequency histograms were spectrally matched.

The resolution of the images was reduced by a factor of 3 and 6, for the MSS and TM/ETM+ images, respectively. The resulting pixel size of 171 m is processing easily, as the sub-scene size was reduced from approximately 1.5 million pixels to only 40,000 pixels.

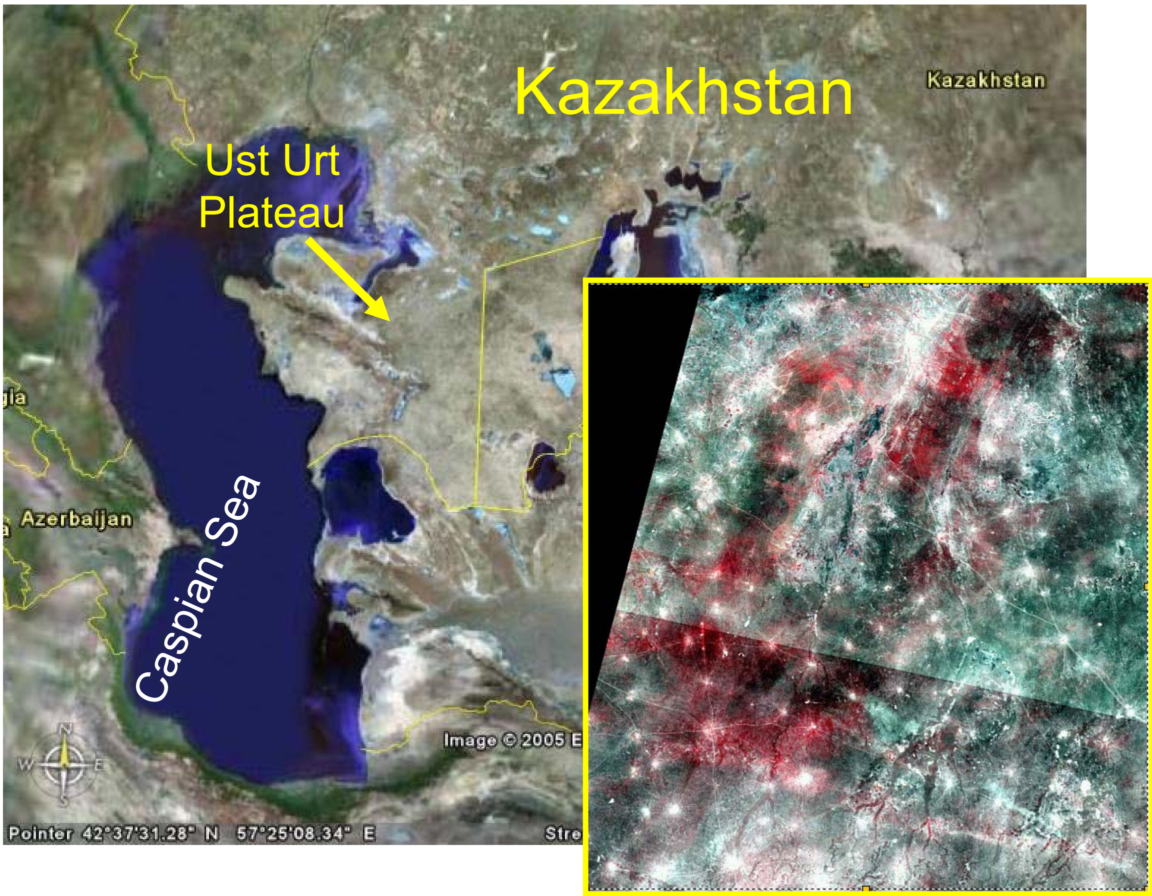

Figure 1. Location of the study site in Ust-Urt plateau, Kazakhstan. The inset shows the area in Landsat-TM image (RGB=4,3,2).

2.3 Geostatistical analysis

Geostatistical technique, namely *kriging*, was used to model and map the spatial variations of the soil brightness values in the study site. Spatially dependent variation may be treated statistically and described through a number of parameters derived from a *semi-variogram* that is the function relating the *semi-variance* to the directional distance between two samples. The semi-variance is defined as half the mean squared difference between two samples, a given direction and distance apart (Equation (1)). The direction and distance are defined by the vector *h* that is commonly referred as the *lag*.

$$\gamma(h) = \frac{1}{2N(h)} \sum_{i=1}^{N(h)} (z_{xi} - z_{xi+h})^2 \tag{1}$$

where γ(h) is the semi-variance at lag h, N(h) is the number of sample-pairs a distance h away, and Zi is the value of the regionalized variable at location i. In addition to the lag, the variogram is characterized by other three parameters – the nugget, range, and sill. The nugget is variability at zero distance, represents sampling and analytical errors. The range of influence designates the extent of distances, say a, beyond which autocorrelation between sampling sites is negligible. The sill represents the variability of spatially independent samples. An empirical semi-variogram can be calculated from the given set of observations and than fit to several common theoretical models. Once the theoretical semi-variogram has been chosen, several criteria can be applied to determine the correctness of the model and to adjust its parameters.

The rational after choosing the kriging technique was the similarity in spatial structure of most of the above-mentioned variables, gradually increase (or decrease) as a function of the increasing distance from the watering point until reaching the limit of no grazing effects, and the typical shape of the variogram. The kriging technique has recently become very common for analyzing spaceborne data.

2.4 Image differencing change detection analysis

Post-processing change detection method, namely BI Differencing [9], was implemented in order to compute the degree and direction of the changes in each site and between the imaging periods. The general function of the change can be considered as:

$$\text{change} = \begin{cases} 0 \text{ if } \left|BI_t - BI_{t+1}\right| \le T \\ 1 \text{ if } \left|BI_t - BI_{t+1}\right| > T \end{cases} \tag{2}$$

where t and t+1 represent the two time periods and T determines the threshold value. A common way to assess changes is based on determination of a threshold in terms of standard deviation (SD) levels bellow and above the mean of the difference between the BI values (Δ<BI>) of the images under study. In this manner, one can distinguish between changed and unchanged pixels as well as between negative and positive changes. In the current study, one SD from the Δ<BI> was defined as the threshold and steps of one SD, beyond this threshold, were determined the magnitude and direction of the change.

3 RESULTS AND DISCUSSION

Figure 2 presents the false color composite image subsets for the Ust-Urt site in 1975, 1987, and 2000, as acquired from the Landsat MSS, TM, and ETM+ sensors, respectively. Watering points can be recognized as light spots spread over mostly the MSS and TM images, but the ETM+ one. In the latter image many of the watering points disappeared, however, a wide bright area exists in the center of the image (Figure 2C). Figure 3 represents the respective BI products as calculated by Equation (1) with the appropriate BI coefficients. The same features can be seen in these products, but here the brightness levels were equally stretched for the different sensors and ranging between 0.65 and 0.96. High values correspond to bare soil while low values indicate vegetation.

The three BI images were used for the kriging analysis. The first step in this course of action was to establish the empirical semi-variogram based on ca. 40,000 pixels in each image, for the three periods. Subsequently, several theoretical models were examined and the exponential model was selected due to the best cross validation results. Thus, all empirical models were fitted with an equation of the form:

$$\gamma(\mathrm{h}) = \mathrm{C}_0 + \mathrm{C}_1\left(1 - \exp\left(-\left|\mathrm{h}\right|/\mathrm{a}\right)\right) \tag{3}$$

where a is the range, h is the lag, C0 is the nugget, and C0+C1 equals the sill. The fitted exponential models are illustrated in Figure 4. All variograms were processed with 16 lags of 1,000 m each. Visually, the 1975 and 1987 variograms look quite similar, having a typical variogram shape. Note that when the shape of the variogram is more round it can be referred to more symmetrical features in the image. Contrary, the 2000 variogram reaching about the same sill level only after 16,000 km and its shape is more linear than round.

In the next step the kriging interpolation maps were performed based on the exponential models. Since the grazing impact was assumed to be an isotropic feature and the direction has no influence on the spatial variation, the maps were derived from the omni-directional variograms. Figure 5 depicts the final products for the distribution of the BI values for the three periods. In the 1975 and 1987 maps, one can observe the belts around the watering points, indicating progressive land degradation radiating from the wells, i.e., the grazing gradient. The dark-red areas in the images are related to zones where the grazing impact is the predominant features that have strong effects on the spatial variation. These areas can be considered as the center of the grazing impact, denoted as 'sacrifice zone' by [10]. The surrounding light red and yellow belts are referred to a mixed zone where grazing impact and natural variability overlay each other or create a stable balance. This zone can be compared to an edge zone of the grazing impact and highlights rather the principle migration routes of livestock. The zone colored by blue tones is considered as an area where natural variability overbalances the grazing impact, denoted as 'grazing reserve' by [10]. The radial pattern, related to the grazing gradient, does not seen in the 2000 map. Instead, the dominant feature in the middle of the scene is colored in by red tones.

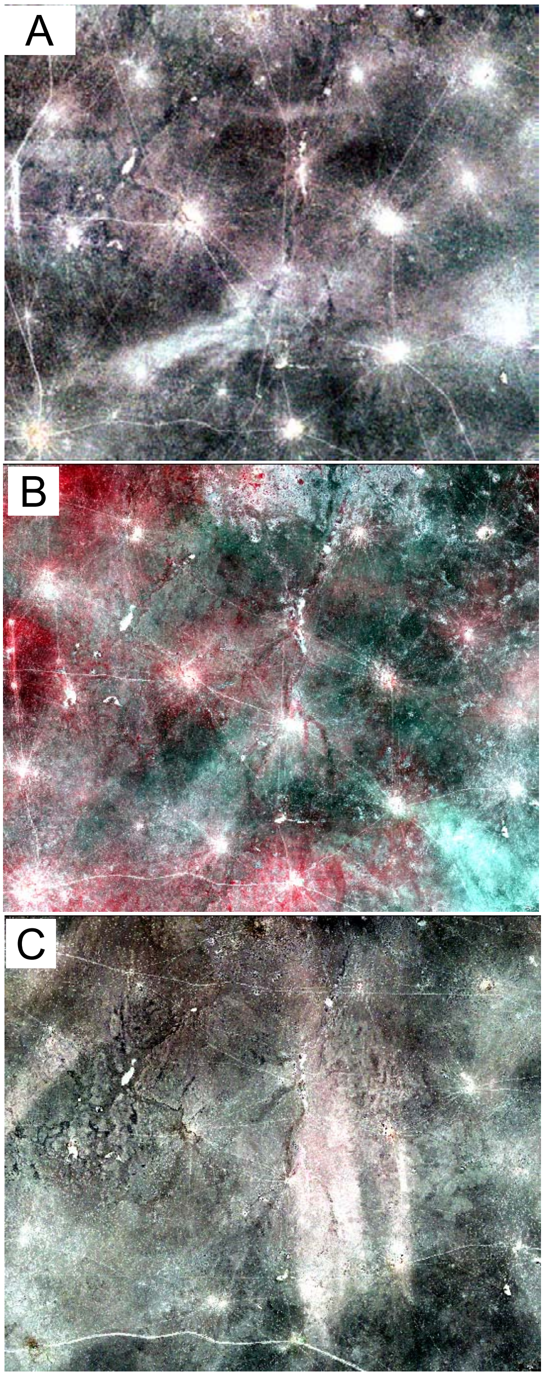

Figure 2. Subset images used for the kriging analysis. (A) Landsat-MSS, 1975; (B) Landsat-TM, 1987; (C) Landsat-ETM+, 2000.

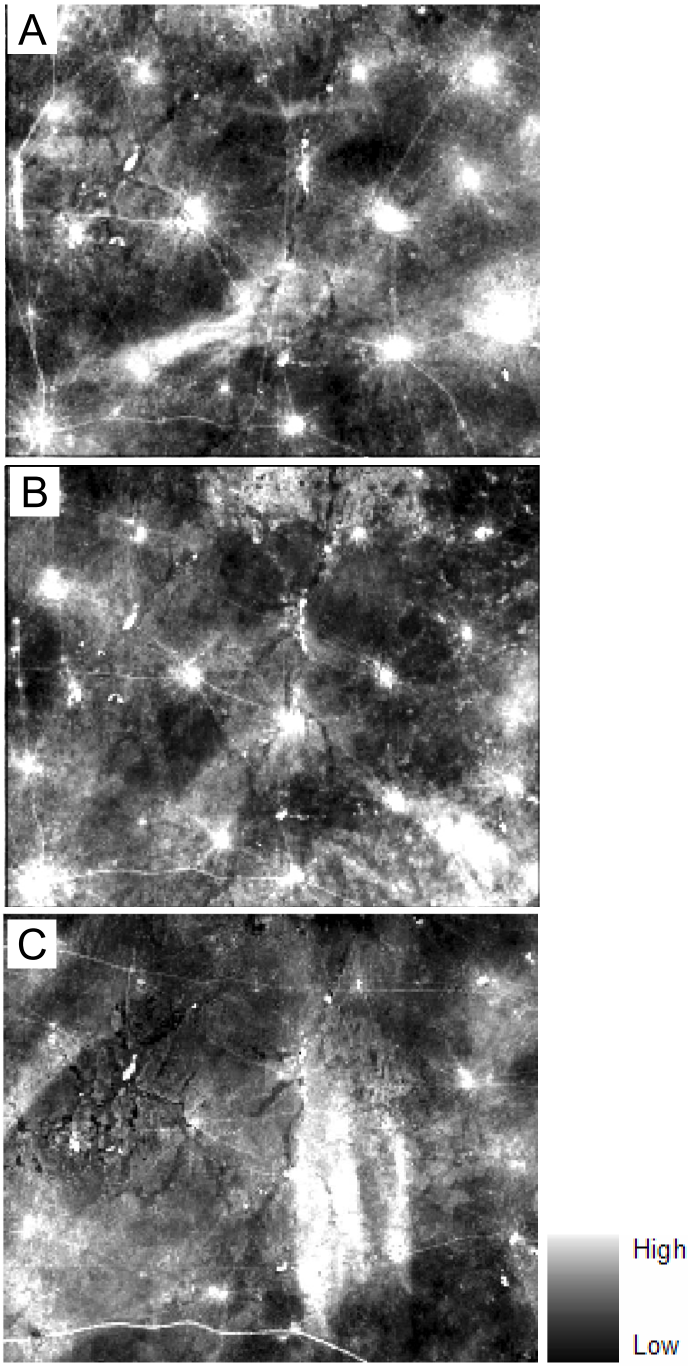

Figure 3. Brightness Index products for the images shown in Figure 2.

Image differencing change detection analysis, based on the BI values, was performed on the two pairs of kriging maps – 1987 vs. 1975, and 2000 vs. 1987. Results are illustrated in Figure 6. Figures 6A and 6C are the change maps while Figure 6B and 6D are the respective frequency histograms of the change categories. The SD lines are also presented. The difference map computed from 1987 and 1975 shows that despite of the area that was considered as 'no change' (64.0%), more area underwent rehabilitation process than degradation (22.1% vs. 13.8%).

A

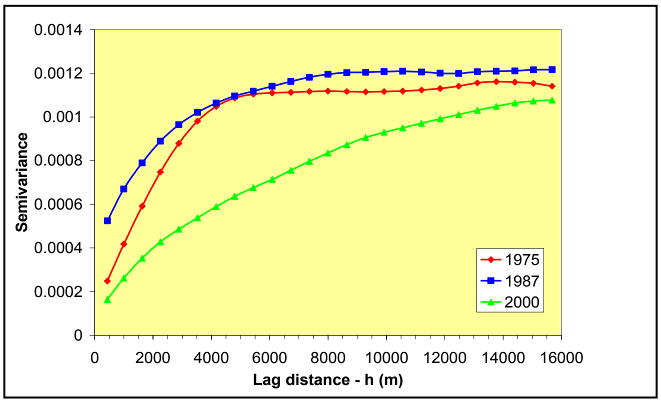

Figure 4. Theoretical variograms based on exponential models for the images shown in Figure 3.

desertification that means utilizing large amount of heavy-duty equipments, large-scale plants, and vehicles that damage the soil surface. On the other hand, following independence of the former Soviet states in 1991, and the imposition of difficult economic conditions with transition reforms, several major socio-economic changes occurred. Strong centralized government subsidy programs terminated, including the practice of guaranteed supplemental forage in cold winters and drought years. Farmers could no longer feed the livestock during the harsh winters, water wells were demolished, pumps were stolen or broken, and there were no longer transportation means to convey the animals to the markets in the central cities [12]. Consequently, drastic decline in livestock populations were observed after 1991 that resulted in much less grazing pressure and hence recovery of the natural vegetation and rehabilitation of the land.

As opposed to this favorable land cover change, degradation process characterizes considerable portion of the area (30.7%) during the second period (2000 vs. 1987), while rehabilitation occurred only in minor part (12.5%).

Two different processes govern the landuse and land cover changes in the Ust-Urt Plateau. On one side desertification processes has developed due to recent exploration and exploitation of the gas and oil reserves in the region [11]. Consequently, large areas went through intensive technological

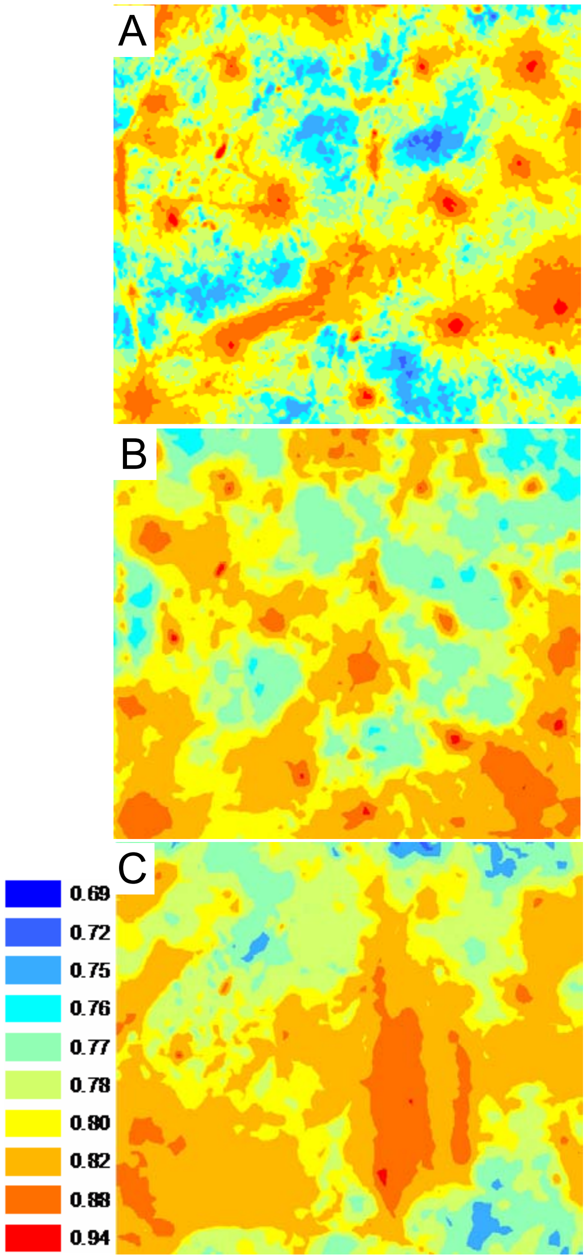

Figure 5. Kriging maps of Brightness Index values, established upon variograms presented in Figure 4, for the images shown in Figure 3.

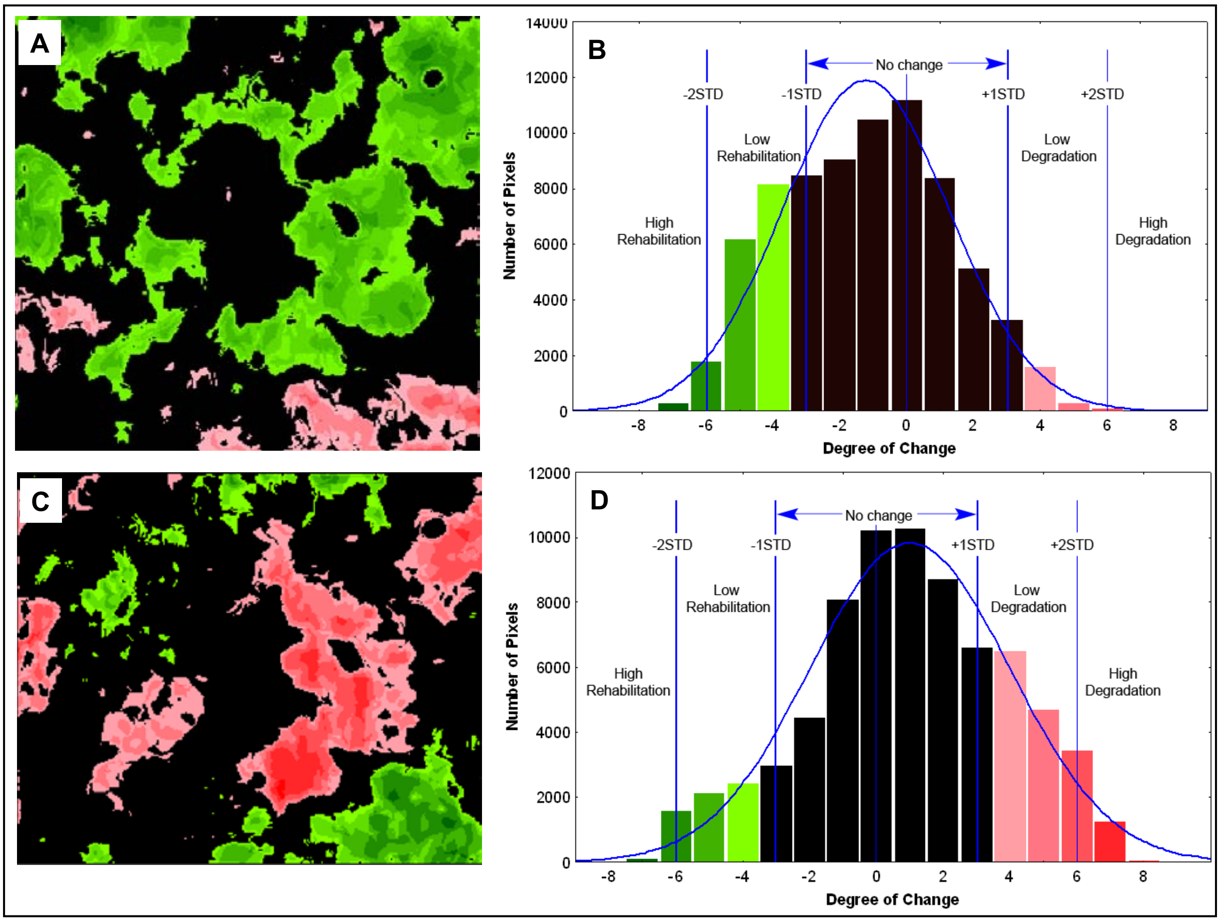

Figure 6. Change detection products, based on the Brightness Index Differencing algorithm. (A) Difference image between 1987 and 1975; (B) Frequency histogram for the degree of change between 1987 and 1975; (C) Difference image between 2000 and 1987; (D) Frequency histogram for the degree of change between 2000 and 1987.

4 CONCLUSIONS

The Tasseled Cap-derived Brightness Index (BI) was selected to describe the spatial surface patterns since it produced the best contrast. This index was originally designated to examine soil properties and its advantage is in the ability to compare between the different sensors having different spectral bands, as it reduces their different spectral bands to one normalized layer of BI values. It was also found that the BI is uniquely independent from the spatial dimension and thus all image frequency histograms were spectrally matched. Geostatistical analysis, based on the Kriging interpolation technique was found to be a suitable method for assessing the spatial patterns of landuse land cover especially around watering points in arid and semi-arid regions. The reason is the similarity between the shape of the variogram and the directional change of many biotic, abiotic, and environmental variables along the grazing gradient. Temporal changes were effectively conducted by the index differencing technique. In the study site, a certain part of the area was degraded due to recent exploration and exploitation of the gas and oil reserves, while another part underwent rehabilitation processes due to dramatic reduce of the grazing pressure after the collapse of the Soviet Union.

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Erosion dynamics and land priority planning: Coupling remote sensing and field observation for assessing land degradation

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ABSTRACT

The coastal area in Lebanon, making roughly 3000 km^2 or about one third of the country, is impacted by both natural and human stresses affecting its resources and the community's quality of life. The mostly mountainous terrain, with torrential sporadic rain and dense population, are inducing an increasing rate of land degradation. Indicators reflecting more semi-aridity are obvious, and enhanced soil erosion is occurring over 64% of the terrain. Thematic mapping of large areas, such as land units classification and predictive erosion risk mapping, is done through remote sensing as an entry to further stages for detailed mapping. Two coastal watersheds of 330 km^2 (Damour) and 140 km^2 (Zahrani) were chosen as pilot areas for this purpose. The criteria for selecting the areas reflect the three components of land degradation: the terrain, the ecosystem and the social fabric. Remote sensing also serves here in allowing fitting of those criteria to the chosen areas. Those criteria are location, size, slope, land cover/use, erosion, population and agro-practices. The predictive erosion risk map is checked in the field of the pilot areas. This will lead to identify and prioritize areas for intervention (hot spot areas). In the Damour watershed, 60.9% and 11% show low to moderate erosion risk and high erosion risk, respectively. In the Zahrani watershed, 62% and 14% show low to moderate erosion risk and high erosion risk, respectively. These are plotted on erosion dynamics maps for arriving at a decision support tool: prioritization mapping for intervention against land degradation. The prioritization criteria are rated from 1 (lowest score) to 3 (highest score). In Damour the obtained classes are: 19%, 65.8% and 4.4% having high priority, medium, and low priority, respectively. In Zahrani the obtained classes are: 12.1%, 66.9% and 15.6% having high, medium and low priority, respectively.

Keywords: coastal Mediterranean, erosion assessment, thematic mapping, watershed, land degradation

1 DIAGNOSTIC REMOTE SENSING

The application of sustainable development principles to fragile ecosystems such as arid and semiarid zones, very common in Mediterranean areas including Lebanon (Fig. 1), must cope with the issues of land degradation and desertification [1, 2]. This requires at least three main measures: preventive, mitigation, and restoration measures. In this connection, the benefits of thematic mapping at reconnaissance scale by remote sensing - such as erosion risk mapping [3] - should be considered mainly in terms of possibilities of selecting areas for priority interventions, where it is possible to carry out more detailed studies. Obviously, the scale used provides the opportunity to assess priority areas for such detailed studies. The torrential rains, which are responsible for severe water erosion processes, act over certain morphological characteristics, i.e. 1. mainly slope and 2. land use/land cover patterns - as described by remote sensing – thus the two represent the two key factors to assess land degradation (Fig. 2).

The slope contributes to the increase in runoff speed and in the erosive power of water. In such situations, the effect of this factor is highly dependent upon the type of land use/land cover and, for agricultural areas, upon agricultural practices for water control [4]. Relying on the various combinations of the two above factors affecting erosion risk, three main scenarios were highlighted in Lebanon for which some general selective management recommendations were provided. These reflect selective areas of significance in land use in the coastal zone [5], they are: Olive plantations, Natural Vegetation, both on steep slopes, and Agricultural areas on moderate slopes. Two pilot areas were chosen to take the predictive erosion, done through remote sensing, another step into the field.

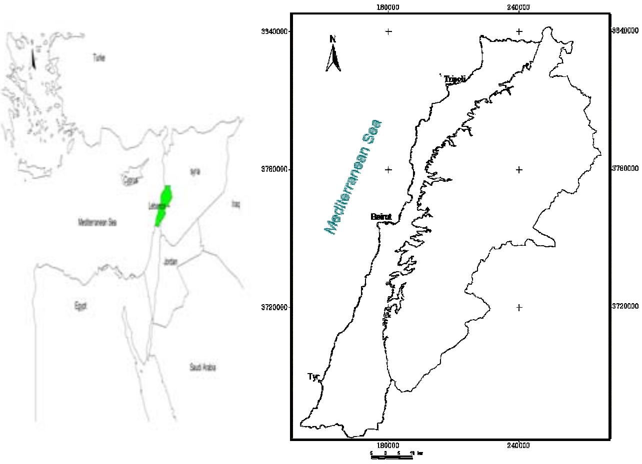

Figure 1. Location of study area in eastern Mediterranean Sea.

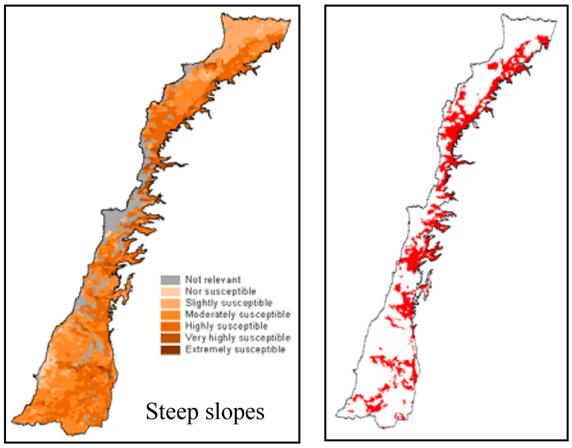

Figure 2. Diagnostic analysis through remote sensing showing steep slopes and potential erosion risk in the coastal area.

2 ANALYSIS OF PILOT AREAS

Damour and El-Zahrani watersheds are both river basins with permanent water courses (Fig. 3). Damour watershed (333 km^2) is situated in central Lebanon, while El-Zahrani watershed (140 km^2) is located in the southern part of the country. They extend from higher elevations in the east, i.e some 1500-2000 m, and go westward, at a relatively short distance (< 25 km), opening their outlets into the sea.

These pilot areas were chosen because their soils (Fig. 3) experienced water-erosion and other degradation, and reflect well the lithological, morphological, pedological, hydrological and climate diversity of Lebanon [6]. In addition, their selection depended on several criteria [7]: 1. Dramatic increase in the population rate, notably in the last two decades and certainly in the coastal stretch. This, accompanied by an increase in several human activities, led to harmful impacts on the environment. 2. Chaotic urban expansion, regardless of any planning control. This stimulates the geo-environmental decline in these basins, and 3. The absence of government solutions, i.e. lack of resources management plans.

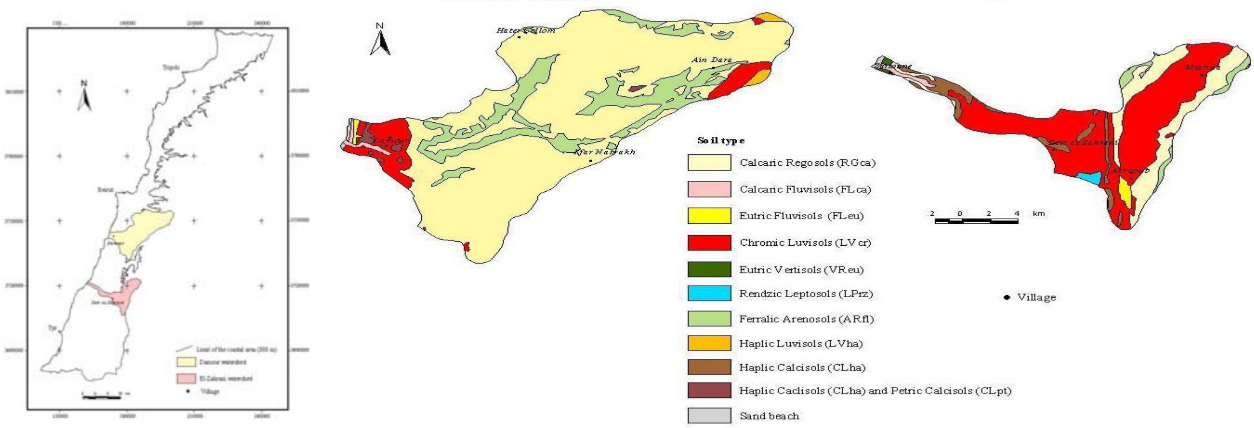

Figure 3. Location of pilot areas – Damour and Zahrani, and their respective soil maps.

Mapping for the detailed analysis [8, 9] was based mainly on the distinction between the following (Tables 1 and 2):

- Stable, non-erosion affected areas, i.e. areas with no evidence of any active erosion processes generating a state of morphodynamic equilibrium, and
- Unstable, affected areas, i.e. areas where one or several active erosion processes occur [10].

Within these categories managed and unmanaged areas were distinguished. The unmanaged areas are the stable forest lands, wastelands where no human intervention is observed. The managed areas belong to the lands with agricultural activities, protection or conservation measures.

Table 1. Stable and unstable areas in Damour.

Type	Erosion situation	Erosion risk/ expansion trend	Code	No. of polygons	Area	
					(Km^2)	(%)
Stable areas (75.3%)	Non-used wasteland	Low to moderate	001	36	2.76	1.9
	Unmanaged areas with potential for forestry use only	Low to moderate	011	98	67.32	45.6
		High	012	1	0.79	0.5
	Managed areas with forestry use only	Low to moderate	031	1	1.10	0.7
		High	032	4	6.97	4.7
	Managed areas with agricultural use	No	040	40	4.93	3.3
		Low to moderate	041	97	18.81	12.7
	Rehabilitated areas	High	062	14	8.60	5.8
Unstable areas (13.9%)	Sediment or excess water	Local	W11	2	0.01	0.0
		Widespread	W12	19	1.55	1.1
	Rill erosion	Widespread	D22	2	1.36	0.9
	Localised gully erosion	Local	C21	17	2.10	1.4
		Widespread	C22	1	0.09	0.1
	Dominant gully erosion	Widespread	C32	1	1.19	0.8
	Localised mass movement	Local	M21	9	10.24	6.9
		Widespread	M22	3	0.39	0.3
	Dominant mass movement	Local	M31	2	0.92	0.6
		Widespread	M32	2	0.96	0.6
	Localised associated processes	Local	P11	2	1.68	1.1
Not relevant				159	15.94	10.8
Total				**510**	**147.73**	**100.0**

Table 2. Stable and unstable areas in Zahrani.

Type	Erosion situation	Erosion risk/ expansion trend	Code	No. of polygons	Area	
					(Km^2)	(%)
Stable areas (80.9%)	Non-used wasteland	No	000	1	4.66	5.0
		Low to moderate	001	13	9.77	10.5
		High	002	3	12.09	13.0
	Unmanaged areas with potential for forestry use only	Low to moderate	011	17	7.47	8.0
		High	012	1	0.76	0.8
	Stable, unmanaged areas with agricultural potential	Low to moderate	021	1	0.34	0.4
	Managed areas with agricultural use	Low to moderate	041	5	0.97	1.0
	Rehabilitated areas	Low to moderate	061	40	39.16	42.1
		High risk	063	1	0.09	0.1
Unstable areas (13.1%)	Localised sediment or excess water	Local	W11	8	5.72	6.1
	Generalised sheet erosion	Widespread	L32	8	2.04	2.2
	Dominant rill erosion	Local	D21	4	1.38	1.5
		Widespread	D22	4	3.69	4.0
Not relevant				54	4.96	5.3
Total				**160**	**93.10**	**100.0**

Accordingly, since the erosional processes reflect an interactive domain from several factors, it is obvious to follow an integrated approach [11]. These factors are inherently dynamic, and therefore dynamic soil erosion maps must be used (Fig. 4).

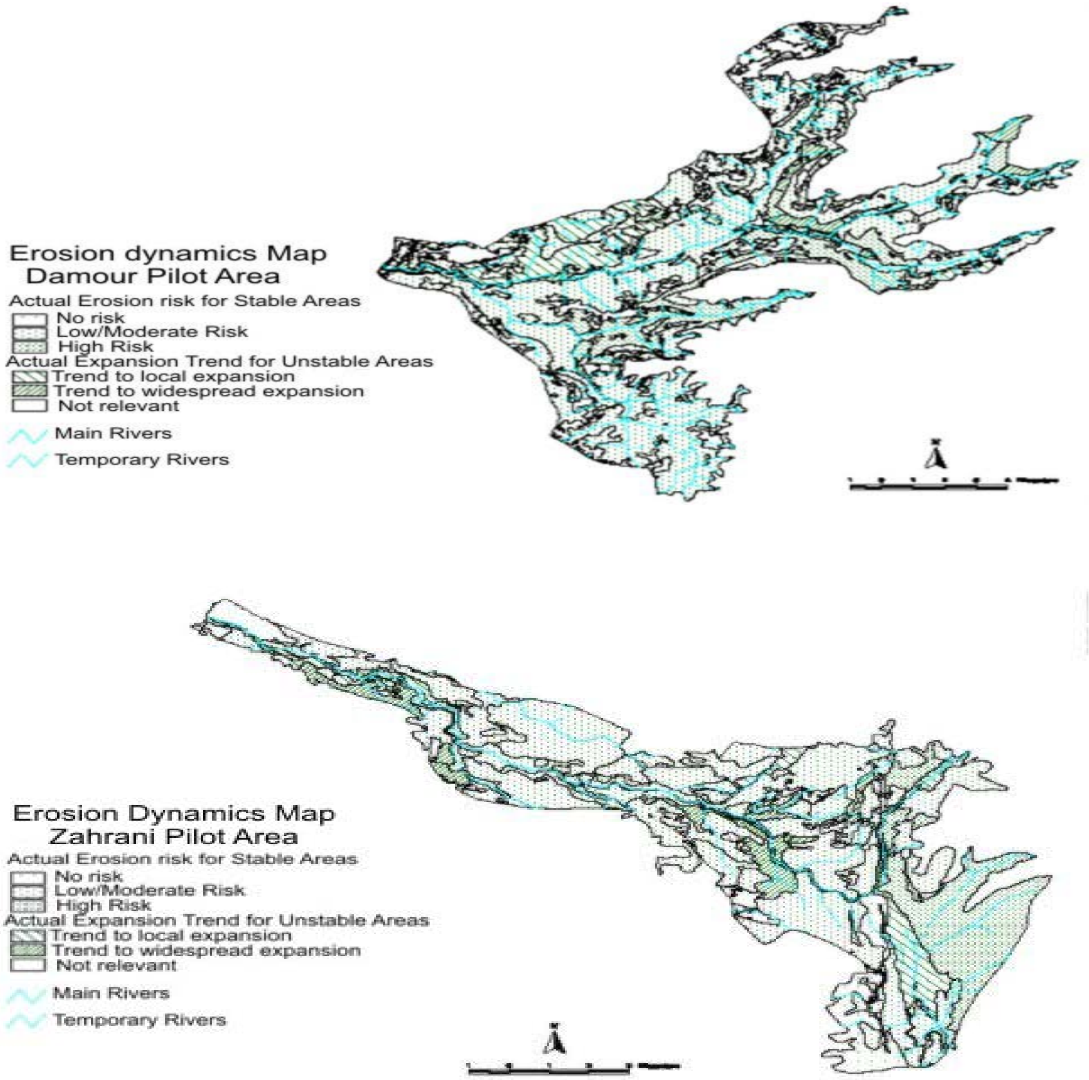

Figure 4. Erosion dynamics maps of the two pilot areas Damour and Zahrani.

3 REMEDIAL MEASURES

The approaches and emphasis to combat land degradation vary from place to place depending on the geography, focus on different practices, socio-economic conditions, different priorities for development, and existing regulations [12]. Based on the currently applied measures in Lebanon and in the two pilot areas recommendations for suitable measures are proposed.

The identification of the currently applied measures was made according to the general distinction between preventive, protective and curative measures, Table 3 reveals the specific themes that are significant when dealing with land degradation . For the environmental concern there are 4 relevant themes: soil, agriculture, forest and surface water. Similarly, for the concern on development the Table reveals the major theme on urban land use. This approach is followed in an attempt to simplify the application of those measures and make them more pragmatic, like contour ploughing and strip planting of hills in the coast of Lebanon mountain to prevent soil erosion.

Table 3. Remedial measures and major concerns for facing soil erosion

Major concern	Themes	Remedial measures			Expected results
		Preventive	Protective	Curative	
Environmental	Soil	Control use of soil cover	Active extension service Reduce slope stretch by introducing barriers	Land use based on soil capability and suitability.	Maintaining balanced ecosystem and durable agricultural production.
	Agriculture	Balanced development of rural areas.	Promote organic farming.	Agri-environmental land management units and agrozoning.	Increased fertilizer use efficiency
	Forest	Prevent forest fires	Optimization of forest maintenance and exploitation	Evaluate the environmental impact of agricultural and industrial activities on biota.	Dense and healthy forest cover
	Water	Prevent run-off from fields	Improved water harvesting practices	Sustainable water management practices	Water sources with acceptable quality
Socio-economic	Health	safe use of low quality water	Regular spatial and temporal monitoring of air, water & soil quality	Activate the role of schools in raising early awareness for hygiene and preventive medicine	Securing a society with healthy conditions
	Quality of living	Laws to prevent environmental degradation	Set & abide by U.N. human development standards	Proper and modern education system	Decrees for environmental laws are issued and implemented
Development	Urban	Controlling chaotic construction	Building codes and land zoning regulations	Upgrading existing infrastructures	Improve living standards in order to reach a decent level

Table 4. Summary of the management plan for the two pilot areas

	Problems & remedial measures			***Institutional/administrative arrangements***	***Monitoring indicators***
	Problem	***Priority***	***Remedial measures***		
Damour	▪ unstable areas	***	▪ stabilise by terraces or forestation & protect	▪ National level: ▪ capacity building ▪ establish databases ▪ enforce regulations ▪ apply ICZM notably land use zoning & protection ▪ Local level: ▪ public participation/NGOs ▪ environmental awareness ▪ agro-co-operatives ▪ enhance agro-sector & socio-economic incentives	▪ areas are stabilised ▪ forest regenerated ▪ water ways controlled ▪ degraded sites are recovered ▪ terraces maintained & cultivated ▪ agro-sector is recovered ▪ improved living
	▪ burnt forest	***	▪ rehabilitate		
	▪ torrential water passage	**	▪ construct protection measures		
	▪ quarried sites	**	▪ rehabilitate		
	▪ degraded terraces	**	▪ rehabilitate/cultivate		
	▪ wasteland	*	▪ cultivate or reforest		
Zahrani	▪ wasteland	***	▪ cultivate		
	▪ degraded terraces	***	▪ rehabilitate/cultivate		
	▪ quarried/disturbed sites	**	▪ rehabilitate		
	▪ excessive interference	**	▪ control & protect		
	▪ unstable areas	*	▪ stabilise & protect		

4 MANAGEMENT AND PRIORITY INTERVENTION

Indeed, the two pilot areas are different in terms of their grade of priority of problems. In Damour, the foremost problems are unstable areas and burnt forests, while in Zahrani they are wasteland and degraded terraced land. This is reflecting directly the socio-economic status. Furthermore, the project noted with great concern the remedial measures, whether being taken now (as observed in the field from pilot areas and elsewhere) or that are possible and commendable. Obviously, many of those measures need training and capacity building, so one has to understand the factors influencing institutional response to capacity building, as well as encourage public participation for an effective implementation of those remedial measures. This is what Table 4 tries to present as an outline of the needed aspects for a management plan for the two pilot areas, Damour and Zahrani. The plan reveals the problems, their remedial measures and priorities for intervention. The prioritization is crucial in view of the limited resources. Thus the priority maps (Fig. 5) help to implement the plan within the available resources.

In Damour watershed, about 19% (28.12 km2) of the assessed area fall into the high priority class, whereas 65.8% (97.19 km2) were classified as medium priority areas and 4.4% (6.48 km2) as low priority areas. The remaining 10.8% (15.94 km2) was urban areas. Amongst the high priority areas in Damour watershed, interestingly, the main part (16.9%, i.e. 24.94 km2) was identified as stable areas. Unstable areas (2.2%, i.e. 3.18 km2) were identified as high priority areas. In the medium priority class, the stable areas are also dominant (55.5% of the whole assessed area, i.e. 81.93 km2). Stable, non-used wastelands with a low to moderate instability risk, and stable, unmanaged areas with potential for forestry use only and a low to moderate instability risk were classified as low priority (3% of the whole assessed area, i.e. 4.41 km2). Also, a small part of unstable areas (1.4% of the whole assessed area, i.e. 2.07 km2), comprising areas with localised landslides with a trend to widespread expansion and dominant associated processes with a trend to local expansion, was considered as low priority.

In Zahrani watershed, about 12.1% (11.45 km2) of the assessed area fall into the high priority class, whereas 66.9% (62.21 km2) were classified as medium priority areas and 15.6% (14.49 km2) as low priority areas. The remaining 5.3% (4.96 km2) were urban areas. Amongst the high priority areas in Zahrani watershed, the main part (10.2%, i.e. 9.62 km2) was identified as unstable areas, also some stable areas (1.9%, i.e. 1.83 km2) were identified as high priority areas. In the medium priority class, the stable areas are dominant (63.4% of the whole assessed area, i.e. 59 km2). These areas mainly comprise rehabilitated areas with terraces and a low to moderate instability risk. A smaller part is unmanaged areas with a low to moderate instability risk, notably those with agricultural potential or with potential for forestry use only. Also some non-used wastelands with a high instability risk fall into the medium priority class. With regard to unstable areas, only a small proportion has been identified as medium priority (3.5% of the whole assessed area, i.e. 3.21 km^2). These areas either showed generalised sheet erosion with soil profile removal and a trend to widespread expansion, or dominant rill erosion with a trend to local expansion. Stable, non-used wastelands with varying degrees of instability risk were classified as low priority (15.6% of the whole assessed area, i.e. 14.49 km^2).

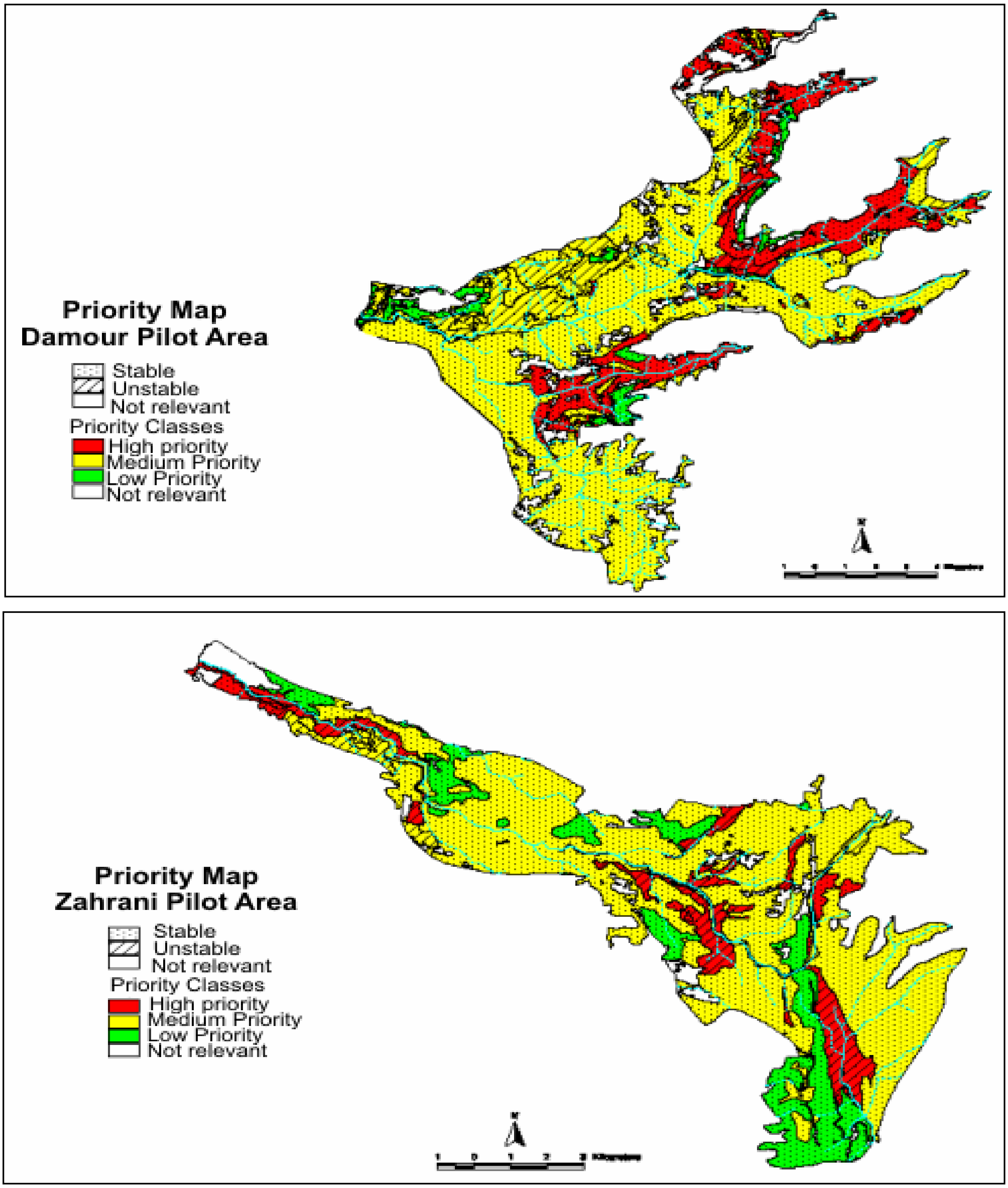

Figure 5. Priority maps for intervention against degradation in the two pilot areas

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Delineating patterns of plant diversity in the Sahel zone of Burkina Faso: Modeling of environmental envelopes with high resolution remote sensing data

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ABSTRACT

Relationships between patterns of plant species richness and landscape degradation were analyzed in the Sahel zone of Burkina Faso, West Africa. To generate predictive models of the distribution of plant species in the study region we used the Genetic Algorithm of Rule-set Production (GARP) modeling system. This system has excellent potential for describing the ecological envelopes and the spatial distribution of species and can be used for many different applications. In combination with environmental layers we used 12,000 geo-referenced collection localities to model species distributions in the Sahel zone of Burkina Faso. Previous studies have usually been conducted at regional scale using climatic and biomass parameters with a low spatial resolution of environmental variables. However informed decision making regarding conservation priorities mainly takes place on the local scale. In a new approach we used derivates of high resolution satellite images (LANDSAT ETM+) as environmental input parameters. Modeled distributions were evaluated with independent test data and selected by expert knowledge and statistical analysis. 138 plant species with at least 15 spatially unique occurrence points were modeled with a high accuracy. Consequently we calculated a map of phytodiversity by combining distribution maps of single species. The highest diversities have been predicted for a little disturbed area of dunes and tiger bush. Our approach has potential for widespread application to assist in conservation priorities and environmental management.

Keywords: Biodiversity conservation, ecological niche modeling, degradation, Sahel zone, West Africa

1 INTRODUCTION

Sustainable use and conservation of biodiversity requires sound scientific information on the geographic distribution of species. Unfortunately information on the geographic distribution of species is often not available in sufficient detail in countries with high levels of biodiversity. Therefore environmental management increasingly relies on easily assessable indicators and operational tools to achieve its primary goal of conservation and sustainable utilization of natural plant resources [1]. Predictive modeling can be one of these operational tools. It can provide detailed information on species distribution with relative low effort and costs by extrapolating evidence of species occurrence via spatially continuous information of environmental gradients to the whole landscape. There are various approaches to predicting the spatial distribution of species [2, 3]. In this study we used the Genetic Algorithm of Rule-set Production (GARP) modeling system [4, 5] to model the spatial distribution of plant species richness in West Africa. This Algorithm has excellent capacities for describing the ecological envelopes and spatial distribution of plants [6], insects [7, 8], reptiles [9], birds [10, 11] and mammals [12, 13, 14]. Insights on species distribution were applied for conservation management [15], to model the invasion of neophytes [6, 16] and to predict changes in species distribution according to climate change [11, 8].

Austin (1980) [17] suggested three kinds of gradients which influence plant distribution patterns. These were resource, direct and indirect gradients. From a mechanistic point of view it is desirable to use direct or resource gradients like climatic variables (temperature, rainfall) for predictive modeling of plant distributions [2, 18, 19]. A disadvantage of this data is that they are usually only available at coarse spatial scales and have no predictive power at fine spatial scales. Data on indirect gradients (e.g. topographic variables) are often available at fine spatial scales, show good correlations with observed species patterns [2] and thus can be used as alternative environmental data sets for predictive modeling. While digital elevation models are widely applied in predicting plant distribution patterns[20, 21], other remote sensing data are almost completely neglected. High resolution earth observation satellite systems such as LANDSAT and SPOT could be a very useful data source of indirect environmental gradients. They are relatively cheap, widely available and therefore especially valuable in tropical regions where other data sources at fine spatial scales are difficult to find. In the study region satellite images have been applied in characterizing land cover [22, 23] and its changes [24, 25], vegetation [26] and soil properties [27]. Furthermore the images give insights into agricultural land use intensities [28] and can give hints for characterizing pasture and fire regimes [29, 30]. Each of these parameters can directly or indirectly reflect distributions of single plant species and phytodiversity. Thus LANDSAT data have the potential "to replace combinations of different resource and direct gradients in a simple way" [2] and therefore to predict species distributions at high spatial resolutions. Despite its potential, studies which relate species diversity with LANDSAT reflectance data directly are relatively rare (but see [31]).

In this study we evaluate the potential of LANDSAT satellite images to predict species distribution patterns in the Sahel zone of Burkina Faso, West Africa and to provide an easy to use tool for assessing plant diversity at landscape scale. The objective is to model the distribution of plant species with different ecological characteristics in a precise and realistic way.

2 METHODS

2.1 Study region

The study region is situated within the "Réserve sylvo-pastorale et partielle de la faune du Sahel", Burkina Faso, West Africa. It is the largest protected area of the country, encompassing 1,600,000 ha in the Sahel zone. The main landscape units are dunes and pediplane, interspersed by Inselbergs and usually temporary water courses and lakes.

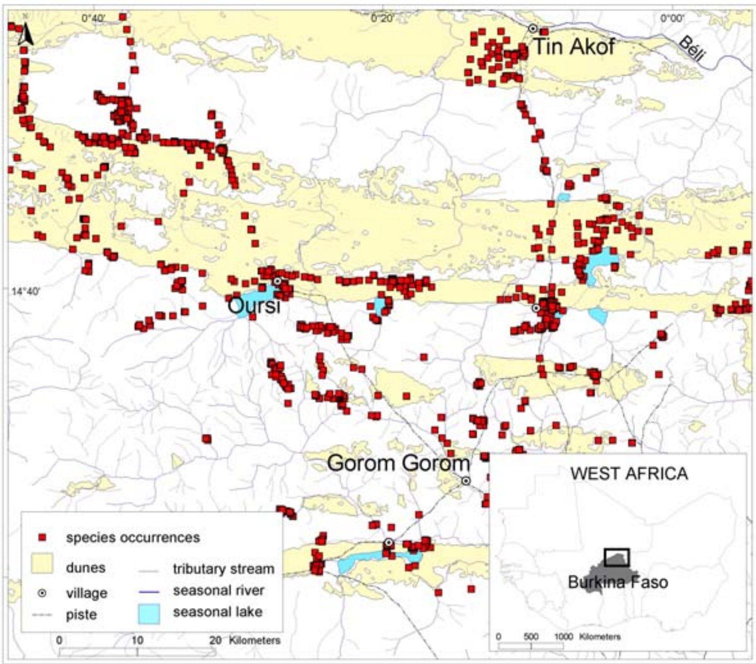

Figure 1. Map of the study area in north Burkina Faso near Oursi and Gorom Gorom. Red quadrangles indicate the location of phytosociological relevés. Pleistocene sand dunes were classified with LANDSAT images.

Anthropogenic impact in the form of pastoralism and agriculture has been shaping the vegetation for more than 5000 years. Numerous studies on Sahelian desertification [32] have focused on this area. For the whole country, Guinko (1984) [33] cites 1054 plant species, Lebrun et al. (1991) [34] 1203 species, but the low collection intensity and comparisons with similar floras allow for estimations of about 2,000 species ([35], unpublished own data from a preliminary checklist), with the highest species richness in the Sudanian zone [19]. White (1983) [36] estimates 1,200 species for the entire Sahelian zone. There are no endemics known for Burkina Faso and only a few for the Sahel as a whole [36]. Due to the long dry season, more than half of the region's flora are therophytes [19].

2.2 Species data

Our data comprises 1461 phytosociological relevés from the Sahelian zone of Burkina Faso distributed over all landscape and vegetation units. Additionally 800 specimens from the West Africa Collection of the Herbarium Senckenbergianum (FR) completed the data, resulting in 13,380 spatially unique species occurrence points for 354 species. For determination of plant species we used Hutchinson & Dalziel (1954-1972) [37].

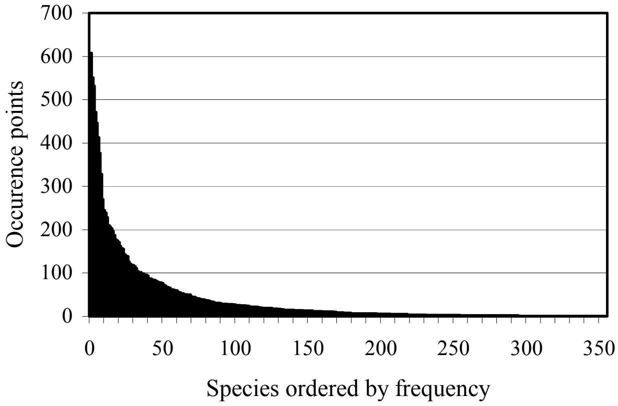

Figure 2. Occurrences per species. Only species with more than 15 occurrence points (i.e. occurring at 15 or more sites) were considered in the modeling process. The two most frequent species are the legumes Zornia glochidiata and Alysicarpus ovalifolius.

2.3 Environmental data

Land cover data were derived from two LANDSAT satellite images (spatial resolution 30 meters) from the year 2000 in the middle of the rainy and the beginning of the dry season. The digital signatures of bands three to six were converted into reflectance values according to Chavez (1996) [38]. To account for vegetation properties different vegetation indices and a tasseled cup transformation (TC) were calculated. Information on soil were derived from the ratio of band5/band7 and band5/band4, which reflect properties of clay and ferrous minerals. Spatial heterogeneity of the landscape was assessed by using the spectral variance of the reflectance values, which were calculated in a moving window (size 3x3 pixel) approach. Topographic data were obtained from the digital elevation model of the Shuttle Radar Topographic Mission (SRTM). All coverages were resampled to a spatial resolution of 60 meters. Combinations of coverages were tested with multiple linear regression analysis to determine which coverages contribute significantly to the predictions of the spatial distribution of species. In a jack-knifing process with a reduced species number (20 species) combinations of layers were tested to check which layers had a significant impact on the accuracy of the models. In a multiple linear regression extrinsic omission and commission errors were used as dependent variables, and inclusion or exclusion respectively of a coverage were used as independent variables. In a stepwise procedure only those coverages which improve the ability of the models to predict species distribution remain.

2.4 Modeling approach

To generate predictive models of species distribution in the study region we used the Genetic Algorithm of Rule-set Production (GARP) modeling system [4],[5]. In the modeling process species occurrences points and environmental data coverages are used to develop models that describe the species distribution in ecological space. A set of rules, which are created by artificial intelligence algorithms, relate the species occurrence and environmental variables via diverse algorithms such as environmental ranges and logistic regression. The rules are determined in an iterative process of generation, testing and selecting rules, which are successful in maximizing accuracy measures based on independent data. Ecological envelopes define the environmental ranges where a species is likely to be found. Consequently the envelopes are then projected onto the whole landscape to predict geographical species distributions. Because of the inherent nature of genetic algorithms GARP does not produce singular results. For each species multiple distribution maps will be produced. Maps which do not reflect the species distribution pattern very well have do be identified with appropriate error measures and excluded in a subsequent evaluation process. Two types of errors can be distinguished: underprediction error (omission error) and overprediction error (commission error) [39]. Whereas the omission error can be easily tested with species occurrences evidence from commission error has to be interpreted with caution. Commission error can reflect either real overprediction of species distribution but also apparent overprediction in regions where a particular species has not been sampled but is in fact there. GARP calculates both error types in an iterative resampling process by comparing model results with intrinsically or extrinsically defined test points (for a detailed description see [40, 5]). We preferred extrinsic testing with statistically independent test points. Therefore we equally divided the species occurrence points in two statistically independent data sets: one for modeling and one for testing. We only modeled plant species with at least 15 spatially unique occurrence points (138 plant species with 12,438 spatially unique occurrence points), to achieve reliable model results following the recommendations of Stockwell and Peterson (2002) [41]. Due to the stochastic nature of genetic algorithms GARP produces several solutions with the same optimization criteria. To account for the variability of possible solutions we created 20 incidence maps representing absence (0) and presence (1) for each species Consequently a careful evaluation of the resulting distribution maps was conducted following the proposed methodology of Anderson et al. (2003) [40] to select optimal models. Therefore we first defined the upper limit of the extrinsic omission error. After visual interpretation of model results an upper limit of 20% seemed to be acceptable. All models with higher omission error values were rejected. In a second step the true proportion of the species' potential distribution was approximated as the mean commission value for those models, which fall below the defined omission limit. We ranked the remaining models according to their omission and commission values and superimposed the best five models to create a composite map showing the number of optimal models in each pixel. After visual interpretation and evaluation of modeled maps for fifteen species a threshold of at least 3 models predicting presence was determined as appropriate for gaining a suitable balance between omission and commission errors. To obtain the distribution pattern of plant diversity incidence maps were added together. Processing of maps written in ASCII format were conducted with visual basic scripts implemented in Access 2000 software.

3 RESULTS

3.1 Floral composition

The flora of the study region comprises 354 species according to our synthesis of collection and observation data. More than half of these are therophytes (62%), frequent especially among the four largest families Poaceae, Fabaceae, Cyperaceae and Convolvulaceae. Phanerophytes (18%) are mainly from Mimosaceae (especially Acacia), Capparidaceae, Combretaceae and Euphorbiaceae, chamaephytes (5%) (including some stem succulents) are a small portion of several families. Most geophytes (5%) and hemicryptophytes (5%) are from Poaceae and Cyperaceae.

3.2 Model performance

In total 2760 distribution maps of 138 plant species were obtained. The evaluation of environmental data layers led to the selection of 12 layers. Finally principal components one to four of the wet and dry season image, TNDVI from the wet season image and channel ratios 5/7 and 5/4 of the dry season image and altitude were included. Models regarding measures of landscape heterogeneity had no better predictive ability and were rejected. After the evaluation and selection of best performing models average omission error was reduced to 10%.

3.3 Delineating patterns of phytodiversity

The combination of the 138 single species models led to the map of plant diversity of the study region. The highest values of plant species richness are found in the northwest of Oursi, an area with dunes and tiger bush vegetation, that is only sparsely grazed because water holes and rivers are lacking. Tiger bush provides many different microhabitats with its alternating stripes of dense thickets and bare soil. Lowest values occur on heavily grazed and often incrusted pediplanes in the southern and eastern part of our study area, covered with Acacia and grass savannah. Water courses and depressions usually display higher diversity values than their surroundings, the same is true for the edges of the dunes.

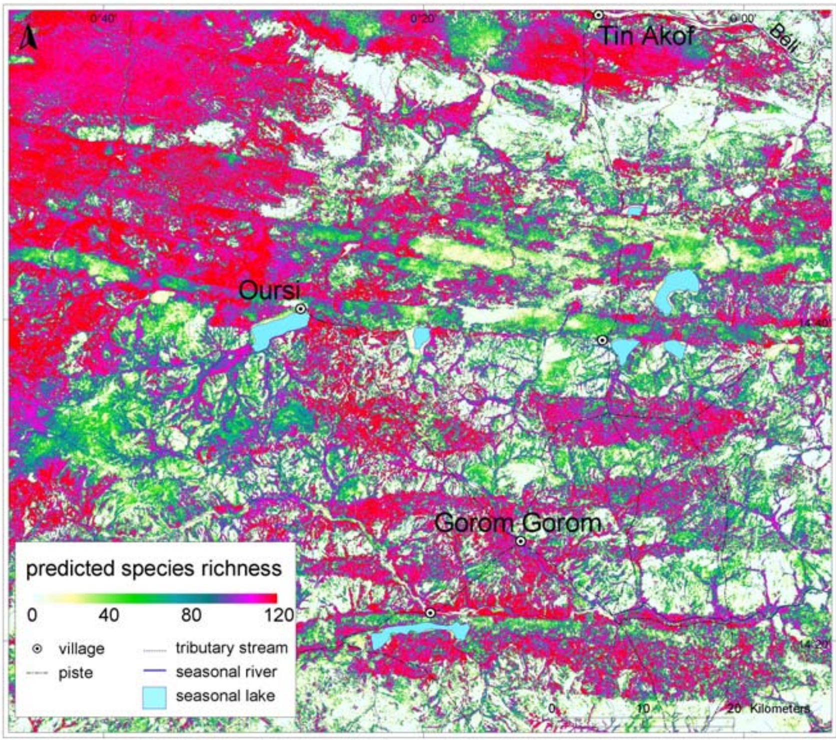

Figure 3. Distribution of plant species richness.

4 DISCUSSION

In this study we use LANDSAT satellite images and field data to predict pattern of plant diversity in the Sahel region of Burkina Faso. We demonstrate the ability of widely available satellite data to model species distributions with high accuracy at fine spatial resolutions. The resulting pattern of plant diversity shows highest values in an area covered to a large extent by tiger bush. This is partly in accordance with Hiernaux and Gérard 1999 [42], whose findings reveal a higher herbaceous plant richness of tiger bush, as compared to the Sahel average, but a lower woody species richness. The area is also less influenced by grazing because of the lack of watering points, and this might also contribute to a higher diversity, since overgrazing and degradation are serious threats to dryland biodiversity [43]. The higher diversity along water courses and depressions may be due to the fact that water is the limiting factor for many species and the neighbouring, more humid Sudanian Zone has a much richer flora [36],[19]. It is also affected by the size of our grid cells, which encompass a whole range of habitats along the water courses, and these will all contribute with various species sets. Using the proposed approach several drawbacks have to be considered. The first concerns the sampling quality of species occurrence, the second the capability of satellite images as explanatory variables and the third the conducted modeling approach.

Two properties of species occurrence points can greatly affect the predictive quality and accuracy of distribution models a) number of occurrence points and b) data completeness. Below a certain threshold the number of species occurrence points does not allow meaningful model training and testing. Even the necessary number is also affected by inherent properties of the species e.g. habitat specificity, niche width and frequency. As a rule of thumb at least 10-20 points are needed for reliable predictions [41]. In most studies frequency distribution of species looks similar, as in our Figure 2 where a relatively small proportion of species is sampled with high frequency, but the majority of species are sampled with very low individual numbers. Thus a majority of species can not be modeled with high predictive accuracy. If the modeling of rare species is the main objective, e.g. for conservation purposes, this approach will prove to be rather unsatisfactory. This point stresses the need for adequate field sampling designed for specific purposes. To make predictive modeling operational, adequate sampling design may help to reduce sampling efforts by maintaining predictive quality of models. Inadequate sampling of species can seriously bias modeling results if the range of possible environmental values is not completely covered. In this study sampling was conducted with high effort and complete coverage of environmental gradients is likely to have occurred. Whether or not this is actually the case will be tested in future studies.

Scale is of considerable importance for environmental modeling. Coarse scale environmental data layer cannot account for local variations in species distribution. Only high resolution satellite images can assess the spatial and temporal variability of environmental gradients directly and can reflect a considerable proportion of the environmental envelopes of species. However, relationships between causal factors determining species distributions and the signal of satellite images remain unclear, and use of indirect gradient data could only lead to robust results if indirect gradients are directly correlated with causal gradients [2]. Additionally findings can only be applied within a discretely defined area. Extrapolation of findings from one area to another must be performed with caution. This requires species occurrence data across the whole range of environmental gradient values.

GARP does not produce single solutions for each modeled species. To cope with the huge quantity of modeled maps we applied an unsupervised evaluation approach to select the best performing models. The underlying assumption of an inverse association between omission and commission error was not tested for all modeled species. This has to be examined in more detail in future studies. GARP models are static and cannot reflect the stochastic nature of landscape. They assume that species live in equilibrium with their environment and do not distinguish between transition states of species occurrence. Here modeling of species distribution with field data sets from consecutive years could give insights to temporal changes in distribution pattern.

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LEIS, a tool for diagnostic and prevision of anthropogenic pressure on natural vegetation: an overview

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ABSTRACT

The desertification process including natural resources degradation is a major obstacle to development in tropical and Mediterranean zones with strong drought constraints, leading to a wide range of possibly irreversible catastrophes: famine, land neglecting and migration. This process comes from a combination of different modes of resource management from different actors at different scales, interacting with strong climatic constraints in arid and semi-arid zones.

In Africa and particularly in the circum-Saharian zone it is urgent to deliver useful information for decision making focused on improving the natural resource management at a local level in order to slow down the desertification process. For that purpose within the ROSELT/OSS program [1], has been developed the LEIS (Local Environmental Information System), an original tool coupling GIS and models to be able to establish a complete diagnostic of natural resource use allowing prediction of future evolutions.

The aim of the LEIS is to model the functioning of an observatory territory at local scale taking into account both biophysical data and socio-economic data together using an integrated spatial approach. Acknowledging the dynamic interactions of these two set of factors the integrated spatial approach is the core of the tool and the conceptual models derive from it. Using minimum kit dataset the modeling of the functioning establishes a diagnostic: spatial description of the uses and resources interactions into spatial references units (SRU) and quantitative estimation of vegetation pressure spatialised on these SRU. The modeling structure allows some prevision to be made when setting a scenario of evolution of parameters. The forecated diagnostics can be compared for different scenarios while succession of diagnostics in a long term monitoring helps to analyse vegetation pressure evolution and to build realistic scenarii. Before being communicated to local or national authorities, as useful information for a better evaluation of desertification risks, balances maps can then be aggregated according to administrative units or to some biophysics units depending on a specific interest.

Implemented under the same GIS software platform, the LEIS tool couples a geographic database and spatialisation models. In this article we will describe especially the integrated spatial approach which is the core of the tool and the conceptual model deriving from it. Aiming at expressing the prospective potential of the tool, the spatialisation models implemented will be explained.

Keywords: desertification, environmental modeling, landscape, use, natural resource

1 INTRODUCTION

The desertification process including natural resources degradation is a major obstacle to development in tropical and Mediterranean zones with strong drought constraints, leading to a wide range of possibly irreversible catastrophes: famine, land neglecting and migration. This process comes from a combination of different modes of resource management from different actors at different scales, interacting with strong climatic constraints in arid and semi-arid zones.

In Africa and particularly in the circum-Saharian zone it is urgent to deliver useful information for decision making focused on improving the natural resource management at a local level in order to slow down the desertification process.

In this context, the Roselt Programme (Observatory Network for Long-Term Ecological Monitoring), set up by the Sahara and Sahel Observatory (OSS), organises scientific monitoring of the environment in order to characterize the causes and effects of land degradation, and to have a better understanding of the mechanisms which lead to desertification. The network is made up of a number of observatories which operate in a network at regional level in Africa, in the geographical zone of the OSS. This zone comprises three sub regions: North Africa, West Africa and East Africa. ROSELT aims at providing reliable data on land degradation in arid areas and pertinent biophysical and socio-economic indicators of desertification, as well as to assess the state of the environment within the OSS zone. As a network ROSELT aims at harmonising methodologies of data collection and data analysis as well as data or metadata circulation.

For that purpose within the ROSELT/OSS program [1], has been developed the LEIS (Local Environmental Information System), an original tool coupling GIS and models to be able to establish a complete diagnostic of natural resource use allowing prediction of future evolutions. This tool, proposed and developed by the Regional Operator of Roselt (IRD, US Desertification, n°166) is issued from a close working attitude and procedure between methodological research, modelling, computing and information system conceptions, and applications in 9 observatories of 9 circum-Saharian countries, involving numerous researchers from the north and from the south.

The aim of this paper is to give an overview of the tool presented in a poster session of RGLDD 2005, as much in its conceptual aspects as in its implementation.

2 GENERAL PRINCIPLES OF LEIS

The global methodology of the LEIS [4] is to combine biophysical data and socio-economic data together using an integrated spatial approach (fig.1). To be able to distinguish in the landscape the respective parts of factors coming from the above mentioned domains, the spatial approach considers intersecting two planes of distinct information, one linked to uses expression and the other one to natural resources, then defining Spatial References Units (SRU) as a functional description [2]. The renewable natural resources are extracted simultaneously or successively in time for diverse uses (agricultural, pastoral, forest, others) in the majority of arid and semi-arid zone. Therefore, multi-use balances (availability minus extraction of natural vegetation) and anthropogenic pressure indices computations are based and output on this functional mosaic description of the landscape (SRU).

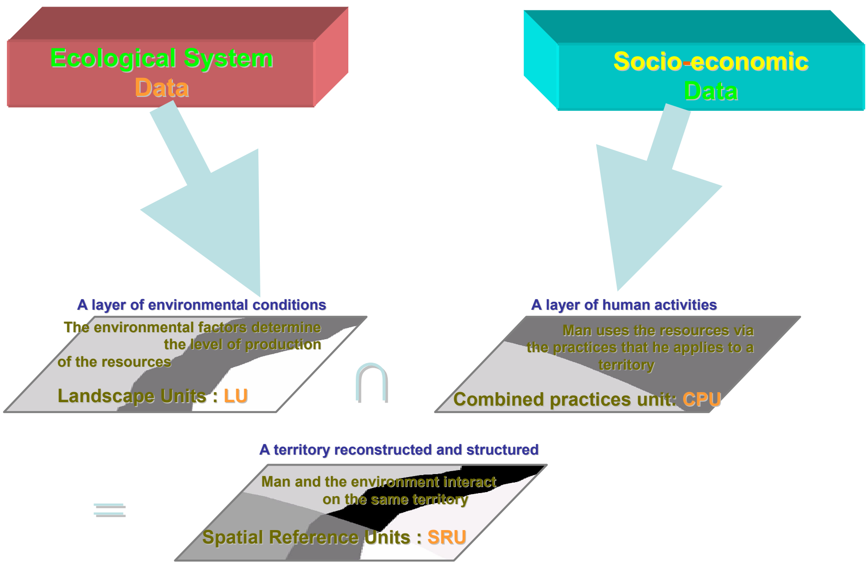

Figure 1. LEIS integrated spatial approach.

The established modeling at a defined period leading to the SRU map and the balances maps resulting on them constitute the diagnostic. The plane of resources expression is built using classical GIS methods on a range of different layers, while the plane of uses comes from spatialisation models of the exploitation practices. These models encompass the originality together with the prediction ability of the tool. From an established Diagnostic, scenarii of evolution of main driving parameters such as population, production (depending on "climate parameters") allow prospective to be made i.e. leading to new balances maps based possibly on new SRU.

3 MODELING PRINCIPLES OF LEIS

Landscape Units:

The plan of natural resources or plan of environmental conditions distinguishes from a biophysical point of view homogeneous areas called landscape units (LU). The Landscape Units are immediately visually perceptible. From a methodological point of view, the LUs are the result of the interaction of three main categories (fig.2) of factors: physical, biological (Land Cover) and human (Land Use in the sense of international classifications: forests, pasture land, cultivated land, etc.).

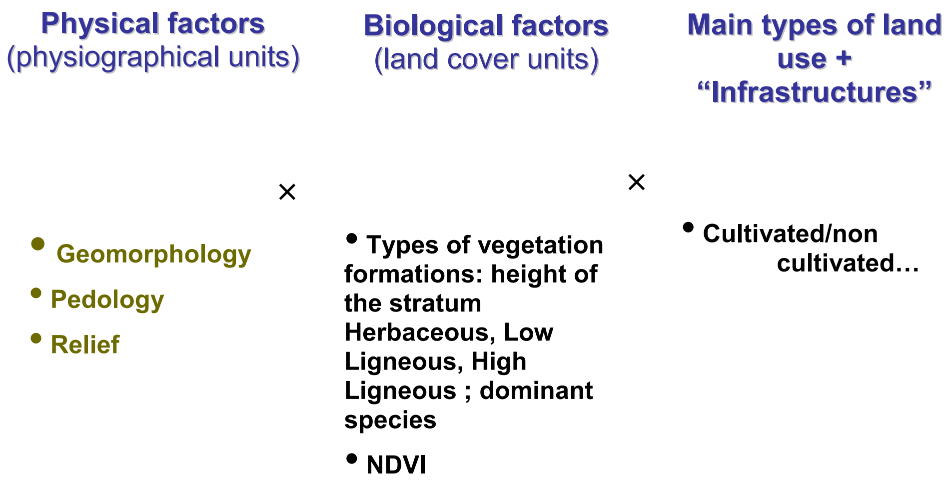

Figure 2. Landscape Units decomposition.

The construction of landscape units calls for classical cartographical methods, combining ground surveys, the use of aerial photos and the processing of satellite images. The limits of landscape units, at a certain level of aggregation or de-aggregation, are common to different biophysical disciplines, at a scale compatible with the changes that Roselt is looking to study. They should have a relative stability in time for given period of diagnostic modeling. The length of this period (usually between 2 and 5 years) is a compromise between the biophysical stability functioning, and the stability of socio-economic functioning. The characteristics of each type of Landscape Unit determine a level of resource production.

Combined practices units

The plan of uses expression or plan of human activities delimits areas, with the same relative stability of LU, which are homogeneous from the point of view of resource exploitation practices.

An exploitation practice is a concrete action of natural resources exploitation led by an exploitation unit according to its exploitation strategy, the biophysical constraints and the objective of production. A class of "combined practices" expresses the simultaneous or successive combination of exploitation practices at the same spatial unit, seasonally or annually

These spatial units are called "Combined Practices Units" (CPU).Contrary to Landscape Units, they are not necessarily visible in the landscape. They are not constructed from processed satellite images and ground surveys, but made from models of spatial distribution practices.

The construction of Combined Practices Units is realised in two main stages:

- the development of the combined practices typology on an observatory territory
- the development of the CPU map via a combined practices spatial distribution model

A particular Combined Practices class issued from the typology associates one or more natural resource exploitation practices, in time and space, with one or more usages. This combination of practices can be applied to different places on the observatory territory on homogeneous spatial units [5]. The occurrences of all the combined practices classes structure the landscape in the Combined Practices Units.

The fundamental principle of the spatial distribution model is as follows: the combined practices are potentially applied to a given location by one or more groups of agents according to:

- the local biophysical characteristics at this location;
- and the expectation that the production of exploitation at this location (average production per exploitation cycle), will contribute to the satisfaction of a type of need

The equation (eq.1) illustrates this principle and is the actual model for agro-pastoral modeling: at pixel px the Combined Practice CP giving the maximum production depending on soil for the minimum effort depending of artificialisation and distance to the CA is spatialised.

$$CP(px) = \arg\max_{cp}\left(\frac{\Pr oduction(cp, Soil(px))}{Effort(d(px, CA), Artif(cp))}\right) \quad (1)$$

Balances evaluation

The methodology consists in applying an analytical approach by usage with a calculation and spatialisation of the balances on the SRUs for each activity. Availabilities of resources are derived from the LU characteristics while extractions of resources are modelled and spatialised according to the current usage and the structuring usage which built the CPU.

4 MODEL BUILDING STEPS

While establishing a diagnostic at a given period, choices of modelling and data collection is needed at each step of modelling (fig. 3).

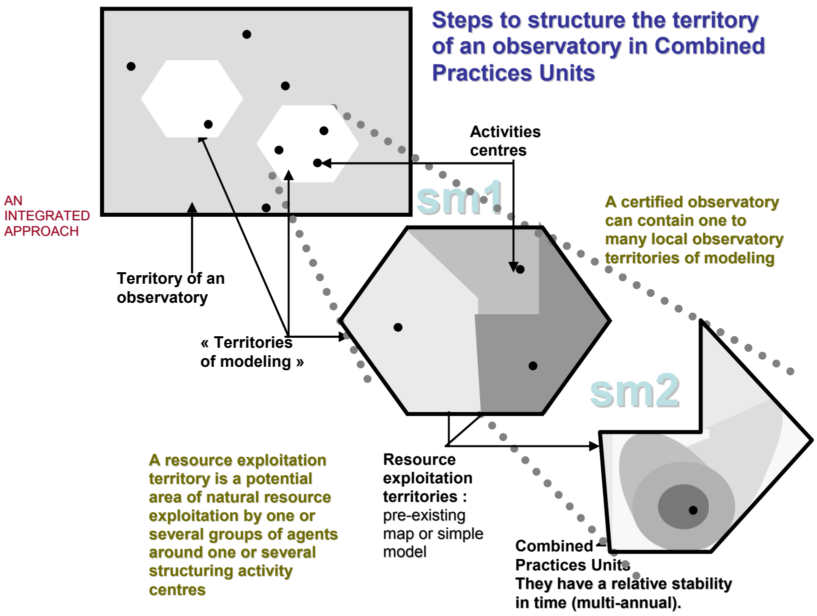

Figure 3. Combined Practices spatialisation models (sm1 and sm2).

(i) Step of spatialisation of the combined practices (CPU) fig3:

- step of delineation of territories of potential exploitation (fig3.sm1)
  According to the choice of activity centres (AC e.g. villages, wells, encampments …) linked to the structuring activity and the choice of the weights (e.g. population, oldness, …) expressing land competition between CAs a weigthed Thiessen algorithm builds a mosaic of polygons of potential exploitation territories around each selected CA;
- typology of combined practices: an agro-socio-economic survey [5] allows to establish a typology of combined practices associated each strategic groups surveyed within the observatory

According to a principle (eq 1.) of maximum production for minimum effort the combined practices are spatialised (fig3.sm2) within each potential territory of exploitation using an algorithm computing the localised production (depending on soil or pastoral quality) and the effort (depending on distance and artificialisation of the combined practice); the extent of exploitation is then limited in order to reach satisfied needs (wished or expected production from the agents)

(ii) Step of definition of the SRU (fig1):

The intersection of CPU and LU is done once the Landscape Units are built following some guidance (see Roselt/OSS, SD3) on choosing the different maps expressing the landscape.

(iii) Step of balances and index computation:

For each usage a set of CAs together with their specific parameters linked to extraction of resources is chosen. According to a simple model of extraction linked to LU preferences availabilities and extractions are spatialised on the SRU (fig 4).

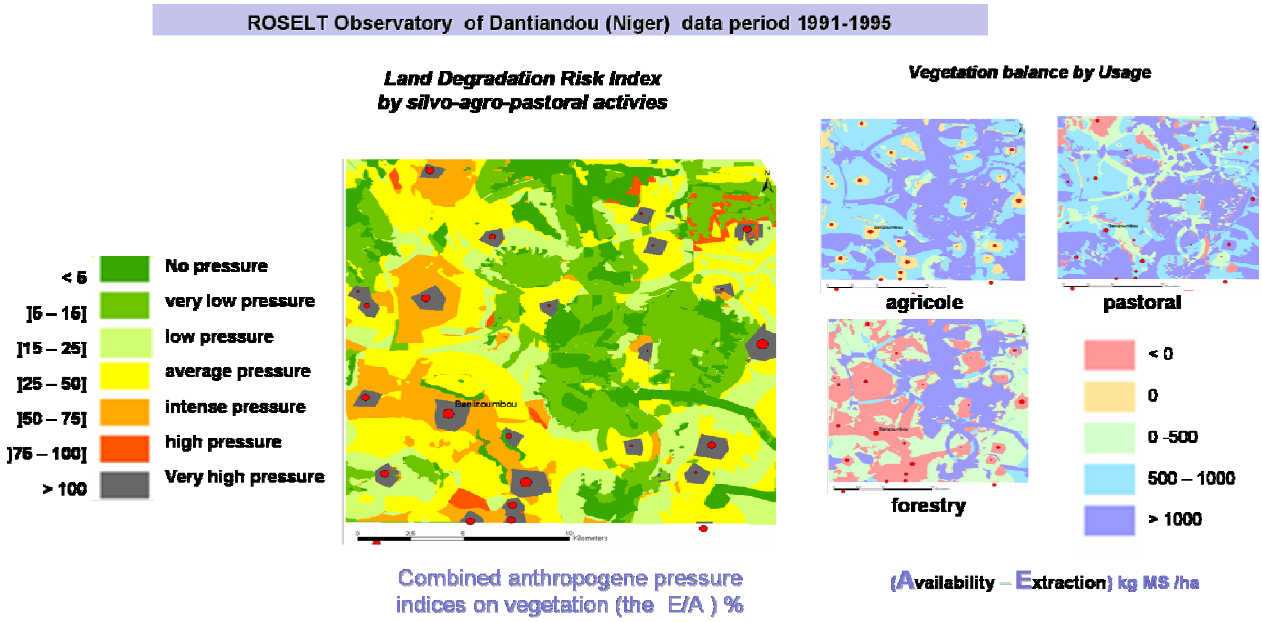

Figure 4. Diagnostic example of vegetation pressure reported on the SRU.

The combined anthropogene pressure indice on vegetation can be considered as a degradation *risk indicator* of the plant resource. It measures the pressure on the environment with regards to the resource (in this case plant resource). It is absolute in the sense that its calculation is made on each SRU without reference to the functioning of the whole observatory. It is therefore comparable from one observatory to another. Whatever the availability of the resources, a quotient greater than 100% potentially indicates, for this unit, a resource extraction that is greater than the availability, which explains the risk. The greater the value of the index, the greater the risk.

(iv) Step of scenarii:

Using the driving parameters of the modeling such as population, productions (natural vegetation and or agricultural production) needs it is possible to forecast, according to the chosen scenario of evolutions, the SRU and the balances on this SRU built in the same way as the previous steps (fig 5).

The analysis of tendencies to degradation through diagnoses constitute in oneself a decision making tool. The exploration of these tendencies, through simulations of scenarii, increase the capacity of decision-making aid. Long-term monitoring will allow us to compare the results of simulations with collected data, and thus to validate or readjust the models implemented in the LEIS.

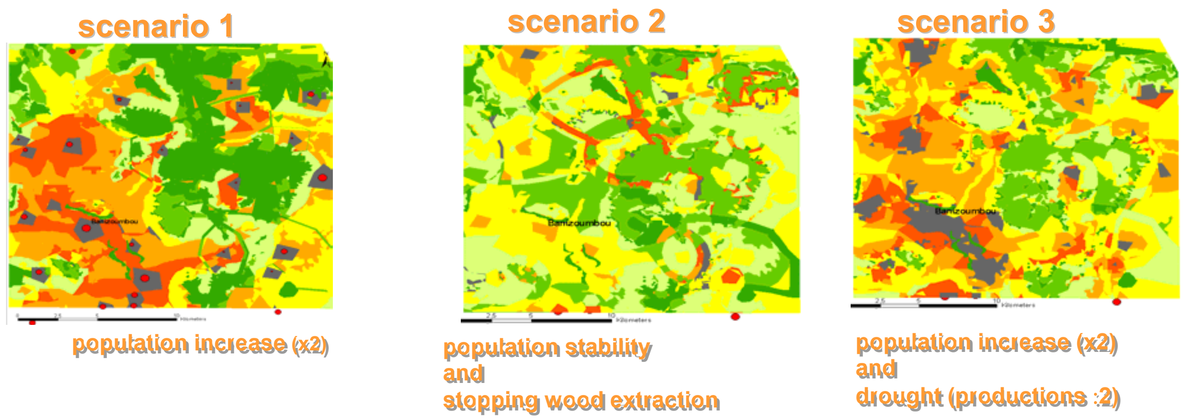

Figure 5. Examples of scenario previsions (Combined extraction index, same legend as fig. 4.).

The analysis of tendencies to degradation through diagnoses constitute in oneself a decision making tool. The exploration of these tendencies, through simulations of scenarii, increase the capacity of decision-making aid. Long-term monitoring will allow us to compare the results of simulations with collected data, and thus to validate or readjust the models implemented in the LEIS.

5 IMPLEMENTATION OF THE GIS TOOL

Implemented under the same GIS software platform, the LEIS tool couples a geographic database and spatialisation models. The geographic database is organised in relational database management system; its structure was formalised using UML [3] to conceptualise and represent the spatial area-resource-usage interactions at local scale (fig. 6).

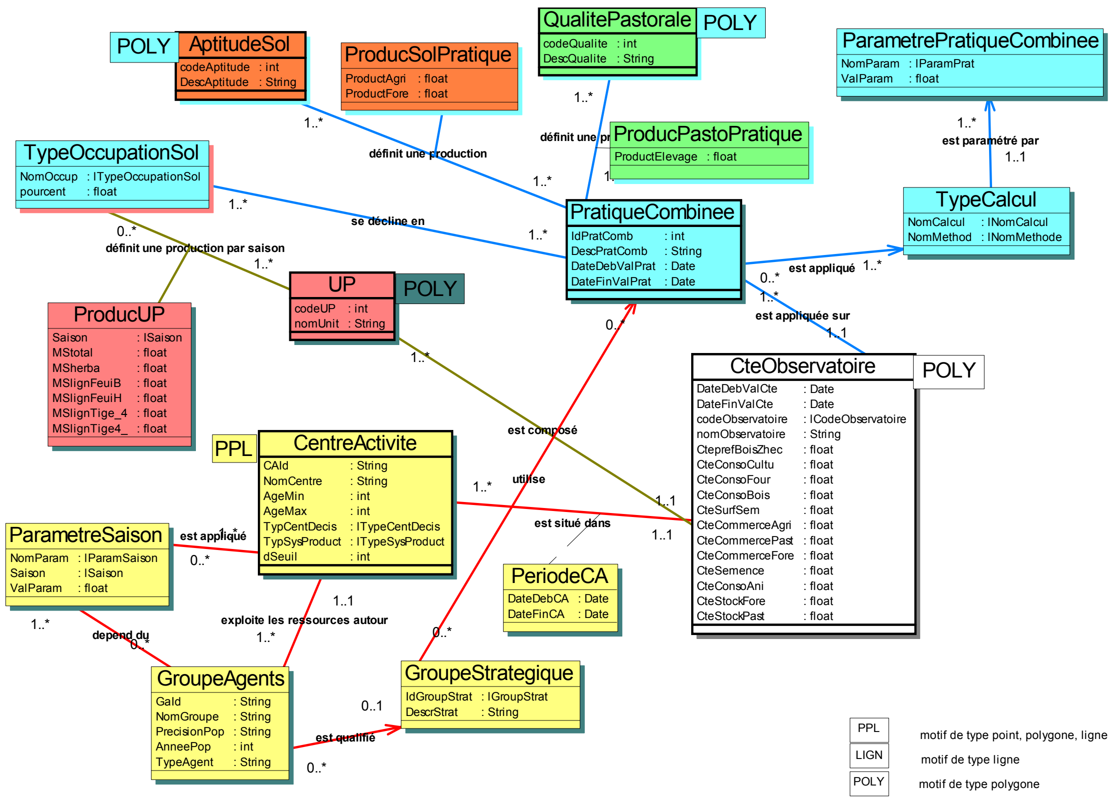

Figure 6. UML diagram of the input data of the LEIS.

Different concepts represented as classes on the diagram of fig 6 are colored according to their different categories but are fully described in the SD3 [4] . The geographic database or GIS database represents here the minimum dataset required to start modelling and grows as the modelling goes from step to step. An extension of ArcGIS has been developed to implement the different steps described above. This extension (fig7) use one GIS database for each observatory but can manage and build more than one modelling at the same time. This allows different tries at different steps in order to initiate validation of the results according to different choices of modelling.

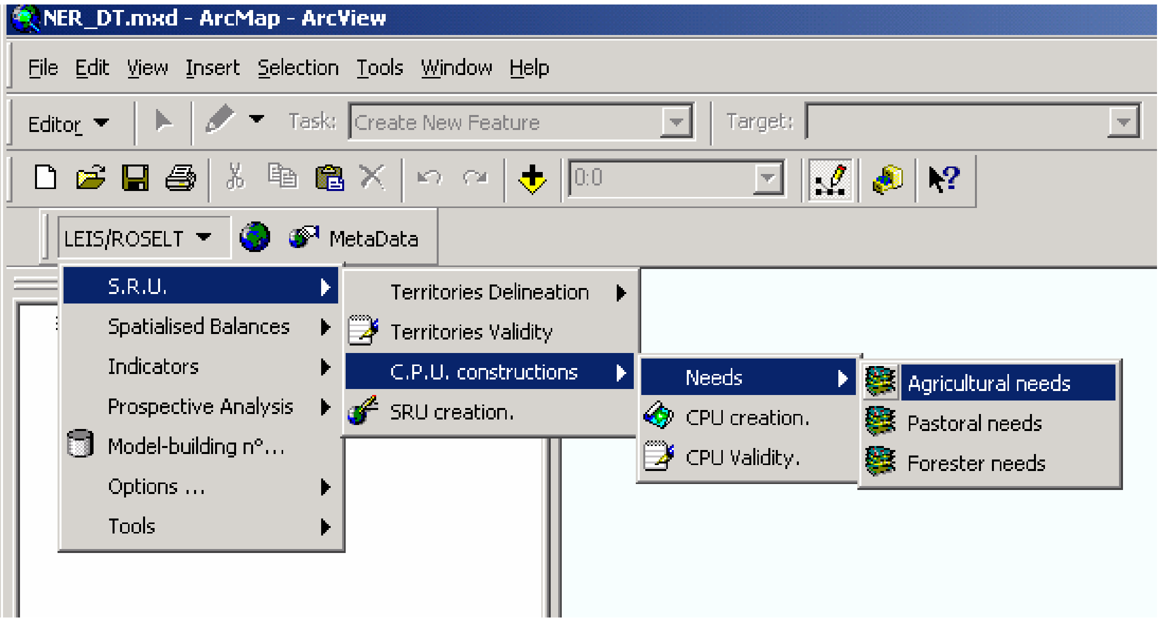

Figure 7. General User Interface of the LEIS tool.

Despite the friendly general user interface LEIS is a tool dedicated to scientists of the domain.

6 PERSPECTIVES AND CONCLUSIONS

Within ROSELT network a full diagnostic prototype of LEIS according to the modelling descriptions here exposed shortly has been done in 4 observatories out of 11 pilot observatories. The results of modelling are actually in validation process. The others observatories are at different stages progressing to establish it. The interest in the tool and the dynamical impact on the other monitoring activities of the network from scientists up to decision makers make the LEIS an important decision tool of the environmental monitoring setting. The integrated approach and methods to establish resource/use spatial balances provide:

- an organisation of data collected from biophysical and socio-economical origins in an integrated schema;
- an activity timetable for the observatories;
- an adaptation of the sampling and data collection for the study of long-term man/environment interactions;
- a harmonisation of the data, of its survey and processing methods from one observatory to another;

All these integrated methodologies contribute to build the national networks of observatories for environmental surveillance in the NAP/CD (National Action Plan to combat desertification) context. The definition of the methodologies to change scales, from the local level to the national level, and reciprocally, are a major current concern of the ROSELT network.

To fully exploit the modelling aspects allowing prospective better scenarii have to be used. First of all realistic climatic scenarii and their impacts on natural vegetation productions as well as crop productions have to be implemented. This is the improvement looked for within the European programme DeSurvey in linking the LEIS model with the PATTERN model [6] which was used in MODULUS a tool similar to LEIS but focused onto policy decision making in the Mediterranean area.

ACKNOWLEDGMENTS

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A GIS-based landscape characterization to assess soil erosion and its delivery potential in the highlands of northern Ethiopia

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ABSTRACT

Land degradation (erosion, soil nutrient depletion, sedimentation) in the highlands of northern Ethiopia is a very serious problem affecting one of the most vulnerable people on earth. Conservation measures are extremely important to reduce further degradation, reclaim the degraded areas and improve land productivity as well as food security of the subsistence farmers. Identification of very high risk areas of degradation is equally critical to prioritize areas of intervention in light of the economic situation of the people and the nation as whole. Currently, distributed erosion models are being used to identify hot-spot areas of erosion and prioritize areas of intervention. However, such models require distributed data for calibration and input, which limit their wide application to data-scarce regions. An alternative approach that can help assess a watershed's propensity to erosion and sediment delivery using easily available data is therefore necessary. To this end, the concept of Similar Erosion Risk Potential Units (SERPUs), areas characterized by similar erosion risk potential due to similarity in geomorphologic processes and human practices, was introduced. The concept of the SERPUs revolves around systematically classifying catchments such that similar attributes, potentials, constraints and processes could be found. To achieve this, five basic terrain attributes, i.e., slope, lithology, surface cover type, surface cover condition and gullies were derived from field surveys, topographic maps and satellite images. These attributes were used as proxies to assess erosivity, erodibility, frictional resistance, degradation level and connectivity and delivery efficiency of catchments, respectively. Each landscape unit within catchments was classified into five categories of erosion potential and mapped (very high, high, medium, low, and very low) considering the above attributes. A systematic combination of the five maps in a GIS and performing simple overlay analysis enabled the categorization of landscape units into similar runoff and erosion potential. The resulting maps were compared with potential sediment source area maps derived from field surveys. The analysis shows that the SERPUs could categorize landscape units into different levels of erosion risk and help to identify areas that require priority in conservation measures relative to others.

Keywords: Landscape characterization, Similar Erosion Risk Potential Units, erosion severity levels, GIS, overlay, northern Ethiopia.

1 INTRODUCTION

Land degradation is a serious problem in Ethiopia leading to a loss of soil fertility, sedimentation and pollution of rivers and lakes, disruption of the hydrologic cycle, increased severity of the impact of droughts, and a further reduction in the ability to produce food and other biological resources demanded by the increasing human and animal population [1]. Soil erosion is the most serious mechanism of land degradation, in which its on- and off-site impacts threaten the food security of people and the national economy of the country [2, 3]. This raises the need for the evaluation of suitable methods of land-use management that can lead to a reduction of soil erosion and subsequent sediment delivery. An efficient control of erosion and sediment yield requires proper assessment of the main controlling factors and identification of hotspot areas [5].

Watershed management programs for effective conservation planning must identify the landscape positions that experience high erosion and contribute the most sediment as these critical units need preferential conservation measures [6]. High-potential source areas of sediment can be identified by combining attributes that enhance susceptibility, which is a result of interaction of natural terrain attributes and anthropogenic practices. Soil erosion models represent an efficient means of assessing the physical processes and mechanisms governing soil erosion rates and identifying critical areas of sediment sources [7].

However, application of distributed erosion models is not an easy exercise due to problems of cost, complexity and representativeness. The physical based models require too much data for calibration [8]. They also have problems in their predictive power [9] making their potential application as policy tools less suitable [10]. The fact that a priori knowledge of the area prevails in the selection of parameters also introduces a strong degree of subjectivity in the calibration of process-based models [11], which means that the successful application of such models is more dependent on the intervention of the user, rather than the model design itself [12]. Applications of less data-demanding empirical models do not form a viable alternative either, since their empirical basis impedes an extrapolation beyond their data domain with confidence [13, 10]. The site-specificity, parameter limitations, and problems of representativeness of empirical models require that considerable research be made to predict erosion before reliable use of the models can be made [14]. The application of distributed models, mainly to data scarce and remote regions, is therefore, complicated.

The above issues call for the necessity of an alternative approach that can help assess a watershed's propensity to erosion and sediment delivery using easily available data. To this end, the concept of Similar Erosion Risk Potential Units (SERPUs), areas characterized by similar erosion risk potential due to similarity in geomorphologic processes and human practices, was introduced in this study. The concept of the SERPUs revolves around systematically classifying catchments into homogeneous zones such that similar attributes, potentials, constraints and processes could be found. It is likely that entities with the same erosion process dynamics consist of certain associations of system characteristics and system inputs.

In line with the above, studies by Flügel [15], Marker et al. [2001] and Flügel et al. [4] show that different erosion process dynamics are associated with specific assemblies of components from the basin's natural and human environment. Consequently, Marker et al. [16] and Flügel [15] conceptualized Erosion Response Units (ERU), which are heterogeneously structured terrain units having homogeneous erosion process dynamics characterized by a slight variance within the unit, if compared with neighbouring ones, whose erosion response is controlled by their physiographic properties and the management of their natural and human environment. Fargas et al. [17] developed a method to identify sites of similar sediment emission risk through qualitative ratings of basic terrain data that can be related to erosive processes. Recently, hydrologically similar surfaces (HYSS), which are distributed homogeneous runoff response units within a catchment based on key runoff producing variables of land use, slope and geology, were conceptualized by Bull et al.[18]..

The SERPUs in this study were derived based on the assessment of basic environmental factors of catchments that could define the susceptibility to erode and connectivity to deliver following similar principles cited above. The aim was to present a simple method of predicting hot-spot areas of erosion using runoff potential produced by combining key catchment attributes in a GIS and performing simple overlay analysis. Since the SERPUs will be derived based on catchment attributes that determine erosion processes, they can represent potential areas of similar erosion problems and serve as proxies to erosion potential maps. The approach is attractive as it focuses on basic terrain information that are relevant to erosion, transportation and deposition processes but relatively easy and less costly to acquire.

2 STUDY SITES

For this study, two catchments with an area of 14 to 19 km^2 were selected in the Tigray region, northern Ethiopia (Figure 1). The landscape of the two catchments is rugged terrain dissected by gullies. Land use/cover is composed of arable land, pasture, and scattered bush/shrub. The major lithology is shale intercalated with limestone with mountain tops covered by sandstone. Soils are dominantly leptosols on the upslope positions, cambisols on the middle slopes and vertisols at downslope locations. The terrain, lithology, and surface cover attributes of the catchments facilitate erosion processes. The two catchments are thus among the most eroded in the region.

3 METHODOLOGY

3.1 Data acquisition

To generate the SERPUs, data on topography, land use/cover (LUC), lithology, and gullies were required. Terrain data such as slope were generated from DEMs derived from 1:50000 scale topographic maps [1]. LUC-related data were derived from ASTER images. Lithology data were extracted from 1:25000 scale geomorphology maps [20] and field surveys. Potential locations of gullies were derived using terrain indices [21]. All the maps were held in grids of 10-m cell size. GIS was then used to perform reclassification and overlay analyses. The procedures employed to derive the variables needed to create the SERPUs are discussed below.

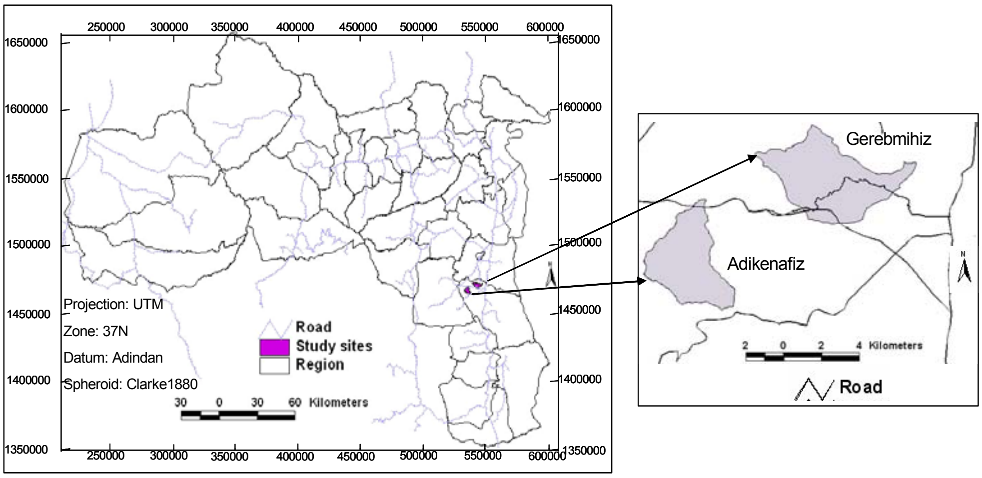


Figure 1. Location of Adikenafiz and Gerebmihiz catchments, Tigray region, northern Ethiopia.

3.2 Catchment stratification based on slope

In this study, slope steepness was considered to serve as a proxy to estimate erosivity potential. The slope maps of the catchments were thus classified into potential runoff assuming that runoff velocities are greater on steeper slopes, resulting in high runoff and erosion potential. On flat slopes, runoff thresholds will be high, since water is likely to pond and infiltrate, resulting in little or no runoff [18]. Considering other variables to be constant, it was assumed that slopes less than 5° are likely to produce low amounts of runoff followed by slopes 6 - 10°, 11 - 15°, 16 - 25° and > 25°. The slope maps were reclassified based on these thresholds and the slope classes were ranked from 1 to 5, with 1 representing slopes with the lowest runoff potential and 5 the highest (Table 1).

3.3 Catchment stratification based on lithology

In an environment where acquisition of soil data is difficult, surface lithology could serve as a proxy to estimate erodibility potential. The potential soil loss from different lithologic groups was thus estimated based on their material composition and the sensitivity to weathering of different lithologic surfaces [17, 22]. Generally, areas dominated by sandstone, basalt, and metavolcanics can have high permeability and contain less loose materials that can be transported by water and therefore do not yield much sediments. On the other hand, shale/marl, siltstone and clay-stone-dominated areas could have low permeability and low infiltration capacity, which encourage runoff and sediment yield (e.g. [17, 23]).

Based on the above premises, sandstone-and basalt- dominated landscape units were ranked to have very low and low soil loss potential. Dolerites were characterized to have medium soil loss depending on the degree of fracturing. Limestone and siltstone were generally characterized as having medium infiltration and runoff potential. Sites with alluvial/colluvial materials were categorized to have medium to high soil loss potential while shale- and marl- dominated landscapes were classed into high to very high potential soil loss category (Table 1).

Table 1. Classification of parameters used in the overlay analysis to generate SERPU.

Rank	Parameter			
	Slope	**Lithology**	**Land cover**	**TSAVI***
1	0- 5°	Sandstone, metamorphic, basalt	Dense cover/enclosures	> 0.1 (Good cover)
2	5 – 10°	Hard rocks of lime stone, dolerites	Bushes/shrubs	0.04 – 0.1 – (Open cover)
3	10 – 15°	Limestone intercalated with shale	Scattered bushes/shrubs	0–0.04 (medium cover)
4	15 – 25°	Colluvium/alluvium deposits	Cultivated/cropland	-0.02 – 0 (Poor cover)
5	> 25°	Shale, marl, debris flow	Degraded bare land	< - 0.02 (Very poor cover)

* Transformed Soil Adjusted Vegetation Index. Classes show landscape sensitivity to degradation.

3.4 Catchment stratification based on surface cover types

Data on surface cover type enables assessing the resistance to erosion of terrain units due to surface protection. When land is covered with vegetation, total roughness can be high, which can increase infiltration and runoff threshold (e.g. [18]). When the land has poor surface cover, its roughness will decrease, resulting in a low runoff threshold and a quick response to rainfall [24, 18]. Forest parcels do not generate runoff and are therefore hydrologically isolated, while arable land areas can be considered as being hydrologically continuous [25]. Overgrazing can result in surface crusting and increase runoff potential (e.g., [26]. Accordingly, areas with dense cover were considered to have low runoff potential while bushes and shrubs were assigned with medium runoff potential. Agricultural cropping lands were considered as sites having high runoff potential and barren degraded lands were assigned with very high runoff potential (Table 1).

3.5 Catchment stratification based on fraction of surface cover

Land-cover maps generated from satellite images may not be able to show the status and fractional degree of the surface cover. Different vegetation indices are available to assess the fractional degree of surface cover and its relative level of degradation. The transformed soil adjusted vegetation index (TSAVI) is considered the most efficient method for calculating vegetation cover ratios in semi-arid environments of sparse vegetation [4]. The TSAVI significantly reduces the effect of soil background reflectance and is conducive to map degradation level. This index was used to differentiate the fractional degree of vegetation cover, where negative values indicate a smaller vegetation cover (higher degradation and hence higher erodibility). The results of the TSAVI were categorized into five classes, with high negative values representing poor surface conditions and positive values indicating good surface conditions (Table 1). Natural breaks of the TSAVI histogram were used to define each TSAVI class (Table 1).

3.6 Predicting gully networks and spatial patterns

Field observation indicates that gully erosion and bank collapse are important processes of soil loss in the catchments. Gullies directly contribute to soil loss due to bank collapse and increase catchment connectivity which encourages efficient sediment delivery [27]. They increase runoff by channelling overland flow and producing an efficient route way for the channelled flow [18]. Automatic delineation of ephemeral gullies from DEMs combined with results of the above factors can thus improve the prediction of the most critical areas of erosion within a catchment. To achieve this, the potential locations and spatial patterns of gullies were analyzed following the method proposed by Thorne et al. [21] and Moore et al. [28]:

$$A_s \tan\beta > 18$$

$$\ln\left(\frac{A_s}{\tan\beta}\right) > 6.8\,,$$

where A_s = the unit contributing area ($m^2\ m^{-1}$); β = the local slope (tan); $A_s \tan\beta$ and $\ln(\frac{A_s}{\tan\beta})$ = the stream power index and the wetness index, respectively.

The potential locations of ephemeral gullies may be predicted when both of the above two conditions are satisfied. The locations representing gullies were coded as 5 while those with no gullies were coded as 1. The results of the above equations were tested using locations of gullies collected during field surveys and there is a good agreement between the two, except that minor differences related to small ephemeral gullies were observed, mainly on the upslope parts of the catchments.

3.7 Producing maps of SERPUs

A simple GIS overlay was used to combine the above reclassified erosion potential maps due to differences in basic terrain attributes (Figure 2). During reclassifying and overlaying, the following limitations were noted: (1) that different weights were not given to the ranks of the different factors, i.e., rank 1 for factor 1 has equal weight to rank 1 for factor 2, and so on; (2) that the ranks were not field calibrated and do not show soil loss rates other than differences in potential soil erosion risks; (3) that the threshold values for the compound gully indices were not calibrated for conditions of the study areas. However, since the main interest was to identify landscape positions with different sensitivity to erosion and sediment delivery potential due to similarity in basic geomorphic and anthropogenic factors, the above assumptions may not have marked impact on the interpretation of results.

Bearing in mind the above weaknesses, the maps containing slope, LUC type, fraction of surface cover and lithology with classes from 1 to 5 were overlaid, resulting in a continuous map of runoff potential with values ranging from 4 – 20. Addition was used to combine the maps in order to keep the size of classes smaller and manageable. After the overlay was performed, each of the maps was reclassified into five categories of erosion potential using equal interval threshold values of 4-7 (very high), 8-10 (high), 11-13 (medium), 14-16 (low), 17-20 (very low). Each category was then coded from 1 to 5, higher values indicating locations with high potential erosion due to the prevalence/suitability of the runoff generating factors.

To derive the final SERPU for each catchment considering also sediment delivery potential, the five category erosion potential maps (class 1 to 5) were combined with the gully erosion potential maps (class 1 and 5) resulting in maps with continuous values ranging from 2 (very low category with no gully) to 10 (very high category with gully). These maps were finally reclassified into five categories of soil loss and delivery potential using threshold values of 2-3 (very low), 4 (low), 5-7 (medium), 8 (high), and 9-10 (very high) to produce the SERPUs. The final reclassification schemes were based on the possible combinations of processes observed. For instance, in conditions where (very) high erosion potential due to the four factors (Table 1) and gullies coincide, the new reclassification resulted in a (very) high category, and so on.

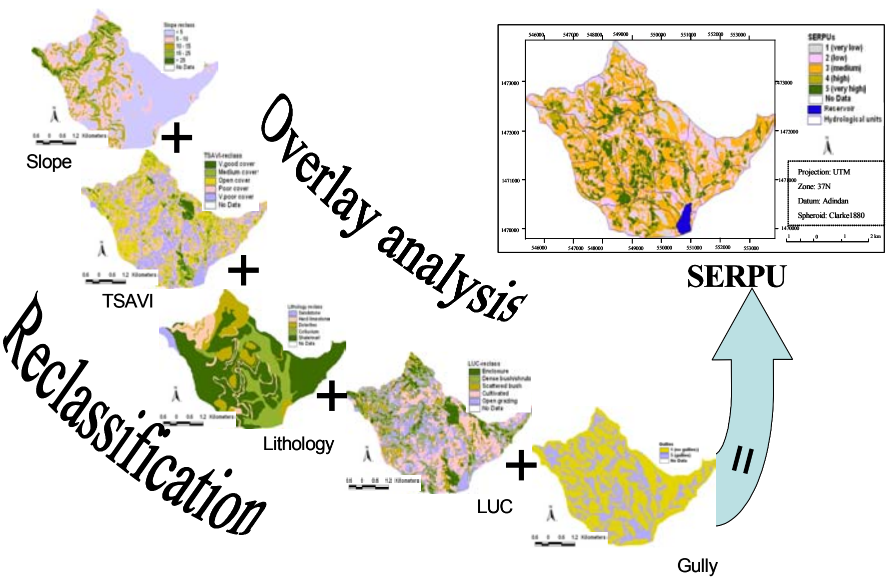


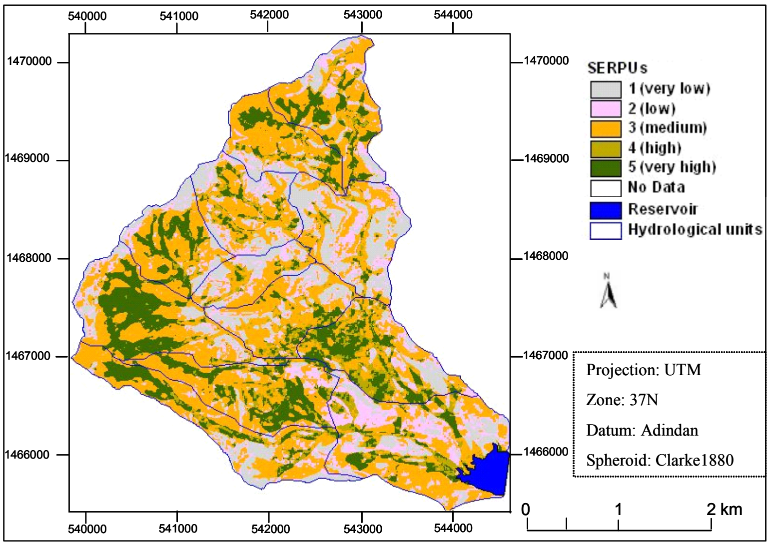
Figure 2. Reclassification and overlay steps conducted to generate SERPUs.

4 RESULTS AND DISCUSSION

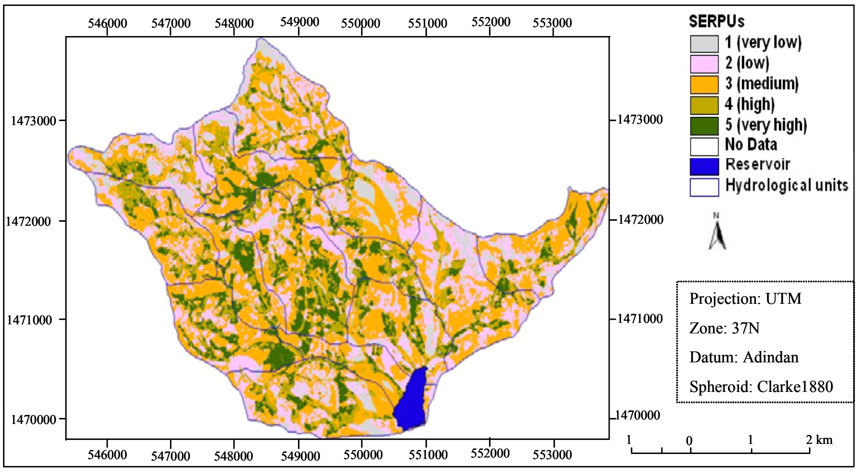
Figure 3 shows the final SERPU maps created for the two catchments. The SERPUs indicate the relative sensitivity of landscapes with respect to key parameters that affect erosion and provide rough information about the critical zones of soil loss and potential delivery. Areas with high values represent locations where sediment availability and transport capacity are not limiting. Such hotspot areas experiencing high erosion risk and sediment delivery potential could be prioritized for management intervention.

The areas experiencing (very) high erosion risk account for about 25% of the catchments while the (very) low ones cover about 30% of the catchments. When the extreme high and low soil loss and sediment delivery potential areas are compared, it can be observed that the former account over 15% while the later account about 10%. This may suggest that areas sensitive to erosion are more common than relatively stable ones in the two catchments. Generally, the landscape positions where steep-slope, poor surface cover, erodible lithology, and gully erosion coincided show high erosion risk compared to others. The interesting observation of the results is that steep slope areas do not necessarily represent high soil loss risk compared to gentle slope ones. This is because most of the very steep sites are either dominated by resistant lithology, which can be characterized as having high runoff but low soil material to be detached and transported and/or dense bush cover, which reduces soil loss and downstream delivery. On the other hand, the landscape positions characterized by (very) high soil loss risk and delivery potential correspond with observed gully and rill erosion where surface cover is poor and erodibility is high. Mostly, the low slope positions are intensively cultivated and overgrazed, leading to accelerated erosion and widespread gullies

[29]. In an environment where soil erosion is a serious problem and gullies are prominent features, the SERPUs could therefore serve as important sources of information to pinpoint locations where further studies and planning for conservation measures need to be prioritized.



(a)



(b)

Figure 3. Areas susceptible to different levels of erosion risk based on SERPUs for (a) Adikenafiz and (b) Gerebmihiz catchments in northern Ethiopia, Tigray.

To evaluate the accuracy of the SERPUs, field surveys were conducted to characterize landscape units into different categories of erosion and sediment yield potential based on evidence of erosion and degree of catchment connectivity [29]. The field surveys enabled the identification and evaluation of areas that are active sources of sediments and their delivery efficiency to adjacent stream channels and reservoirs. The landscape units assigned with ranks from 1-5 (1 representing highest soil loss and decreases further) based on field observation coincide well with corresponding erosion risk areas of the SERPUs. The agreements between the field survey results and SERPU maps were better for the high and low erosion risk areas compared to the medium ones. This means that the SERPUs were efficient in identifying areas characterized by high soil loss and sediment delivery potential and/or those which are less sensitive. This shows that the approach employed to generate the SERPUs could be used as an efficient method for the identification of hot-spot areas of erosion for planning relevant management interventions. However, adequate field calibration of the SERPUs factor reclassification schemes and proper verification of results should be a crucial step before results are used for intervention.

5 CONCLUSION

The results presented in this study demonstrate that areas of similar erosion risk (SERPUs) can be derived employing basic information on terrain, LUC, lithology, and gullies. The SERPUs enable the identification of hotspot areas of erosion that require appropriate conservation measures to reduce on-site soil loss and its off-site delivery. Generally, the SERPUs show that the potential sediment source areas are located within the proximity of gullies with poor surface cover and erodible lithology.

The advantage of the SERPUs could be that they do not demand complicated data and can be used to locate areas more vulnerable to erosion with minimum calibration. Their limitation could be that they are more of qualitative and slight manipulation of reclassification approaches could change the risk level of catchment subunits. Through calibrating the semi-qualitative scorings with field data and conducting detailed sensitivity analysis for the reclassification schemes, it may be possible to attach adequate physical meaning to the ranks (reclassified maps) of each factor and use the approach for prioritizing areas of intervention in more quantitative way.

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Using AVHRR satellite and herbivore time series data to model kangaroo dynamics at two scales in Australian rangelands

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ABSTRACT

Kangaroo populations in South Australian sheep rangelands have increased historically, and are being managed by hunters according to quota allocations. Predictive modelling of kangaroo abundance would help reduce the costly need for aerial surveys. A property-scale statistical model was developed in parallel with a regional model to compare the suitability of property-scale models with regional scale models. To overcome uncertainty about the most suitable forage surrogate data for the model, the forage surrogate values were validated against pasture photos ranked according to forage content. The kangaroo dynamics model at both scales performed best with a term measuring the minimum forage growth flush per season, and alternatively rainfall data. Model parameters and mapped model accuracy were also compared to look for characteristic processes at different scales. Contrary to expectation the property-scale model did not perform better than the regional model, and several reasons and possible improvements were identified.

Keywords: Rangelands, herbivore dynamics, longterm time series, NDVI, scale.

1 RATIONALE

Land degradation in rangelands occurs as a result of overgrazing by a combination of domestic and native herbivores, its' extent being influenced by climate and rainfall and their effects upon vegetation and, in turn, herbivore dynamics [1]. Consequently understanding the dynamics of native kangaroo populations is a prerequisite for understanding and predicting their contribution to land degradation.

The primary objective of this research is to establish a modelling capacity of kangaroo population dynamics for the purpose of population management at fine resolution, using longterm time series of NDVI and herbivore data.

In Australia's sheep rangelands, kangaroo population density has increased substantially since pre-European times. Contributing factors are the increased provision of water for livestock, the exclusion of the main predator, the wild dog, and the modification of native vegetation to pasture for introduced livestock, which increased the availability of palatable green shoots for kangaroos [1, 2].

Kangaroo management aims to achieve a commercial harvest of kangaroos while bound to maintain viable kangaroo populations. For the South Australian sheep rangelands there are regional scale kangaroo models which provide generally robust descriptions of the grazing system and its current state at a regional level [3, 4]. However, some management decisions, particularly the issue of harvest quotas, are made at a property scale. Currently kangaroo management is based on the allocation of harvest quotas according to the dynamics of regional survey counts, with some local corrections.

Rangeland landscapes are spatially dynamic mosaics of soil and vegetation (woody and non-woody) resources shaped by edaphic, climatic and disturbance processes operating at a spectrum of spatial and temporal scales [5]. The pattern of precipitation and vegetation growth has a temporally and spatially high variance, causing increased local mortality at any time, and regional kangaroo population losses on a decadal scale [6]. The observation of habitat-precipitation interactions and temporally variable habitat associations [7] also indicates the need for consideration of spatial effects to improve predictability of kangaroo dynamics. Large herbivores interact with the spatially and seasonally dynamic forage resources at several levels of ecological organisation [8].

Spatially explicit population models at a diversity of scales are expected to increase the ability to accurately model spatially and temporally complex landscapes [9]. Particularly for rangelands it is therefore to be expected that models at a finer scale will represent important processes working at that scale, starting with the effects of the spatio-temporal heterogeneity on populations at fixed locations, but potentially also providing the prerequisite to model other aspects of kangaroo dynamics, particularly nomadic movement.

1.1 Objectives

- The most suitable forage surrogate variables for modelling kangaroo population dynamics will be selected by comparison with a time series of ranked photo records from permanent monitoring plots.
- A regional-scale, and a property-scale kangaroo model will be developed using the forage surrogate variables.
- Accuracy, structure and utility of the property scale model will be compared to the regional scale model.

Initially the models will be developed for red kangaroos only, which occur throughout the study area. The higher resolution model will also serve the purpose to allow more detailed analysis of spatial and temporal dynamics, which cannot be researched using coarse-scale regional models.

2 APPROACH

2.1 Study area

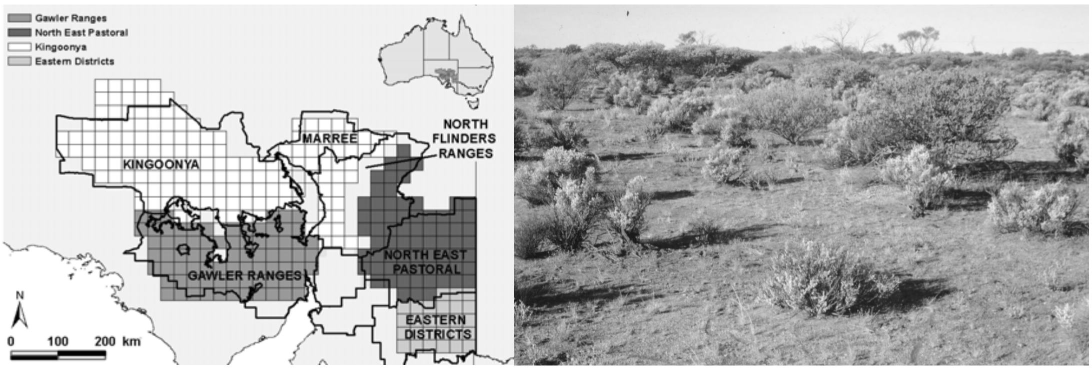


Figure 1. a) The stratification of the survey area was based on the current soil conservation districts (bold borders), but the North Flinders Ranges were split between North East Pastoral and Kingoonya, and Marree was included with Kingoonya. The extent of the modelling grid is determined by the extent of consistently surveyed landscape. b) The vegetation cover of the sheep rangelands of South Australia is low and highly variable.

The study area (figure 1a) receives between 150 mm at the northern extreme, and 450 mm rainfall in the south and in proximity of the Flinders Ranges. Consequently the vegetation cover is sparse (figure 1b), and also highly variable. Apart from the Flinders ranges, which were excluded from the survey due to survey limitations, as well as difference in landscape character, topographic relief is low. As this study focuses on population dynamics, and their dependency on the dynamic variables forage, competition from sheep, and harvest pressure, static habitat variables were only considered to support or modify the stratification of the study area. The patterning of vegetation greenness data supports a stratification similar to existing soil conservation districts [10], but the North Flinders Ranges district was split and added to the adjacent districts to the east and west, thus creating more homogeneous as well as contiguous strata.

2.2 Model data set

The kangaroo population in the sheep rangelands of South Australia has been surveyed annually, since 1978 [11]. The aerial survey follows east-west line transects at 25 km spacing. The survey cells are 5 km long and 400 m wide.

Both Advanced Very High Resolution Radiometer (AVHRR) normalised differential vegetation index (NDVI) [12] data and rainfall data were used as a surrogate for forage data. Rainfall data was interpolated from point records using inverse distance weighting. AVHRR NDVI images have been available since 1981. The data used in this study was available as monthly maximum value composites of the normalized difference vegetation index (NDVI) [13]. The data consists of two combined time series, one from the NOAA global area coverage, the second fully processed by the ERIN unit of Environment Australia. The available data resolution was 5 km. The green flush, or integral under the NDVI curve over time minus the annual base level, is an established measure of seasonal pasture production [14]. The value was calculated for six-monthly reference periods, and several variations according to biological patterns of the kangaroo population dynamics were compared. The reference period was generally the driest time of the year, which is most highly correlated with the rate of change of the red kangaroo population over the following year.

During data exploration it was found that neither the original NDVI values, nor the green flush, showed any useful correlation with our target variable of kangaroo rate of change. Closer examination of the NDVI time series showed frequent occurrence of cloud-contaminated pixels, which were filtered out. Additionally a new NDVI background was calculated, as a smoothed minimum determined by a three-year minimum moving window. The objective of NDVI transformation is to separate the dynamics of ephemeral and woody vegetation from NDVI time series [15]. The approach here was chosen because of its robustness, suitable even in the absence of regular seasonal rhythms.

The correlation between herbivore population dynamics and NDVI data, required for any population studies based on NDVI data, or any other forage surrogate data, was found to be weak, so that the selection of forage surrogate variables for the population model was, in the absence of quantitative pasture monitoring data for the South Australian sheep rangelands, carried out using a time series of pasture photos. Photo time series, of about 20 images per photo monitoring point, for 20 monitoring points per property, were available for two properties of contrasting soil and vegetation conditions (Bulgunnia and Mulyungarie), for the years from 1992 to 2002, for a range of seasons. The forage bulk on the photos was ranked for each photo point, and the ranks correlated with the alternative, corresponding forage surrogate variables. Building on the findings of the photo record analysis, ecological reasoning relevant for kangaroo population dynamics was applied to the variable generation and selection. The variable measurement period was extended, to cover the driest time of year with the most severe seasonal forage deficits, which is most likely to affect kangaroo population dynamics.

Annual domestic livestock data, and kangaroo harvest records, were converted from property records. Both were redistributed to the modelling grid in annual time steps. Harvest data was converted to harvest rates, normalised by the geometric mean of kangaroo density. This posed the problem of occasional zero observations, and additionally noise due to the stochastic nature of kangaroo survey counts. The most effective harvest rate variable was calculated at 100 km resolution, to reduce the temporal (and spatial) noise.

Geostatistical analysis [10], input data resolution from property scale between 5 and 100 km, to 5 km for survey and NDVI data, suggested a modelling scale of 25 km as a trade-off between input data resolution, avoiding autocorrelation and improving statistical independence of data points, and intended model application at the scale of large properties. A lower model resolution would be affected by a lack of environmental homogeneity per pixel, due to the irregular shape of the study area, and the dissection of the landscape by the Flinders Range and several salt lakes.

2.3 Ricker model

The effects of intra-specific competition, forage, domestic herbivores, and harvesting on the population dynamics of red kangaroos at a property scale (25km), and at regional scale, were analysed using a spatially explicit regional population dynamics model [3].

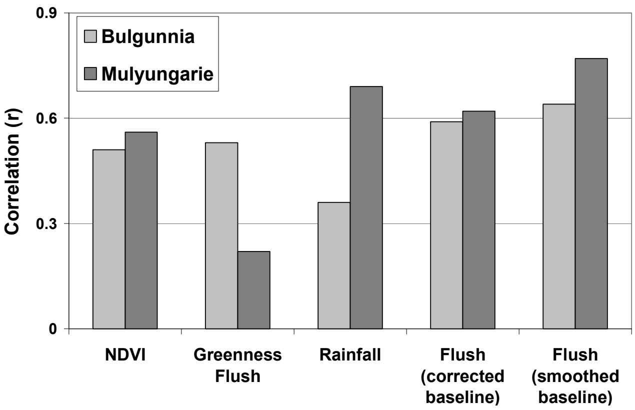
$$N_{j,(t)} = N_{j,(t-1)} e^{(a_i + b_i Y_{j,(t-1)} + c_i V_{j,(t-1)} + d_i S_{j,(t-1)} - H_{j,(t)} + E_{j,(t)})} \tag{1}$$

Equation 1 shows the Ricker population dynamics model, with population density N, regional indices i, spatial indices j, and parameters and terms for the intersect, scaled population density Y, forage surrogate data V, livestock S and harvest H. The t stands for an annual time step. All terms in the exponential term are scaled to permit comparison of parameter weights. The parameters a-d will be kept regionally consistent for all cells in one region. For the regional model the effect of regionally independent and shared parameters were compared. The model will be evaluated using a Likelihood approach. The same model structure was essentially used at regional and at 25 km scales, with both forage surrogate data types. The models for both scales were based on the same sheep, harvest and abundance data. The model fits, and parameters determined by the models, were compared for both scales. The Akaike Information Criterion AICc was used to compare models at each scale, and Pearson’s correlation coefficient was used to compare models between scales. Correlation coefficients for the predicted and observed time series, and ratios of mean annual absolute residuals to annual mean densities for the best models at either scale were mapped to assess issues of spatial fit.

In dependence on the scale, or number of time series to be modelled, different considerations had to apply to data preparation and model structure. At regional scale, data preparation consisted of aggregation of the more detailed data. The main differences in model structure examined here were whether the parameters specific to each variable could be used jointly for all regions, or had to be specific to a region. At property scale, data preparation was more laborious. Zero values in population observations had to be avoided for calculation of harvest ratio, and were replaced by a low non-zero value. Missing property-based data had to be imputed, to ensure maintenance of spatial contiguity. To maintain a predictive potential for the models, the harvest rate was calculated using currently reported harvest data and the geometric mean of previous years' density values.

3 FINDINGS

3.1 Selection of forage variables



The comparison of the correlation of forage surrogates (figure 2) shows clear differences. Initially the correlation with untransformed NDVI provided more consistent correlation than for the green flush calculation, and for rainfall data. Correcting the baseline for the flush calculation to eliminate cloud contamination, and subsequent use of the temporally smoothed baseline further improved the correlation of the green flush measures with forage bulk in the photo series. The fit also applied more consistently to both properties.

Figure 2. The chart shows a comparison of the correlation coefficients r between all compared forage surrogates and the photo ranks according to forage bulk.

3.2 Model selection regional scale

One best model for each forage surrogate was selected for each scale. Table 1 shows the regional models compared for this study. The minimum flush (min. flush) model uses the lowest growth flush of three consecutive 6-month periods beginning in December, thus providing a good measure of forage supply in the dry season. The flush (6m) uses the growth flush calculated for a fixed February to July window, and the rainfall data is for the months January to June. There was a clear advantage of using the NDVI variables in this comparison. For all forage surrogates the model using independent parameters for the intersect and forage performed better than the model for regionally shared, and for all parameters regionally independent. The growth flush data based on the NDVI is the best input data for regional models. For the regional models (figure 3) the NDVI flush model was found to have the highest fit between the regional observation and prediction, measured with Pearson's correlation coefficient, as well as the lowest AICc value, clearly identifying it as the preferred model.

Table 1. Model performance for 4 model structures and 3 forage variables. The * marks the best models according to the lowest Akaike information criterion, AICc, which generally coincided with the highest correlation coefficient r.

Model	r_{avg}	AICc
Min. flush model (regional parameters)	0.70	-77.4
*Min. flush model (individual parameters for forage and drift)	0.71	-87.8
Min. flush model (individual parameters)	0.70	-77.4
Flush(6m) model (regionally uniform parameters)	0.70	-82.9
Flush(6m) model (individual parameters for forage and drift)	0.70	-84.2
Flush(6m) model (individual parameters)	0.70	-74.9
Rainfall model (regionally uniform parameters)	0.66	-77.6
*Rainfall model (individual parameters for forage and drift)	0.68	-82.6
Rainfall model (individual parameters)	0.66	-69.5

Table 2. The best two regional and property-scale models used for the visual comparison of model fit, showing correlation patterns for all four regions.

Model	Fit	Eastern Districts	Gawler	North-west	North-east
Regional minimum flush model	r	0.67	0.72	0.64	0.82
	ravg	0.71			
	AICc	-87.84			
Regional rainfall model	r	0.71	0.58	0.61	0.82
	ravg	0.68			
	AICc	-82.57			
Property-scale flush model	r	0.67	0.59	0.68	0.84
	ravg	0.70			
	AICc	2748	5715	12354	3186
Property-scale rainfall model	r	0.78	0.56	0.70	0.81
	ravg	0.71			
	AICc	2764	5730	12486	3149

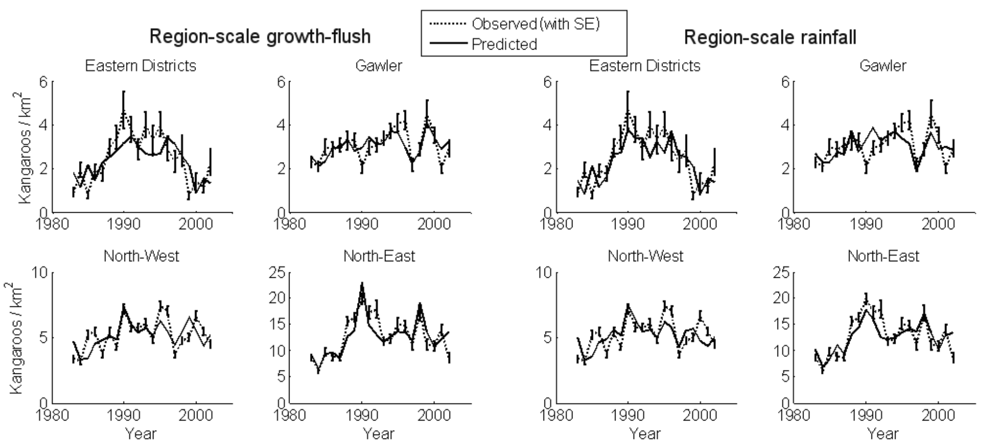


Figure 3. The two best regional kangaroo dynamics models using green flush, and rainfall, for the four regions of the study area. The lines for observed and predicted refer to the mean values for each region. The model fit clearly varies from region to region, and between models, and falls outside of some of the standard error bars for the regional survey counts (vertical bars).

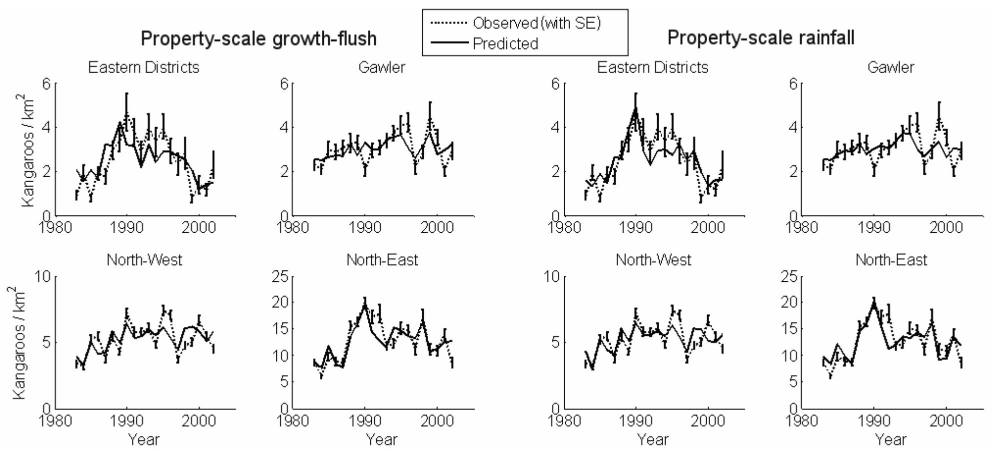


Figure 4. a) The best property scale flush model, using the minimum flush variable. b) The property scale models for 6-monthly rainfall and green flush perform similarly. Both models have numerically and visually similar fit.

3.3 Model selection property-scale

For the property-scale model the differences between rain and NDVI models were not consistent for all regions, and overall differences were negligible (figure 5b). The difference between property-scale models using untransformed data was negligible. In terms of model fit there is no clear choice between rainfall and NDVI flush as input data.

Table 3. Comparison of parameters between models.

Model		Intersect	Abundance	Forage	Livestock
Regional, shared parameters between regions	flush	0.12	-0.26	0.11	0.14
	rain	0.12	-0.27	0.08	0.17
Regional, two shared parameters between regions	flush	0.13	-0.25	0.07	0.14
	rain	0.13	-0.27	0.07	0.16
Property-scale, independent parameters between regions	Flush	0.04	-0.93	0.07	0.09
	Rain	0.09	-0.94	0.14	0.09

The subset of models permits a comparison of parameters (table 3). The model was constructed without a parameter for the harvest variable, so it is assumed that the effect of harvesting is constant per population growth rate [3]. The most noteworthy difference between scales are the quadrupled parameter values for density dependence, but also the doubling of the parameter for rainfall data in the property–scale model.

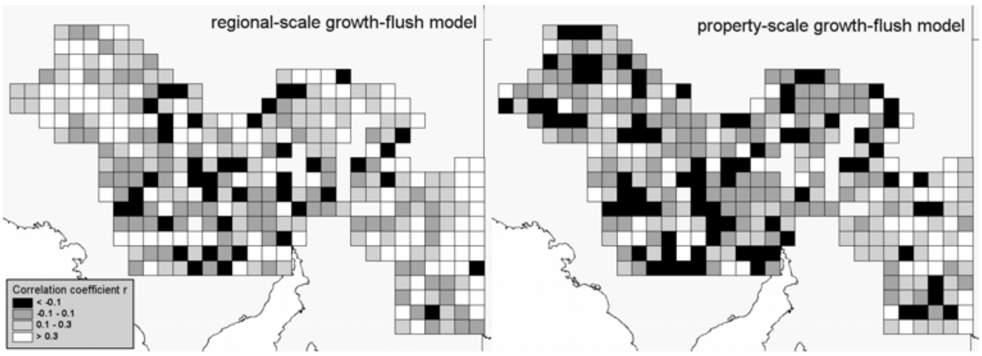


Figure 5. The panel shows the spatial distribution of the temporal correlation coefficients of the best green flush models for regional and property scale.

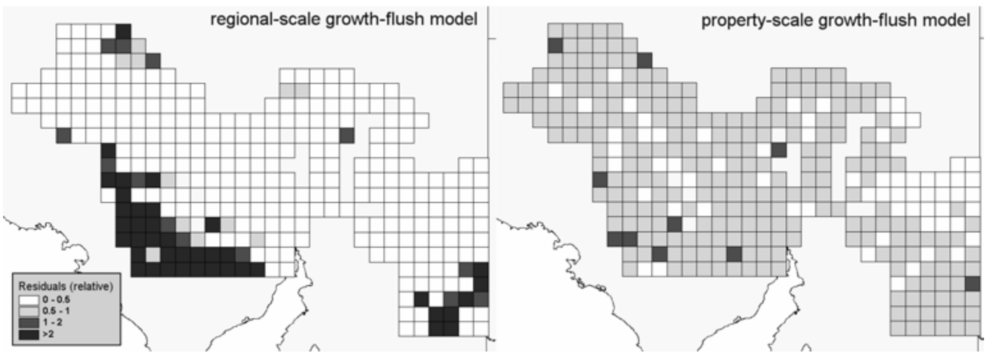


Figure 6. The panel shows the relative residuals of the best green flush models for regional and property scale. The regional model has clear difficulties coping with heterogeneities, particularly at the margins of the study area.

As shown in table 2, all models have region-specific strengths and weaknesses. The spatial comparison of the temporal correlation coefficients (figure 5) shows strong similarity between rainfall and green flush models. Modelling spatial detail with the property scale model makes the spatial model fit noisier. The average correlation coefficient for the regional map is 0.18, compared to 0.08 for the property-scale prediction. For comparison, the correlation of the aggregated observations and predictions for both scales is identical. The panel showing the residual ratios (figure 6) highlights a clear difference between the regional scale map and the property scale map. Using the property scale output will produce a pattern of prediction more representative of the heterogeneity of the study area.

4 DISCUSSION

It is likely that the pasture photo ranking chosen to validate is influenced by the bulk of the vegetation as much as the overall greenness. Kangaroos prefer green shoots and are therefore not attracted by dry forage bulk. However, this is as close as we could come to validating the time series of NDVI (and rainfall) data, as no forage sampling has been carried out for the study period. The most highly correlated variables from the photo series validation were then used as input variables for the model at both scales, finding that both the flush and rainfall variables were performing quite similarly. Based on the ecological rationale that a drought-adapted animal is most likely to be most severely affected by the severity of the dry season, it is not surprising that the calculation of the minimum 6-month flush in a nine-month window around summer and autumn provided the best green flush variable. Surprising is the good performance of rainfall data, which does not seem to be affected by the order of magnitude difference which exists in the rain gauge network density throughout the study area.

The regional model is more stable in its parameter selection, i.e. easier to parameterise, parameters are less noisy, and make more ecological sense. The data acquisition and preparation is much simpler. The property-scale model however required preparing the data to a reasonable level of spatial accuracy at that scale, and operators are confronted by issues of zero observations and low values, and spatial inaccuracies in the input data, particularly when referenced to properties with changing boundaries.

The best regional model is based on the minimum 6-month flush in a nine-month window, but the fit of the rainfall model is not much worse. The characteristics of the property-scale models are similar for both forage data types. The accuracy of the time series fit (using spatially aggregated data) is quite similar for regional and property scale, as is the visual fit of the time series plots. Once mapped on cell-by–cell basis, the average temporal correlation for the property scale model is lower than the regional model, suggesting that the data is very noisy. However, mapping residuals from the regional model demonstrates that the regional models are less suitable for modelling population dynamics in more heterogenous regions, when the property-scale model does produce more realistic residual to observation ratio values, which would be more recognisable by local landholders for instance.

The comparison of parameters shows that the forage surrogate data contributes less to the model than kangaroo density, or implied harvesting, but of similar magnitude to the livestock data. This helps to explain why both rainfall and green flush perform so relatively similarly at both scales. It would be desirable for forage to have a higher parameter loading into the model, to be able to predict droughts more effectively. The high loading for kangaroo density compared to the regional model, combined with the stochastic nature of kangaroo observations, makes it likely that increased density dependence is artificially introduced into the property-scale model, which would result in a noisier model, and could possibly be overcome with some form of data smoothing, both spatially and temporally. The aggregated regional performance of the property-scale model is the same as for the regional model, so it remains to be examined, what strategy of noise reduction, whether smoothing of population data, or smoothing of output, such as spatial resampling to a coarser scale, will provide less noisy predictions.

5 CONCLUSION AND FUTURE RESEARCH

The Ricker model predictions of kangaroo dynamics at both scales were very stable given the noisy nature of the rangeland data. Furthermore it was possible to build true predictive models for kangaroo dynamics, by only using data from previous time steps, with the exception of the routinely collected kangaroo harvest data. So it would be technically possible to replace expensive aerial surveys with modelled output from property-scale models.

This study provided a clear demonstration of the difficulties involved in finding a workable compromise between utilising the information content in spatially detailed data, without suffering the adverse consequences of the statistical noise in the data. The finding that forage variables do not have the highest relative contribution to the predictions provides a cautionary note: extreme population crashes are less likely to be predicted accurately due to the model exposure to noise in population data, and the relative stability of human-managed variables. The results of the property-scale model have been shown to be useful as a comparison to regional predictions.

Not all avenues to overcome problems with noisy input and output data have been exhausted, and the property-scale model is a starting point for future examination of spatial effects such as nomadic behaviour of kangaroos, or a more detailed examination of rangeland dynamics at property scale.

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Assessing rangeland degradation in heterogeneous Mediterranean environments. A case study in the County of Lagadas/Greece

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ABSTRACT

The county of Lagadas in northern Greece is typical for heterogeneous Mediterranean rangelands embedded within a patchwork of land use types. In front of the spatial configuration of the landscape, major changes in rural economies experienced in past years have caused competing demands of different stakeholders on the utilization of natural resources. Their sound management needs to incorporate both ecological and economic aspects, a spatially explicit assessment of the present state and detailed knowledge of relevant processes. In particular, the presence of land degradation effects calls for consideration of the temporal dimension. Given the availability of extensive local-scale remote sensing data archives, such as the Landsat-MSS7TM/ETM+, the required spatial and temporal coverage may be addressed. In the present study, temporal and spatial trends were analyzed using linear trend analysis techniques, and grazing pressure and its effects were investigated using a cost surface modeling approach. It was found that stability, degradation and regeneration of vegetation are present in close proximity and their spatial pattern was found to be largely determined by socio-economic factors. Most importantly, the different interpretation approaches were found to contribute to a general understanding of the various feedback loops operating in the watershed of Mygdonia valley.

Keywords: Mediterranean rangelands, grazing, land degradation, time series analysis, .cost surface

1 INTRODUCTION

Many Northern Mediterranean countries experienced dramatic land use changes during the second half of the last century, which lead in many cases to an unstable state of ecosystems. In particular, large areas of Mediterranean rangelands are now affected from transitional processes that cause conflicts between past and present land uses or economic and ecological priorities. Heavy overgrazing in some parts, the accumulation of woody biomass triggered by the abandonment and undergrazing of rangelands in others are causing substantial management problems in preserving the cultural landscapes that have evolved in the Mediterranean over the past centuries. The area of the county of Lagadas in Northern Greece is characterized by a heterogeneous mosaic of a variety of ecosystems and land uses, which is typical for Mediterranean rangelands [1]. Surrounding the central Mygdonia valley with its two large freshwater lakes, agricultural areas are interwoven with rangelands. The latter are mainly used for livestock husbandry and vegetation is dominated by *Quercus coccifera* shrublands and grasslands. Like in many Mediterranean countries, Greece becoming a member of the European Union in 1981 significantly affected rural activities, and resulted in widespread land abandonment in less accessible rural areas, especially in the higher zones. While in some regions numbers of grazing animals dramatically increased as a result of the European Common Agricultural Policy, the communities of the Lagadas area show a more stable behavior. Nonetheless, grazing regimes have been reported to have changed from transhumance to more sedentary systems with a tendency towards intensification in more concentrated areas.

It has been shown that multi-temporal time series of satellite imagery can be utilized to quantitatively assess the development of vegetation cover with time Hill et al. [2] and Hostert et al. [3] have linked such temporal information with animal statistics to provide a large-scale assessment of grazing related degradation. This study follows different pathways to assess temporal and spatial trends in the rangelands of Lagadas, and aims at identifying major drivers behind these. A suite of interpretation approaches will be followed that address different spatial aggregation levels, with the overall goal to characterize major processes determining the state and function of the rangelands and their surroundings in the county of Lagadas.

2 DATA SETS AND PROCESSING

A long time-series of 15 Landsat-5 TM and Landsat-7 ETM+ data has been acquired for the test area, covering the years 1984 to 2000 with one image per year. These have been selected to represent the period of maximum photosynthetic activity in the rangeland. In addition, a late summer scene was acquired for 2000 to represent only woody vegetation. All scenes were supplied from Eurimage© as system-corrected products. A digital elevation model (DEM) at 30 m grid resolution was made available from Geoapikonisis Ltd., which had been derived from photogrammetric analysis of digital aerial photographs. Finally, a standard Quickbird image product from late summer 2003 for a smaller subset of the test area was available, consisting of four individual tiles from two dates.

The Landsat images were geometrically corrected to match the standard Greek reference system. First, a master image was referenced based on a map scale 1:50,000. Subsequently, large sets of ground control points were retrieved using a procedure based on correlation windows [4]. All images were corrected to sub-pixel accuracy and non-systematic distortions introduced by local relief conditions were accounted for using the DEM. In a next step, all scenes were fully radiometrically corrected. To correct for decaying sensor sensitivity, sensor calibration was based on a time-dependent function calculated from vicarious calibration experiments [5]. A modified 5S code [6] was employed to correct for atmospheric effects based on pre-defined standard atmospheres and aerosol scattering phase functions [7, 8]. Local variations of illumination conditions due to topography were specifically incorporated in the radiometric correction chain by characterizing the sun-surface-sensor constellation for each pixel and for the time of image acquisition using the DEM [9, 10]. A set of pseudo-invariant targets was used to iteratively parameterize the radiative transfer models and ensure the quantitative consistency of the data set. Spectral Mixture Analysis (SMA) was used to infer proportional cover of photosynthetically active green vegetation for each pixel and all scenes [11, 12]. Assuming that the spectral variance in an image may be represented by a limited set of reference surface types (spectral endmembers), a library of spectral reflectance measurements carried out in field and under laboratory conditions was used. With reference to the spectral contrast in the scenes and the spectral dimensionality of Landsat-TM/ETM+ [13], a variable four endmember model was set up, consisting of a spectrum for green vegetation, developed soil and the abundant gneiss bedrock. In addition, an artificial shade spectrum was introduced to account for albedo variations. In the adopted pixel-adaptive SMA approach, the vegetation and shade components were forced to be included in the model for each pixel, while the soil and bedrock spectrum were used alternatively. Finally, a shade normalization was carried out.

Using time series analysis techniques, spatial patterns of temporal trends may be derived from multi-temporal data sets. Such trends involve transient and cyclic components as deterministic elements, and memory effects as well as stochastic elements [14]. Given the available long-term time series, the transient component was calculated for each pixel by means of a linear regression. The temporal development of SMA-derived green vegetation cover for each pixel was characterized using the gain and offset of the resulting function and different statistical parameters, such as correlation coefficient, t-tested significance etc. As locations with stable vegetation cover may exhibit low variations and hence not show a significant trend, the root mean squared error (RMSE) was introduced to account for such cases. Furthermore, a threshold analysis was included in the time series analysis, allowing identifying dates of major changes in temporal development, which may for instance result from short-term disturbances, such as fires, clearings etc.

The Quickbird tiles were geometrically corrected using the sensor model the rationale polynomial coefficients supplied with each tile. The DEM was included in the correction process and sub-pixel accuracy was attained by incorporating a number of field-measured GPS positions. The resulting data set provided a geometric resolution of 0.6 m for the panchromatic band and 2.4 m for the multi-spectral bands.

From the DEM, major landscape mesoforms were inferred using a methodology proposed by del Barrio et al. [15]. This methodology was refined and adapted to conditions of the Lagadas test site. Different topographic variables were computed from the 30 m DEM: slope, profile and plan curvature, catchment size, wetness index, sediment transport index, distance to the nearest stream, and insolation factor. An unsupervised classification of the resulting matrix yielded 23 preliminary classes, which were aggregated to 10 superclasses using sequential agglomerative hierarchical non-overlapping (SAHN) cluster analysis. They were then related to landscape mesoforms or relief categories and the resulting data set was employed in different contexts.

A detailed habitat map was derived from the visual interpretation of digital aerial photographs acquired at a scale of 1:20,000 in 1980, which was complemented and updated by numerous field surveys. This vector data set provided information on major vegetation formations following the Natura 2000 habitat code, which was extended by structural information on the average cover of the different habitats [16].

In addition to the spatial data layers, animal census information for sheep, goats and cattle was available for the communities of Lagadas County at decadal intervals from 1961 to 1991 and for 2002.

3 TRENDS AT COMMUNITY SCALE

In a first analysis step, it was investigated whether the available data permit to identify degradation of vegetation on the aggregation level of communities. To do so, animal census information was used to calculate effective stocking rates for sheep and goats excluding agricultural areas for 1981, 1991 and 2002. During the time covered, stocking rates have changed in different directions and at different magnitudes. According to the last estimates, stocking rates are low in the communities adjacent to the Mediterranean Sea, and most of the elevated regions in the North-West and North are classified as ‘lightly grazed’ or properly grazed (0-0.5 and >0.5-1.5 animals/ha, respectively) according to Tsiouvaras et al [17] and Chouvardas & Papanastasis [18]. On the other hand, there is a dominance of communities where livestock grazing is expected to exceed the carrying capacity and which were rated ‘heavily grazed’ or even ‘very heavily grazed’ (>1.5-2.5 and >2.5 animals/ha, respectively).

Then, the linear function characterizing vegetation development for each pixel was employed to calculate vegetation cover for these dates. Doing so reduces the influence of phenological variations that might still be present in some parts despite careful selection of image acquisition dates. Then aggregated vegetation cover estimates (excluding agricultural areas) were related to the respective stocking rates for each community (fig. 1).

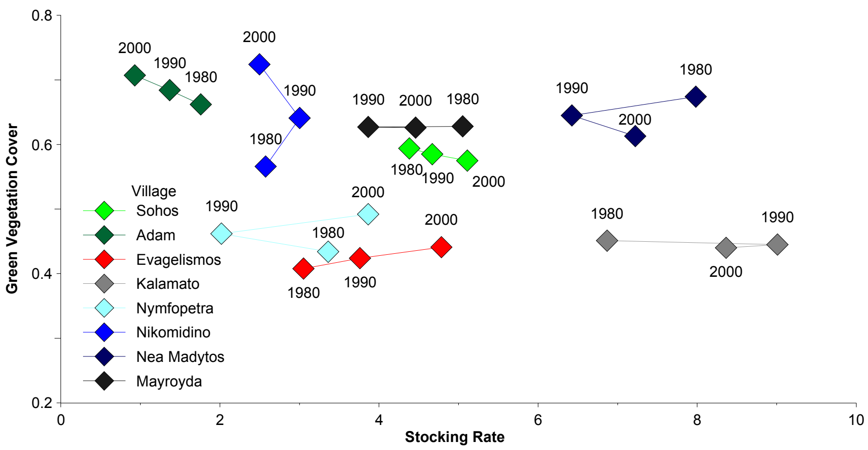

Figure 1. Temporal trajectories of the relation between vegetation cover and stocking rate at three decades. Vegetation cover estimates were derived using the linear trend function for each pixel.

Given the availability of both shrubs and herbaceous species, goats act as browsers with a preference for woody vegetation, while sheep act as grazers mostly living on herbaceous species. Flocks in the area are predominantly mixed and the cover estimates derived using this analysis were expected to show a continuous negative relation to stocking rates. In practice, this effect was only detected in 11 out of 47 considered cases (represented in fig. 1 by Sohos and Adam). To the contrary, 9 communities show a continuous positive relation, where increasing stocking rates relate to increasing vegetation cover (Evagelismos). Opposed to these examples, 19 of the remaining communities show discontinuous temporal trajectories, with a change in the direction of the relation between cover and stocking rates (Nymfopetra, Nikomidino, Nea Madytos). Finally, 8 communities exhibit stable vegetation cover regardless of changes in animal density (Kalamato, Mayroyda). When interpreting these trajectories, the general level of stocking rates needs to be considered. For instance, even a decrease is not expected to result in a recovery of vegetation cover if this decrease occurs above thresholds for proper grazing. The adverse effect may be observed if stocking rates increase on a low level. In most cases, however, it is suspected that with the changes in grazing schemes from transhumance to sedentary systems, a degree of ‘artificiality’ is introduced that is responsible for decoupling grazing pressure from generalized indicators, i.e. community-aggregated vegetation cover. Most of all, concentration processes appear to be the driving forces. With the establishment of a large number of sheds – frequently in the direct vicinity of villages – and shepherding becoming a daytime profession, animal grazing periods were reduced to some hours a day in close proximity of the sheds. Concerning nutrition, this is compensated for be the provision of feedstuffs in the sheds, which is to a large part sustained by products from intensive irrigated farming in the plain of Mygdonia valley. In the lower elevations adjacent to the main valley, this is aggravated by an increased accessibility that facilitates transportation of feedstuffs to the sheds. On the other hand, in communities where large areas are used for agriculture, only little generic grazing areas exist, translating in high stocking rates and preventing rotation grazing schemes to set aside rangelands for regeneration purposes. A further aspect is the general change in socio-economic frameworks, partly resulting from the political setting after Greece joined the European Union in 1981. In the test area this has led to migration to urban and touristic centers of Thessaloniki and the Chalkidiki peninsula, as well as a general ageing of the rural communities.

Concluding, an overall degradation of vegetation as a result of increasing animal numbers can not be generally stated. A variety of factors are affecting the rangeland vegetation through their determination of grazing impact. As a result of these, the community aggregation level does not seem appropriate to characterize grazing dynamics under the given grazing scheme, as the required direct relation is only maintained in a limited number of cases.

4 PIXEL BASED TREND ANALYSIS

The analysis at the community aggregation level suggests that a more spatially differentiated approach may be needed to resolve the spatial heterogeneity of processes in Mediterranean rangelands. In order to make full use of the time series of vegetation cover estimates, a linear trend analysis was carried out to infer temporal trends and different statistical parameters were calculated for each pixel. From an ecological point of view the same magnitude of change may have different ecological implications depending on the general level on which it manifests. Hence, a per-pixel degradation index was formulated similar to the work of Hostert et al. [19], which combines information on the direction of the trend and the associated change rate with information on the overall average vegetation cover estimate. Results show an intimate mixture of areas with positive and negative development as well as stable regions. Excluding settled and agricultural areas, the following class proportions were found (fig. 2).

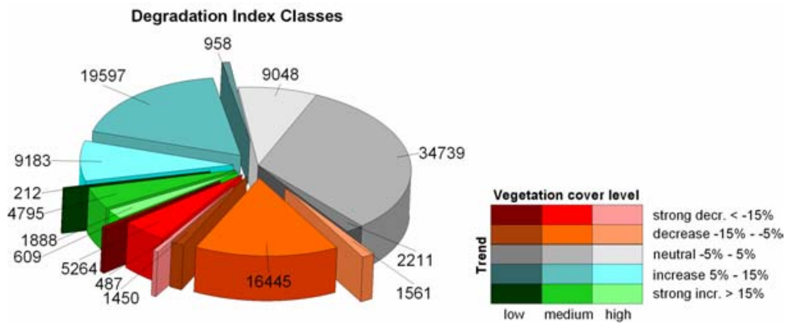

Figure 2. Distribution degradation index classes (pixel counts).

Less than half of the full area is showing stable conditions on medium to high cover levels. It is especially striking that on a generalized level, positive and negative trends appear almost balanced and that moderate trends dominated over strong trends. From a visual analysis of the map representation of the degradation index, different distinct patterns emerged, which were further analyzed for a set of target areas. Figure 3 shows target area 3, a major grazing area north of Lake Koronia.

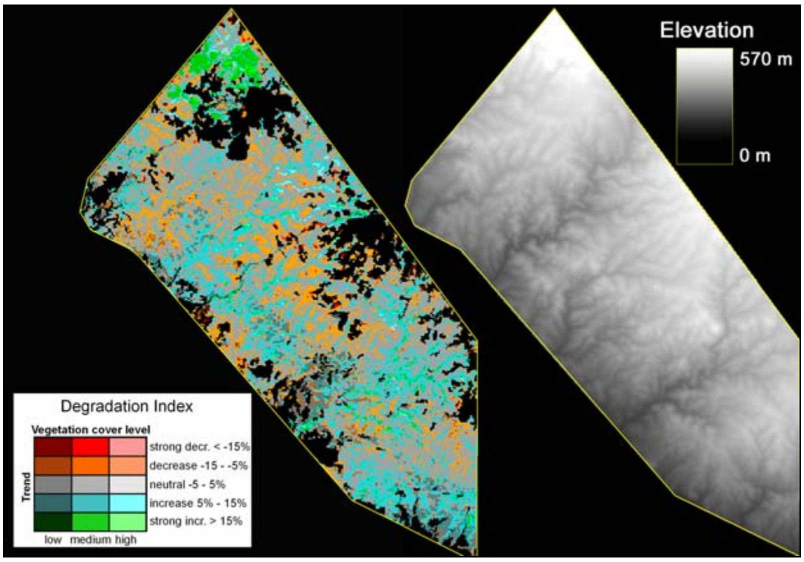

Figure 3. Degradation index (left); corresponding DEM-subset (right).

In the northernmost corner of this focus area strongly positive trends appear in a homogeneous patch that relates to afforestation measures. Apart from this, the degradation index image shows a matrix of pixels with neutral status mostly on a medium cover level. Embedded, a reticulate pattern of moderate to strongly positive trends is evident in the whole investigation area. The DEM suggests that these strands correspond to narrow valleys and gorges. On the other hand, areas with negative trends show a more patchy structure and are often located on plainer areas in between the reticulates. Both positive and negative developments are mostly on a moderate level although strongly positive trends do appear as well. Using data derived from the digital elevation model, such as elevation zones or information on landscape mesoforms, the influence of topography on this pattern was further investigated (fig. 4).

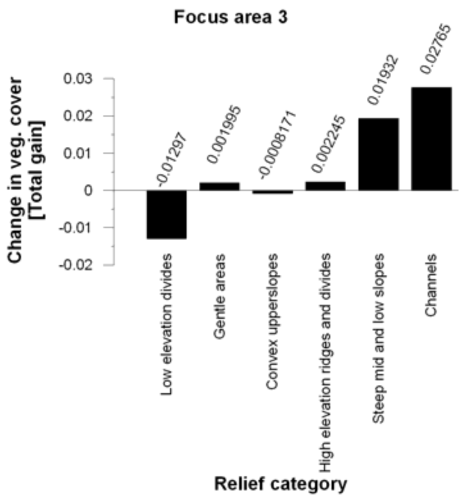

Figure 4. Average total gain from linear trend analysis for the relief categories in target area 3.

The 'steep mid and low slopes' and 'channels' classes differ significantly from the other classes in showing positive overall trend gain as well as degradation index values indicating a positive development. Apart from clearing for agricultural use, grazing is assumed to be the major factor responsible for shaping rangelands. Given the present, sedentary grazing scheme in Lagadas County, these results suggest that the obvious dependency of temporal trends on topography must be interpreted as a proxy for the accessibility of these different locations. This relates in particular to the fact that shepherding is now a daytime profession where animals are only grazed relatively close to their sheds for a limited time each day. In addition, the average age of shepherds was reported to increase as a result of urban migration processes. In front of this, in many locations grazing is confined to plain areas or gentle upper slopes that are easiest to reach, while steep areas are avoided by the shepherds. The resulting development of woody vegetation cover in less accessible areas in turn results in a thickening of the shrub canopy, thus even further deterring flocks from accessing these areas. This analysis highlights the potential of a spatially differentiated analysis of multi-temporal satellite imagery, especially if auxiliary data and process knowledge allow for a synoptic interpretation to identify major drivers behind apparent trends.

5 SPATIAL TRENDS IN RANGELANDS

In the previous sections, temporal trends in rangelands were identified based on the development of photosynthetic active vegetation cover. Complementing this temporal approach, a second interpretation strand focused on the investigation of spatial trends in rangelands resulting from grazing. A grazing-gradient approach proposed by Pickup & Chewings [20] for large and homogenous Australian rangelands was adapted to the heterogeneous conditions encountered in the test area. It was based on the concept of piospheres as a manifestation of the impact of a spot disturbance on the vegetation cover of a landscape [21, 22]. In semi-arid areas, where water is the limiting factor, watering holes were frequently employed to represent points of livestock concentration (PLC) as starting points for the analysis. In the case of the Lagadas rangelands, water supplies are abundant, such that animal sheds and gathering points are the starting points for livestock moving through the shrublands. Using Quickbird VHR data, these could be precisely located.

Common to most studies is the application of circular buffers around PLC to retrieve information on a target indicator. As this would imply a random distribution of animals around the PLC it is not suitable for the heterogeneous rangelands in the test area. Hence, a cost distance approach was implemented based on simulated friction surfaces to predict the spatial pattern of livestock roaming. In a first step, major factors determining grazing patterns as a result of the preferences of shepherds and their animals were identified with the help of local experts. In accordance with conclusions from the previous sections, four major factors were identified: distance, topography, attractiveness and accessibility. Using a set of functional hypotheses, these need to be 'translated' into regular grids of friction or cost values scaled to the same range. These represent the effort required for animals and shepherds to travel through these cells under consideration of a given factor [23].

The factor 'distance' incorporates knowledge about common walking distances of shepherds and their flocks. This implies that the probability of moving further depends on the distance already accumulated between their current position and the shed. An exponential function was applied to a distance grid calculated around the PLC. The function was scaled to reach a cost of 100 at a distance of 2000 m, but in this case higher costs were allowed as shepherds may still walk further. The factor 'topography' directly incorporates knowledge of shepherds' customs as discussed in the previous sections. This factor was modeled in a straightforward manner based on slope calculated from the DEM and steep areas were assigned highest costs.

While the first two factors are more related to the influence of shepherd decisions, the 'attractiveness' and 'accessibility' factors directly reflect animal's grazing behavior. Based on knowledge of the preferred diet of goats and sheep, information available through the habitat GIS could be translated into friction values, where lowest values indicate the highest nutritional value. Given the different demands of browsers and feeders discussed before, mixed rangelands with an open structure composed of shrubs and herbaceous vegetation are most attractive to mixed flocks. Hence, sparse shrublands were rated most attractive ahead of grasslands and recently abandoned fields, while beech and thermophilous oak forests were assigned highest friction values. Also, villages, barren fields or agricultural areas were assigned a maximum friction value of 100. Accessibility might be related to different parameters, such the accessibility of patches, the network of roads and tracks etc. In this study, 'accessibility' was used to model the openness of the landscape. It is exclusively related to the structural land surface type, accounting for the fact that animals are deterred by dense vegetation. Again, the necessary information was derived from the habitat GIS, leading to low costs for grasslands, which increase with increasing shrub cover and attain maximum values for agricultural fields and villages. Although these are masked from the gradient analysis, it is important to ensure they act as barriers in the model.

Once the four factors were spatially represented, an integrated friction surface was calculated by averaging using factor-specific weights based on questionnaires of grazing experts. Using the PLC as starting features, a pushbroom algorithm [23] was applied to calculate for each pixel the accumulated cost to reach the nearest PLC (fig. 5). The resulting data set can be considered a spatial representation of the cost of movement of shepherds and their animals. Hence, high accumulated costs correspond to a low probability of a location being grazed.

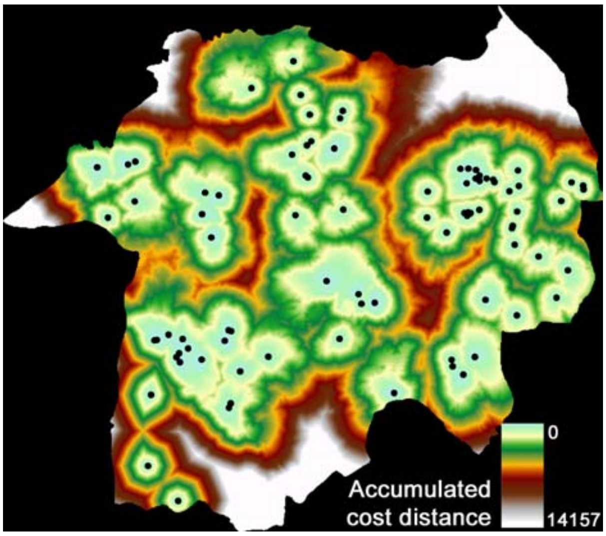

Figure 5. Accumulated cost distance surface based on integrated friction surface and calculated using a pushbroom algorithm starting from points of livestock concentration (PLC depicted as black dots).

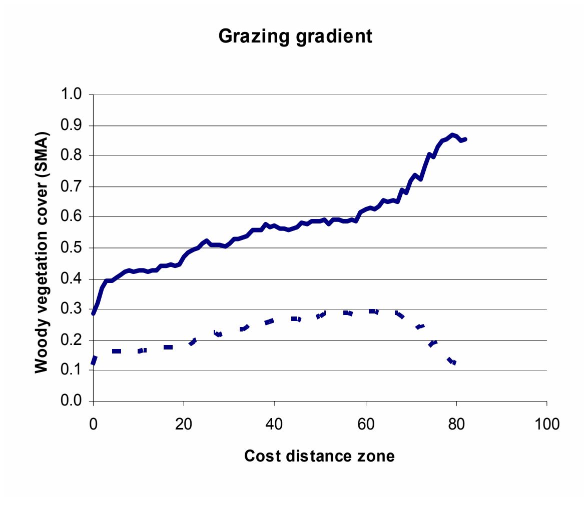

Figure 6. Gradient of woody vegetation cover (solid line) and standard deviation (dashed line) derived from piospheric analysis.

In a next step, a categorization using a threshold of 100 cost units allowed to convert this accumulated cost distance surface into discrete zones approximating the expected spatial distribution of livestock animals. These could be intersected with spatial information on the target indicator of grazing pressure to assess the presence of spatial trends. In this study, a late summer Landsat-5 TM image acquired in 2000 was used to infer information on green vegetation cover. Due to the acquisition date, it was presumed that only shrubs as an indicator of grazing are still photosynthetically active. After masking agricultural and urban areas, average cover, standard deviation and the number of pixels contained within a zone were extracted from this image for each buffer zone (fig. 6).

The result shows a distinct gradient of increasing cover estimates with increasing cost distance away from the PLC, from which different zones can be inferred. The first represents the 'sacrifice zone' [24] in the direct vicinity of the sheds where the effects of grazing are most prominent due to an almost permanent grazing pressure (zone 10). This pressure impacts the environment through trampling effects as well as the fact that grazing animals prefer young shoots and smaller plants. This effectively prevents regeneration of vegetation in the direct surroundings of sheds. Adjacent, the almost indifferent section corresponds to the major grazing area most frequently visited (zone 10-22). Here, a continuous increase in cover is noted. Grazing animals only manage to feed on shrubs until they reach a certain height and often graze compact shrub canopies from the outside. Hence, shrubs in this region frequently exhibit umbrella-like structures and the landscape often exhibits a high degree of fragmentation. It is also the zone from which shepherds still manage to return their flocks to the shed during the hot midday hours. The next section of the gradient corresponds to the areas that are more remote from sheds and/or difficult or unattractive to reach (zone 22-60). Consequently, the gradient shows again a strong increase in cover estimates directly related to decreasing grazing pressure. Especially at high altitudes, this absence of grazing may result in the development of dense *Quercus coccifera* shrublands or even the evolvement of open forests. This may for instance be the case in the previously discussed open forest zone in the Northeast of the pilot area (beyond zone 60).

6 CONCLUSIONS

The present study demonstrated the potential of integrating extensive multi-temporal remote sensing data sets with auxiliary spatial and non-spatial information and expert knowledge to characterize the impact of grazing on heterogeneous Mediterranean rangelands. Following a rigorous quantitative pre-processing scheme, different interpretation pathways were followed to assess the present state of rangelands and characterize ecosystem dynamics and processes that may lead to a degradation of resources. In the Lagadas rangelands, complex interactions could be observed resulting from a transition of traditional land utilization schemes to more intensified uses and sedentary grazing systems. Although no general degradation of resources was detected on the community aggregation level, a spatially differentiated analysis on a per-pixel basis revealed distinct patterns resulting from a combination of physical and socio-economic factors. As a consequence, stability, degradation and regeneration of vegetation operate in close proximity. This has widespread consequences, as the thickening of the shrub layer in the valleys ultimately reduces runoff towards the lakes. In addition, large quantities of water are being extracted for irrigated agriculture in Mygdonia valley, where different crops are produced to be used as additional feedstuffs for animals. As a consequence, the water table of Lake Koronia has been reported to be reduced by 90 % in the past years which severely affects the local fishery economy. This feedback loop underlines the complexity of present day utilization of Mediterranean rangelands interwoven with other land use types.

Cost distance modeling was used to model spatial dynamics determining livestock distribution and grazing behavior. Incorporating major driving factors for shepherd’s decisions and animal’s diet preferences, this approach could be used to infer a distinct spatial trend in dependency of sheds and gathering points as livestock concentration points. When interpreting the effect of livestock grazing on the landscape structure, it is important to note that due to the long history of human resource utilization in Mediterranean rangelands, there is a mutual determination of utilization schemes and landscape structure. For instance, the locations of sheds are to a large degree responsible for the grazing effects manifested in the grazing gradient, but of course the landscape structure does also partially affect the selection of shed locations. Based on the parameterization of the cost surface model, the approach is particularly suited for local grazing management, as it allows calculating multiple least-cost paths to simulate the effect of management interventions, such as the introduction of rotation schemes, prevention of grazing through fencing etc.

As a result of these analyses, a number of management recommendations were suggested to the local authorities. In the rangelands, it was suggested to advocate rotational grazing schemes of varying intensity and promote sparse to medium dense shrublands. This is in accordance with ecological objectives, as such mixed and open landscapes were shown to represent high floral and faunal diversity [16]. In addition, agricultural land use should be extensified to support a more balanced hydrological system and reduce nutrient input to the lake ecosystems. Obviously, the promotion of healthy rangelands in the Lagadas area would require adaptations of the local socio-economic framework, necessitating the provision of specific incentives for an extensive, sustainable resource utilization, which could be complemented by the establishment of alternative sources of income through an increased touristic use of the area.

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Prediction of desertification processes in the semi-arid zone of Israel: Integrating field data, remote sensing observations and GPS technologies in dynamic spatio-temporal models

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ABSTRACT

Desertification and land degradation in drylands occur due to a combination of drought and mismanagement of the land. The role of habitat preference by grazers in these processes is difficult to study because of the complexity involved with animal tracking and mapping of spatio-temporal variation in vegetation status. We addressed this problem by integrating remote sensing, GIS and GPS technologies. In this paper we assess the time spent by goat herds in different herbaceous plant habitats in a semi-arid environment. We use a predictive habitat distribution model which is spatially and temporally explicit, based on rules formulated using fuzzy logic. The results show that biomass availability together with a number of abiotic factors (slope decline and orientation, distance from the corral and topographic sub-slope units) could not fully explain the pathway of herds in the study area. However, the effect of these variables on the time spent by the herd in the different habitats was significant. Specifically, the time spent by the herd at a given location (grid cell) along the herding route was positively correlated with the biomass availability in the cell. Furthermore, the time spent by the herd in a cell responded non-linearly to the distance from the corral, being relatively low at near and far distances, and high at intermediate distances. The results imply that additional factors in these systems have a strong influence on the grazing habits. We suspect that one of the most notable among these factors is the effect of shepherd, which will be studied in future research.

Keywords: Ecosystem functioning, Semi-arid regions, GIS, Habitat prediction, Fuzzy logic, animal tracking, GPS.

1 INTRODUCTION

It has been widely agreed that desertification and land degradation in drylands occur due to a combination of drought, i.e., worsening in climate conditions, and human mismanagement of the land [1], [2]. Desertification may cause economic and societal damages as well as environmental damages due to changes in ecosystems functioning: studies have shown that variation in habitat conditions, rainfall distribution and grazing pressures cause disparity in vegetation production [3].

In the early days of statehood, fifty years ago, desertification was not a major problem in the semi-arid part of Israel and therefore was not of major public concern. However, in recent decades high intensity gully erosion both in agricultural areas and rangelands, mainly in the loess areas, may be a sign of emerging desertification and potential future risks [4]. Therefore, there is a need for reliable indicators as well as efficient monitoring systems to measure the magnitude and rate of desertification processes in this dramatically fluctuating climatic region. Geographic information technologies provide excellent infrastructure for: 1) observations through the use of remote sensing data for measuring vegetation and soil characteristics; 2) tracking of herd movements by GPS; and 3) dynamic spatio-temporal ecological and environmental modeling. The use of this framework for studies of ecosystem functioning as an indicator for desertification is particularly useful due to the dynamic nature of vegetation and the immediate response of ecosystems to land degradation and desertification.

Annual production of herbaceous vegetation plays a key role in understanding ecosystem functioning. In semi-arid regions this production responds significantly to spatial and temporal changes in rainfall amounts, soil moisture distribution and variation in grazing pressures. Therefore, semi-arid regions are characterized by a patchy pattern that has not yet been fully analyzed at the slope and watershed scales.

In this study, we demonstrate how an integrated use of remote sensing images, geographic information techniques, ecological modeling and tracking animals with DGPS, in a geo-computation framework, can serve for studying the effect of available biomass and abiotic factors on the grazing habits of goat herds.

2 THE STUDY AREA

The study area (figure 1) – Lehavim site ($31^0 20'$ N, $34^0 45'$ E) – is a Long-Term Ecological Research station and is located in the northern Negev with an average rainfall of 280 mm per annum. The mean annual temperature is 20.5^0C, with a maximum of 27.5^0C and a minimum of 12.5^0C (The Meteorological Survey of Israel). The terrain is hilly and the dominant rock formations are Eocenean limestone and chalk with patches of calcrete. Soils are brown lithosols and arid brown loess partly covered by a microphytic crust. Alluvial soils predominate in the wadi floors. The phytogeography of the region is Irano-Turanian, but Mediterranean and Saharo-Arabian species can also be found. Vegetation at the site is characterised by scattered dwarf shrubs and patches of herbaceous vegetation, mostly annual, which spread between rocks and dwarf shrubs. The dwarf-shrub community develops a steppe-like landscape with diffuse vegetation. The dwarf shrub community is dominated by *Sarcopoterium spinosum* (L.) Spach, *Corydothymus capitatus* (L.) Reichenb. and *Thymelaea hirsuta* (L.) Endl.. The herbaceous vegetation appears shortly after the first rains and persists as green forage for 3-4 months, depending on the amount and distribution of rainfall. The herbaceous vegetation is highly diverse, mostly composed of annual species that represent 56% of the regional flora. Forest plantations, natural reserves, and wheat fields surround the Lehavim site.

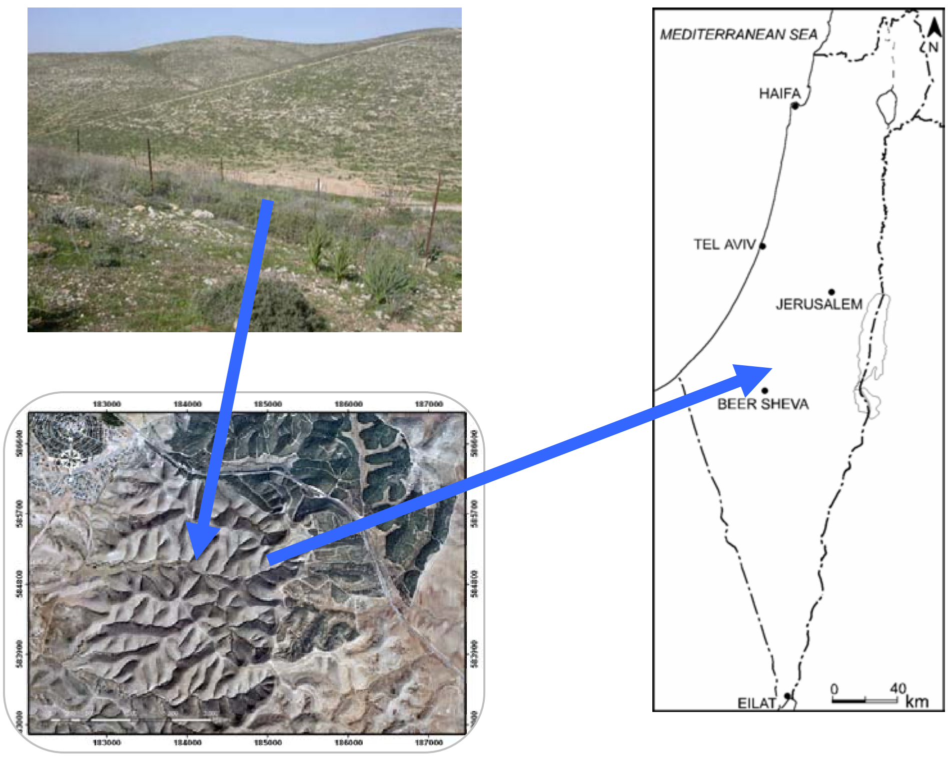

Figure 1. The study area – Lehavim LTER site.

3 METHODS

The geoinformatics scheme that we suggest (figure 2) includes the following four components: a) Remote sensing was used for data collection of rock coverage and for validation of model predictions using vegetation index layers; b) Process-based ecological modeling using fuzzy logic was developed to predict herbaceous vegetation production; c) environmental GIS database was developed to apply explicitly in space the fuzzy rules of the ecological model; and d) the time spent in each grid cell by goat and sheep herds was monitored with Differential GPS (D-GPS). The following sections detail the role of each component of the scheme.

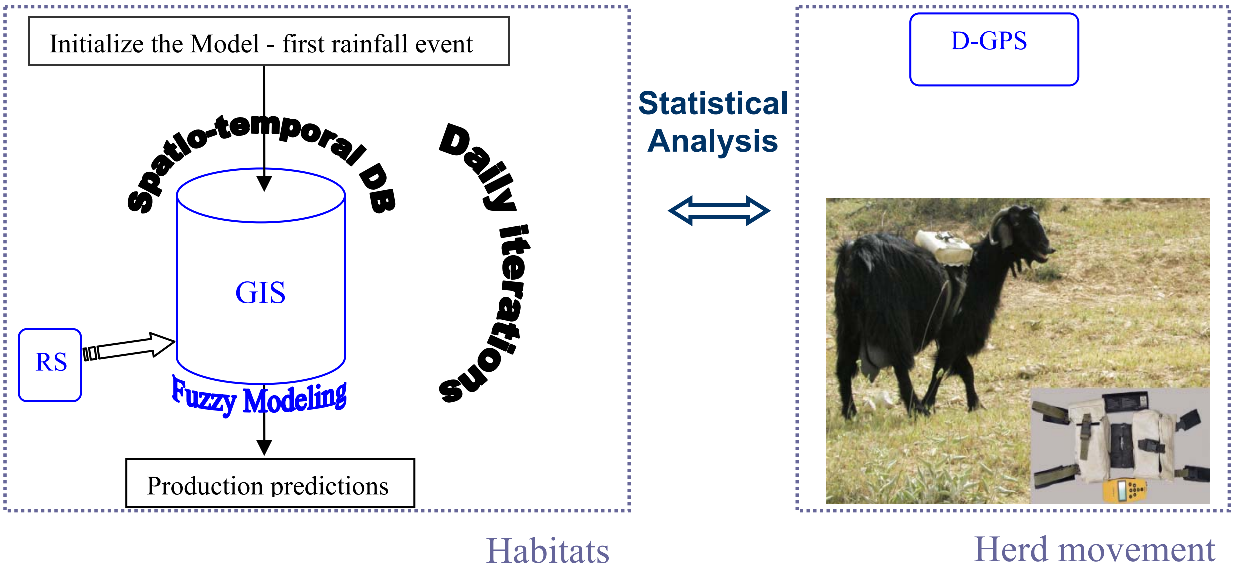

Figure 2. The geoinformatics scheme.

Remote sensing

High resolution (1.25 x 1.25m pixel resolution) aerial photos were used to classify three land cover categories: rock and stone; bare soil; and vegetation. We used maximum likelihood classification and similarly to previous studies, confusion matrices show very good correlation between image processing results and field measurements (>90%). Next, the frequencies of rock class were calculated to each cell of 25X25m fishnet that covered the entire study area. To calculate the recent version of the Normalized Difference Vegetation Index based on [5], 4-meters resolution IKONOS data from winter 2003 was used. All images were geometrically corrected to the Israel New Grid with root mean square error values of less than one pixel.

Ecological modeling and GIS database

The ecological model used in this study is described in detail in [6] and only the key principles of the model are presented here.

The quality of habitats in the study area is assessed based on their potential to produce herbaceous vegetation biomass. This potential may vary with time – due to variation in climatic conditions – and with space – due to variation in environmental and biotic conditions. We assume that in Mediterranean, semi-arid and other water-constrained environments, water availability and temperature play a crucial role in determining herbaceous vegetation productivity and, therefore, the model is formulated based on the hypothesized water requirements of herbaceous vegetation in the study area. The model has done so through formulating spatial and temporal variation in four indirect variables: rock cover, radiation, runoff (Dynamic TOPMODEL) and soil catena sub-slope units (figure 3). (Presumably rock cover and catena do not change over time). In addition, we added to the model three climatic variables: temperature, evaporation and rainfall amount. These are all measured on a daily basis from standard meteorological stations of the Israeli Meteorological Survey.

The radiation, sub-slope units and runoff were predicted using digital elevation data with 25X25m cell resolution, while soil characteristics, extracted from a field survey, were also combined in the runoff model. Rock and soil surfaces differ significantly in their response to rainfall and therefore, dynamics of the saturated zone cannot always be approximated by quasi-steady state representation. We divided the study areas into eight sub-slope units and assigned soil properties from soil survey and from literature to each of the sub-slope units along the catena. For calculating soil water deficit, the cells in the study area were divided into sink and source units. Solar radiation in the study areas was predicted based on three DEM-based variables: slope orientation and decline and hillshade; and the sun altitude and azimuth.

The database was integrated to apply the rules of water requirements through the use of fuzzy logic. Fuzzy logic is a theory in formal mathematics that enables arriving at a definitive solution for problems that are complex, uncertain and unstructured. Therefore, it is a most suitable modeling platform for hydrological and ecological phenomena [7]. The fuzzy logic production model allows prediction of two processes in the life span of the herbaceous plant: 1) germination - modeled in daily iterations until the conditions are satisfied; 2) primary production - modeled from germination until the end of the rainy season.

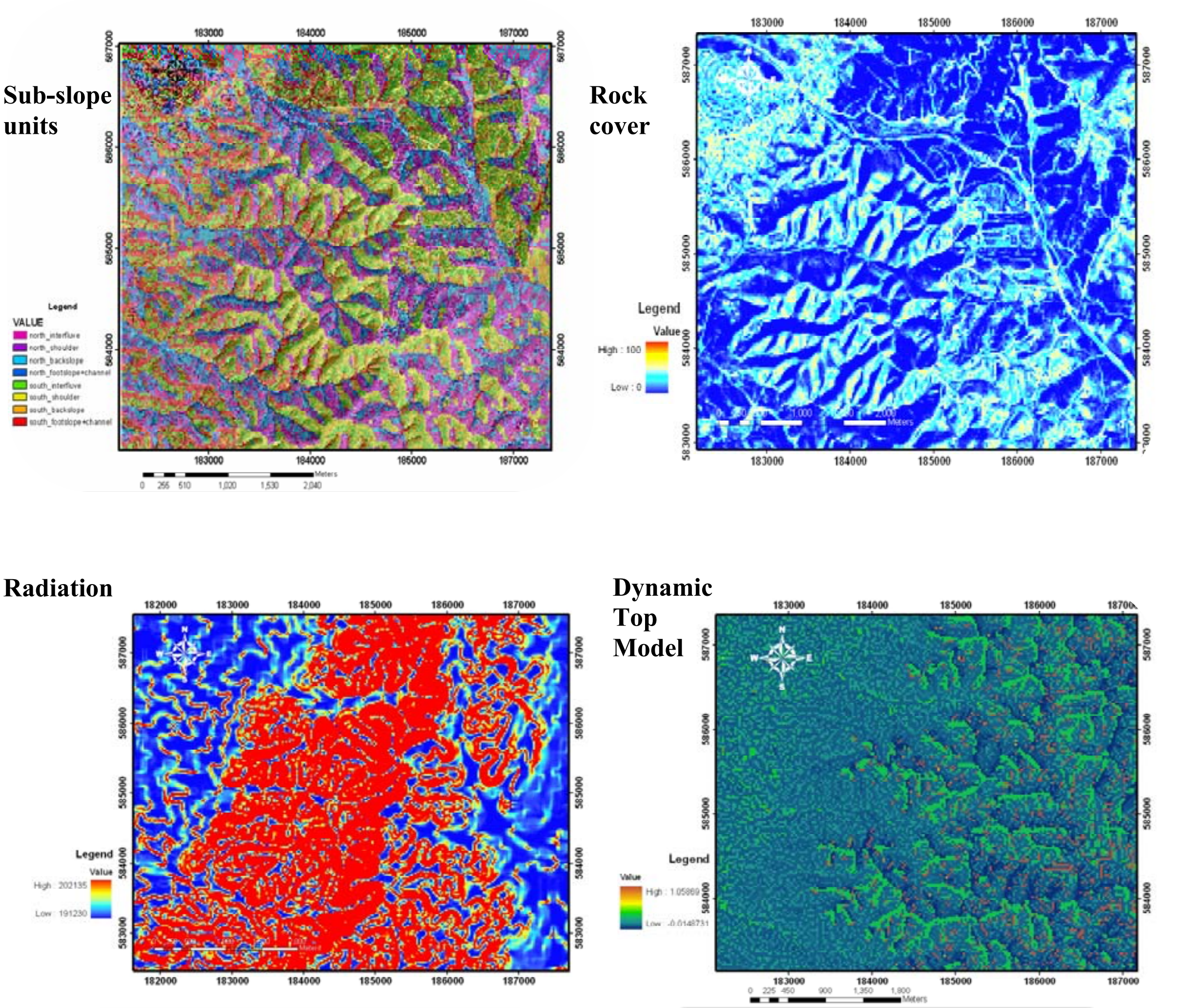

Figure 3. The GIS database layers as applied to the Lehavim study area.

Herd movement

The use of GPS for testing the effect of grazers habits on vegetative landscape is an emerging field [8]. Nevertheless, we could not find in the literature studies that actually tested the effect of herds of sheep and goats on herbaceous vegetation production in semi-arid areas and thus on desertification and land degradation.

The GPS analyses done here sought to represent spatial variation in grazing pressures. The study site is populated by two herds: a herd of sheep (400 animals) and a herd of goats (200 animals) which are shepherded across the landscape, with a fixed night corral and watering point. The herding routes were tracked on 78 days in the green season of 2003 (Feb to May), using a GPS rover unit (Trimble GEII Explorer) harnessed to one goat in the herd, and scheduled to record a position every 15 sec. The routes were overlaid on GIS raster layers containing data on vegetation production, at a resolution of 25x25 m per cell. For each GPS location, 25 min of animal presence was accrued to each of the 8 closest raster grid cells, based on number of animals and the estimated area occupied by the stationary herd. Statistical analyses of observed versus expected distributions are planned using filtered data sets to reduce autocorrelation.

Statistical Analysis

We have used multiple regression analysis to examine the relationship between: available biomass, distance to the corral, slope decline and orientation, sub-slope units and the speed of the herds in each cell. The speed of the herd movement in the cells represents the herd initiative to spend time at the cells. Data was analyzed using the software package JMP version 5.1 (SAS Institute Inc., Cary, NC).

4 RESULTS

The production model was executed and tested against actual field measurements (using the harvest method) acquired during three years 2003-2005. The results show good agreement and the coefficients of determination of the correlations between predicted and measured values vary between 0.5 to 0.74 with $p < 0.001$ in all cases.

The total area of the Lehavim Bedouin demonstration farm is approximately 5 Km^2. Within this area the actual pathway length of the goat herd was 14 Km while the actual pathway length of the sheep herd was 40 Km.

Figure 4 shows the pathways of sheep and goat herds in the study area during the growing season of 2003.

The figure shows that both goats and sheep cover large parts of the study area and therefore most cells of the study area are visited by either sheep or goats. However the question that arises is how much time the herd spends in each cell (and thus what are the cells that experience higher grazing pressures). We tested statistically the relationship between available biomass as predicted by the model for the dates: 25/Feb/2003-03/March/2003 and speed of the goat herd in each cell. Table 1 shows that the effect of the abiotic factors on the speed of the herd was found significant or near significant in all cases.

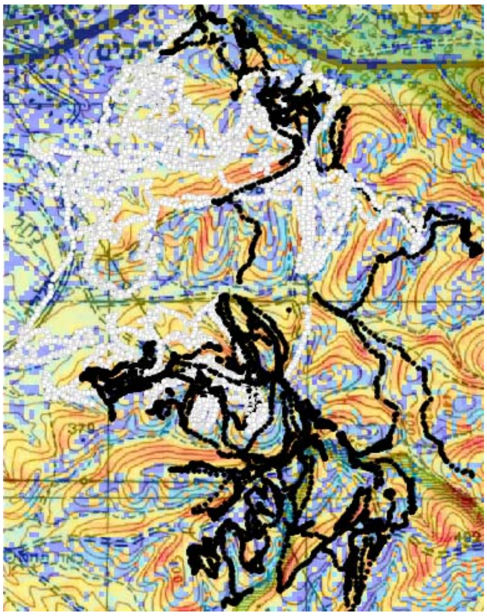

Figure 4. Annual routes of sheep (white dots) and goats (black dots) annotated on the background of the biomass model output from March 2003 (reddish colours represent low biomass potential while bluish colours represent cells of high biomass). For visual interpretation purposes, the biomass layer is overlaid, with 30% transparency, on a topographic map.

Table 1. Levels of significance of the six environmental variables in a multiple regression analysis of speed of herd movement in the cells of the study area. The available biomass is the model prediction for mid winter 2003.

Variable	*P* value
Available biomass	0.0616
Distance to corral (linear)	< 0.0001
Distance to corral (quadratic)	< 0.0001
Aspect	< 0.0001
Slope	0.0609
Sub-slope units	0.0248

Furthermore, when the number of variables in the model was reduced to available biomass and distance to corral, the level of significance of available biomass increased (table 2).

Table 2. Levels of significance and regression coefficients of the three independent variables in a multiple linear regression analysis of the speed of herd movement. The available biomass is the model prediction for mid winter 2003.

Variable	*P* value	Coefficient
Available biomass	0.0003	-0.006004
Distance to corral (linear)	< 0.0001	-0.000195
Distance to corral (quadratic)	< 0.0001	9.41e-7

The negative coefficient for Available biomass means that the herd moves more slowly in cells in habitats of higher quality i.e., the herd spent significantly more time in areas of high available biomass and vice versa. The response to Distance to corral was curvilinear: speed of herd movement declined with increasing Distance to a minimum at approximately 1100 m, and increased thereafter.

5 CONCLUSIONS

The integration of remote sensing, ecological modeling, GIS and GPS technologies in a geo-informatics scheme allows mapping and analyzing grazing pressure imposed by a herd of goats and sheep during the grazing season. The use of the geo-informatics scheme supported the hypothesis that biomass availability, distance from the corral and slope decline and orientation all significantly affect the time spent by sheep herd in each cell of a semi-arid study area. As expected, the herd spent significantly more time in cells of higher biomass. The distance from the corral was also found to be an important factor: both in the vicinity of the corral and in the remoter areas the herd tended to spend less time in each cell along its herding route (higher speed), while there was a tendency for greater residence times (lower speed) in cells located at intermediate distances from the corral. Future development of the geo-informatic framework shown here will allow to locate cells that suffer higher grazing pressures and to estimate if they suffer deterioration in primary production. Such an analysis will help to locate areas of higher desertification risks. Further study will also need to be done on the effect of the shepherd on the time spent by the herd in each cell. We suspect that a large part of the variation that could not be explained by the variables that we studied may be attributed to the preferences of the shepherd.

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Erosion risk assessment with readily available data in the East African Highlands

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ABSTRACT

Effective soil and water conservation strategies in the East African Highlands require the concentration of resources on limited spatial areas. An important criterion for selecting such areas is the extent of erosion. Limited data availability in these regions complicate a regional scale erosion assessment. This study was conducted to assess the potential of accurately mapping erosion risk for a 70-km^2 area in northeast Tanzania with readily available data. Slope was derived from an SRTM DEM and NDVI was calculated from a Landsat image. Both slope and NDVI were automatically divided in five classes of equal occurrence and were combined using a minimum operator to construct the final erosion risk map. The accuracy of the map was determined with qualitative field estimates of erosion risk, and was 81 % when allowing a one-class difference. High and low erosion risk areas could therefore be properly identified by the simple method using only readily available data. Possibilities to apply the method to other regions within the East African Highlands exist.

Keywords: Soil erosion; soil and water conservation; erosion risk mapping; qualitative data integration; remote sensing; regional scale; East African Highlands.

1 INTRODUCTION

Sustainable agricultural production is negatively affected by soil erosion in the East African Highlands [1]. To maintain production levels of subsistence and cash crops, it is required to control erosion effectively within this region. Conventional intervention was achieved through a top-down approach in which farmers were expected to adopt pre-defined measures of erosion control. Due to limited success, national soil and water conservation programs increasingly realized the need for the participation of farmers. To achieve increased participation, several East African countries used the Catchment Approach [2], in which a conservation plan is drawn up together with farmers for limited focal areas (100-200 ha). Although the Catchment Approach has proven to be more effective than conventional approaches, substantial time and dedication of extension staff is required. Due to limitations of personnel and money of the involved institutions, the participatory approach can only be applied to a limited amount of focal areas. However, selected areas can have a demonstration effect for a larger region. Focal area selection is thus of high importance. Partly the selection criteria depend on social factors, like the presence of a capable village leader and local requests for support. Nevertheless, a criterion of high importance is the extent of erosion.

Therefore, spatial maps showing the extent of erosion are required at a spatial scale ranging from 50 to 10.000 km^2. These maps can assist in prioritizing soil and water conservation efforts, and consequently direct the application of the Catchment Approach to specific areas. Spatial data are indispensable for constructing such maps. A major difficulty in the East African Highlands is however the poor availability of spatial data at this scale. This impedes the application of erosion models with high data requirements [3], besides the difficulties that exist in applying models developed for a small scale to larger regions [4]. Qualitative approaches can therefore be more effective for erosion mapping in regions with a limited data availability [5]. These approaches can benefit from the increased availability of satellite imagery and image-derived products, like digital elevation models (DEMs), that are currently available to the general public, often at low cost.

The objective of this study was to assess the potential of accurately mapping erosion risk for a 70-km^2 area in northeast Tanzania, using a simple qualitative integration of readily available data.

2 MATERIALS AND METHODS

2.1 Study area

The 70-km^2 Baga watershed is situated in the West Usambara Mountains, which are located in northeast Tanzania (figure 1). These mountains form part of the system known as the East African Highlands, which comprises highland areas in Burundi, Ethiopia, Kenya, Rwanda, Tanzania, and Uganda. Because of a favourable climate and fertile soils, the West Usambara Mountains are very important for food, fibre, and fodder production. However, high population pressure (> 100 persons/km^2) has resulted in overexploitation of natural resources causing deforestation and land degradation [6]. Currently, soil erosion is widespread and forms the major limiting factor for agricultural production [7].

The Baga watershed has two rainy seasons, locally named short rains (October-November), and long rains (March to May). The elevation within the watershed is between 1200 to 2000-m above mean sea level. The terrain is highly dissected with local slopes of up to 90%. The major land use is agriculture, mostly composed of small-scale fields. The common cultivation is maize, intercropped with beans, banana, cassava, and sugarcane, whereas the combination of coffee, banana, and coco yam occurs on some steeper slopes. In the valley bottoms irrigated vegetables are grown. Furthermore large tea plantations occur and mountain rain forest is present in the higher parts of the watershed. Dispersed trees of different species are found in varying densities throughout the watershed. Sheet and rill erosion are the most common erosion forms, occurring on fields cultivated with annuals. At the onset of the long rains when high-intensity rainfall occurs and soil cover is poor, erosion is most severe [8]. Very few soil and water conservation measures were encountered in the Baga watershed.

2.2 Data and preprocessing

Data sources used included a Shuttle Radar Topography Mission (SRTM) C-band DEM, a Landsat-7 Enhanced Thematic Mapper (ETM+) image, and field data. Furthermore an orthophoto covering the study site was applied for precise georeferencing of the Landsat image.

The SRTM DEM has a 90-m resolution and is freely available from the internet. Minor data voids occurring in the study area were filled by interpolating neighbouring elevation values. Slope was calculated from the DEM using the slope definition of Zevenbergen and Thorne [9]. Subsequently the slope map was reprojected to a UTM projection, using nearest neighbour resampling and a 30-m output pixel size. This pixel size was chosen to match the Landsat image.

The Landsat image was recorded on 6 February 2003, which is slightly before the start of the long rains. At that moment many agricultural fields were bare, causing high erosion risks at the onset of the rains. To reduce geometric distortions caused by the steep topography, the Landsat image was orthorectified. For this purpose, elevation was obtained from the DEM and viewing geometry was derived from the path definition of the Landsat-7 satellite. The geometric reference was determined by visually relating 23 control points on the orthophoto to the same points on the Landsat image. The root mean square error of the orthorectification model was 23.6-m, which is lower than the image resolution. Nearest neighbour resampling was applied with an output pixel size of 30-m. The 30-m pixel size was maintained to represent small-scale differences in vegetation cover.

Figure 1. The Baga watershed: location, boundary, and drainage pattern.

After radiometric calibration of Landsat bands 3 and 4, the Normalized Difference Vegetation Index (NDVI) was calculated. Besides being indicative of the amount and status of the vegetation, the NDVI has the additional advantage that it reduces the topographic effect [10]. This effect implies that spectral reflection is different for similar vegetation covers due to variation of slope gradient and aspect in rugged terrain. A good reduction of the topographic effect was obtained for the Baga watershed.

A detailed field survey was performed between late February to early March 2003, before the start of the long rains. The reasons for the timing of data collection were: (1) the high erosion risk at the onset of the long rains, (2) the easier accessibility of many locations as compared to the wet season, and (3) the availability of a good Landsat image acquired at nearly the same moment. During the survey, a Global Positioning System (GPS) marked the location of 151 points with an approximately uniform vegetation cover. The following characteristics were recorded at these locations: land cover, fractional vegetation cover, slope gradient, presence of soil and water conservation measures, and erosion indicators. A qualitative field estimate of erosion risk (E_f) was made jointly with an extension officer on a scale ranging from 1 (very low) to 5 (very high), based on the recorded surface characteristics and the presence of local erosion indicators [11]. Used indicators of high erosion risk included stony soil, exposed roots, and the presence of rills and pedestals.

2.3 Erosion risk mapping

The soil erosion risk map was constructed through a qualitative integration of the slope map and the NDVI map. The slope map was automatically classified into five classes that each occupy 20 % of the study area. A factorial erosion risk class (E_{slope}) ranging from very low (1) to very high (5) was assigned, in which E_{slope}=5 corresponds to the 20 % steepest slopes in the area. The flat and very gentle slopes obtained E_{slope}=1.

The NDVI provided a good measure of fractional vegetation cover (FVC), since regression analysis between field estimates of FVC and the corresponding NDVI values yielded a relationship with an R^2 of 0.80. The NDVI was therefore used directly as an indication of FVC. A similar automatic classification as for slope was executed, resulting in a factorial erosion risk class for NDVI (E_{NDVI}). In this case E_{NDVI}=1 related to very high NDVI, and E_{NDVI}=5 to the 20 % lowest NDVI-values in the study area.

The erosion risk map was constructed by taking the minimum of E_{slope} and E_{NDVI} at each location:

$$E_T = \min (E_{slope}, E_{NDVI}) \quad , \tag{1}$$

where E_T is the final erosion risk class. The minimum-operator was chosen because it was assumed that the lowest of the two factorial risk classes determines the final erosion risk. This assumption was made because a good vegetation cover on a steep slope and a poor cover on a very gentle slope both result in a relatively little erosion. The method therefore pinpoints to locations where steep slopes and limited vegetation cover coincide.

2.4 Accuracy assessment

The 151 field estimates of erosion risk (E_f) were compared against mapping results (E_T) for the same locations. The frequency of occurring combinations of E_f and E_T were tabulated in a confusion matrix. Two quantitative measures of accuracy (A) were defined. The first (A_1) strictly defines a correct classification as one where $E_f = E_T$. For the confusion matrix, this means dividing the sum of the diagonal elements by the total number of observations (=151):

$$A_1 = \frac{\sum_{i=1}^{5} u_{i,i}}{\sum_{i=1}^{5} \sum_{j=1}^{5} u_{i,j}} \cdot 100\% \quad , \tag{2}$$

where $u_{i,j}$ represents a specific number in the confusion matrix. The second (A_2) is more flexible and assumes that a one-class difference is acceptable. This may be justified because the classes are not defined on a nominal scale, but on an ordinal scale. If we state that a classified pixel with a certain E_T is considered correctly classified when there is a maximum difference of one class (e.g. E_T=2 is correct if E_f is 1, 2, or 3), we obtain the following equation:

$$A_2 = \frac{\sum_{i=1}^{5} u_{i,i} + \sum_{i=2}^{5} u_{i,i-1} + \sum_{i=1}^{4} u_{i,i+1}}{\sum_{i=1}^{5} \sum_{j=1}^{5} u_{i,j}} \cdot 100\% \tag{3}$$

3 RESULTS AND DISCUSSION

Slope was calculated from the DEM and the values were divided in five equal-frequency classes (E_{slope}). The class limits of E_{slope} are presented in table 1. The interval for each range is different because the only classification criterion was that each class occupies 20% of the study area. The resulting factorial erosion risk map for slope is shown in figure 2a. For NDVI a similar division was made (table I) resulting in the factorial erosion risk map for NDVI (figure 2b). Applying the minimum-operator to the E_{slope} and the E_{NDVI} map following Equation (1) resulted in the final erosion risk map (figure 2c). The highest erosion risk classes are present on the steep slopes where

Table 1. Ranges for factorial erosion risk classes based on 20 % occurrence of each class in the Baga watershed.

Factorial erosion risk classes E_{slope} and E_{NDVI}	slope range	NDVI range
very low (1)	< 14.6	> 0.735
low (2)	14.6 – 20.9	0.695 – 0.735
medium (3)	20.9 – 26.2	0.649 – 0.695
high (4)	26.2 – 32.9	0.580 – 0.649
very high (5)	> 32.9	< 0.580

annuals are cultivated, and which were bare or had limited vegetation cover at the time of image acquisition. Low erosion risk is found both in the valleys and on the higher-located forested areas.

Table 2 shows the confusion matrix comparing the 151 field estimates of erosion risk (E_f) with the classification results (E_T) for the same locations. The accuracy was calculated from this matrix using Equation (2) and (3): A_1 is 38 % and A_2 is 81 %. A clear cause for misclassifications could not be determined. However, the accuracy is quite high, at least when a one-class difference is allowed. This implies that the presented method allows for a proper identification of high and low erosion risk areas within the Baga watershed.

The accuracy assessment was based on a limited number of field estimates of erosion risk that were acquired before map construction. The qualitative estimates of erosion risk are not fully objective figures. Although clear criteria were set, it is possible that persons have different opinions on the importance of a particular factor at a specific location. Therefore, only five classes were assigned and all estimates were done together with a local extension officer. It was believed that this resulted in a proper qualitative estimate of erosion risk. For accuracy assessment, a one-class difference seems permissible however when using such ordinal qualitative estimates.

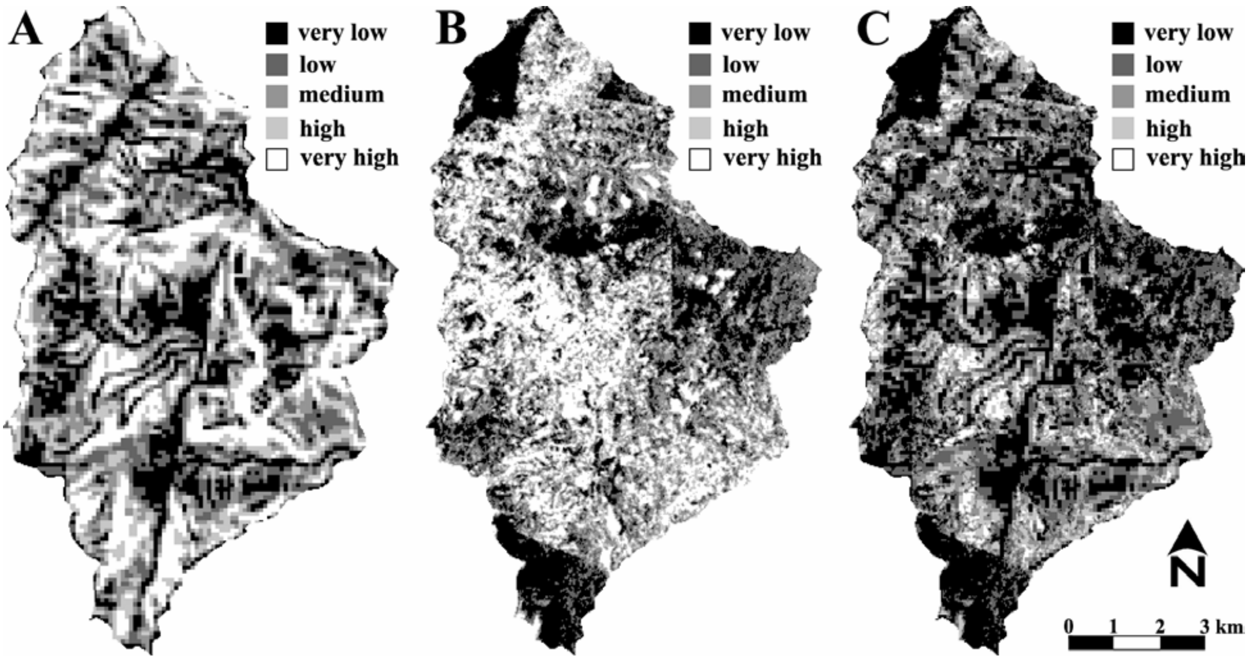

Figure 2. The Baga watershed with (a) factorial erosion risk map for slope (E_{slope}); (b) factorial erosion risk map for NDVI (E_{NDVI}); and (c) final erosion risk map (E_T) created by taking the minimum of E_{slope} and E_{NDVI}.

Table 2. Confusion matrix for validation of the erosion risk map comparing classification results (E_T) with field estimates of erosion risk (E_f).

		Field estimate of erosion risk					
		Very low	Low	Medium	High	Very high	Total
Erosion risk map	Very low	25	16	9	3	1	54
	Low	8	13	6	5	2	34
	Medium	5	13	9	8	1	36
	High	0	3	8	8	3	22
	Very high	0	0	0	2	3	5
	Total	38	45	32	26	10	151

Validation of erosion risk maps may also be achieved through the execution of erosion measurements or quantitative surveys that repetitively measure rill volumes. The problem is that such measurements are extremely labour intensive and require long time series of data for making proper assessments. Mostly it is therefore unfeasible to perform measurements at many different locations, which explains an often-encountered lack of validation activities in regional erosion studies. However, validation is of utmost importance to assess the validity of the methods applied for the region and scale under study. We think that the qualitative field estimation of erosion risk provides a good alternative for validating regional erosion maps and mapping methodologies. Timing of the field surveys needs to be in balance with the image date and the erosive season.

The presented methodology uses only slope and NDVI for mapping erosion risk. Other factors such as soil properties and rainfall intensity are also important for the occurrence and intensity of erosion processes. Nevertheless the spatial distribution of erosion risk is well represented in the Baga watershed with only these two factors, following the A_2-value of 81 %. Partly this may be explained by the relationship that exists between vegetation cover, here described by NDVI, and factors like soil type, rainfall characteristics, and land management. Therefore, NDVI implicitly accounts for these factors. For a different part of the East African Highlands, in Kenya, Okoth [12] found that the factors slope and vegetation cover had a high correlation with the presence of erosion features. This implies that the presented methodology may produce good results in different parts of the East African Highlands. However, more validation work would be required within this region. Also in other regions the methodology may be useful for quickly identifying high erosion risk areas by locating areas where relatively steep slopes coincide with limited vegetation cover, especially when there is limited data availability. For environments where spatial variability of erosion processes is importantly determined by other factors (e.g. soil properties) this approach will probably fail, unless a strong correlation exists between the soil properties and vegetation cover.

The end product for this specific case study is not necessarily the erosion risk map (figure 2c). As stated in section 1, the rationale for the mapping exercise was to assist the Catchment Approach by producing an erosion risk map which can serve for focal area selection. The presented map shows where potential problem areas are located. For the local extension agency, it may be of more interest to construct a map showing erosion risk per focal area, through e.g. averaging the pixel values within that area. Because focal areas were not clearly defined and do not always coincide with hydrological catchments, the raster map was maintained. However, this base map can be processed further for increased utility and improved readability by involved agencies.

Readily available data were applied for mapping erosion risk. We chose to use data sources that can be easily obtained by a large public to enhance the applicability of the methodology. This choice resulted in the use of a 90-m SRTM DEM, and a 30-m Landsat image. Therefore, a slight mismatch exists in the scale of the data. The decision was made not to degrade the Landsat data to 90-m because of the high spatial variability of vegetation in the Baga watershed. Ideally a higher resolution DEM would be applied, e.g. derived from ASTER or ERS interferometry. However, archive ASTER data were scarce for the study area and ERS interferometric DEMs are not easily obtainable by a large public. Nevertheless, the slope pattern was properly represented with the 90-m DEM, judging from the acquired field knowledge on the watershed. Here, slope pattern is more important than the exact slope gradients, because no physically-based modelling is performed. We therefore do not believe that the scale difference has a major influence on the results, as can also be observed from the good accuracy of the map.

Timing is a critical issue for erosion risk mapping. For vegetation cover analysis, a satellite image must be properly timed with respect to the most erosive season. In this case a Landsat image was obtained that was recorded slightly before the long rains, which is thought to be the most erosive period. However, fieldwork indicated that more fields were cleared for cultivation of annual crops between the image date and the start of the rains. Nevertheless, availability of good-quality satellite images is rather low in the region due to frequent cloud cover, and thus the selected image is the optimal solution. Other optical images like ASTER or SPOT serve equally well for applying the presented methodology to other areas. Although SAR imagery allows for the assessment of vegetation characteristics, radar lay-over and shadow effects limit its applicability in mountainous regions.

A static site-dependent representation of erosion risk in a region is provided by the presented methodology. The division of classes of equal occurrence fully depends on the type of slopes and vegetation covers that are present at a certain moment in a certain region. Changing the boundaries of the region would change the classification, and changing the image date would impact the class boundaries. The methodology is not intended for the monitoring of changes in erosion risk, due to e.g. improved soil and water conservation. Nevertheless, possibilities for this exist using multi-temporal imagery of the same season. In that case however, class limits should be defined for one reference image (e.g. the oldest acquisition). The requirement of having 20 % occurrence of every factorial erosion risk class should then be loosened for subsequent dates, but the same class limits as the reference should be applied.

4 CONCLUSIONS

A simple qualitative method integrating the factors slope and NDVI allowed for an accurate mapping of erosion risk in the Baga watershed. These factors could be easily derived from an SRTM DEM and a Landsat image, which are readily available for a large public. Therefore, options for applying the method to other data-poor regions exist, especially within the East African Highlands. The method is however highly dependent on the extent of the region considered, because it automatically defines slope and NDVI classes based on the occurrence within a region. The primary application of the method is the quick identification of high erosion risk areas and can thus assist in the application of the Catchment Approach. The optical satellite image used for the mapping should be acquired during or just before the most erosive season, when soils are less protected and high rainfall intensities occur.

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Integration of Remote Sensing Technique and GIS for Erosion Risk Modeling in Syrian Coastal Zone

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1 INTRODUCTION

The issue of land resource conservation is strongly talked about now days in Syria like in all Mediterranean countries. However, in the coastal zone, where there are many natural resources have to be protected and conserved. Protection of natural resources will enhance the environment on one hand and decrease the degradation process on another hand.

Degradation processes are accelerated not only by natural assets of the landscape (relief, geomorphology, geology, soil characteristics, rain intensity, …) but also strongly by people intensity and human practices and activities (intensive agriculture, industry, transportation….).

The study aims to:

- Implement a methodological approach and capacity building program, for supporting plans to control and manage land degradation in Syrian coastal zone.
- Provide the country planners and decision makers with thematic maps (erosion risk map) which help them to know the level of sensitivity to erosion over the coastal zone and this help them to take the right decision how to control the erosion.
- Provide the country planners and decision makers with technical tools for acquiring knowledge on causes and effects of land degradation in the coastal area as well as with management tools for supporting the implementation of sustainable development action in the area.
- Produce an overall sustainable improvement of the environmental knowledge and awareness in the country.
- Apply a capacity building program not only for the team of the study but also for the expert and managers who are working in the environmental sector.
- Produce guidelines and recommendations for supporting coastal land degradation control and management; know how to transfer the methodological monitoring approaches, provision of thematic maps, technical report and comprehensive database.

2 GENERAL DESCRIPTION OF THE STUDY AREA

Syrian coastal zone is located in the northwestern part of country. Bordered in the west by the Mediterranean Sea with coastline of about 220 km. The region can be viewed as a major natural resource and “transitional” in character, linking the Mediterranean Sea with arid zones of the interior Syria and the Arab world.

The coastal zone of Syria covers about 4190 km^2 (2% of the national territory). The region is divided into two governorates; Latakia governorate in the north with 2300 km^2 total area and Tartous governorate in the south with 1900 km^2 area.

Geomorphologically: the coastal region can be divided into 5 main geomorphologic areas: shore –line, coastal plain, hilly areas, river valleys, and mountainous areas.

Pedologically: Syrian costal soils can be divided into main five groups: coastal plain soils, piedmont soils, summit soils, river bed soils, and forest soils (Darwish, 1986 -GORS, 1991). The natural land cover of the coastal area in Syria is similar to those natural covers in all Mediterranean countries (Nahal 1984).

Climatically: the entire coastal region belongs to the Mediterranean humid or subtropical types of climate, with the amount of rainfall and temperature gradually increasing from the west to the east and decreasing from the higher to the lower slopes of the coastal mountains and from north to south down the Bassit block (PAP/RAC, 1990). Thus, a general characteristic of the coastal zone is a combination of high temperature and medium amount of rainfall. The annual temperature averages for Latakia and Tartous are almost 20 ^{o}C, as compared to 12.5 ^{o}C for Slenfeh, in the mountainous hinterland (Eid, 2004).

3 MATERIALS AND METHODOLOGY

3.1 Materials

3.1.1 Satellite image

LandSat 7 satellite image (2 views) covering the whole study area was used. This satellite image is collected in year 2001, spring time. It used to produce land cover / land use map.

3.1.2 Arial photos

1/20.000 scale B/W aerial photos of the coastal area. 2 stripes (from the southern part of the study area) were not available. These photos were used to understand the land form. They used in parallel with the Landsat image to draw the land form map.

3.1.3 Topographic maps

Topographic maps for the study area are available at the 1/100 000 and 1/50 000 scale.

They used to produce different GIS layers.

3.1.4 Climatic Data

Some climatic data were collected to link the relationship between the erosion process and the climatic factors.

3.1.5 Softwares

ArcView 3.1 and ArcGIS 8.3 were used to conduct the GIS works.

ERDAS 8.3 was used to do the satellite image rectification, enhancement and analysis.

Microsoft Access and Excel softwares were also used for loading and processing data collected during the field survey.

3.2 Methodology

The study was conducted at the reconnaissance survey scale. It is particularly suitable to the assessment of land resources over a large area. It is based on an holistic approach (from the Greek "*olos*" meaning "all complete"), according to which, the main components of the land, such as soil, vegetation, geomorphology, hydrology are not considered individually but as a whole, they are mapped simultaneously, taking also into account interactions between them and man's influence, so that the earth surface is the result of something more than just the sum of all of them. The result of the applied methodology is a Land Unit Map, where the basic cartographic unit is represented by a portion of land composed by a series of "land facets", each of them considered to be homogeneous, at the scale of the survey, with regard to its geomorphologic, pedologic and vegetation characteristics.

Satellite images are essential tools of the survey as they provide a reliable basis for the interpretation of the landscape and allow a "landscape guided" approach (Zonneveld, 1979) to the survey of natural resources, resulting in a better cost-effectiveness of the work.

The methodology consists from the following steps:

1. Preparation of land unit map:

   Based on the available materials, the land unit map was prepared. Every land unit consist from three parts: land system, land form and land use / land cover.

   The proposed land system types, land form types and land cover / land use types are as following:

   Land System types: Lower coastal Plain, Coastal Plateau, Steep Hills and Upper Plateau.

   Land form types: Flat topped Crest, Sharp Crest, Rounded Crest, Fluvial Scarps, Badlands, Undulating Planation Surface, Flat Planation Surface, Erosion Slopes, Accumulation Foot Slopes, Accumulation Plain, Alluvial Valley, Fluvial Valley, Beaches and Coastal alluvial plain

   Land Cover / Land Use types: Closed forest, Closed maki, Open forest, Open forest & shrubs, Open forest and olives, Open forest and fruit trees, Open maki, Open maki and olives, Citrus plantations, Citrus plantations and olives, Citrus plantations and field crops, Greenhouses and field crops, Olives, Olives and open maki, Olives and citrus plantations, Olives and greenhouses, Olives and open forest, Olives and field crops, Olives and fruit trees, Field crops, Field crops and citrus plantations, Field crops and greenhouses, Fruit trees, Fruit trees and open forest, Fruit trees and field crops, Urban, Sandy soil, Rock outcrops and olives, Quarries and Reservoirs.

Simplified Land cover / land use types: Closed natural vegetation, Opened natural vegetation, Olive plantation (dominant), Opened natural vegetation with olives, Fruit trees (dominant), Field crops (dominant), Green houses (dominant), Barren land, Water bodies and Urban.

2. Reconnaissance survey

   Field work was done and forms of reconnaissance survey and erosion risk estimation were filled. The field work was done according to stratification method for selecting the sites for the investigation.

3. Data processing and Preparation of Erosion risk map:

   Data collected were processed on spatial computer program and the results were transferred to GIS layer and used to produce the erosion risk map.

   The erosion risk map was created; based on the land unit map and analysis of the relating factors. Values were given to these factors and transferred into rating numbers automatically. Sum of the rating numbers for all these factors were also calculated automatically. The extracted values referred to the soil erosion risk. Six soil erosion risks were put. Rating of factors is explained in table 1.

Table 1. Factors contributing in soil erosion and their rating.

Factors	**Rating**	**Factors**	**Rating**
Land cover/Land use types	1 - 12	Vegetation density	1 - 12
Slope gradient	1 - 32	Conservation practices	-6 - 0
Slope length	1 - 8	Depth of rills and gullies	0 - 4
Slope form	1 - 3	Spacing of rills and gullies	1 - 6
Soil texture	1 - 8	Mass movement area	1 - 3
Soil depth	0 - 3	Mass movement rate	1 - 4
Surface sealing	0 - 4	Sheet erosion activity	0 - 4

Six risk classes were suggested according to the rating sum. The following table explains this process.

Table 2. Ranking of the erosion risk classes.

Class Description	Rating sum	Erosion Risk class
Not relevant	-999	0
Not or insignificantly susceptible	> 0 - 17	1
Slightly susceptible	> 17 - 25	2
Moderately susceptible	> 25 - 33	3
Highly susceptible	> 33 – 49	4
Very highly susceptible	> 49 - 65	5
Extremely susceptible	> 65 - 100	6

4. RESULTS AND DISCUSSION

Based on the applied methodology areas and locations of different water erosion risk classes were achieved. See table 3 and figure 1.

Knowledge on causes of land degradation process is needed in order to select, for each management area, the most appropriate actions for natural resources conservation.

Obviously, the scale used for such a study does not allow a detailed analysis of causes of land degradation with relevant preventive or corrective measures, although it provides the opportunity to assess priority areas for more detailed studies and to get an overview of the main causes of land degradation processes in these areas.

Table 3. Areas of every erosion risk class.

Risk Class	Risk Class ID	Count	Area KM^2
Not relevant	0	127	108.1610
Insignificant susceptible	1	81	604.2125
Slightly susceptible	2	124	470.2043
Moderate susceptible	3	431	1336.5183
High susceptible	4	311	918.6685
Very high susceptible	5	50	158.1534
Extremely high susceptible	6	36	48.2834
TOTAL		**1160**	**3643.7631**

The torrential rains which are responsible for severe water erosion processes act over an environment where certain morphological characteristics (mainly slope) and land use/land cover patterns - as described by the reconnaissance survey - represent the two key factors to assess land degradation and to be taken into account for devising land management programmes.

Among other factors which play an equally important role, some soil features - such as soil texture - can be mentioned as affecting in turn water infiltration rate and, as a consequence, the infiltration/runoff ratio.

4.1 The role of Morphology in soil erosion

The morphological asset plays a key role in characterizing erosion-prone areas and in defining management plans aimed at facing similar phenomena.

Table 4 presents areas of different risk classes with different land system types.

Table 4. Areas of erosion risk classes categorized according to land system types.

LAND SYSTEM TYPES	LS code	Not relevant	Risk 1	Risk 2	Risk 3	Risk 4	Risk 5	Risk 6
Upper Plateau	A	1.29	0.00	23.11	22.90	0.00	0.00	0.00
Steep Hills	B	10.09	80.11	107.15	1032.39	738.91	124.00	30.74
Coastal Plateau	C	4.60	11.37	27.48	249.20	161.73	34.15	17.55
Lower Coastal Plain	D	92.19	512.73	311.87	32.02	18.03	0.00	0.00
Sum		108.17	604.21	469.60	1336.51	918.66	158.15	48.28
Form total		2.97	16.58	12.89	36.68	25.21	4.34	1.33

4.1.1 Slope gradient

Slope gradient is undoubtedly the most important factor in assessing erosion risk is the slope value. It weights half or more the total erosion risk value (up to 32 of rating sum). High slope gradient is mainly occurs in the steep hills. This land system class comprises 58.28% of the total study area with an area of 2123.39 km^2 (table 4). It is either covered by natural vegetation with different level of degradation or by some agricultural uses mainly olive plantations. 42% of the steep hills in the study area (893.65 km^2) classified as highly, very highly or extremely highly susceptible to erosion (table 4).

4.1.2 Slope length

Slope length becomes a critical factor in the land system of Lower Coastal Plain, where it plays a quite important role on more gentle slopes in which the presence of long slopes with quite uniform slope gradient contributes to the increase in runoff speed and in the erosive power of water.

Lower coastal plain in the study area comprises 26.54% of the total study area with an area of 966.83 km^2. Most of this land is arable land and used for agricultural purposes (crops, green houses, citrus…)

The highest three risk classes falling in the lower coastal plain comprises 1.86% (18.03 km^2) of this land system category; while the lowest three risk classes comprises 88.60% (856.61 km^2) of this land system category (table 4).

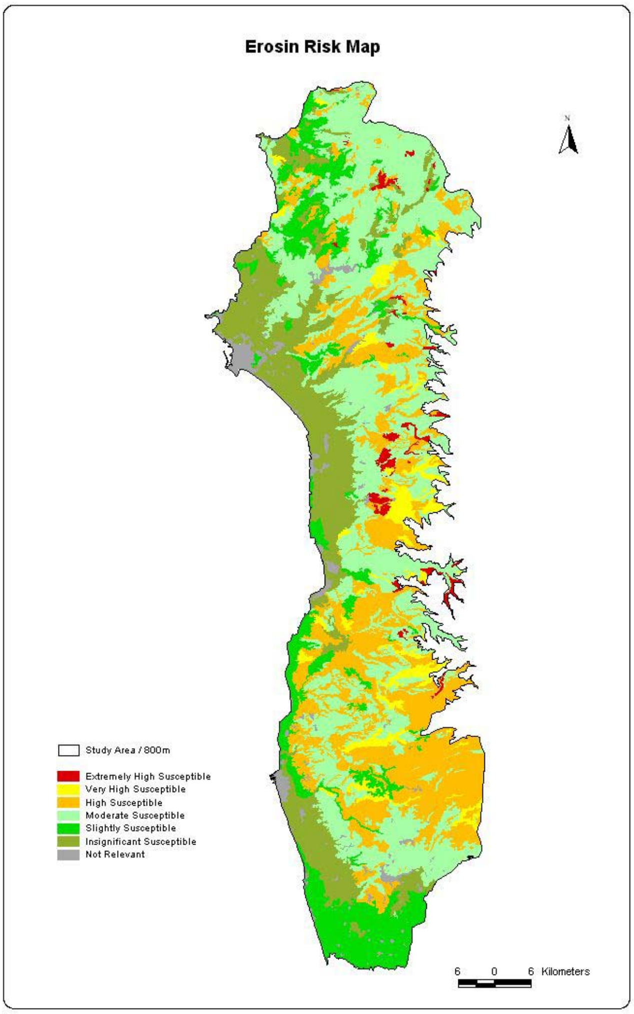

Figure 1. Erosion risk Map.

4.1.3 Slope gradient and slope length together

Coastal plateau comprises 13.94% from the total study are with an area of 507.7858 km^2. It is recognized with different levels of slope gradient and slope length. So, in this land system type two factors together can influence the erosion process.

The highest three risk classes falling in the coastal plateau comprise 42.17% (213.42 km^2) of this land system category (table 4).

Analyzing the data of land forms with different erosion risk classes showed similar results to these ones from analyzing the land system types with different erosion risk classes (Data are not presented).

4.2 Land cover / land use

Syrian coastal zone shows, besides a quite high variety of morphological characteristics, some dominant patterns of land use types characterised by specific erosion risk classes.

Table 5 shows the data of erosion risk classes based on simplified land cover / land use types (data of the detailed land cover / land use types are not presented). Table 6 shows the simplified land cover / land use classified based on land system types.

Table 5. Areas of erosion risk classes categorized according to simplified land cover / land use types

Simplified LU / LC types	Risk 1	Risk 2	Risk 3	Risk 4	Risk 5	Risk 6
Opened natural vegetation	0.00	48.68	42.02	78.77	41.77	23.87
Closed natural vegetation	66.21	22.17	316.08	0.01	0.00	0.00
Olive plantation (dominant)	24.62	106.62	779.45	537.12	59.34	0.00
Opened natural vegetation & olives	22.01	5.88	56.21	266.55	45.44	0.00
Fruit trees (dominant)	445.67	70.76	130.28	18.03	11.61	6.87
Field crops (dominant)	35.45	215.49	0.00	0.00	0.00	0.00
Green houses (dominant)	10.25	0.00	0.00	0.00	0.00	0.00
Barren land	0.00	0.00	12.47	18.20	0.00	17.55
Water bodies	0.00	0.00	0.00	0.00	0.00	0.00
Urban	0.00	0.00	0.00	0.00	0.00	0.00
Sum	604.21	469.61	1336.52	918.67	158.15	48.28
% Form total	16.58	12.89	36.68	25.21	4.34	1.33

4.2.1 Olive plantations

41.37% of the study-area (1507.16 km^2) is marked by olive plantations. It is characterised by a plant spacing which usually leaves more than a half of the soil surface unprotected from erosive action of raindrops. Table 4-4 shows that the most of those plantations are located on hilly areas, with morphologies varying from gently undulating to steep slopes and calcareous soils particularly rich in gravel and stones. About 40% of this land use type falls in the highest three erosion risk with an area of 596.46 km^2 (16.37% from the total study area).

The estimation relying on field survey data, shows that about 70% of slopes covered with olive plantations are managed through some kind of conservation practices; mainly contour stones terracing, but also bench terracing and contour ploughing. Such practices play a key role in decreasing the surface water runoff and, as a consequence, in reducing erosion risk. Obviously, the effectiveness of such measures depends on slope steepness.

Olive plantations are very often irregularly mixed with natural vegetation formations such as open maki, and, locally, smaller reforested surfaces. This mixed natural vegetation with olives comprises about 11% from the total study area. These formations have various degrees of degradation. 78.77% from this land cover / land use type is falling in the highest three risk classes and with an area of 311.98 km^2 (8.56% from the total study area).

4.2.2 Arable land

At lower altitudes, on the coastal plain, land use is characterised by tree plantations (especially citrus), intensive farming and field crops (see table 6).

Arable land is also a critical area to be highlighted in the coastal plain, where high slope length - even if with a moderate slope gradient – combined with agricultural activities (in particular, citrus and other fruit trees plantations and field crops) may induce erosion phenomena.

5.34% from the fruit trees is falling in the highest three risk classes and with an area of 36.5 km^2. While, 94.66% of this land type is falling in the lowest three risk classes with an area of 646.71 km^2.

The same trend is occurring with the other agricultural land use types (field crops and green houses).

Table 6. Areas of simplified land cover / land use falling in different land system types.

LS / LC	Upper Plateau	Steep Hills	Coastal Plateau	Lower Coastal Plain	Not Relevant	
Land cover / Land use Types	**A**	**B**	**C**	**D**	**NR**	**SUM LC**
Open Natural Vegetation	0.852	205.553	24.760	3.941	0.000	235.106
Closed Natural Vegetation	3.411	391.995	2.360	6.702	0.000	404.468
Olives Plantations	41.749	1002.842	368.703	93.864	0.000	1507.158
Open Natural Vegetation / Olives	0.000	369.151	19.818	7.117	0.000	396.086
Fruit Trees (Dominant)	0.000	128.181	54.866	500.161	0.000	683.208
Field crops (Dominant)	0.000	0.000	0.000	250.940	0.000	250.940
Green Houses (Dominant)	0.000	0.000	0.000	10.253	0.000	10.253
Barren Land (Dominant)	0.000	15.588	30.965	7.444	2.083	56.080
Water Bodies	0.000	0.000	0.000	0.000	19.807	19.807
Urban	0.000	0.000	0.000	0.000	80.491	80.491
SUM LS	46.012	2113.310	501.472	880.422	102.381	3643.590

4.2.3 Natural vegetation cover

61.42% from the opened natural vegetation is falling in the highest three risk classes with an area of 144.4 km^2. The density of vegetation cover of this land use type has different percentage, ranging from 30% to 70% with different slope gradient and different level of degradation.

The closed natural vegetation has different trend to erosion risk. About 100% of this land cover type is falling in the in the lowest three erosion risk classes with an area of 404.46 km^2. The vegetation cover of this land cover type is very dense with coverage percentage more than 80%. This dense vegetation cover absorbs the rain drops strikes and minimizes the influence of the water on the soil even if this vegetation locates in very sloppy area. About 78% of this land cover type is falling in the risk class 3 with an area of about 316 km^2. This is because most of this cover type is falling in very sloppy areas.

Based on the field survey evidence, most of these lands still pure natural areas with very dense vegetation cover. Decreasing the vegetation density in this area can increase rapidly the erosion risk to higher classes. So, in such areas vegetation cover plays more important role than the slope gradient in generating water soil erosion.

4.2.4 Urban

90 % of the urban areas are falling in the coastal plain with an area of 80.48 km^2 (about 11% of the lower coastal plain) which is the fertile arable land. This land is used from hundreds years for agriculture. It is highly recommended to keep this land for agriculture and look for less fertile land for urban expansion.

5 RECOMMENDATIONS FOR THE WHOLE STUDY AREA MANAGEMENT

General recommendations can be set into two main groups. The first one is related with *technical* issues that deal directly with erosion phenomena it self. The other one is related with *policies and strategies* that design for sustain management of the land.

5.1 Recommendations at the technical level

- Protect, conserve and improve the vegetation cover on steep and very steep slopes.
- Establishing natural protected areas.
- Replanting fruity forestry species in the cleared forested areas.
- Establishing contour stones terracing in sloppy areas and maintain these terraces annually and when needed.
- Establishing bench terracing and contour ploughing in less sloppy areas.
- Improve and support the modern irrigation systems.
- Increase the projects of water catchments (dams, reservoirs and tanks).
- Stop the practice of burning the agricultural residuals, especially on the boarders of the forest land in order to minimize forest fire risk.
- Restricting the urban expansion in the forest land and arable land.

5.2 recommendations at the policies level

Policies should be followed in order to protect the soil from erosion. The following are some political recommendations which can improve the land conservation:

- Creating a cohabitation relationship between the local people and the forest land.
- Strictly applying the environment and forest law.
- Establishing land degradation monitoring system (monitoring indicators).
- Improve the awareness system about land degradation and land conservation.
- Improve the capacity building for the related staff who are working in the field of land degradation and natural resources conservation.
- Conducting researches and projects at national level related to land degradation and sustainable development and involving the participation approach in such kind of projects.
- Involvement in international projects which are relevant to land degradation.
- Improve the governmental agricultural loan system in order to facilitate the construction of the terraces and applying modern irrigation system.
- Avoiding construction of new roads through forest lands unless, it was for actual and necessitous needs and if can't be helped, applying geometric basis for constructing.

6 CONCLUSION

Preparation of management plans for degradation affected areas cannot disregard the guiding principles of sustainable development. Management and conservation of natural resources can be well done by the usage of modern technologies and the orientation of institutional efforts so to ensure the continuous satisfaction of human needs for present and future generations. The application of sustainable development principles makes preservation of natural and genetic resources possible. This will lead to environment development and results viable economy and acceptable society.

The application of sustainable development principles to fragile ecosystems such as arid and semiarid zones, very common in Mediterranean areas, must cope with the issues of land degradation and desertification.

This is why, it is essential to combat desertification through the application of three main measures:

- **Preventive measures**, in order to prevent the occurrence of degradation phenomena in lands which are not yet degraded, or which are only slightly degraded;
- **Mitigation measures**, in order to rehabilitate the productivity of moderately-degraded lands;
- **Restoration measures,** for soil recovery and land reclamation in seriously degraded areas.

Therefore, the main objectives to be taken into account in the development of land management plans should be: *technical objectives* by applying of preventive, mitigation and restoration measures to relevant areas and *Political objectives* by developing of policies which aim to promote sustainable development and reduce the impact of human pressure on land resources.

If we want to make these objectives applicable, we should translate them into strategies and actions, both at national and international levels.

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