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Land use change syndromes as a framework for integrating satellite observations into the assessment of dryland degradation

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Zusammenfassung

Als direkte Reaktion auf Klimaveränderungen, Wasserverknappung und Übernutzung von natürlichen Ressourcen werden Landdegradation und Desertifikation in den Trockengebieten der Erde als eine der größten Bedrohungen für die globale Umwelt eingeschätzt. Das Monitoring der Veränderungen sowie die mittel- bis langfristige Entwicklung von Gegenmaßnahmen sind Thema zahlreicher Forschungsprojekte und nationaler und internationaler Konventionen und Programme.

Um nachhaltige Landmanagementstrategien zu entwickeln, ist es unabdingbar, die oftmals sehr komplexen Landdegradationsprozesse vollständig zu verstehen. Dazu gehört neben dem Verständnis der eigentlichen Prozessdynamik auch die Identifizierung der zugrundeliegenden klimabedingten oder anthropogenen Ursachen. Zudem ist es essentiell, betroffene und gefährdete Gebiete in ihrer räumlichen Ausdehnung und Struktur zu erfassen. In den letzten Jahrzehnten wurden zahlreiche Indikatoren und Methoden entwickelt, um sowohl den Grad als auch die Veränderungen der Landdegradation abzuschätzen. Da die biologische Produktivität eines Ökosystems Rückschlüsse auf dessen Zustand zulässt, eignen sich fernerkundlich abgeleitete Vegetationsparameter, um großflächige Analysen durchzuführen. Aufgrund der Komplexität dieser Problematik wurde bisher kein umfassendes Beobachtungsverfahren entwickelt, über das Landdegradation auf regionalem Maßstab operativ und konsistent abgeleitet werden kann.

Im Rahmen der vorliegenden Arbeit wurde, basierend auf dem Konzept „Syndrome des globalen Wandels“ ein Interpretationsrahmen entwickelt, der die vielfältigen anthropogen verursachten Umweltveränderungen mit ihren Ausprägungen (Symptome) typischen Ursache-Wirkungsmustern (Syndrome) zuordnet. Angesichts der hohen klimatischen Variabilität in Trockengebieten ist es notwendig, lange Beobachtungszeiträume zu analysieren, um die klimabedingten Schwankungen der Ökosystemproduktivität zu erfassen und von anthropogen verursachten Landnutzungsänderungen zu unterscheiden. Um diesen Anforderungen Rechnung zu tragen, wurde ein Ansatz entwickelt, der basierend auf der Zeitreihenanalyse von temporal hochaufgelösten Satellitendaten des NOAA-AVHRR-Sensors und unter Einbezug von interpolierten Niederschlagsreihen sowie der demographischen Entwicklung die dominanten Landbedeckungsveränderungen und ihre mutmaßlichen Ursachen ableitet.

Der Syndrom-Ansatz wurde beispielhaft für Spanien für den Zeitraum von 1989 bis 2004 umgesetzt, und berücksichtigt die spezifischen sozio-ökonomischen Rahmenbedingungen

der mediterranen EU-Mitgliedsstaaten. Dieser geographische Raum eignet sich besonders für die Implementierung eines solchen Ansatzes, da es dort in den letzten Jahrzehnten zu großen sozio-ökonomischen Transformationsprozessen kam, die einen massiven Landnutzungswandel verursachten.

Die dominanten Landnutzungsänderungen, die in Spanien detektiert wurden, sind die Zunahme von Biomasse in marginalen Gebieten aufgrund von Sukzessionsprozessen und Aufforstungsmaßnahmen („Landfluchtsyndrom“), eine weitere Intensivierung der Landwirtschaft in den fruchtbaren Agrargebieten, die oftmals auch mit einer Ausweitung von Bewässerungsmaßnahmen einhergeht („Übernutzungssyndrom“) und als dritter großer Transformationsprozess der Trend zur Urbanisierung mit einem Schwerpunkt in den touristisch geprägten mediterranen Küstengebieten („Urbanisierungssyndrom“, „Massentourismussyndrom“).

Obwohl keine großräumige „klassische“ Landdegradation (im Sinne reduzierter biologischer Produktivität) festgestellt wurde, zeigt eine Betrachtung der Ökosystemfunktionalität, dass die erfassten Änderungen gleichzeitig positive und negative Auswirkungen auf Ökosysteme besitzen können. Nach Abwägung der Vor- und Nachteile können diese Veränderungen auch als eine Art Degradation des Ökosystems bezeichnet werden. So erhöhen Aufforstungsmaßnahmen zwar das Potential zur CO_2-Sequestierung, gleichzeitig steigt aber die Waldbrandgefahr. Neben den Veränderungen innerhalb eines Ökosystems müssen ebenso die Wechselbeziehungen zu benachbarten Ökosystemen betrachtet und bewertet werden. Angesichts des Klimawandels wird in Spanien dabei vor allem die Wasserversorgung der Landwirtschaft, des Tourismusbereiches und der Bevölkerung im Vordergrund stehen.

Im Hinblick auf die häufig kleinräumigen Landschaftsstrukturen im mediterranen Raum liegt es nahe, dass die Detektierbarkeit von Prozessen stark von der Beobachtungsskala abhängt. Anhand einer Untersuchung zweier Zeitreihen (NOAA AVHRR und Landsat-TM/ETM+) konnte der Einfluss der geometrischen und temporalen Auflösung quantifiziert werden. Es zeigte sich, dass die höhere zeitliche Auflösung von Systemen wie NOAA AVHRR und MODIS Einschränkungen durch ihre schlechtere geometrische Auflösung zum Teil zu kompensieren vermag. Allerdings bleibt die Detektion einer Reihe kleinräumiger Veränderungsprozesse auf lokal eingegrenzte Studien mit räumlich feiner aufgelösten Systemen wie Landsat, SPOT etc. beschränkt.

Der in dieser Arbeit implementierte Syndrom-Ansatz kann Verantwortlichen im Bereich der Politik und des Landmanagements wertvolle Informationen über die Ausdehnung und die räumlichen Muster von dominanten großräumigen Landnutzungsveränderungen geben, die Grundvoraussetzung sind, um die Wichtigkeit verschiedener Ökosystemfunktionen abzuwägen und Strategien für nachhaltige Landnutzung zu entwickeln und umzusetzen.

Contents

Chapter I

Introduction

1 Land degradation in the context of global change

Degradation of terrestrial dryland ecosystems, also termed desertification, is recognized as one of the major threats to global environment impacting directly on human well-being (Millenium Ecosystem Assessment 2005a). The global relevance of this issue is underlined by the fact that the "United Nations Convention to Combat Desertification" (UNCCD), adopted in 1994, has since been ratified by 193 countries. The understanding of land degradation processes including their causes and consequences on ecosystem functioning as well as the identification of affected areas and regions at risk are a prerequisite to develop strategies to mitigate and avoid land degradation by implementing appropriate management actions. According to this, over the past decades many national and international research initiatives were concerned with developing strategies to assess and monitor land degradation and desertification aiming at supporting stakeholders at all management levels ranging from local farmers, policy makers at national level to stakeholders at supranational level. Within the frame of the Millenium Ecosystem Assessment (MEA; 2001-2005) and the 10-year strategy plan of the UNCCD (2008-2018) discussions about the current status of land degradation science, its gaps and consequential necessary developments were emerging.

The land degradation and desertification subject is part of the more broadly perceived debate on global change. Thereby, climate change and its environmental and economic consequences have been and still are major environmental issues of global concern being discussed worldwide. A focal point is the assessment of anthropogenic contribution to the measured climate change as well as the definition of binding agreements, for instance concerning CO_2 emissions, which evokes intense discussions. It is assumed that since 1850 a high proportion of anthropogenic CO_2 emissions results from land use (Houghton and Hackler 2001). However, land use is often perceived and managed as a local environmental issue but viewing it in sum shows that it is rather a concern of global dimension (Foley et al. 2005).

Land use practices have not only affected global and regional climate due to the emission of relevant greenhouse gases but also by altering energy fluxes and water balance (Foley et al. 2005). Moreover, land use changes and associated alterations of habitat structure as well as release of substances like fertilizers, pesticides, air pollutants and waste water impact on various ecosystems goods and services, amongst them biodiversity, substance flows, water and air quality, soil properties and disease vectors (DeFries et al. 2004; Foley et al. 2005;

Lambin et al. 2006; Vitousek et al. 1997). In addition, even seemingly “unaffected” areas are also influenced and altered indirectly through pollutants and climate change (DeFries et al. 2004; Foley et al. 2005).

Human activities have transformed a major part of the earth’s terrestrial ecosystems to meet rapidly growing demands for food, fresh water, timber, fibre and fuel. On that account, over the past 50 years the earth's surface has changed more rapidly and extensively than ever before (Millenium Ecosystem Assessment 2005b). Recent studies suggest that human-dominated ecosystems are now covering more of the earth’s surface than (at least almost) unaffected ones (Foley et al. 2005; Vitousek et al. 1997). Ellis and Ramankutty (2008) argued that the current classification and description of biomes totally neglects or at least confines the influence of humans to few classes like urban built-up areas or cropped land. They proposed and implemented an approach which includes the human influence on ecosystems/biomes and termed them “anthropogenic biomes” or “anthromes”. The results indicate that major parts of the world are covered by biomes that are intensively or extensively influenced by humans (around 75% of the ice-free land and even ca. 90% of terrestrial net primary productivity (NPP)). This is in accordance with findings of Sanderson et al. (2002) who estimated that ca. 80% of the earth’s surface is directly or indirectly influenced by humans. The consideration of human impact is also a central point of the integrated “syndrome approach” developed by Schellnhuber et al. (1997) that seeks to map global change syndromes based on “functional patterns of human-nature interaction” (Lüdeke et al. 2004). In this framework, syndromes (as a “combination of symptoms”) describe bundles of interactive processes and symptoms which appear repeatedly and in many places in typical combinations and patterns, e.g. the damage and destruction of natural landscapes caused by large-scale projects is termed “Aral Sea Syndrome”.

It is indisputable that the use and alteration of ecosystems by humans aiming at obtaining goods and services is essential and therefore, should not be evaluated as a bad thing per se. Rather, it should be of interest to use and manage ecosystems in a sustainable way to secure the functioning of ecosystems and therewith, also the future provision of essential goods and services. Important goods and services comprise the provision of food, raw materials, fresh water and energy but also other services like disease control, flood control, recreation and maintenance of biodiversity. According to the concept developed for MEA (2005b) ecosystem services can be grouped into four major classes (compare figure 1) that influence human well-being directly or indirectly: supporting, regulating, provisioning, and cultural services.

Whereas in the past, conservation of ecosystems was given priority to maintain ecosystem services, in the face of global change it cannot be assumed that the future behaviour of ecosystem responses to changes will be the same as in the past (Chapin III et al. 2010). Instead, the challenge of future land use management will include the assessment of trade-offs between acute human needs and the long-term capacity of ecosystems to provide goods and services (DeFries et al. 2004; Foley et al. 2005). Thus, humans face the challenge to maintain the benefits of ecosystems but at the same time to reduce adverse environmental consequences. Furthermore, it will be of major importance to foster the resilience capability of ecosystems and thus, move towards a resilience-based ecosystem stewardship (Chapin III et al. 2010).

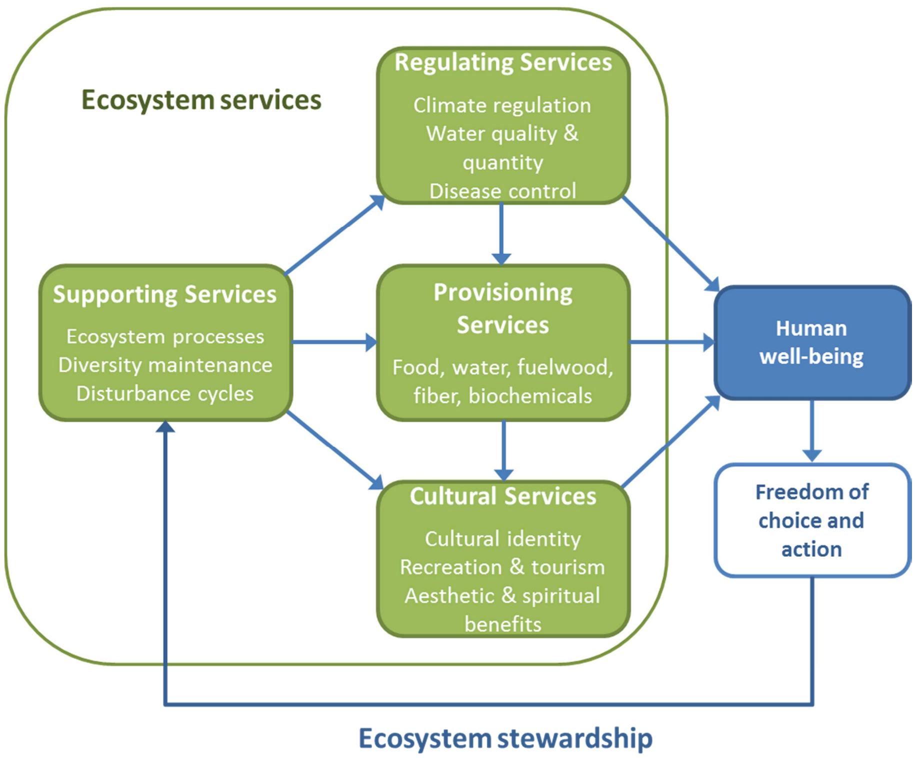

Figure 1: Linkages between ecosystem services (as defined by the Millenium ecosystem assessment (2005b)), well-being of society, and ecosystem stewardship. Supporting services are the foundation for the other categories of ecosystems services that are directly used by humans. They involve the fundamental ecological processes that control structure and functioning of an ecosystem and are often unperceived by society. Regulating services influence the provisioning services that are directly used by humans, which on the other hand affect cultural services (after Stafford Smith et al. 2009).

Sustainable management of ecosystems requires information concerning the actual conditions and furthermore, alterations of ecosystems in relation to reference states. Such information allows for a thorough analysis of ecosystem functionality and enables rating trade-offs between ecosystem services which policy decisions (where necessary considering climate change scenarios) could impose by inducing land use changes (DeFries et al. 2004). Thereby, it is essential to consider that ecosystem responses to land use changes vary in time and space and moreover, analysis should encompass covering larger areas with sufficient spatial resolution to ensure that ecosystem responses are detected. Depending on external factors like climate, geology, soil and topography the same land use changes can evoke varying ecosystem responses (DeFries et al. 2004).

Earth observation by remote sensing is one tool that essentially contributes to the assessment and monitoring of ecosystems from local to global scale. Hence, information extracted from remote sensing data can be employed (i) to assess the extent and condition of ecosystems and (ii) moreover, to monitor changes of ecosystems conditions and services over long time periods (Foley et al. 2005; Turner II et al. 2007). Thus, despite being a rather young discipline the use of earth observation data contributed fundamentally to the understanding of dynamics and responses of vegetation to climate and human interactions (DeFries 2008).

2 Dryland degradation

Even though land use changes are affecting almost all terrestrial ecosystems, drylands are considered as most vulnerable to the degradation of ecosystem goods and services. Thus, water scarcity, overuse of resources, and climate change are a much greater threat for ecosystem functioning in drylands than in non-dryland systems, and can result in a sustained, significant decline of the provision of ecosystem goods and services (Millenium Ecosystem Assessment 2005a).

Drylands cover about 41% of the earth's land surface, comprising hyper-arid to dry sub-humid climate zones and they are home to about one third of the global population (Safriel et al. 2005). Drylands are characterized by low mean annual precipitation amounts compared to potential evaporation (ratio of mean precipitation to potential evaporation less than 0.65) and furthermore, a high variability of inter-annual rainfall amounts (Safriel et al. 2005; Thomas and Middleton 1994).

Hence, water availability and the tolerance to periods of water scarcity, respectively, are key factors in dryland productivity (Stafford Smith et al. 2009). In response to water scarceness and prolonged drought periods fauna and flora of dryland ecosystems have adapted to these conditions following manifold strategies, for instance the development of drought-avoiding (annual grasses) or drought-enduring (cactuses) plant species (Hassan and Dregne 1997). In consequence of this adaptability, dryland ecosystems are characterized by a high degree of resilience and biodiversity. Even though the population levels of fauna and flora may vary noticeably in short-term due to environmental influences (e.g. extended drought periods), their long-term species composition remains relatively constant (Hassan and Dregne 1997). For thousands of years humans also developed strategies to use the goods and services provided by drylands (compare table 1) in a sustainable way, thereby considering the level of aridity. Amongst these are water harvesting techniques, crop rotation, irrigation practices, transhumance and nomadism. The countries that comprise drylands differ largely in their socio-economic development ranging from agrarian via industrialized to service oriented societies. This development stage defines to a large extent the land use systems, also termed transition state, and in consequence, the process framework of land use/cover changes (DeFries et al. 2004). Population growth, socio-economic transformations and climate change often exert increased pressure on dryland ecosystems which is beyond their resilience capacities, thus causing land degradation and desertification.

Table 1: Key dryland ecosystem services (after Millenium Ecosystem Assessment 2005a).

	Supporting Services Services that maintain the conditions for life on earth	
	- Soil development (conservation, formation) - Primary production - Nutrient cycling	
Provisioning Services Goods produced or provided by ecosystems	**Regulating Services** Benefits obtained from regulation of ecosystem processes	**Cultural Services** Nonmaterial benefits obtained from ecosystems
- Provision derived from biological productivity: food, fibre, forage, fuelwood, and biochemicals - Fresh water	- Water purification and regulation - Pollination and seed dispersal - Climate regulation (local through vegetation cover and global through carbon sequestration)	- Recreation and tourism - Cultural identity and diversity - Cultural landscapes and heritage values - Indigenous knowledge systems - Spiritual, aesthetic and inspirational services

The terms land degradation and desertification received worldwide attention in consequence of the prolonged Sahel drought in the 1970s and 1980s which caused a humanitarian catastrophe. In the framework of a UN conference on desertification in 1977 a “Plan of action to combat desertification” was approved, and in 1994 the “United Nations Convention to Combat Desertification” (UNCCD), which has by now been ratified by 193 countries, was adopted and brought into force in 1996. The definition of the terms desertification and land degradation was subject to highly controversial debates. A detailed review of this debate cannot be given here but an interesting review focussing on the changing context of this debate is given by Hermann and Hutchinson (2005). They describe the change in the perception of desertification from a mere climate-driven desert encroachment to a complex concept of degrading ecosystem functioning induced by biophysical and socio-economic drivers.

A nowadays widely accepted definition of desertification and land degradation is provided by the UNCCD. Accordingly, desertification is defined as “land degradation in arid, semi-arid and dry sub-humid areas, resulting from various factors, including climatic variations and human activities” (UNCCD 1994). Thereby, land degradation means “reduction or loss, in arid, semi-arid and dry sub-humid areas, of the biological or economic productivity and complexity of rainfed cropland, irrigated cropland, or range, pasture, forest and woodlands resulting from land uses or from a process or combination of processes, including processes arising from human activities and habitation patterns” (UNCCD 1994). This definition clearly reflects that land degradation and desertification include manifold and complex processes that cause a sustained decrease of ecosystems services and moreover, that all terrestrial ecosystems in dryland areas are at risk. Nevertheless, this also leaves room for interpretation and uncertainties concerning the terminology as outlined in Vogt et al. (2011) and hence, also arouses different perception of the processes that lie behind these two terms.

In the field of desertification science, the assessment and monitoring of land degradation affected areas has been one of the major objectives of scientists. Assessment thereby is referred to as the evaluation of a status, e.g. estimation of land condition, whereas monitoring includes a temporal component, e.g. repeating land condition assessments at defined time intervals to observe changes (Del Barrio et al. 2010). The identification of affected areas and regions at risk aims at supporting the understanding of land degradation processes and developing strategies to mitigate and avoid land degradation by taking suitable management decisions. Nevertheless, until today no comprehensive operational monitoring system could be established (Verón et al. 2006). This can be partially attributed to the complexity of the topic which requires inter-disciplinary, multi-scale approaches and moreover, to a gap between the focus of scientific research and the needs of users (Vogt et al. 2011); related to these issues, the question how to assess land degradation in a comprehensive way is still under debate.

In the past decades the understanding of the scientific community has undergone a shift concerning the key factors that are required to allow for adequate assessment and monitoring of land degradation. In the beginning assessment and monitoring was confined to the derivation of bio-physical parameters that are linked to land degradation. Underlying forces and drivers, human-induced or climate-driven were not considered in these assessments. Only in the late 1990s new theoretical concepts were developed that explicitly included climatic and human drivers as well as spatial and temporal scale issues as mandatory factors that should be considered to understand and identify land degradation processes (Vogt et al. 2011).

Puigdefabrégas and Mendizabal (1998; 2004), perceived land degradation as a process which origins from disturbances of a dryland ecosystem, climatic- or human-driven. In case disturbances exceed the resilience capacity and furthermore, counter measures cannot be applied or are not implemented, substantial loss of the carrying capacity and potential productivity is caused in consequence (figure 2).

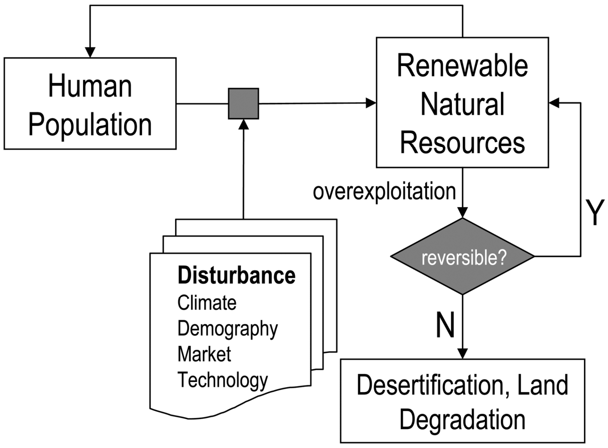

Figure 2: Impact of disturbance on the feedback loop between natural resource availability and their use/exploitation by human activities (after Puigdefábregas and Mendizabal 1998).

Numerous case studies throughout the world dealing with land degradation built the basis for a meta-analysis performed by Geist and Lambin (2004) that aimed at revealing the main mechanisms that trigger land degradation processes. Their analysis concludes in the understanding that these processes which often manifest themselves in land use/cover changes (LUCC) are governed by proximity causes (immediate human and biophysical actions) that for their part are depending on underlying drivers (fundamental social and biophysical processes). Figure 3 illustrates the dependencies of LUCC from proximate causes and underlying drivers. Furthermore, alterations of ecosystem services caused by LUCC can again alter underlying drivers, proximate causes and even external constraints, hence, resulting in a feedback loop. Policy plays an important role in avoiding positive feedback mechanisms which can accelerate unsustainable land use (Reid et al. 2006).

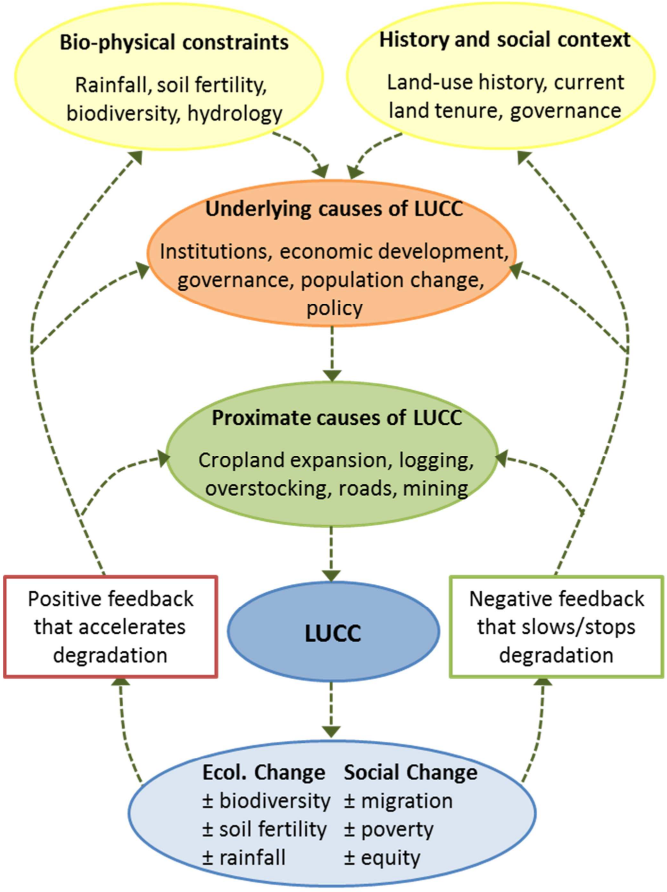

Figure 3: Conceptual model illustrating the feedback loop of Land Use/Cover Changes, its consequences and the underlying and proximate causes (after Reid et al. 2006).

A comprehensive framework designed to capture the complexity of land degradation and desertification was provided by Reynolds et al. (2007). They introduced the term “Drylands development paradigm” (DDP), which “represents a convergence of insights and key advances drawn from a diverse array of research in desertification, vulnerability, poverty alleviation, and community development” (Reynolds et al. 2007). The DDP aims at identifying and synthesizing those dynamics central to research, management, and policy communities (Reynolds et al. 2007). The essence of this paradigm, which consists of five principles, builds the assumption that desertification cannot be measured by solitary variables but that it has to include the simultaneous consideration of biophysical and socio-economic data (Vogt et al. 2011). Thereby, a limited number of “slow” variables (e.g. soil fertility) are usually sufficient to explain the human-natural system dynamics. These slow variables possess thresholds and if these thresholds are exceeded the system moves to a new state. “Fast” variables, for instance climatic variability, often mask the slow variables and thus, aggravate the assessment of the slow variables which is a prerequisite to understand the ecosystem behaviour. Moreover, it is important to consider that human-natural systems are “hierarchical, nested, and networked across multiple scales”, both the human component, e.g. stakeholders at different levels, and the biophysical component, e.g. slow variables at one scale can be affected by the change of slow variables operating at another scale (Reynolds et al. 2007).

Spatially explicit assessments that were not specifically designed in the context of land degradation but have accordingly been adapted and implemented, are for instance the concept of human appropriation of net primary production (HANPP, Erb et al. 2009a; Haberl et al. 2007) and the syndrome approach (Schellnhuber et al. 1997). HANPP represents the aggregated impact of land use on biomass available each year in ecosystems as a measure of the human domination of the biosphere. Global maps of the parameter were prepared based on vegetation modelling, agricultural and forestry statistics, and geographical information systems data on land use, land cover, and soil degradation (Erb et al. 2009a; Erb et al. 2009b; Haberl et al. 2007). In a global study Zika and Erb (2009) estimated the annual loss of NPP due to land degradation at 4% to 10% of the potential NPP of drylands, ranging up to 55% in some degraded agricultural areas.

The syndrome approach has been developed in the context of global change research (Cassel-Gintz and Petschel-Held 2000; Lüdeke et al. 2004; Petschel-Held et al. 1999). It aims at a place-based, integrated assessment by describing global change by archetypical, dynamic, co-evolutionary patterns of human-nature interactions instead of regional or sectoral analyses. In this framework, syndromes (as a “combination of symptoms”) describe bundles of interactive processes (“symptoms”) which appear repeatedly and in many places in typical combinations and patterns. Sixteen global change syndromes were suggested and distinguished into utilisation, development and sink syndromes. Geist and McConnell (2006) summarized that the objective of the approach is to define the mechanisms of environmental changes with a high level of generality based on expert knowledge related to local cases. Downing and Lüdeke (2002) applied the approach to land degradation. Out of the set of global change syndromes they identified the syndromes that are of relevance in dryland areas and linked vulnerability concepts to degradation processes. The syndrome concept is considered as a suitable interpretation framework that allows for an integrated assessment of land degradation (Sommer et al. 2011; Verstraete et al. 2011).

The paradigm shift in the concept of land degradation from the mainly biophysical perspective to a multi-disciplinary approach is also reflected in the focus of the scientific projects promoted within the framework programmes of the European Union like, for instance, MEDALUS I-III (Mediterranean desertification and land use, 1991-1999), ModMED I-III (Modelling Mediterranean ecosystem dynamics, 1994-2001) and DESERTLINKS (2001-

2004). With a main focus on remote sensing, DeMon (Satellite Based Desertification Monitoring in the Mediterranean Basin, 1992-1995) focussed on the identification of spatial indicators that proved to be suitable to assess land degradation processes based on remotely sensed data. Thereby, vegetation- and soil-related parameters appeared to be the most appropriate surrogates (Hill et al. 1995; Pickup and Chewings 1994). In subsequent projects such as DeMon II (1996-1999) and GeoRange (Geomatics in the Assessment and Sustainable Management of Mediterranean Rangelands, 2001-2003) the assessment of human-nature systems was considered evermore important and led on to the development of integrated multi-disciplinary concepts. Thus, remotely sensed patterns of land degradation processes were linked to socio-economic and geographic data to identify the determinants of ecosystem change (Hostert et al. 2003; Röder et al. 2007; Röder et al. 2008). Whereas these projects aimed at setting up operational observation methods at local to regional scale, projects like LADAMER (Land degradation assessment in Mediterranean Europe, 2002-2005) and DeSurvey (A Surveillance System for Assessing and Monitoring of Desertification, 2005-2010) were concerned with developing integrated frameworks including expert knowledge from all kinds of disciplines to assess land degradation at regional to global scale, partially in response to the European Global Monitoring for Environment and Security (GMES) initiative. In addition to the scientific progression, there was increasing awareness that dissemination of results to end-users from local to international level as well as their integration in the design of the products is a crucial aspect of land degradation research.

3 The contribution of remote sensing to dryland observation

Various scientific disciplines contribute valuable information that enhances the understanding of land degradation and desertification at different temporal and spatial scales. These include studies ranging from the plot scale to global assessments as well as the collection of biophysical or socio-economic data and the implementation of models to predict land use changes in future decades. Exhaustive coverage of these methods is beyond the scope of this thesis. Rather, a short overview of the contribution of remote sensing to set up a dryland monitoring system shall be given. This section briefly outlines considerations regarding the requirements to set up a comprehensive observation system for dryland areas and the contribution of earth observation data.

Due to the nature of dryland areas, long term observation data are necessary that are capable of observing the long-term status of ecosystems and reveal human-induced land cover changes affecting these areas. It is of major relevance that the observation period captures the climatic variability (Reynolds et al. 2007; Vogt et al. 2011). In this context, remote sensing data offer the possibility to obtain long term observations across large areas in an objective and repetitive manner (Graetz 1996; Hill et al. 2004).

Consequently, three major components are of particularly importance to set up a comprehensive observation system for dryland areas that can support decision makers to understand consequences of their actions and help to prevent unsustainable use of ecosystems goods and services:

- assessment of actual land condition, i.e. the capacity of an ecosystem to provide goods and services
- monitoring of land cover changes and assessment of their implications for land condition and, in consequence, the functioning of ecosystems
- separating climate-driven and human-induced land use/cover processes

Neither the condition of ecosystems nor the processes affecting them can directly be measured by earth observation data. Rather, suitable indicators have to be identified (Verstraete 1994) that (i) can be related to the status and processes and (ii) can be derived in standardized and replicable way.

A great variety of ecological, economic, social and institutional indicators were proposed to monitor the condition and degradation of drylands (Brandt et al. 2003). The proposed ecological parameters include, amongst others, soil properties including soil compaction, increased rate of movement of soil particles and corresponding change in the soil-bedrock relation as well as parameters related to vegetation such as vegetation cover, plant composition and land use.

Earth observation systems nowadays comprise a great variety in terms of spatial, temporal and spectral resolution of the provided data (Hill et al. 2004). In consequence, for remote sensing data a range of approaches and models were developed that allow for the derivation of a variety of biophysical parameters that are appropriate for employment in dryland observation (Hill 2008; Lacaze 1996). Depending on the spatial and spectral characteristics of the remote sensing data these qualitative and quantitative measures include vegetation indices related to greenness, vegetation cover, landscape metrics, pigment and water content, soil organic matter of the topsoil etc. (Blaschke and Hay 2001; Hill et al. 2004; Jarmer et al. 2009; Small 2004; Ustin et al. 2009).

One of the key factors that describe the functioning of an ecosystem and is also referred to in the definition of desertification and land degradation of the UNCCD, is the biological productivity of ecosystems (Del Barrio et al. 2010). Parameters related to greenness, vegetation cover and biomass can thus serve as proxies to assess and monitor the productivity of ecosystems.

Considering the need of long-term observations to adequately assess and monitor dryland areas, sensors which provide continuous long-term observations of the earth' surface, are the preferable choice. As mentioned above, numerous space-born sensors exist, but only few satellites are fulfilling the two criteria of collecting data (i) covering a long time period and (ii) providing a regular global coverage. These systems can be distinguished in two major groups, the first one providing coarse scale resolution data with a high temporal resolution, the second working at medium spatial resolution at the expense of temporal resolution.

Coarse scale sensors like NOAA AVHRR, SPOT Vegetation, MODIS and MERIS acquire images of the earth's surface on a daily basis providing a spatial resolution ranging from 250 to 1000 m. NOAA AVHRR provides the longest back-dating global time series spanning from 1981 to 2006: the "Global Inventory Modelling and Mapping Studies" (GIMMS) data set with a pixel size of 8x8 km², which consists of bi-monthly measurements of the NDVI (Tucker et al. 2005).

An alternative NOAA AVHRR dataset covering the Mediterranean is the "Mediterranean Extended Daily One-km AVHRR Data Set" (MEDOKADS). The data archive consists of 10-day maximum value composites of full resolution NOAA AVHRR channel data and supplementary data. It covers the whole Mediterranean region from 1989 to 2004 with a spatial resolution of about 1 km² (Koslowsky 1996, 1998).

A prerequisite for long-term observation analysis are well-calibrated data archives. This is especially demanding in case of the NOAA AVHRR data archives as pre-processing comprises the correction of effects caused by orbital drift of the sensor (i.e. changing overpass time) as well as the inter-calibration of the spectral channels between the different AVHRR sensors employed to create the archives. A detailed description of the thorough pre-processing scheme applied to create for instance the MEDOKADS archive may be found in section 2.1 of chapter IV, Furthermore, a comprehensive explanation of the dataset is given in Koslowsky (1996, 1998) and Friedrich (2009).

Due to the limited spectral properties of the NOAA AVHRR sensor, the derivation of biophysical parameters is limited. Usually the Normalized Difference Vegetation Index (NDVI) is employed, a commonly used vegetation index calculated from the red and near-infrared spectral information which is related to greenness (Rouse et al. 1974; Tucker 1979).

The NDVI is known to have weaknesses due to its sensitivity to soil background especially when vegetation cover is low (Price 1993). Nevertheless, for temporal analysis it seems to be of more decisive importance to employ a robust and consistent measure (Udelhoven and Hill 2009). Thus, results of a comparison of trend analysis of several simple and enhanced vegetation-related parameters derived from Landsat TM/ETM+ time series indicated that the derived trends from linear regression only differ slightly (Sonnenschein et al., 2011).

Data products derived from the MODIS-sensor provide a better spatial and spectral resolution and consequently, enable the derivation of more enhanced biophysical surrogates that can be employed in land degradation observation. Moreover, the sensor properties facilitate the provision of a consistent high quality data archive. However, in comparison to the NOAA AVHRR data sets the archive is confined to a shorter observation period starting in 2000. Hence, despite its weaknesses, the longer retrospective view of the NOAA sensor makes it a valuable information source that should not be abandoned.

Landsat TM and ETM+, respectively are providing data of the earth’s surface with a spatial resolution of 30 m since 1982 (Goward and Masek 2001). The Landsat program consists of a series of optical sensors that record the reflected radiance in six spectral bands (complemented by one band in the thermal domain) ranging from visible to middle infrared which allows the derivation of several surrogates related to vegetation properties (Fang et al. 2005). The temporal revisit rate of the sensor is 16 days and could theoretically provide a time series of earth observations with similar density compared to those provided by coarse scale sensors. But also in many dryland areas cloud cover impedes the acquisition of utilizable images. Thus, often only few images of sufficient quality can be acquired per season. The creation of such a time series is complicated by the fact that the images should originate from comparable phenological stages. Therefore, many studies investigating time trajectories of vegetation based on Landsat time series are confined to one observation per season (Hostert et al. 2003; Röder et al. 2007; Röder et al. 2008). Furthermore, to reduce noise imposed from the observation geometry long term monitoring requires accurate geometric and radiometric correction of the data to enable a meaningful analysis, necessitating a rigorous pre-processing scheme for all the time series images (Röder et al. 2007; Röder et al. 2008).

Although both sensor types provide data that allow for adequate dryland observation, considering the special needs of an operational observation framework trade-offs between geometric and spectral level of detail, areas covered and temporal resolution have to be balanced.

3.1 Assessment of land condition

Land degradation maybe defined as a long-term loss of an ecosystem's capacity to provide goods and services. Therefore, as outlined in the preceding section a major component of a comprehensive dryland observation system is the assessment of land condition which can be linked to ecosystem status. Even though land degradation is recognized as a severe threat, only few global land degradation assessments have been carried out until today (Millenium Ecosystem Assessment 2005a; Vogt et al. 2011).

The first global assessment of land quality was provided in the frame of the GLASOD project (Global assessment of Human-Induced Soil Degradation, 1987-1990) where human-induced soil degradation (extent, type and grade) was mapped at a scale of 1:10 million based on expert judgement (Oldeman et al. 1990). Another well-known global assessment was provided by Dregne and Chou (1992) who also integrated information on vegetation status based on secondary sources. Whereas the map provided by GLASOD indicated that 20% of soils in drylands are degraded, the approach of Dregne and Chou estimated that 70% of dryland areas are affected either by degradation of soil or vegetation. A more recent study (Lepers 2003) prepared for the Millenium Ecosystem Assessment covered about 60% of all dryland areas. Several data sources including remote sensing data were integrated in the analyses and land degradation was estimated at 10%. One of the major points of criticism concerns the subjectivity of these studies which impedes their operational use or comparability (Millenium Ecosystem Assessment 2005a).

In recent years approaches have been developed that aim at bridging this gap. These approaches base the assessment of land condition primarily on biological productivity of ecosystems. Briefly speaking, the concept is based on the fact that land degradation which might be caused by a wide variety of climate- and human-induced processes, results in a decline of soil potential to sustain plant productivity (Del Barrio et al. 2010). Using the example of rangelands, figure 4 clearly illustrates the dependence of biological productivity from grazing pressure, rainfall and soil properties. In this respect, soil properties like water holding capacity and nutrient supply are essential factors that directly affect primary productivity. On-going overgrazing drives feed-back loops between vegetation and soil, resulting in a degradation of these soil properties that involves a sustained decrease of the soil's capacity to sustain primary productivity. In consequence, the ecosystem's capacity to utilize local resources like soil nutrients and water availability in relation to the potential capacity can be called land condition. This in turn allows drawing conclusions on the degradation status of observed areas (Boer and Puigdefabregas 2005). Hence, biological productivity is considered a suitable surrogate to assess land condition. Vegetation-related parameters derived from remote sensing are surrogates of primary productivity and thus, provide the opportunity to assess land condition and its development for large areas.

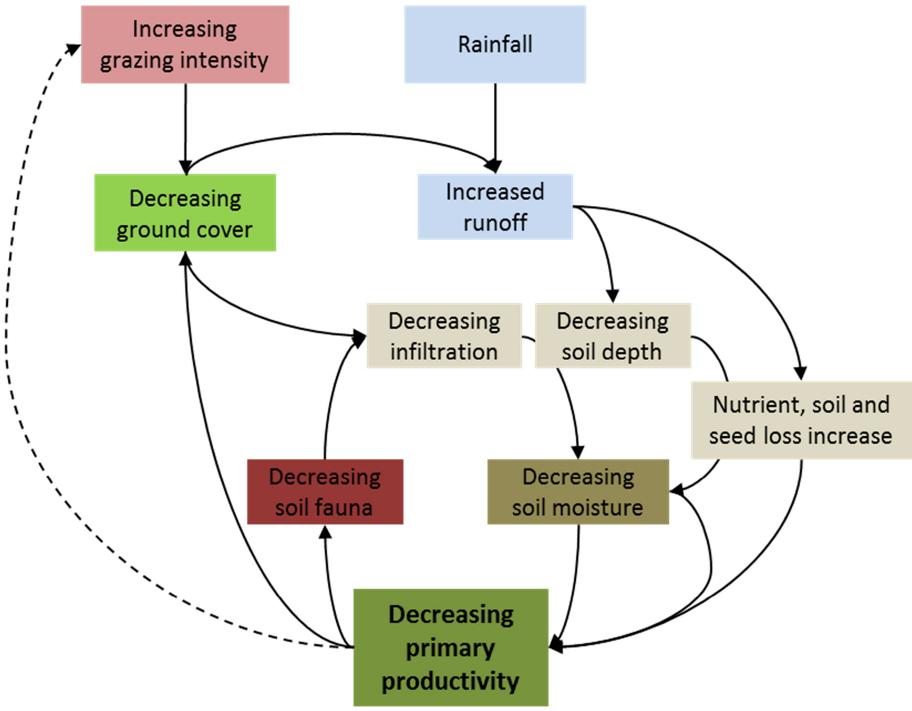

Figure 4: Aspects of landscape function using the example of grazing. Changes of ground cover which are in short-term mainly driven by rainfall variability and grazing pressure can affect soil properties negatively. If thresholds are crossed the cycle moves towards a new state characterized by degraded soil characteristics and a long-term loss in productivity. Even though a negative feedback exists to grazing intensity, management interventions often weaken this mechanism by maintaining constant stock numbers (after Stafford Smith et al. 2009).

Boer and Puigdefabrégas (2005), for instance, conceptualized and implemented a spatial modelling framework to assess land condition at local scale based on climate data as well as on NDVI data derived from the Landsat sensor. The approach is based on the assumption that besides the fact that water availability is the major limiting factor of productivity in arid and semi-arid areas, local water balance, which results from rainfall, soil properties and vegetation, reflects land condition. Based on this theoretical concept they proposed a long-term ratio of mean actual evapotranspiration to assess land condition.

At coarse scale, approaches were developed recently that make use of the concept of rain use efficiency which was introduced by Le Houérou (1984). RUE is defined as ratio of NPP to precipitation over a given time period and may be interpreted as “proportional to the fraction of precipitation released to the atmosphere” (Del Barrio et al. 2010). In the framework of the LADA project (Land Degradation Assessment in Drylands) Bai et al. (2008) proposed a methodology to assess and monitor land condition by deriving RUE based on the global NOAA AVHRR GIMMS dataset. The implemented methodology and the results were criticized (Wessels 2009) because the specific approach does not take into account that rainfall is not a limiting factor in more humid areas and moreover, RUE values are dependent from precipitation amounts and thus, impede the direct comparison of RUE values from regions of diverging aridity level.

Del Barrio et al. (2010) presented an alternative concept which takes this dependency of RUE on aridity into account. The approach was implemented for the Iberian Peninsula based on 1 km NOAA AVHRR NDVI data and spatially interpolated climate data. Due to the fact that the Iberian Peninsula shows a strong climate gradient, the derived RUE values were in a first step de-trended for aridity to ensure the comparability of the derived data between different climatic zones. In a next step, statistically derived boundaries of minimum and maximum RUE were employed to calculate relative RUE values. Based on the assumption that healthy and undisturbed vegetation is characterized by a maximum RUE value the relative RUE can be treated as a measure for land condition. Besides assessing the land condition based on RUE the study furthermore comprises monitoring of changes in primary productivity considering effects caused by climatic changes. Results of the study indicate that land condition of the Iberian Peninsula is better than expected. Changes of the productivity suggest that areas already in good conditions improve further whereas degraded areas are static. This approach proofed to be transferable to other regions and thus, bears the potential to build an important component of a comprehensive observation system for drylands. The complete study of Del Barrio et al. (2010) can be found in Appendix A of this thesis. Appendix B comprises a study where this specific approach was transferred to a study area in Inner Mongolia, China (Hill et al. 2010).

3.2 Assessment of land use/cover changes

The approach developed by Del Barrio et al. (2010) provides a comprehensive and operational methodology to characterize land degradation affected areas based on the assessment of ecological responses of vegetation to physical drivers. Another essential component of a dryland observation system is, as outlined at the beginning of this chapter, the assessment of land use/cover changes, their underlying causes and their consequences for ecosystems functioning.

Thereby, land cover is defined by the attributes of the land surface including all aspects such as flora, soil, water and anthropogenic surfaces (Lambin et al. 2006). For thousands of years humans have altered landscapes aiming at improving the amount, quality and security of ecosystem goods and services (Ramankutty 2006). In consequence, land use has been defined as the purpose for which humans employ land cover (Lambin et al. 2006). Changes of land cover always imply changes in ecosystem functions such as for instance primary productivity, soil quality, water balance and climatic regulation (Foley et al. 2005; Steffen et al. 2004; Turner II et al. 2007). The assessment of landscape dynamics forms therefore an essential component for dryland observation as it provides information about the nature and extent of the changes, and accordingly, allows for the evaluation of the consequences for ecosystem functions. More specifically, remote sensing techniques can usually determine land cover but not land use (Lambin et al. 2006). Only by integrating additional information sources, mostly provided by ground truth data, land use can be associated with land cover (Lambin and Geist 2006).

Land cover changes can be distinguished in two major groups: (i) conversions and (ii) modifications (Coppin et al. 2004; Lambin and Geist 2006). Land use transformations usually involve the replacement of one land use/land cover class with another one (e.g. shrub lands with arable land) whereas modifications are usually related to gradual changes within one thematic class (e.g. shrub encroachment within natural ecosystems). The assessment of both conversions and modifications is important to provide a comprehensive picture of land use/cover changes. In particular, the assessment of modifications is a crucial element in dryland areas because land cover changes related to land degradation are often associated with modifications of the landscape (Lambin et al. 2006; Lambin and Geist 2001), these include for instance vegetation cover loss due to overgrazing or primary or secondary succession on abandoned fields and rangelands.

The assessment of land use/cover conversions is often based on land use change detection performed at defined years of interest. Several strategies and methods were developed to optimise the results of change detection analyses. Detailed overviews of change detection techniques and their application, potentials and limits are given in Lu et al. (2004) and Coppin et al. (2004).

The detection and monitoring of modifications is often more challenging because changes of biophysical properties have to be observed and distinguished from inter-annual variability. This is especially important for dryland areas where primary productivity is dependent on the highly variable climatic conditions in terms of rainfall (Turner II et al. 2007). A suitable means to assess gradual changes of land cover is provided by time series analysis of remote sensing archives (Udelhoven 2006; Udelhoven 2010), because it provides tools to delineate inter-annual variability from long-term trends. Time series analysis of land cover based on remote sensing data archives requires consistent long-term series of biophysical parameters connected to surface properties. As outlined in chapter I (section 3) surrogates linked to vegetation performance are suitable parameters that describe ecosystem properties and therefore, are often employed to investigate modifications of land cover. It was already mentioned before that at present only two broad remote sensing data sources, are qualified for long-term observations which at the same time meet the condition to regularly cover the entire earth. These are medium spatial resolution data provided by the Landsat sensors and coarse scale resolution earth observation archives provided by sensors like NOAA AVHRR and MODIS.

Whereas the Landsat data provide high geometric detail of vegetation cover at local scale and often match the scale of driving management practices (Cohen and Goward 2004; Lambin et al. 2006), coarse scale resolution data are more suitable to cover large areas, because they have a much higher temporal repetition rate. These data archives therefore permit the detection of changing parameters connected to vegetation cover as well as the deduction of changes in phenology (Andres et al. 1994; Bradley and Mustard 2008; Brunsell and Gillies 2003). The trade-offs between the two data types and the resulting implications will be more thoroughly studied in chapter II of this thesis.

A wealth of studies exists that assess landscape dynamics in dryland areas based on these data sources. Using a variety of data processing and analysis concepts, these give useful insights on process-patterns on regional to global (Eklundh and Olsson 2003; Geerken and Ilaiwi 2004; Herman et al. 2005; Lambin and Ehrlich 1997; Tucker and Nicholson 1999; Wessels et al. 2007) as well as local scales (Hill et al. 1998; Paudel and Andersen 2010; Petit et al. 2001; Röder et al. 2007; Röder et al. 2008; Washington-Allen et al. 2006).

A meaningful interpretation of time series results also has to include land use information (Vogt et al. 2011). With regard to land degradation the decrease of primary productivity not necessarily implies land degradation processes (e.g. transformation of forest to arable land is associated with decreasing primary productivity but can be nevertheless profitable and sustainable) and in turn, an increase of productivity is not always a consequence of improving land condition (e.g. bush encroachment in abandoned rangelands impacts several ecosystem functions negatively and can be considered as degradation).

Besides the pure assessment of land use/cover changes by monitoring biophysical parameters, the identification of the underlying causes is a crucial element. Time series analysis allows for the discrimination of “real” land cover changes and changes caused by inter-annual variability but moreover, the underlying and proximate causes of human-induced changes have to be identified (Geist 2005; Reynolds et al. 2007). Only in this manner the coupled human-natural character of land cover changes can be understood and an identification of the mechanisms that drive land degradation is possible. This knowledge provides the foundation to support the development of sustainable land management strategies.

In Australia, where extensive rangelands are used for grazing, operational monitoring systems covering major parts of the continent have been set up during the last decades (Wallace et al. 2006; Wallace et al. 2004). Several regional projects use parameters derived from Landsat time series to monitor land cover changes and land condition, which are integrated in the Australian Collaborative Rangeland Information System (ACRIS) on a nationwide level and serve as information source for farmers as well as for national institutions (Bastin et al. 2009). Several prevailing local studies aimed at linking the biophysical dimension of land use/cover changes to the human dimension based on several techniques (Améztegui et al. 2010; Lorent et al. 2008; Millington et al. 2007; Serra et al. 2008). Recently, LADA has been implementing a global land degradation information system (GLADIS) which provides information on land degradation with a spatial resolution of 8x8 km². Their interpretation of ecosystem changes including RUE, NPP and climatic variables bases on an integrated land use system map. This map entails information about the main proximate causes of land use/cover changes such as land cover, livestock pressure and irrigation. The major drawbacks of this approach concern the derivation of the RUE and the NPP from the GIMMS NOAA AVHRR dataset (Wessels 2009) and moreover, the spatial resolution that hampers the detection of land cover changes (Vogt et al. 2011). Nevertheless, Vogt et al. (2011) emphasize that this assessment is a first important step towards an integrated assessment.

4 Objectives and Methods

The preceding sections illustrate that the definition and perception of land degradation and desertification have undergone a substantial transformation. While in the beginning the biophysical assessment of degradation processes which often focussed on soil degradation was the primary objective of many research initiatives, in recent years the necessity to look at the mechanisms of human-environmental systems as a prerequisite to create a comprehensive understanding of land degradation processes has been recognized. This is considered essential to understand the impacts of land degradation on the provision of ecosystem goods and services and thus, its impact on human well-being (Millenium Ecosystem Assessment 2005a). This development was fostered by the establishment of different conceptual frameworks (compare chapter I, section 2) raising awareness that multidisciplinary approaches have to be established to assess both the bio-physical and human nature of land degradation. In recent years great efforts were put into developing methodologies to enhance the understanding of coupled human-environmental systems. Thereby, earth observation has proved a useful discipline to provide surrogates for important biophysical indicators, like for instance primary productivity, that consequently can be employed to long-term observation of land condition and land cover changes at different spatial and temporal scales (compare chapters I, sections 3.1 and 3.2). One of the major challenges remains, which is to link these observations with socio-economic data to connect the mere biophysical and socio-economic information into combined information of the land change processes and their underlying causes. This proves especially crucial for large scale assessments of land degradation from a national to global scale due to its complex nature.

The work presented in this thesis was implemented within the EU-funded projects LADAMER and DeSurvey (compare chapter I, section 2). Both projects aimed at providing comprehensive approaches to capture land degradation and desertification across large areas, and under consideration of their complexity ranging from the assessment of land condition (Del Barrio et al. 2010) and land cover change processes to the implementation of complex land use change models (Van Delden et al. 2010) and vulnerability concepts (Ibáñez et al. 2008) in a multi-disciplinary consortium.

The main research intention of this study was to contribute to the establishment of a large-scale framework for an integrated, consistent and widely applicable observation system that particularly considers the dominant process background of drylands. The syndrome concept was considered a suitable approach to integrate time series analysis results with other bio-physical and socio-economic data to derive major human-induced transformations related to land degradation. The approach developed serves as a complementary methodology to the 2dRUE approach developed by Del Barrio et al. (2010) which was also developed in the framework of the two EU-projects. Based upon the considerations of the preceding chapters with regards to the requirements of dryland observation, the aim to set up a reproducible analytical approach was pursued by following objectives:

1. Analysis of the possibilities and limits of monitoring land system change processes with remotely sensed data at different spatial and temporal scales

2. Distinction of climatic fluctuations and socio-economic forcing
3. Implementation of a regional monitoring framework that is capable to identify land cover change based on hypertemporal remote sensing time series (conversions and modifications)
4. Expanding the remote sensing based syndromes approach by incorporating other data sources

Each of these objectives was addressed by specific journal publications which are presented in chapters II to V.

The first objective was pursued by comparing the results of time series analysis of a Landsat TM/ETM+ time series and the NOAA AVHRR MEDOKADS archive for a study area in Greece. The intention of this study was to characterize the impact of spatial and temporal scale on the patterns resolved and in this way evaluate the potentials and limits of NOAA-AVHRR and Landsat-based data archives regarding the assessment of land use and land cover change. This is of high relevance to enable the assessment of the constraints of products derived from these data archives and thus, be able to interpret the derived results in a meaningful way. This is immensely important in relation to the dissemination of results.

A major concern of assessing and monitoring dryland areas is the distinction between land cover changes, especially modifications, caused by human intervention and by climatic fluctuations (objective 2). In the study presented in chapter III a quantitative link between a vegetation index and rainfall as well as temperature anomalies was established based on distributed lag models (Eklundh 1998). The study aimed at assessing the variability of NDVI anomalies that is explained by rainfall and temperature patterns and moreover, to derive the duration of the dependency. This approach was implemented for monthly and annual data for the Iberian Peninsula for the observation period 1989 to 1999.

In chapter IV a framework was set up to identify the spatial extent of major degradation-related land use change syndromes for the Iberian Peninsula based on coarse scale hypertemporal remote sensing data for the observation period 1989 to 2004. Based on the MEDOKADAS archive a first attempt tried to interpret patterns of land cover changes in semi-natural areas in relation to potential socio-economic drivers and subsequently to identify the spatial extent of major degradation-related land use change syndromes.
Chapter V expanded this syndrome approach by explicitly incorporating climatic and socio-economic data. Results of the NDVI-rainfall relationship served to distinguish areas where NDVI variability is explained to a high degree by climatic variability and furthermore, population density data which were used as a proxy for socio-economic developments in Spain. All these objectives were pursued for test areas situated in the European part of the

Mediterranean basin. This region is particularly appropriate for the development of prototypes because the region has undergone major political and socio-economic transformations within the last century. With the accession to the European Union (EU) countries like Portugal and Spain have moved within a short period from low economic development levels compared to other western European countries to industrialized societies. This development was strongly driven by policy instruments of the EU, such as the Common Agricultural Policy (CAP), the European Regional Development Fund, the European Social Fund and the Cohesion Fund. Comprehensive literature of the physical, historical and socio-economic developments of the Mediterranean and the Iberian Peninsula, respectively, can be found in Wagner (2001) and Breuer (2008) (both in German). Moreover, the Mediterranean is considered most prone to adverse impacts in Europe in the face of climate change (Schröter et al. 2005) making it necessary to develop management strategies that foster sustainable land use. Another important fact that should not be omitted is the wealth of existing studies that support the evaluation and interpretation of the achieved results.

Chapter II

Dryland observation at local and regional scale - comparison of Landsat TM/ETM+ and NOAA AVHRR time series

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Dryland observation at local and regional scale — Comparison of Landsat TM/ETM+ and NOAA AVHRR time series

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ABSTRACT

During the last few decades, many regions have experienced major land use transformations, often driven by human activities. Assessing and evaluating these changes requires consistent data over time at appropriate scales as provided by remote sensing imagery. Given the availability of small and large-scale observation systems that provide the required long-term records, it is important to understand the specific characteristics associated with both observation scales. The aim of this study was to evaluate the potentials and limits of remote sensing time series for change analysis of drylands. We focussed on the assessment and monitoring of land change processes using two scales of remote sensing data. Special interest was given to the influence of the spatial and temporal resolution of different sensors on the derivation of enhanced vegetation related variables, such as trends in time and the shift of phenological cycles. Time series of Landsat TM/ETM+ and NOAA AVHRR covering the overlapping time period from 1990 to 2000 were compared for a study area in the Mediterranean. The test site is located in Central Macedonia (Greece) and represents a typical heterogeneous Mediterranean landscape. It is undergoing extensification and intensification processes such as long-term, gradual processes driven by changing rangeland management and the extension of irrigated arable land. Time series analysis of NOAA AVHRRR and Landsat TM/ETM+ data showed that both sensors are able to detect this kind of land cover change in complementary ways. Thereby, the high temporal resolution of NOAA AVHRR data can partially compensate for the coarse spatial resolution because it allows enhanced time series methods like frequency analysis that provide complementary information. In contrast, the analysis of Landsat data was able to reveal changes at a fine spatial scale, which are associated with shifts in land management practice.

1. Introduction

Over the past 50 years the earth's surface has changed more rapidly and extensively than ever before (Millenium Ecosystem Assessment, 2005b). There is compelling evidence that the alteration of almost all terrestrial ecosystems is largely due to human activities (Steffen et al., 2004; Vitousek et al., 1997).

Drylands make up 40% of the earth's land mass and support nearly 2 billion people (White & Nackoney, 2003). Despite low and highly variable rainfall drylands support the majority of the world's livestock, produce significant amounts of cereals, roots and tuber crops, and are centres of biodiversity. Drylands, which comprise arid, semi-arid and sub-humid climate zones, are highly vulnerable to land degradation processes (Millenium Ecosystem Assessment, 2005a). As these processes often take place over long time periods, dryland observation requires explicit consideration of the temporal dimension. In this context, a particularly important issue is to identify areas affected by land use conversions and modifications (Lambin et al., 2006). Remote sensing offers the opportunity to monitor large land surface areas with a high repetition rate. Remotely sensed parameters related to vegetation cover/biomass are suitable surrogates to detect landscape processes, because change of vegetation amount can be linked to the underlying causes, e.g. socio-economic or climate change. Time series analysis is an appropriate tool for vegetation monitoring, in particular as it may differentiate between actual changes and inter-annual variations of phenology caused by external factors, e.g. changing climatic conditions (Turner et al., 2007). Moreover, it allows for the discrimination of gradual and abrupt changes in vegetation cover and, given the time series is dense enough, the detection of changing phenology as an indication of changing plant communities (Andres et al., 1994; Brunsell & Gillies, 2003). To increase the reliability of the derived trend parameters and to decrease the influence of phenological variations it is favourable to use data covering a long time period and providing information at dense time steps.

Two long-term archives of satellite data meeting these requirements are available. Since 1984 Landsat TM and ETM+ provide data with a geometric resolution of 30 by 30 m, while on regional to global

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scale NOAA AVHRR acquires data at 1 km^2 geometric resolution. Sensors with better spectral and geometric properties, like the ASTER sensor at local scale, or MODIS, MERIS and SPOT Vegetation at regional scale, are so far limited to only short time spans up to 10 years.

Landsat time series have been employed to detect land cover changes by means of time trajectories. Studies of Viedma et al. (1997, 2006), Röder et al. (2008a) as well as Kennedy et al. (2007) addressed the detection of disturbances, fire and logging events respectively whereas Hostert et al. (2003a) and Röder et al. (2008b) monitored subtle changes in vegetation cover to evaluate the impact of changing grazing management practices by means of trend detection in Mediterranean study areas. In Australia, where extensive rangelands are used for grazing, operational monitoring systems have been set up covering major parts of the continent during the last decades (Wallace et al., 2006, 2004). Several projects, e.g. the Land Monitor Project (Caccetta et al., 2000), Queensland's Statewide Landcover and Trees study (Danaher et al., 1998) and the National Carbon Accounting — Land Cover Change Project (Caccetta et al., 2007) use parameters derived from Landsat time series to monitor land cover changes and land condition, and the State level programmes are integrated in the Australian Collaborative Rangeland Information System (ACRIS) on a nationwide level (Bastin et al., 2009). The software package “VegMachine” is a simple tool developed for pastoral producers where satellite imagery and expert knowledge are combined to assess the health status of grazing grounds and, thus, support management decisions (CSIRO, 2009). It is very likely that the application of Landsat TM time series will further increase due to the free distribution of the Landsat data archive by the end of 2008 by USGS and advances in automated data pre-processing chains.

At regional scale there is abundant literature documenting the use of NOAA AVHRR data archives to perform time series analysis in regions all over the world and addressing a wide range of applications. Besides trend analysis more enhanced methods like frequency analysis were implemented to exploit phenological information as well (e.g. Eklundh & Olsson, 2003; Geerken & Ilaiwi, 2004; Herman et al., 2005; Heumann et al., 2007; Lambin & Ehrlich, 1997; Olsson et al., 2005; Piao et al., 2009; Tucker & Nicholson, 1999; Wessels et al., 2007).

Although both sensor types provide data that allow for land use change detection in general, there is a trade-off between geometric and spectral level of detail, area covered and temporal resolution. On the one hand, the Landsat data provide high geometric detail of vegetation cover at local scale and often match the scale of driving management practises (Cohen & Goward, 2004). On the other hand, the coarse scale resolution data acquired by the NOAA AVHRR sensors have a much higher temporal repetition rate and therefore permit the detection of changing parameters connected to vegetation cover as well as the deduction of changes in phenology. It is therefore of great importance to characterize potentials and limitations associated with the assessment of land use conversions and modifications using both systems.

The problem of the appropriate geometric scale for the detection of specific patterns by means of satellite data was addressed in several remote sensing studies. For example, high resolution data were degraded to simulate the effect of a coarser geometric resolution on pattern recognition and landscape parameters (Benson & MacKenzie, 1995; Justice et al., 1989; Kavzoglu, 2004; Pax-Lenney & Woodcock, 1997; Townshend & Justice, 1988). To the contrary, a smaller number of studies explicitly included the temporal dimension in their studies. Townshend and Justice (1988) provided a bi-temporal study where Landsat MSS data were degraded to a range of resolutions to recommend a suitable geometric resolution of the MODIS instrument. Goetz (1997) compared NDVI, surface temperature and biophysical variables for a mixed grassland site for NOAA AVHRR, Landsat TM and SPOT-HRV data and found that the temporal evolution of atmospherically corrected NDVI images was consistent among sensors. Bradley and Millington (2006) investigated biomass burning regimes in Bolivia and Peru taking into account spatial and temporal issues. In this study fire data of different coarse scale fire products were compared to multi-temporal Landsat TM data covering a time period from 1987 to 2000. For the period from 1985 to 2006 Pouliot et al. (2009) examined NDVI trends from 1 km AVHRR data for Canada and showed that the positive trend of NDVI detected for parts of Canada could also be detected in corresponding Landsat scenes. Ludwig et al. (2007) developed a scaling procedure to choose the appropriate scale for assessing landscape health.

However, no study exists where time series analysis was performed at different scales to evaluate the interaction of spatial and temporal scale in a systematic manner for a study area. In this paper, we provide information on the comparability and applicability of two important and commonly used satellite data archives for land cover monitoring.

This intention will be addressed using time series of the Normalized Difference Vegetation Index (NDVI), a spectral index related to green vegetation (Rouse et al. 1974; Tucker, 1979), derived from Landsat TM/ETM+ and NOAA AVHRR archives, which were compared for a heterogeneous Mediterranean study area in Central Macedonia (Greece). NDVI was chosen due to the limited spectral resolution of the NOAA AHRR sensor, which constrains the derivation of enhanced vegetation parameters. Enhanced parameters are usually derived from the Landsat TM series to monitor changing vegetation condition, for instance fractional vegetation cover derived using spectral mixture analysis. The Landsat TM time series comprises the period from 1984 to 2000 and consists of annual data acquired within a time window that represents the period of maximum photosynthetic activity of semi-natural areas Röder et al. (2008b). At regional scale the “Mediterranean Extended Daily One Km AVHRR Data Set” (MEDOKADS) was available (Koslowsky, 1998), providing decadal NDVI data for the period from 1989 to 2005.

In particular, the objectives of this study are to:

- characterize the impact of spatial and temporal scale on the patterns resolved;
- evaluate the potentials and limits of NOAA- and Landsat-based data archives regarding the assessment of land use and land cover change; and
- determine whether the higher temporal resolution of the NOAA AVHRR sensor can partially compensate for the coarse geometric resolution.

These objectives will be addressed through a series of experiments which are illustrated in the workflow presented in Fig. 1. The first experiment (Section 4.1) focuses on objective 1 and subdivides into two steps. In a first step, a spatial degradation of the Landsat time series is performed and the patterns of the derived regression coefficients of a linear trend analysis are evaluated (Section 4.1.1). Next, a direct comparison of a spatially degraded Landsat TM time series and a temporally reduced NOAA AVHRR time series was performed to assess the direct comparability of the linear trend analysis (Section 4.1.2). In the second experiment (Section 4.2) addressing mainly objectives 2 and 3, the Landsat TM/ETM+ and NOAA AVHRR time series data archives are compared in their original resolution to evaluate their maximum information contents for land cover change detection. Based on the individual results, we conclude on perspectives of multi-scale analyses to detect landscape processes (Section 5).

2. Study area

2.1. Physical setting

The study site is located in Central Macedonia, Greece and covers about 6500 km². It comprises mainly the eastern part of the prefecture of Thessaloniki, the south western part of Serres and the northern part of Chalcidice (Fig. 2) and is a typical heterogeneous Mediterranean

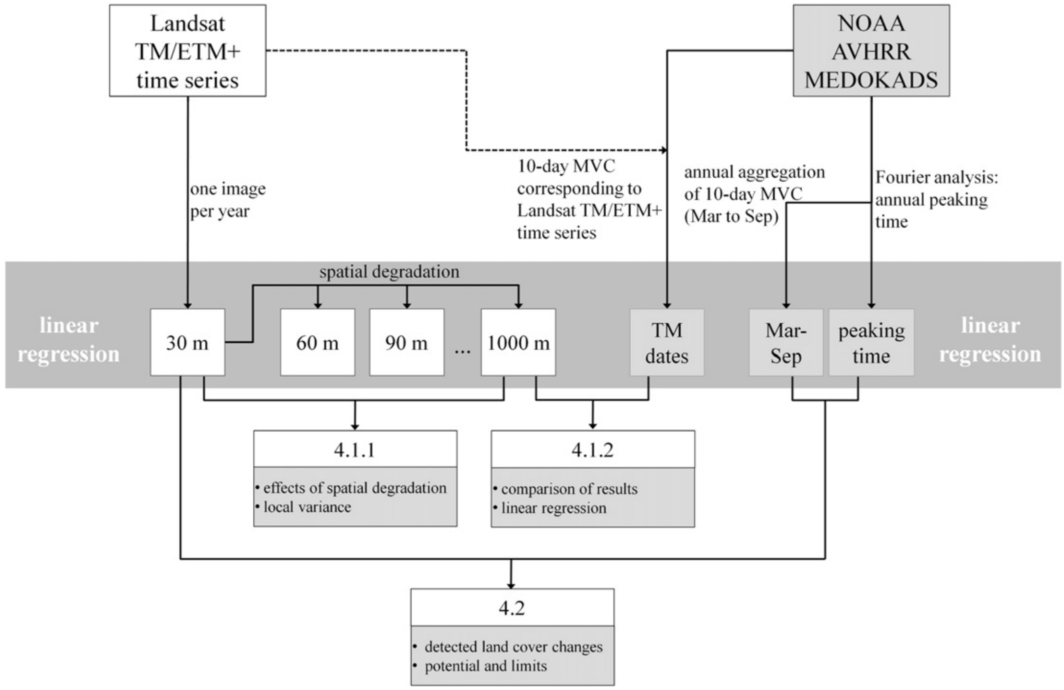

Fig. 1. Workflow of the presented research. Section 4.1.1 treats the effects of spatial degradation on the identification of different types of land use change. Results of the comparison of the spatially degraded Landsat TM/ETM+ time series and the temporal degraded NOAA AVHRR time series are described in Section 4.1.2. Section 4.2 deals with the potentials and limits of both time series to detect land cover change in a heterogeneous Mediterranean test area.

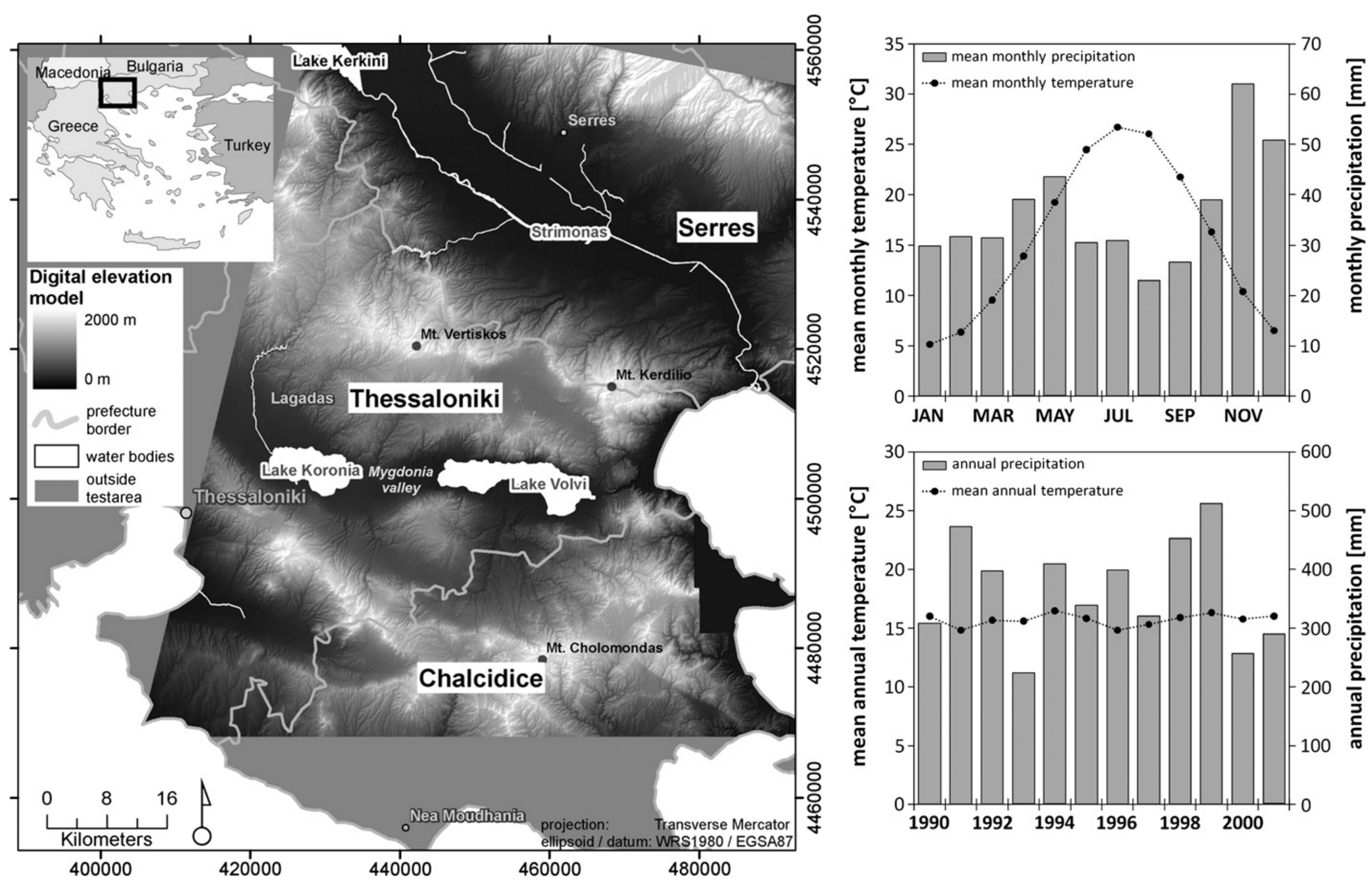

Fig. 2. Hill-shaded digital elevation model (DEM) of the study area provided by Geoapikonisis Ltd., Athens, Greece (left), mean monthly temperature and monthly precipitation (right above) and mean annual temperature and annual precipitation (right below) for Loutra Thermis.

landscape that has undergone several land use and land cover changes during the last decades. It is characterized by long-term, gradual processes which are mainly driven by changing grazing management practices. Moreover, the study site is affected by the extension of irrigated arable land.

Main water bodies are the freshwater lakes Koronia and Volvi, which are situated in the Mygdonia basin, as well as the artificial lake Kerkini and the Strimonas river in the northern part of the test site. Mygdonia Valley and the Strimonas Basin are enclosed to the North by the Kerdilio and Vertiskos mountains which reach up to 1100 m asl, and by the mountain range of the Hortiatis, Holomondas and Stratoniko mountains to the South.

The climate of the study area belongs to the thermo-Mediterranean to meso-Mediterranean zone and can be classified as sub-arid to sub-humid (Konstantinidis & Tsiourlis, 2003). Precipitation ranges from 450 mm in the lowlands to 800 mm in high altitudes. Usually, the rainfall distribution is bimodal with the maximum peak in early winter and a minor peak in spring, while minimum precipitation occurs in summer. The range of the monthly mean temperature is relatively high, e.g. in Serres the maximum difference amounts to more than 20 °C, January being the coldest month with about 4 °C and July being the warmest with 26 °C mean temperature (HNMS (Hellenic National Meteorological Service), 2007).

The potential natural vegetation of the region is mainly dependent on altitude and corresponding climatic conditions. In the thermo-Mediterranean lowlands forest of evergreen, sclerophyllous species (*Quercion illicis*) are dominant. Thermophilous oak forests (*Quercion confertae*) are characteristic in the meso-Mediterranean zone in altitudes between 300–400 m and 600–800 m asl, whereas continental vegetation (*Fagion mosaicae*) follows in higher altitudes characterized by sub-Mediterranean climate (Konstantinidis & Tsiourlis, 2003).

2.2. Socio-economic setting

Today, the highest areas of the mountain ranges are mostly covered by thermophilous forests, but in the undulating terrain between the high mountains and the agricultural plains the forests are substituted by sclerophyllous shrubland and grasslands, resulting from anthropogenic intervention through cutting and grazing activities for more than 10,000 years. Dominant species are the kermes oak (*Quercus coccifera*) and the holm oak (*Quercus ilex*). Due to the constitution of the grazing land Greek livestock husbandry is dominated by sheep and goats whereas cattle breeding is of minor importance. In contrast to traditional transhumance practised over centuries, at the present time sedentary grazing management systems are prevailing (Legg et al., 1998).

Mygdonia as well as Strimonas Valley are the main agricultural areas for the region and they are characterized by high proportion of intensive irrigation practices, making Strimonas Valley one of the major irrigation zones of Greece. Land use maps of the years 1986 and 2001 based on multi-seasonal datasets (Stellmes et al., 2007) illustrate the aforementioned developments in the study area (Fig. 3).

3. Data and methods

3.1. Data basis

3.1.1. Landsat TM time series

The Landsat time series provided by Röder et al. (2008b) is one component of the comparison of local and coarse scale resolution data. For the period from 1984 to 2000, yearly images with comparable phenological stages were acquired, except for 1989 and 1991 when cloud cover prohibited the acquisition of appropriate images. Mid-summer images were chosen since these best represent the period of maximum photosynthetic activity (Table 1). The time series was acquired to detect land use/cover changes and to evaluate the land degradation status within the semi-natural to natural rangeland areas of the county of Lagadas. In the frame of this study the Landsat TM/ETM+ data of the common overlapping time period of both time series (1990 to 2000) were used for the comparison.

In the study of Röder et al. (2008b) vegetation abundances had been derived from spectral mixture analysis (SMA) and used for trend analysis. Due to the limited numbers of reflectance channels of the NOAA AVHRR sensor, implementing a SMA-like method is restricted. Hence, to facilitate the comparison of the two time series, we used the NDVI, calculated from the Landsat TM/ETM+ red and near infrared bands.

Time series analysis and land use change detection requires accurate geometric and radiometric correction of the data to enable a meaningful analysis, necessitating a rigorous pre-processing scheme for all the Landsat TM/ETM+ images. The geometric correction was performed semi-automatically using a 2D-cross-correlation approach to retrieve a large number of ground control points and integrating digital elevation data to account for topography-induced distortions (Hill & Mehl, 2003). The resulting root mean square error (RMSE) did not exceed 0.3 pixels.

The implemented radiometric correction comprised sensor calibration and full radiative transfer modelling based on the 5S Code by Tanré et al. (1990). It included the correction of topography-induced illumination variations considering diffuse and direct radiance terms calculated from the digital elevation model and the radiative transfer code (Hill et al., 1995). A high radiometric consistency of the time series was achieved, and variations between integrated reflectance spectra of pseudo-invariant areas did not exceed $\pm 2\%$ (Röder, 2005; Röder et al., 2008b). Areas affected by cloud cover or shadow were masked and excluded from time series analysis.

3.1.2. NOAA AVHRR time series

The top-of-atmosphere (TOA-)NDVI data used in this study were derived from the "Mediterranean Extended Daily One Km AVHRR Data Set" (MEDOKADS). This data set is compiled from the NOAA-11, -14 and -16 sensors and is distributed by the Institute of Meteorology, Free University of Berlin. Covering the time period from 1989 until present, the archive comprises full resolution NOAA AVHRR channel data and supplementary data in latitude longitude presentation with a resolution of 0.01° in both directions. MEDOKADS is registered using an orbital model and a supervised fine-navigation with a final accuracy of about 1 km (Friedrich & Koslowsky, 2009). The geometric resolution was resampled to 1 km^2 when projecting the archive to the same UTM projection as the Landsat data. MEDOKADS-AVHRR channels 1 and 2 data are corrected for sensor degradation and orbital drift effects that cause non-linear changes in the measured signal (Friedrich & Koslowsky, 2009; Koslowsky, 1996, 1998). The applied correction method is based on the intercalibration of any AVHRR-instrument to the NOAA-11 instrument, which in turn is intercalibrated to NOAA-9 using the absolute in-flight calibration results published by Rao and Chen (1994). To derive degradation coefficients for the AVHRR instrument, time series from invariant desert areas are used to inter-calibrate the instruments of the successive satellites NOAA-11, NOAA-14 and NOAA-16 (Koslowsky, 2003). Starting with carefully evaluated post-launch calibration coefficients for the NOAA-11 satellite (Rao & Chen, 1994), cloud free daily series from the test areas were adjusted to nadir viewing conditions, radiometrically corrected using the LOWTRAN code, BRDF corrected, and finally smoothed by a 30 day median filter. The comparison of the generated NOAA-11 reference series with data from succeeding sensors was accomplished using piecewise regression analysis to take temporal varying sensor degradation effects into consideration.

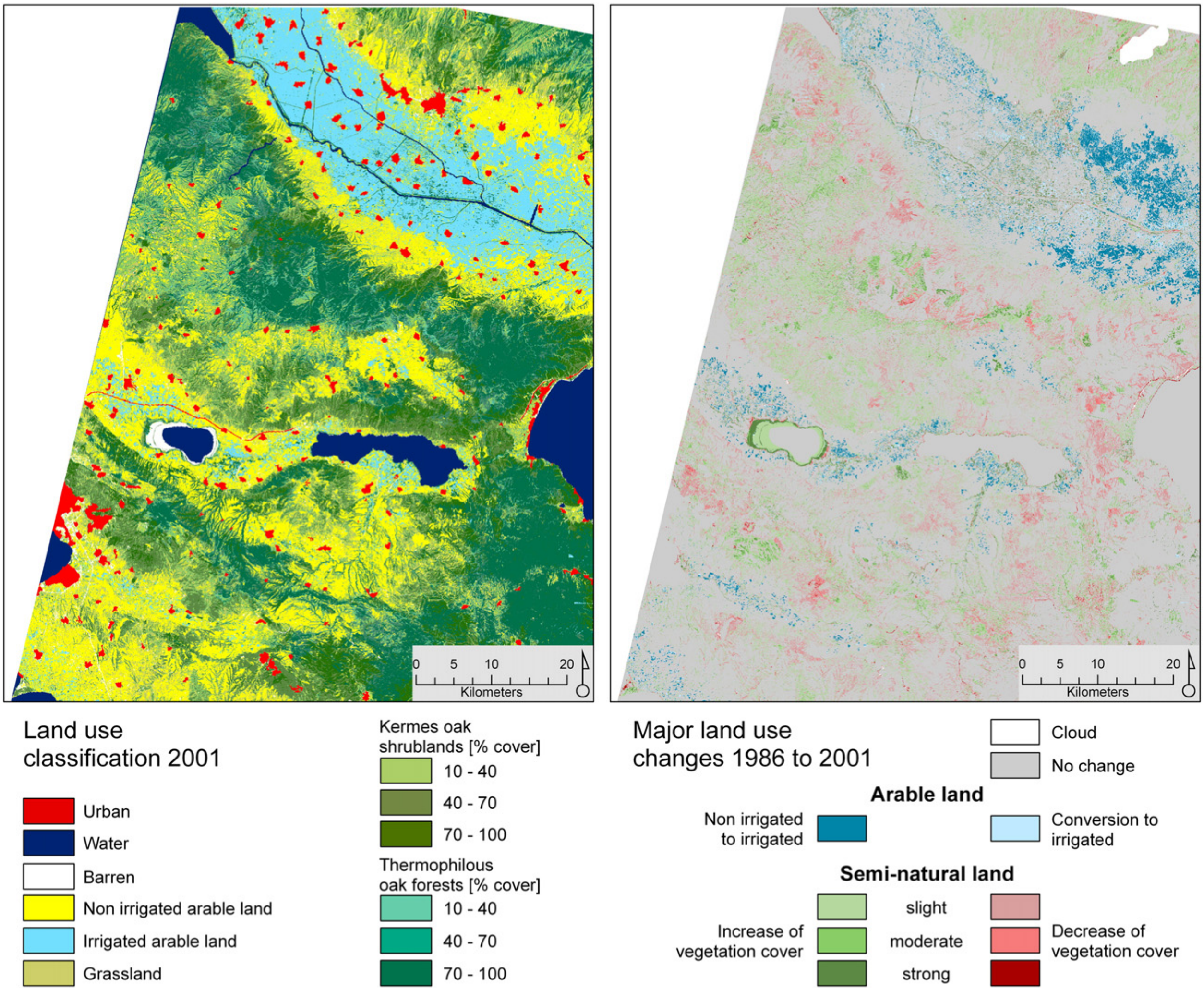

Fig. 3. Land use classification 2001 (left) and major land use changes between 1986 and 2001 (right) of the study area.

For the period from 1990 to 2000 10-day maximum value composite (MVC; Holben, 1986) data were employed for frequency analysis whereas the trend regression of the NDVI series was restricted to yearly aggregated MVC values from March to September to exclude uncertainties due to large sun-zenith angles, which occur during the winter period. Additionally, a temporally degraded NOAA AVHRR time series was created comprising the 10-day MVC corresponding to the dates of the Landsat TM/ETM+ time series. This temporally degraded time series was used for a direct comparison of spatially degraded Landsat TM/ETM+ data.

3.1.3. Additional data

A digital elevation model for the geometric and radiometric correction of the Landsat TM/ETM+ images was provided by Geoapikonasis Ltd., Athens, Greece. The DEM was derived from photogrammetric analysis of aerial orthophotographs (Scale 1:40,000) with a geometric resolution of 30 m × 30 m and an estimated vertical accuracy better than 3 m.

The results of the time series analysis were compared to land use change maps to support the interpretation of the time series analysis results. The land use maps were derived from classifications of

Table 1
Landsat-based satellite image time series for the Lagadas area.

Platform	Sensor	Path/row	Acquisition date	Proc. level	Resampling	Provider
Landsat-5	TM	183/32	04. 07. 1990	System corr.	NN	Eurimage
Landsat-5	TM	183/32	23. 06. 1992	System corr.	NN	Eurimage
Landsat-5	TM	183/32	12. 07. 1993	System corr.	NN	Eurimage
Landsat-5	TM	183/32	28. 05. 1994	System corr.	NN	Eurimage
Landsat-5	TM	183/32	02. 07. 1995	System corr.	NN	Eurimage
Landsat-5	TM	183/32	04. 07. 1996	System corr.	NN	Eurimage
Landsat-5	TM	183/32	21. 06. 1997	System corr.	NN	Eurimage
Landsat-5	TM	183/32	10. 07. 1998	System corr.	NN	Eurimage
Landsat-5	TM	183/32	11. 06. 1999	System corr.	NN	Eurimage
Landsat-7	ETM+	183/32	05. 06. 2000	System corr.	NN	Eurimage

multi-temporal Landsat TM/ETM+ imagery for 1986 and 2001 (Stellmes et al., 2007).

3.2. Spatial degradation of Landsat TM/ETM+ time series and temporal degradation of the NOAA AVHRR time series

Due to the different spectral properties of the reflectance channels, the observation geometry and the pre-processing of the two time series, a comparison of the two sensors was implemented to compare the results under similar conditions. This comparison can be considered as a quality criterion of the regression coefficient derived from time series analysis (compare Section 3.3). The comparison requires two processing steps: (a) the Landsat TM/ETM+ data have to be spatially degraded to the geometric resolution of NOAA AVHRR; (b) the NOAA AVHRR time series has to be temporally degraded to match the Landsat TM/ETM+ time series.

a) Spatial degradation of Landsat TM/ETM+ data
   The Landsat TM/ETM+ time series was degraded from the original resolution of 30 by 30 m to 1 km^2 as follows: In several studies the degradation of high resolution data to a coarser resolution has been investigated (Cola, 1997; Justice et al., 1989; Kavzoglu, 2004). The spectral information of a pixel is usually assumed to originate from the IFOV (Instantaneous Field of View) of the remote sensing sensor. However, a proportion of the spectral signal associated with a given pixel emanates from surrounding areas (Kavzoglu, 2004). The Point Spread Function (PSF) describes the fraction that a specific area attributes to a pixel's spectral signal. It is difficult to derive an accurate model of an in-flight PSF of the NOAA AVHRR sensor due to the interaction of several factors (observation, view angles, optical characteristics, along-track and cross-track asymmetry) (Moreno & Melia, 1994; Oleson et al., 1995). Nevertheless, the PSF of a sensor can be sufficiently approximated by a two-dimensional Gaussian pulse (Billingsley et al., 1983; Kavzoglu, 2004). We applied an approach proposed by Oleson et al. (1995): a Gaussian filter is designed depending on the resolution of the original and the degraded image data and the PSF assumed for the NOAA AVHRR sensor as provided by Bevington (1969; from Oleson et al., 1995). The resulting Gaussian filter to generate simulated NOAA AVHRR data was parameterized with a Full Width at Half Maximum (FWHM) of $\sigma = 13$ pixels and was applied to the radiometrically corrected reflectance bands of the Landsat data. However, the spatially degraded Landsat TM/ETM+ data are expected to have a higher spatial resolution than the NOAA AVHRR data since the real geometric resolution of the NOAA AVHRR data is less than 1 km due to the geometric inaccuracy of the NOAA AVHRR time series. (compare Section 3.1.2).
   Local variance is a measure which was introduced by Woodcock and Strahler (1987) to assess the effects of changing resolution. It is defined as the mean value of the standard deviations of a 3 by 3 moving window. The local variance was calculated for the regression coefficient at different resolutions from 30 to 1000 m to capture the effect of the spatial degradation. If the spatial resolution of an image is much finer than the objects mapped the local variance will be low because most of the values will be highly correlated to their neighbours. If the resolution is much coarser than the objects the local variance will be low as well because many objects will be contained in a single pixel. The local variance increases when the spatial resolution of an image approximates the size of the objects mapped and is therefore an indicator if the spatial resolution is appropriate to detect these objects (Cao & Siu-Ngan Lam, 1997).
b) Temporal degradation of NOAA-AVHRR data
   Following the spatial degradation of the Landsat TM/ETM+ time series, the reduced NOAA AVHRR time series was created extracting the 10-day composites congruent to the dates of the Landsat TM/ETM+ series from 1990 to 2000. We considered 10-day-composites to be more appropriate for the comparison because the daily images were strongly affected by convective cloud cover.
   After the spatial and temporal degradation, linear regression was again performed for both simulated time series (compare following section).

3.3. Time series analysis

A time series consists of different components: systematic patterns (trend and seasonality) and random components (noise) (Schlittgen & Streitberg, 1999; Shumway & Stoffer, 2006). These components can be analyzed depending on the temporal resolution of a time series (Andres et al., 1994; Brunsell & Gillies, 2003). The analysis of the data was carried out using the TimeStats software package, which was specifically developed for analyzing long-term hyper-temporal satellite data archives (Udelhoven, 2006).

a) Linear trend
   One of the most important descriptors of vegetation dynamics is its long-term trend (increase and decrease of vegetation), which is determined by a pixel-wise linear regression analysis. Trend analysis of remote sensing data requires an accurate geometric and radiometric pre-processing to generate homogenous data. Effects of spatial mis-registration and differing radiometric pre-processing on change detection was addressed in several studies (Song et al., 2001; Townshend et al., 1992) and thus pre-processing can influence the derived trend parameters.
   Simple linear regression was performed for both the Landsat TM/ETM+ and the NOAA AVHRR time series to assess gradual changes of NDVI. To rate the fit of the linear trend model, the root mean squared error (RMSE), the p-value of the two-sided Student's t-test, the correlation (r) and the standard deviation (SD) were calculated. Additionally, a z-normalization was performed for the two time series to allow for a direct comparison of the detected trends derived from both time series (Table 2). Subsequently, regression coefficients were calculated for the normalized time series. This transformation was considered useful as the NDVI ranges of the two sensors differ due to the spectral and geometric properties of the sensors. The derived z-transformed regression coefficients were used to scale the original regression coefficients in a comparable way.
b) Seasonal component
   The high temporal resolution of the NOAA-AVHRR time series systems allows the derivation of phenological information, such as the amplitude of the annual cycle and the time of occurrence of the annual maximum NDVI. For the present study, the date of maximum NDVI (peaking time) was derived for each year from decadal MVCs by fitting a Fourier polynomial x_t (Shumway & Stoffer, 2006). Considering the annual (36 10-day composites) and the semi-annual cycles (18 10-day composites) is sufficient

Table 2
Mean NDVI and its standard deviation of the different time series examined. These parameters served for z-transformation of the time series.

	Mean	Std
A: Landsat TM, 30 m	0.48	0.20
B: Landsat TM, spatially degraded, 1000 m	0.48	0.17
C: NOAA AVHRR, agg. Mar–Sep	0.38	0.08
D: NOAA AVHRR, temporally degraded, Landsat dates	0.38	0.13

to approximate the seasonal components of interest, the peaking time was derived from the fitted function:

$$x_t = a_0 + \phi_1 \cos 2\pi \frac{1}{36} + \phi_2 \sin 2\pi \frac{1}{36} + \phi_3 \cos 2\pi \frac{2}{36} + \phi_4 \sin 2\pi \frac{2}{36}$$

where a_0 is the constant of the Fourier polynomial and Φ_n equal the Fourier coefficients.
Subsequently, linear regression was applied to the resulting values of the annual time of maximum NDVI and magnitude to monitor changes of phenology.

4. Results and discussion

The objectives stated in the introduction were addressed based on two major comparison experiments (see also Fig. 1). The first experiment investigated the impact of spatial and temporal degradation on the information content of the respective time series (Section 4.1). Complementary, the second experiment compared the two archives in their original resolution, thus assessing the maximum information content of the different data archives (Section 4.2).

4.1. Direct comparison of Landsat TM/ETM+ and NOAA AVHRR archives

4.1.1. Effects of spatial resolution on the regression coefficient

Fig. 4 illustrates the effect of geometric degradation on the detection of trends of NDVI for four subsets of the test area representing different land cover changes. Subsets A and B (Fig. 4) represent typical rangelands that were characterized by fine scale patterns of negative and positive trends at the original resolution of 30 m. Röder et al. (2007) showed that these patterns are caused by under- and overgrazing due to the changed rangeland management system. The trend gains were furthermore intersected with terrain classes derived from a digital elevation model; easy accessible areas show negative to neutral trends whereas remote and steep terrain is characterized by positive trend gains (Röder et al., 2008b). At a resolution of 1 km² main structures were still preserved, whereas the fine scale patterns of negative and positive trend gains were neutralized. Subset C (Fig. 4) shows a forested area where a vast fire event occurred in 1998 and resulted in a connected patch of negative NDVI trends surrounded by undisturbed forest showing increasing NDVI. This fire event could also be identified within the degraded time series. Subset D represents an almost homogeneous area characterized by increase of greenness. Small patches of negative trends could be identified at the resolution of 30 m but have completely vanished at the resolution of 1 km^2.

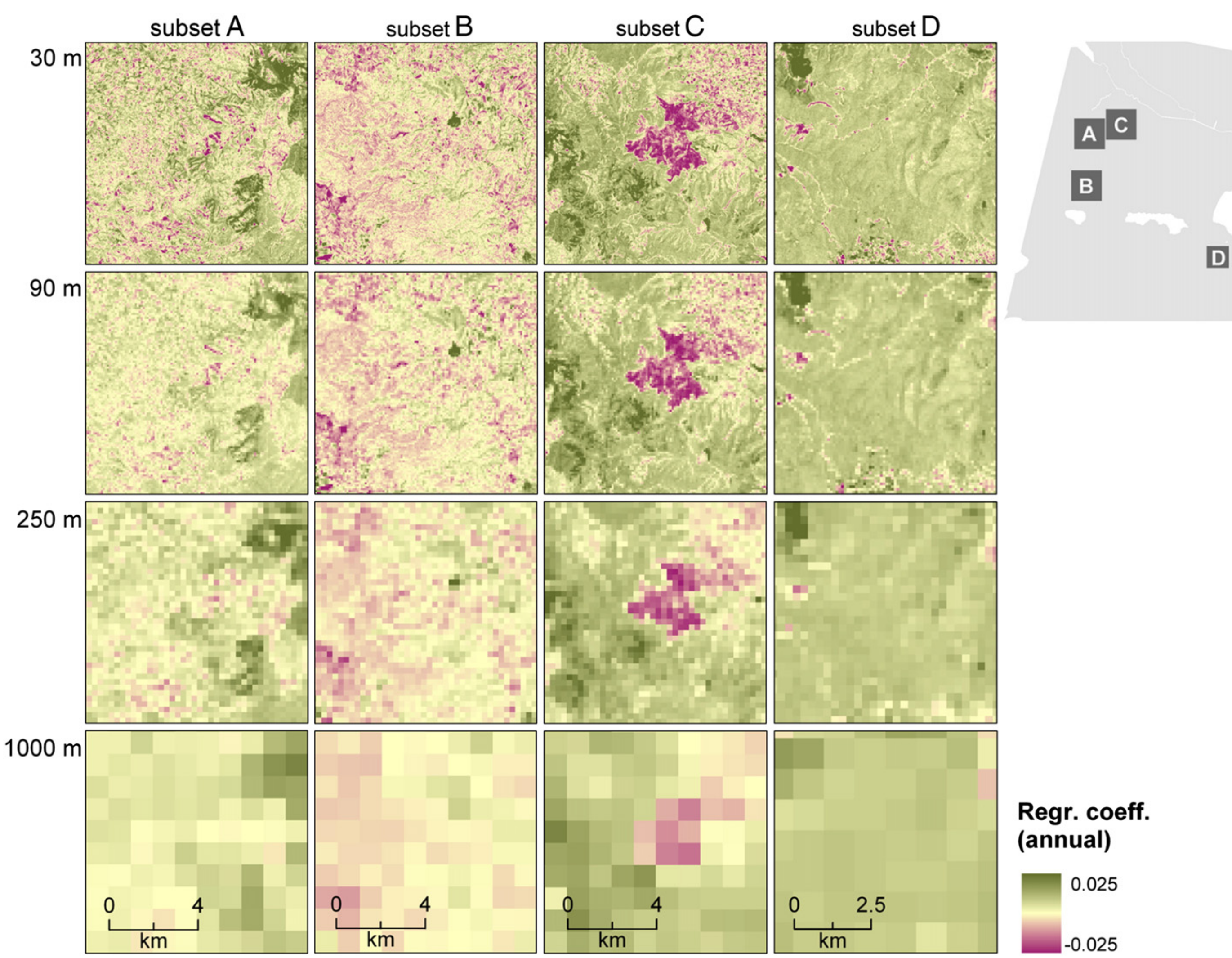

Fig. 4. Effects of spatial degradation of Landsat TM/ETM+ time series from a geometric resolution of 30 m to 1000 m on the derived regression coefficient. The four presented subsets represent different types and scales of land cover change.

Fig. 5 shows the resulting local variances calculated from the regression coefficients for the four subsets and the different geometric resolutions; the corresponding diagrams present the mean value of the local variance and its standard deviation for the original resolution and the step-wise spatially degraded time series up to 1000 m. The variance of subsets A and B (Fig. 5) rapidly decreased from 30 m to 250 m and then remained almost constant up to 1000 m. This indicated that these small-sized patterns are not detectable by low to medium spatial resolution systems, such as NOAA AVHRR, SPOT VEGETATION, MODIS and MERIS. In contrast, the mean local variance and its standard deviation of subset C (Fig. 5) was found to be almost constant for all presented geometric resolutions. This supports the finding that the fire event is large enough to be still detected at a resolution of 1000 m. Generally, subset D is characterized by a lower local variance than subsets A to C (Fig. 5). Even though this subset represents a homogenous area where increasing greenness is dominant, a decrease of the local variance can be detected. This can be attributed to the varying strengths of the detected positive NDVI trends.

The four examples clearly illustrate that the loss of information from fine to coarse scale depends on factors such as the spatial arrangement of the objects, their contrast, their spatial extent and in case of change detection the magnitude of change (Townshend & Justice, 1988). Subsets A to D (Figs. 4 and 5) show that the spatial degradation of the time series still preserves major patterns of change in greenness but subsets A and B (Figs. 4 and 5) reveal that small scale patterns caused by management practices are lost. Whereas Woodcock and Strahler (1987) characterized landscape structures of mono-temporal data by calculating the local variance at different spatial scales, different types of land cover change were determined by the local variance of the regression coefficients. Furthermore, the geometric resolution appropriate for identifying different types of change could be derived.

4.1.2. Comparison of the simulated time series

After assessing the effect of spatial degradation on the recognition of trends of greenness for different types of land cover change, Fig. 6 illustrates the temporal degradation of the NOAA AVHRR time series for a single pixel representing thermophilous oak forest. Due to cloud cover, the Landsat TM/ETM+ data were acquired within a time window from June to August to allow for a preferably dense time series with exception of 1989 and 1991 when no appropriate images could be acquired. The Landsat time series was initially compiled to represent the maximum photosynthetic activity of semi-natural areas. The temporal degradation of the NOAA AVHRR time-series shows that this objective could be achieved. The averaged Landsat TM NDVI was always close to the annual maximum of the NOAA AVHRR NDVI even though the time steps were not exactly one year. Displacements caused by phenological variations and missing values influence the

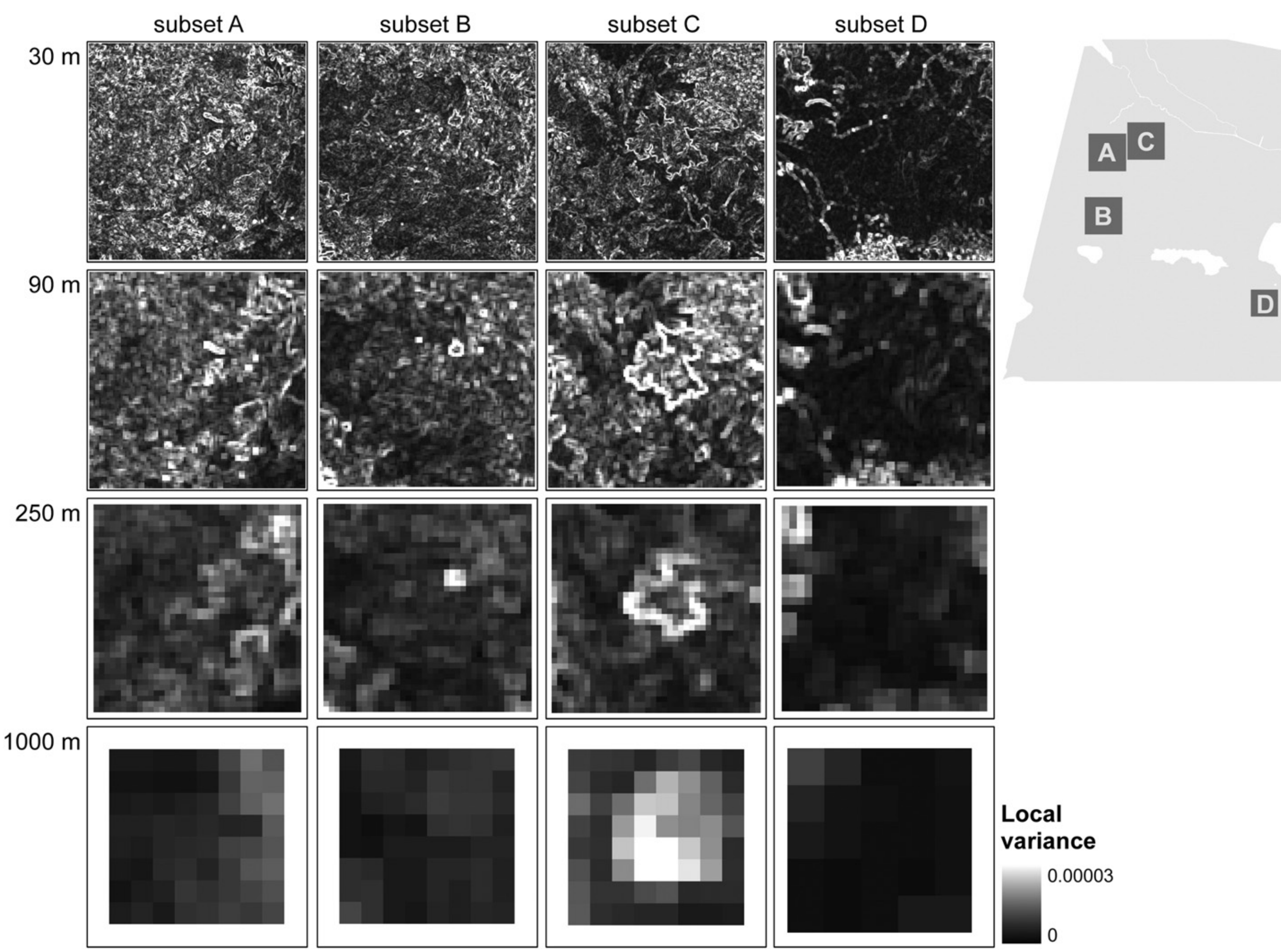

Fig. 5. Local variance derived from the regression coefficients of the subsets introduced in Fig. 4 for several geometric resolutions (above), mean local variance (black line, dots represent the spatial resolution simulated) and standard deviation of the local variance (grey area) of the four subsets (see following page).

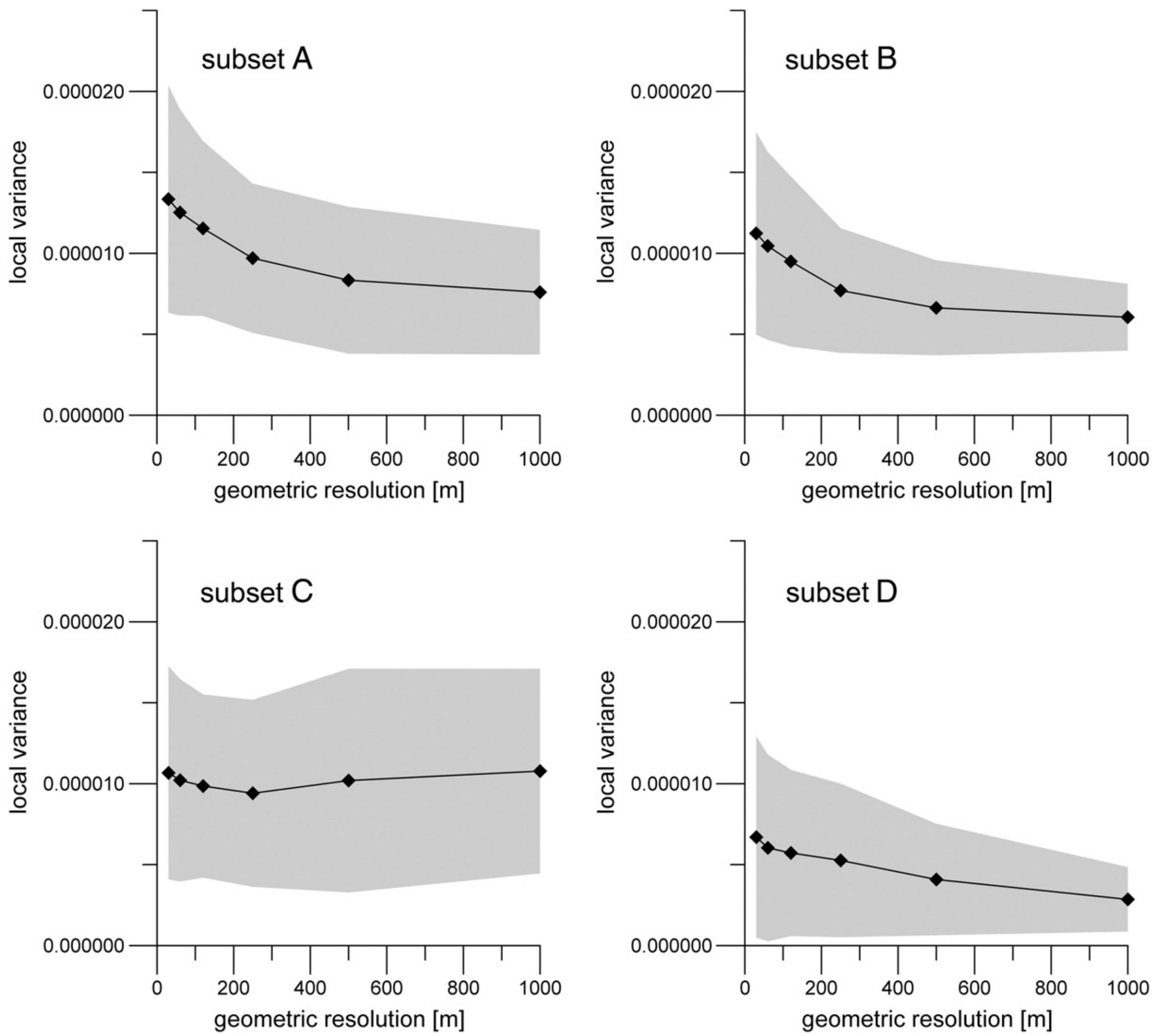

Fig. 5 (*continued*).

derived time series parameters. The effect is larger at the beginning or end of the series, while errors in the middle of a time series reduce significance of the derived trends without compromising the overall direction of the trend function. Given an appropriate number of images in a time series, sensitivity analyses indicated resulting trends to be very robust (Hostert et al., 2003b; Röder et al., 2008a).

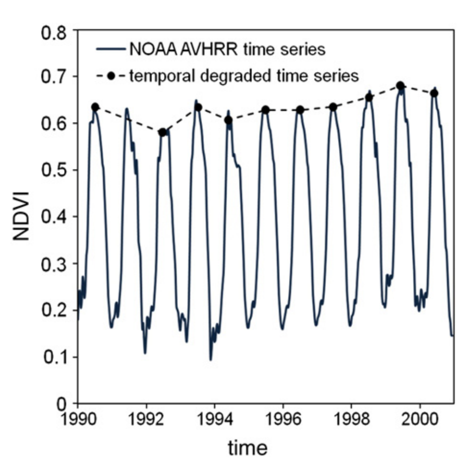

Fig. 6. Temporal degradation of the NOAA AVHRR time series. Original NDVI time profile (continuous line) and time series extracted congruent to the Landsat TM/ETM+ time series (dashed line).

Fig. 7 presents the derived regression coefficients, their significance and the correlation for the time series (original and degraded) examined in this study. Because of the different spectral characteristics of both sensors and the degradation of the Landsat TM time series (spatially) and the NOAA AVHRR data (temporally), the NDVI range differs and leads to regression coefficients that are not directly comparable. Therefore, the colour scheme of the regression coefficient has been scaled by implementing a *z*-normalization of the examined time series (compare Section 3.3 a).

The comparison of the geometrically degraded Landsat and the temporally degraded NOAA time series revealed similarities in the overall patterns of trends, even though they are not congruent (Fig. 7, maps B and D). This visual impression is supported by Fig. 8 which shows the scatter plot of the regression coefficient derived from the trend analysis of the simulated data. The correlation and the r^2 of the linear regression of the trends were modest with 0.75 and 0.57, respectively. The gain of the regression function derived from the time series (Fig. 8, left) diverged from a 1:1-relationship. As mentioned before the major reason for the deviation is the different range of the NDVI values calculated from the two sensors. Differing pre-processing and spectral characteristics cause a higher NDVI range derived from

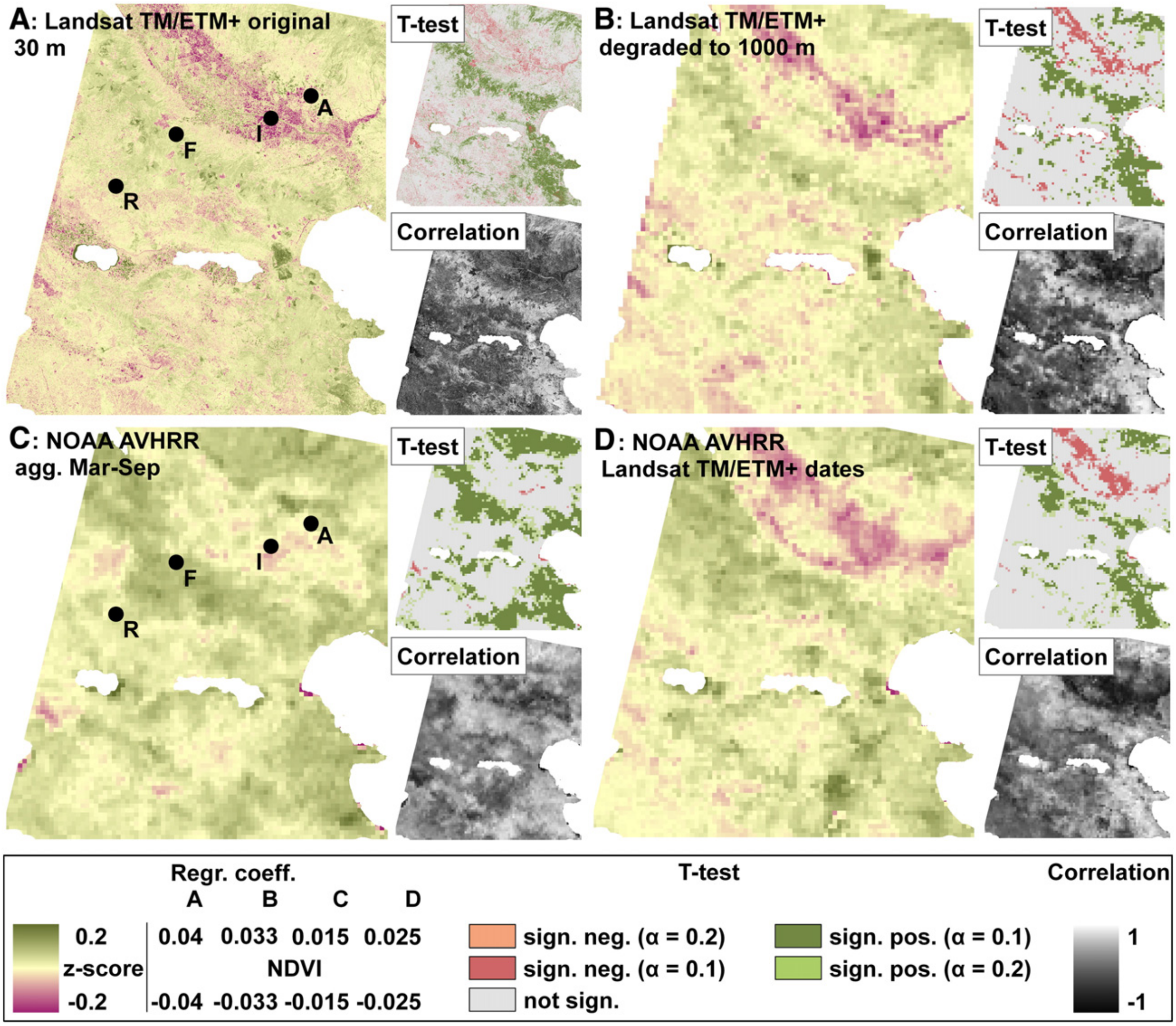

Fig. 7. Regression coefficients derived by simple linear regression, significance of the regression coefficients and correlation calculated from the (A) original Landsat TM/ETM+ time series (above left), (B) the spatially degraded 1 km Landsat TM/ETM+ time series (above right), (C) the annually aggregated MEDOKADS NOAA AVHRR 10-day MVC composites from March to September (below left) and (D) the temporally degraded MEDOKADS NOAA AVHRR archive covering the 10-day MVCs corresponding to the Landsat TM/ETM+ time series (below right). Abbreviations used in panels A and C correspond to time profiles shown in Fig. 9 which represent rangeland (R), forest (F), and irrigated (I) and arable (A) land.

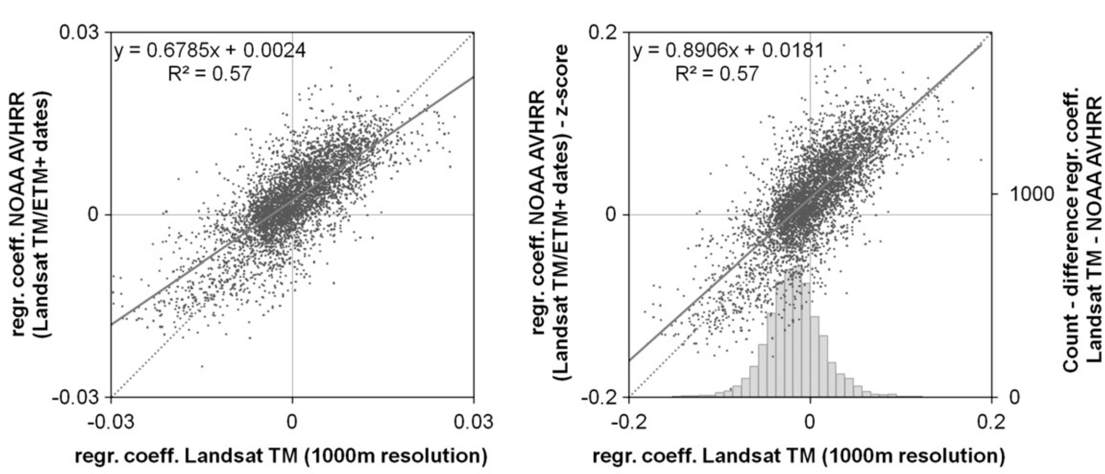

Fig. 8. Scatterplot and linear regression of the regression coefficients derived by linear regression from the spatially degraded Landsat TM/ETM+ and the temporally degraded NOAA AVHRR time series, original data (left) and z-normalised data and histogram of the differences between the regression coefficients (right).

the Landsat sensor than from the NOAA AVHRR and subsequently, impose a systematic deviation from the 1:1-relationship. Therefore, the regression coefficients of the *z*-standardized time series (Fig. 8, right) show a relationship that is close to the 1:1-line.

The moderate coefficient of determination is caused by several factors that increase the noise of the relationship. These include the spatial resampling where the PSF was approximated using a two-dimensional Gaussian filter instead of the true PSF function, and errors in the geometric registration of the Landsat and the NOAA AVHRR time series. In this context an effect is worth mentioning, which was already described by Townshend and Justice (Townshend & Justice, 1988): the interannual geometric registration of the simulated 1 km^2 Landsat time series is more precise than the NOAA AVHRR data because it is based on the geometric corrected 30 m data. Additional errors can be assigned to the different overflight times. Whereas the single date Landsat TM/ETM+ images are recorded in the morning, the NOAA data are 10-day-maximum value composites created from afternoon scenes.

Despite the moderate coefficient of determination the scatter plots show that the regression coefficients were linearly dependent. Furthermore, the *y*-axis intercept of the regression function was close to zero with a small positive bias of the Landsat TM/ETM+ regression coefficients. Hence, the turning point between positive and negative trends, i.e. the trend direction, was comparable for the simulated time series. This finding implies that despite the different pre-processing procedures, the calculated regression coefficients are comparable when the time series are standardized, suggesting time-series for both sensors are consistent and well-calibrated.

However, this comparison also reveals several major disadvantages of the NDVI. First, different spectral properties of remote sensing sensors cause a different NDVI range that impedes a direct numerical comparison of derived regression coefficients amongst different sensors. The problem of NDVI interoperability among different sensors operating at different spatial scales was addressed in several studies suggesting cross-sensor relationships which could solve this problem to a certain extent (Martínez-Beltrán et al., 2009; Steven et al., 2003). Nevertheless, NDVI cannot be used as a quantitative measure to assess changes in vegetation, contrary to other parameters like the aforementioned vegetation cover percentage derived using SMA.

4.2. Potentials and limits of information extraction on local and regional scale

Following the direct comparison of the two sensors, a comparison of the changes detected by the original full resolution Landsat and the NOAA data archives was performed. At local scale, the Landsat TM/ETM+ time series with a pixel size of 30 m from 1990 to 2000 was considered. At regional scale, the full hyper-temporal NOAA AVHRR time series was used. Time series analysis of the latter dataset included simple linear trend analysis but also allowed the derivation of phenological parameters such as the annual peaking time.

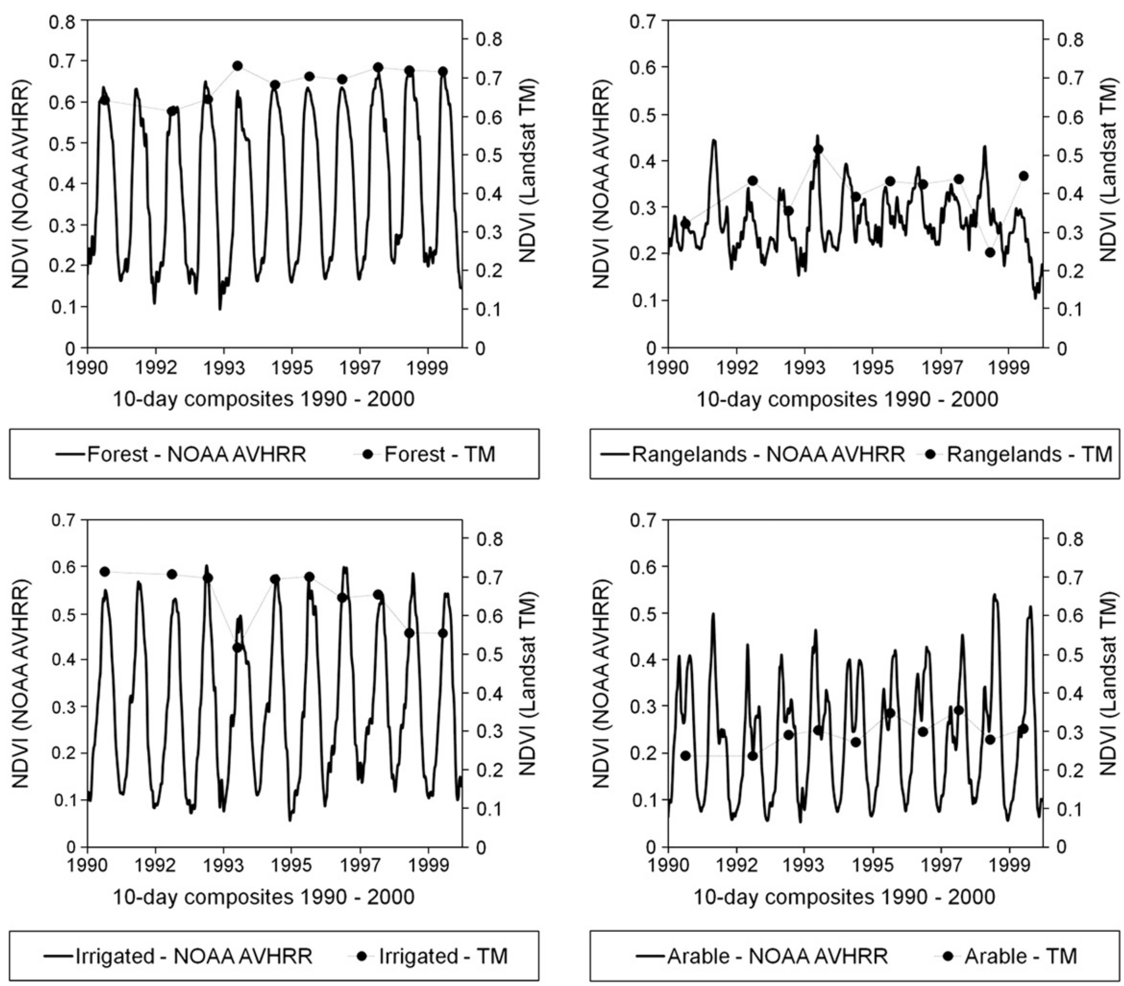

Fig. 9. NDVI time profiles derived from the Landsat TM/ETM+ time series and the MEDOKADS archive for different land use types covering the time period from 1990 to 2000; semi-natural areas (above) — rangeland (R) and forest (F), agricultural areas (below) — irrigated (I) and arable (A) land; for location compare Fig. 7.

4.2.1. Comparison of linear NDVI trend components

Fig. 7, maps A and C show the regression coefficient of the NDVI of the two time series derived by simple linear regression, indicating largely divergent results in some areas. Whereas the trends appeared similar in areas covered by natural vegetation (compare Fig. 3) the areas mainly covered by arable land showed major differences.

Fig. 9 shows four different NDVI time profiles (compare location in Fig. 7), two representing natural vegetation and two arable land, which were extracted from the spatially degraded Landsat TM/ETM+ time series and the 10-day NOAA AVHRR time series. A comparison of the NDVI values of the two sensors for rangelands and forests demonstrated that the observation of the Landsat TM/ETM+ time series coincided well with the maximum NDVI of the NOAA AVHRR time series. Only in 1999 the “rangeland” pixel showed a strong deviation for the Landsat TM/ETM+ time series from the maximum NDVI of the NOAA AVHRR. Even though this may have impacted the derived trend parameter, due to the high number of observation points (the underlying time series starts in 1984), the influence of single erroneous values may be compensated. The NDVI time profiles of the Landsat TM/ETM+ series of arable land showed several big leaps caused by slight deviations from the phenological stages in different years resulting in negative trends. As mentioned in Section 3.1.1, the Landsat TM/ETM+ time series was optimized to detect changes in rangeland areas (dates from June to August) and was able to detect gradual changes of vegetation cover for these areas very well, but changes within other land use classes, such as arable land, could not be tracked reliably as the observation time is not suitable. Fig. 10 underlines these findings by illustrating the spatial distribution of the RMSE of the linear regression model, which is an indication for the goodness of fit of the linear model. The RMSE of Landsat TM/ETM+ time series showed high values for arable land (compare Fig. 3) and cover changes within semi-natural areas caused by sudden changes like for example wild fires. The high RMSE values indicate that a linear regression model is not an appropriate model for these areas. In contrast, the RMSE of the NOAA AVHRR linear regression showed uniformly low values.

A closer examination of the “arable” NDVI profile (Fig. 9) showed a change of the occurrence of the maximum NDVI from spring to autumn. This indicated a shift from non-irrigated arable land to irrigated arable land as derived by the land use change analysis. The observation time of the Landsat time series was located around June, exactly between the maximum levels of vegetation cover of the two cultivation types. Therefore, the detection of these changes by analysis of the Landsat time series is inappropriate due to the selected dates. The simple linear regression performed for the NOAA AVHRR time series of aggregated values from March to September was not able to track these change either, because the mean value of the vegetation cover remained stable even though the land use changed.

4.2.2. Phenological information derived from frequency analysis

One advantage of hyper-temporal data provided from sensors such as NOAA AVHRR is the possibility to detect changes of the phenological cycle, e.g. by means of a Fourier analysis, which is a common means to exploit information about changing vegetation cover (e.g. Heumann et al., 2007; White et al., 2009).

Fig. 11 suggests a pronounced backwards phase shift of up to two months within Strimonas valley indicating the extension of irrigated arable land. Fig. 11 shows the temporal development of the phase for two areas within arable land. The area, which was irrigated throughout the observed period (I) was characterized by a relatively constant phase around June to July at the beginning of the time series. In contrast, the area subject to land use conversion (A) started with a phase around April to May, which until the end of the time series shifted to June to July. Similar to the land use change maps (Fig. 3), which incorporate phenology information, the time series analysis of the annual NDVI cycle was able to track the expansion of irrigated arable land in the Strimonas valley at reduced spatial scale.

The results of Sections 4.2.1 and 4.2.2 suggest that both examined time series deliver valuable information about land use and land cover change. The Landsat TM/ETM+ time series contains information about land use transformation that corresponds to changing land

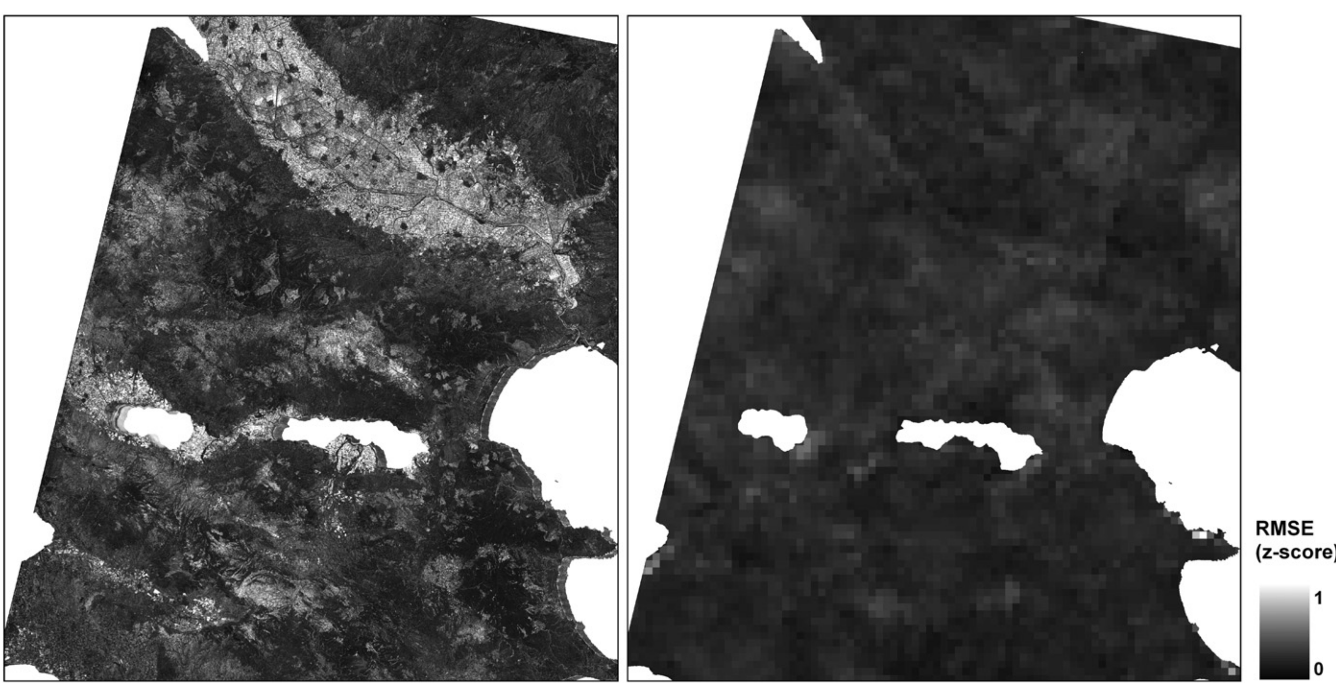

Fig. 10. RMSE of the linear regression model of the 30 m Landsat TM/ETM+ (left) and the annually aggregated MEDOKADS NOAA AVHRR 10-day MVC composites from March to September (right), 1990–2000.

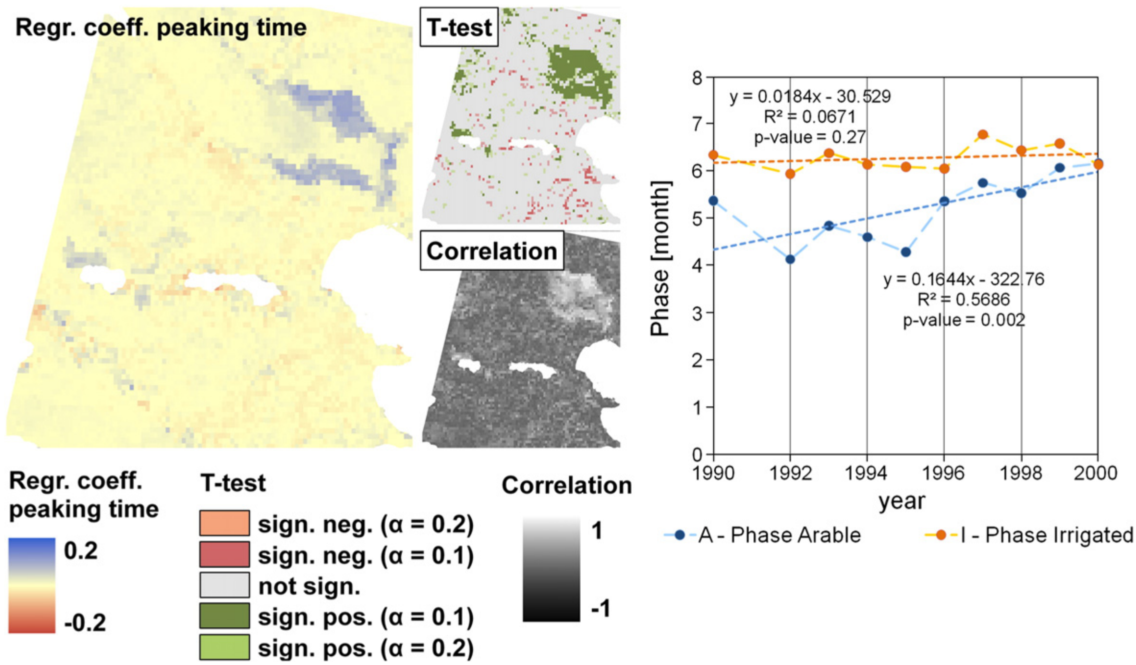

Fig. 11. Regression coefficients, significance of the regression coefficients and correlation calculated by simple linear regression of the peaking time derived from the NOAA AVHRR MEDOKADS archive, 1990–2000 (left); phase derived from the MEDOKADS archive for arable land changing from non-irrigated to irrigated arable land (A) and irrigated arable land (I) covering the time period from 1990 to 2000 (right).

management practises in areas with slowly changing phenologies as stated by Cohen and Goward (2004). However, our study also demonstrated that these time series have to be prepared very carefully with regard to the research question. The extension of the irrigated arable land could not be assessed due to the observation date of the time series. In the case of land cover conversions it is more efficient to approach the problem by a land use change detection analysis based on, for example, multi-seasonal land use classifications (compare Fig. 3). Of course, problems might emerge from the limited revisit capacities which often complicate the set-up of a dense time series due to cloud cover (Aplin, 2006; Bradley & Millington, 2006).

The NOAA AVHRR MEDOKADS archive can reliably reveal land cover changes at coarse scale if these changes affect large areas consistently. The high temporal frequency and the possibility to determine changes of phenological characteristics offer additional information and can partially compensate for the coarse spatial resolution (Pax-Lenney & Woodcock, 1997).

These findings underline that the complex and interwoven land use and land cover change processes that often characterize heterogeneous areas necessitate a comprehensive analysis of data with different scales and approaches (Aplin, 2006; Atkinson & Aplin, 2004; Benson & MacKenzie, 1995).

5. Conclusions and outlook

The aim of this study was to evaluate the interaction of spatial and temporal scale of remote sensing time series in a systematic manner for a heterogeneous Mediterranean study site.

The results demonstrated that, provided comparable geometric and temporal resolution, time series analysis derived from coarse-scale, hypertemporal and fine-scale data archives yield comparable results regarding the direction of trends and their spatial patterns. Even though the individual pre-processing steps applied produced well-calibrated long-term records that are consistent across different scales and resolutions, a direct numerical comparison of the derived regression coefficients is not possible due to different NDVI ranges of the two time series. For this purpose, first a standardization of the data or the application of a transfer function is necessary. Coarse scale hyper-temporal data are able to reveal changes of vegetation cover and the phenological cycle, such as the expansion of irrigated arable land at the expense of rain-fed agriculture if these changes affect large areas consistently. Hence, these data are suitable to monitor land cover change across large areas and identify target areas for further investigation (using for instance local scale satellite data). In that respect, the higher temporal resolution indeed provided an additional level of information and partially compensated for the lower geometric resolution. Yet, it was also shown that fine scale land cover changes cannot be tracked by means of coarse scale time series. For instance, changing vegetation cover in the rangelands of Lagadas could not be identified because the resulting trends were fine-scale patterns of opposite trends, making higher geometric resolution essential. These kinds of land cover modifications can only be assessed by means of high resolution time series as provided by Landsat TM/ETM+ data. Furthermore, the changes detected at local scale can be associated to management practices as the considered scale is comparable.

The degradation study of the Landsat TM/ETM+ time series and the derived local variances suggest that recent medium-resolution sensor systems like MODIS, providing vegetation indices at 250 m resolution, are most likely not appropriate for assessing these small-scale patterns of opposite trends.

These conclusions can be summarized as follows:

- Provided comparable geometric and temporal resolution, the direction of trends and their spatial patterns are comparable for different sensor systems.
- Direct numerical comparison of trend regression coefficients from different sensor systems is not possible due to different NDVI ranges; specifically adapted transfer functions can overcome this problem.
- For larger landscape units, phenological information derived from coarse-scale systems can indicate changes in plant communities or land use intensities.
- In heterogeneous environments, fine-textured patterns of positive and negative trends will be camouflaged by coarse-scale systems regardless of their higher temporal resolution.

This means that utilizing coarse-scale data in heterogeneous, small-structured environments may lead to failure to detect important landscape processes, as shown here for rangeland areas in Northern Greece. Nonetheless, we suggest that this kind of satellite imagery can support the creation of a high resolution time series that is appropriate for such patterns. Thus,

(i) the selection of appropriate dates for an accurate examination of land use change processes by fine resolution data could be supported,
(ii) acquired images could be evaluated for inclusion in a time series with respect to their representation of the chosen phenological status, and
(iii) time steps could be identified which are useful for multi-seasonal land use classifications.

Complementing this study with a focus on gradual processes, further investigations might address other land use change phenomena, such as combination of short-term disturbance and gradual processes typical for areas dominated by wildfires and succession processes. If land use conversions are targeted, change detection based on multi-seasonal image classification should complement trend based approaches.

The findings suggest that a global monitoring system should be established based on EOS data that are selected with careful regard to the research question. Nested approaches could be established to design a comprehensive monitoring system for drylands that make use of data of different geometric detail, temporal frequency and using different approaches adjusted to the existent processes. In this respect, the adequacy of new sensor platforms that provide better spectral and spatial resolution (e.g. MODIS) should be evaluated. Yet, these data archives cover only a relatively short time period compared to the NOAA AVHRR data, such that their future use depends on the continuity of the mission.

Given the increasing availability of Landsat data for minimal cost consideration should be given to the transfer of data-processing and analysis strategies developed for coarse scale hypertemporal time series to local scale archives and enhanced vegetation-related parameters. However, this might be impeded by cloud cover, and would require further work on automating data pre-processing to guarantee spatial and quantitative consistency of the developed time series.

Whichever strategy is being pursued, this study makes a strong case for the continuity of long-term data archives, as only these support a consistent monitoring of terrestrial ecosystems.

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Chapter III

Assessment of rainfall and NDVI anomalies in Spain (1989-1999) using distributed lag models

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Assessment of rainfall and NDVI anomalies in Spain (1989–1999) using distributed lag models

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In this study a link was established between anomalies in climatic and Advanced Very High Resolution Radiometer (AVHRR)/Normalized Difference Vegetation Index (NDVI) data in Spain for the period from 1989 to 1999 on a monthly and annual basis using multivariate distributed lag (DL) models and generalized least-square (GLS) parameter estimation. In most areas significant time-delayed correlation between anomalies of monthly rainfall and NDVI data was confined to an interval of 1 month. Locally higher lag orders of up to 3 months were found. By contrast, relationships between surface temperature and the NDVI were insignificant in the multivariate context at most locations. The multiple correlation coefficients of the DL models achieved 0.6 in the maximum. Regions characterized by the most significant NDVI–rainfall correlations include the southern forelands of the Pyrenees in Catalũna, rainfed agricultural areas in Extremadura, Andalusia, and the western parts of Castilla y Leon. Average ratios of rainfall to potential evapotranspiration (PET) in the sensitive areas ranged between 0.5 and 2, with annual rainfall amounts less than 700 mm. For each land-cover class a linear discriminant analysis (LDA) was carried out to assess the environmental factors that might explain the differences in the NDVI–rainfall relationships. The highest discriminant coefficients and factor loadings were recorded for those factors that recurrently trigger water deficit in the sensitive regions, such as low total annual rainfall, large seasonal rainfall variability, high average PET and surface temperature. On the annual basis the lagged correlation of the NDVI and rainfall data was confined to natural vegetation (grassland and scrubland) areas in western Spain. This region suffered from a severe drought in the early 1990s, after which biomass production lagged several years behind improved rainfall conditions. The approach presented is useful for assessing the influence of climatic variables on the pattern of temporal anomalies in the NDVI or related vegetation parameters.

1. Introduction

Although natural vegetation has developed a great capacity for physiological adaptation and resistance to long droughts and soil moisture below the theoretical wilting points (Kosmas 1999), precipitation is considered the primary limiting factor for plant growth in semi-humid and semi-arid areas (Wang *et al.* 2003, Huxman *et al.* 2004, Reynolds *et al.* 2004, Schwinning *et al.* 2005). Important additional covariates for primary production are soil physical

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properties, plant evapotranspiration, root zone soil moisture, vegetation type, surface temperature, irrigation activity and microclimate (Cihlar *et al.* 1991, Schultz and Halpert 1993, Kawabata *et al.* 2001, Li *et al.* 2002). The temporal dependence of green vegetation biomass, as represented by the Normalized Difference Vegetation Index (NDVI), on climatic variables has been addressed in many studies covering a wide range of environmental conditions, in the USA (Hicke *et al.* 2003, Wang *et al.* 2003, 2004), Africa (Richard and Poccard 1998, Heumann *et al.* 2007, Philippon *et al.* 2007), the Middle East (Schmidt and Gitelson 2000, Geerken and Ilaiwi 2004) and China (Li *et al.* 2002, Xiao and Weng 2007), and at a global scale (Kawabata *et al.* 2001, Brown *et al.* 2006). Among others, the establishment of such relationships has implications for modelling crop yield and primary productivity in semi-arid regions (Gurgel and Ferreira 2003). Furthermore, knowledge about the vulnerability of a region towards climatic variability is useful in establishing early warning systems or to distinguish between climatic and human-induced changes in surface conditions.

The correlation between climatic parameters and the NDVI is strongly determined by the degree of aggregation of the variables in the time dimension (Wang *et al.* 2003). Most of the studies quoted above suggest that the NDVI lags behind rainfall by several weeks or months. The functional relationship between monthly or annually aggregated NDVI and rainfall data was found to be log-linear (Davenport and Nicholson 1993) or linear (Wang *et al.* 2003, Evans and Geerken 2004).

For this study a quantitative link between monthly rainfall and temperature and NDVI anomalies was established in Spain using distributed lag (DL) models (Eklundh 1998, Wei 1990). The results were examined in relation to geographical data such as land-cover, terrain morphology, potential evapotranspiration (PET) and aridity. Two important aspects in using DL models are the total amount of variability in NDVI anomalies that can be explained by environmental factors and the duration of this dependency.

2. Materials and methods

2.1 *Study area*

Mainland Spain is bordered by two mountain chains, the Pyrenees-Cantabric Range belt in the north, and the Betic Cordillera in the south. In its central part, the Iberian Mountains and the Central System define the main hydrographic basins (figure 1).

Rainfall in Spain is characterized by high temporal and spatial variability. In general, almost 50% of Spain receives less than 500 mm precipitation per year and the relative coefficient of variation of the interannual rainfall variability locally achieves 40% (Del Moral and Sauri 1999). The mean annual total rainfall, which decreases from north to south and from west to east, can be used to divide the country into two major zones: (1) the humid Spain of the north down to the central plains (the Meseta), with a warm oceanic climate, continuously mild temperatures and rain throughout the year; and (2) the dry Spain, with warm winters and hot, dry summers.

With regard to seasonal rainfall patterns, Spain is characterized by a large diversity of rainfall patterns. Following Martin-Vide and Cantos (2001), 13 different seasonal rainfall regimes can be distinguished in Spain out of 24 theoretically possible regimes. The four major regimes are: (1) winter maximum→summer

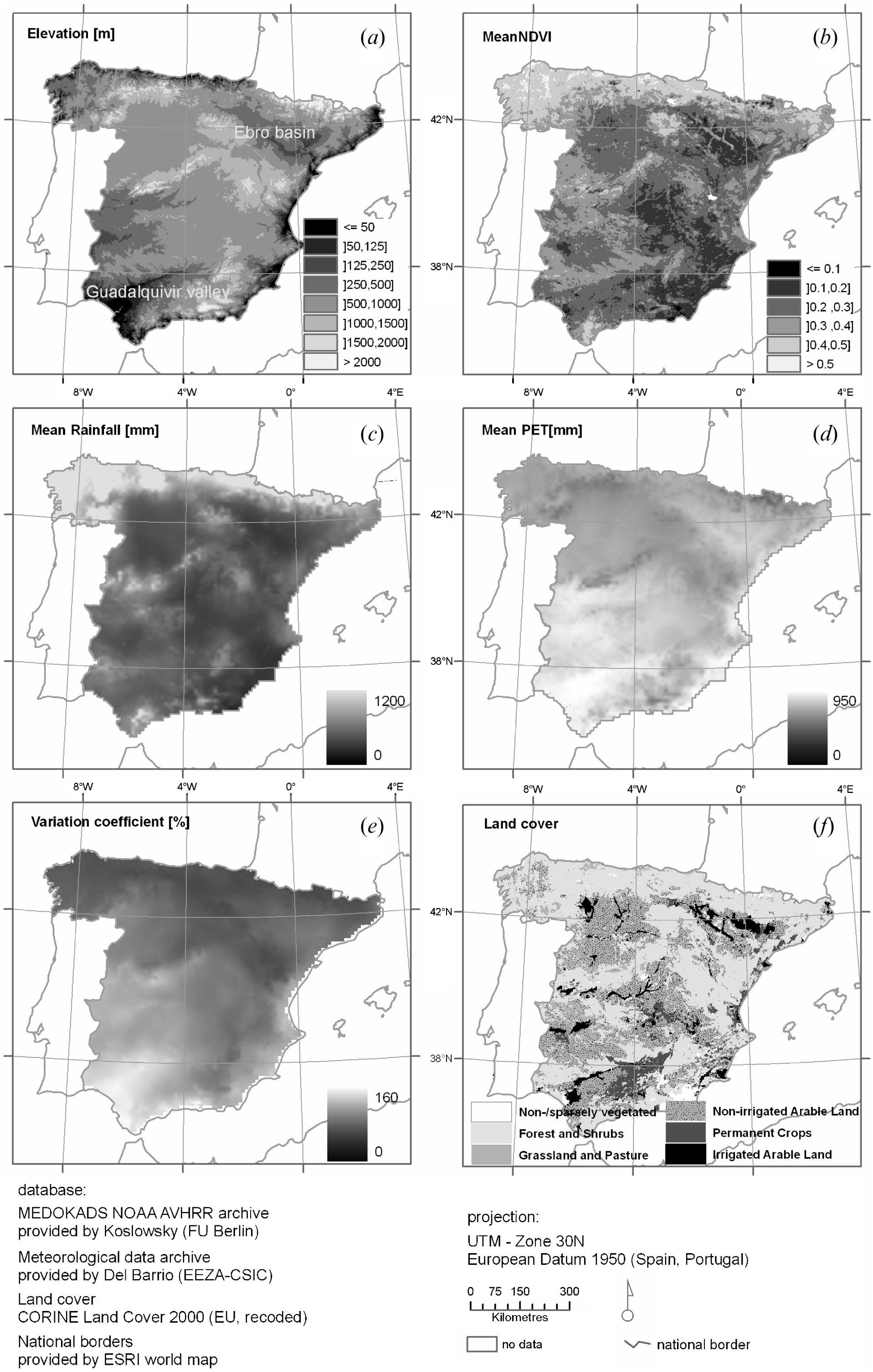

Figure 1. Geographical conditions in Spain: (*a*) elevation, (*b*) mean NDVI, (*c*) mean annual rainfall, (*d*) PET, (*e*) seasonal (monthly) rainfall coefficient of variation and (*f*) land cover. Land-cover classes were derived from the CORINE 2000 classification.

minimum (Atlantic, Cantabria and the southern Mediterranean and the Canaries), (2) summer maximum→winter minimum (Catalonia Pyrenees and one sector of the Cordillera Ibérica mountains (Jiloca-Guadalaviar), (3) autumn maximum→spring minimum (the eastern Mediterranean and the Balearic Isles), (4) spring maximum→autumn minimum (the inland peninsula). Spatial rainfall variability is additionally triggered by intricate relief patterns that generate multiple patches of rainy and dry rain-shadow areas (Wallén 1970). Occasionally climatic anomalies lead to coincidental drought periods in the wetter northern parts of the country and heavy rains in the drier south (Del Moral and Sauri 1999). The percentage of dry days in the northern fringe of Spain with an oceanic climate is less than 65% of the total days. By contrast, the southeast of Spain has more than 85% dry days (Martin-Vide and Gomez 1999).

The alluvial lower terraces of the major river valleys, such as the Guadalquivir and Ebro River, are under permanent irrigation. In the central planes dry farming cultivation is commonly practiced.

The mean annual temperatures of the Iberian Peninsula vary, with the exception of very high altitudes, between 13°C in Cantabria and 19°C on the southeastern coasts and some isolated patches in the south. In the interior the hottest month is July, whereas for most of the Mediterranean coast August experiences the highest temperatures (Wallén 1970).

2.2 *Data*

The top-of-atmosphere (TOA) NDVI data used in this study were derived from the 'Mediterranean Extended Daily One Km AVHRR Data Set' (MEDOKADS). This data set is compiled from the National Oceanic and Atmospheric Administration (NOAA)-11, -14 and -16 sensors and is distributed by the Institute of Meteorology, Free University of Berlin. The archive covers the time period from 1989 to present and comprises full-resolution Advanced Very High Resolution Radiometer (AVHRR) channel data as well as collateral data in latitude–longitude presentation, with a resolution of 0.01° in both directions. The MEDOKADS data are corrected for sensor degradation and orbital drift effects that cause nonlinear changes in the measured signal. The data preprocessing chain is described in Koslowsky (1996) and Friedrich *et al.* (2006): data from every AVHRR instrument comprising the series were intercalibrated to NOAA-11, which in turn were intercalibrated to the NOAA-9 instrument using the absolute in-flight calibration results published by Rao and Chen (1994). The correction coefficients for the two solar channels were derived from cloud-free daily time series from each AVHRR instrument using invariant desert areas and the cloud detection algorithm developed by Koslowsky (1997). The series were adjusted to nadir viewing conditions, radiometrically corrected, adjusted for bidirectional reflectance distribution function (BRDF) effects, and finally smoothed by a 30-day median filter. Before applying the derived correction factors to the MEDOKADS, channel 1 and 2 data were adjusted for Sun height and Sun–Earth distance.

Mean monthly rainfall and temperature data were extracted from a climatic archive of mainland Spain. This archive comprises monthly raster layers between 1970 and 2000 that have been spatially interpolated at a resolution of 1 km from a normalized and georeferenced network of 1661 pluviometric stations and 435 thermometric stations (De Neef 2004). Potential evapotranspiration (PET) was estimated using the method of Hargreaves and Samani (1982). As temporal overlap

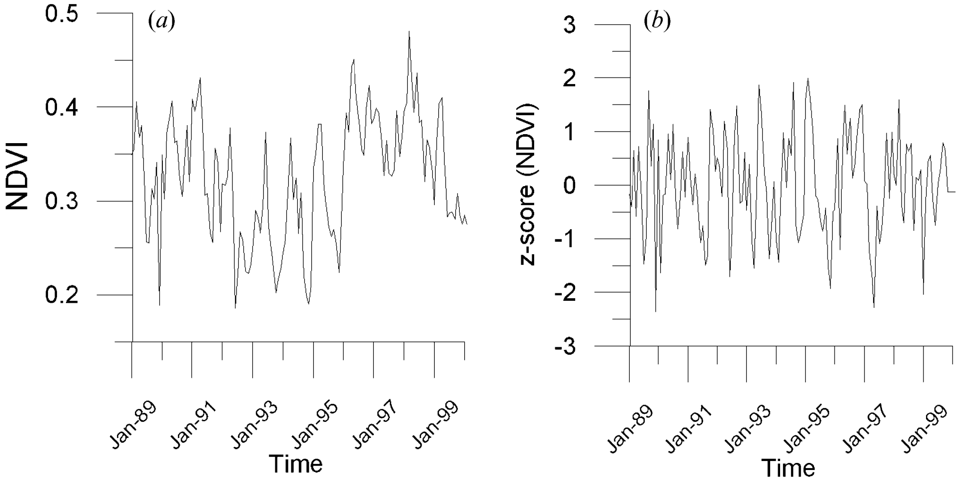

Figure 2. MEDOKADS-NDVI curve from Spain (*a*) before and (*b*) after detrending and *z*-score transformation. Deviations from the zero mean represent monthly positive or negative anomalies, respectively.

between the climatic data and the NDVI archive ranges from 1989 to 1999, the analysis was restricted to the Coordination of Information on the Environment (CORINE) 2000 land cover map (Bossard *et al.* 2000).

2.3 *Methods*

Regression analysis of the series was accomplished at monthly and annual levels. For the monthly models detrended data were used to assure stationarity conditions. One problem in addressing the biomass–rainfall relationship is the influence of seasonal components on the autocorrelation structure. The correlation of two unrelated series containing a common seasonal component gives rise to spurious cross-correlation that arises solely from the similar temporal autocorrelation structure (Chatfield 2004). Therefore, long-term average seasonal effects were excluded from the monthly NDVI and climatic data, and only standardized seasonal anomalies were addressed using seasonal *z*-score transformation:

$$z_{tj} = \frac{x_{tj} - \overline{x_j}}{s_j} \tag{1}$$

where $\overline{x_j}$ and s_j denote the long-term means and standard deviations, respectively, for month j, and t is a time index. The standardized anomalies do not preserve the characteristic differences between the rainfall regimes in Spain that are embedded in the original data. Figure 2 provides an example for an NDVI series before and after *z*-score normalization. At the beginning of the 1990s, figure 2(*a*) shows the enduring influence of a severe drought period on vegetation cover.

To address the multi-annual influence of an extended drought period in the early 1990s on green biomass production, the analysis was repeated using accumulated annual data compiled for the hydrological year in Spain (October to September) without detrending the series.

Regression analysis was carried out using a DL model to assess time-lagged effects of the explanatory variables rainfall amount and mean surface temperature:

$$\mathrm{NDVI}_t = a + \sum_{i=0}^{i=\max} \alpha_i \mathrm{rainfall}_{t-i} + \sum_{j=0}^{j=\max} \beta_j \mathrm{temperature}_{t-j} + \varepsilon_t \qquad (2)$$

where β and α denote the impulse response weight vectors describing the weights assigned to past rainfall and temperature data values. The ε_t are the model residuals and a is the constant term. The maximal number of lags considered in the model (parameter max) was fixed to six months. Other possible explanatory factors for temporal anomalies, such as wildfires, were not considered in the study. Model parameter estimation was conducted following the iterative generalized least-square (GLS) approach described in Shumway and Stoffer (2000) to enable consistent and efficient parameter estimation. The idea of the algorithm is to estimate the model parameters in equation (2) by ordinary least-square (OLS) regression and then to filter the input and output series by an auto-regressive moving average (ARMA) model that has been fitted to the residuals. An iterative process of ARMA filtering and OLS estimation is started until the error series becomes white noise. The procedure converges towards the maximum likelihood solution under normality of the errors.

Regions characterized by significant and nonsignificant NDVI–rainfall relationships, consecutively referred to as sensitive and insensitive areas, were further investigated using stepwise linear discriminant analysis (LDA) for selected CORINE land-cover classes (table 1). The following predictor variables were considered in the analysis: annual means of rainfall amount, surface temperature and PET, mean aridity (rainfall/PET), monthly rainfall coefficient of variation and average peaking times of the NDVI.

3. Results

Figures 1(*b*) and 1(*c*) illustrate the average monthly NDVI and annual rainfall amounts during the observation period. NDVI, PET and rainfall maps show similar spatial patterns. Significant correlations between temporal NDVI and climatic variables, including rainfall and evapotranspiration, have been found for a wide range of environmental conditions (Nicholson *et al.* 1990, Cihlar *et al.* 1991). According to Anyamba *et al.* (2001), the NDVI is an integrated measure of the land surface responses to climate variability at different temporal and spatial scales. The mean annual rainfall during the observation period varied from 200 mm in southeast Spain to more than 1200 mm in Galicia (northwest Spain). With increasing distances from the Atlantic Ocean, the rainfall amount declines from the west to the east and so does the NDVI. Except for parts of Galicia, all areas with more than 1500 mm precipitation are located at altitudes higher than 1500 m (Wallén 1970). The two central plateaus and most parts of the Ebro basin receive rainfall amounts between 300 and 500 mm and the Guadalquivir basin in the south between 500 and 1000 mm.

Figure 3 depicts the results from the monthly DL models. Figure 3(*b*) shows the lag order for the rainfall data; figures 3(*c*) and 3(*d*) illustrate the related *t*-statistics for the regression coefficients at lag zero and one. Figures 3(*e*) and 3(*f*) show the results for the *t*-tests related to the monthly temperature data. Light-coloured locations highlight significant positive (α=0.1) and dark colours negative relationships. White colours represent regions with nonsignificant regression coefficients or where the specified lag order was not achieved.

Table 1. Results from the stepwise linear discriminant analysis (*F*-value IN=3.84, *F*-value OUT=2.71) using a random subset of samples from selected CORINE land-cover classes (each 120 pixels). Regions with significant lagged NDVI–rainfall correlation (class 1) were separated from those without (class 2) using the variables average monthly rainfall amount (Rf), average monthly temperature (*T*), mean monthly potential evapotranspiration (PET), aridity (Rf/PET), rainfall coefficient of variation (Rf-Varkoeff) and annual peaking time of the NDVI (α=10%, lag order $\geqslant$1). Both the standardized discriminant coefficients and the factor loadings are shown (*n*=720). Wilks' lambda for all discriminant models is highly significant (α<0.1%).

	Arable land	Permanent/ annual crops	Agroforestry areas	Forest	Natural grassland	Transitional woodland, scrub-vegetation
Insensitive (km^2)	93 442	128	5263	66 040	12 286	54 053
Sensitive (km^2)	81 464	165	21 542	32 264	13 145	36 036
Standardized discriminant coefficient						
Rf						
T						0.47
PET	0.56					
Greenpeak	−0.55	0.75				
Aridity				−0.23		
Rf-Varkoeff		0.63	1.00	0.91	1.00	0.64
Structure matrix (loadings)						
Rf	−0.44	−0.20	−0.47	−0.50	−0.48	−0.19
T	0.83	0.69	0.72	0.63	0.77	0.86
PET	0.90	0.74	0.73	0.68	0.81	0.90
Greenpeak	−0.90	−0.13	−0.48	−0.67	−0.93	−0.68
Rf/PET	−0.50	−0.03	0.15	−0.43	−0.06	−0.76
Rf-Varkoeff	0.72	0.79	1.00	0.96	1.00	0.93

Figure 3(*b*) demonstrates that the majority of DL models comprise 0 and 1 month of lagged rainfall data. In southern and western Spain locally lagged NDVI–rainfall dependencies up to 3 months occur. Negative correlations at lag zero, which mainly occur in the western part of Spain, can be explained by a coincidence of negative monthly NDVI and positive rainfall anomalies in autumn and winter and a reverse situation in late spring and summer. This might occur occasionally despite removing the average seasonal long-term effects from the series. By contrast, the positive correlations at higher lags demonstrate the dependence of plant biomass production on accumulated previous rainfall amounts. Monthly temperature anomalies explain less variability in the NDVI compared to the rainfall data. At zero time lag, negative correlations in the Ebro basin and in southeast Andalusia are due to the occasional coincidence of positive NDVI and negative temperature anomalies in early spring and the opposite conditions in late summer/autumn. However, at most locations significant relationships disappear at higher-order lags.

Considering a lagged time interval of 1 month, the southern foreland of the Pyrenees in Catalũna and the western Extremadura region (figure 3(*d*)) are characterized by a significant positive rainfall–NDVI correlation. The related areas in Extremadura and the western part of Castilla y Leon are mainly used for rainfed agriculture and locally mixed with agroforestry areas and grazing land (figure 1(*f*)). The multiple correlation coefficients (not shown) of the DL models vary between

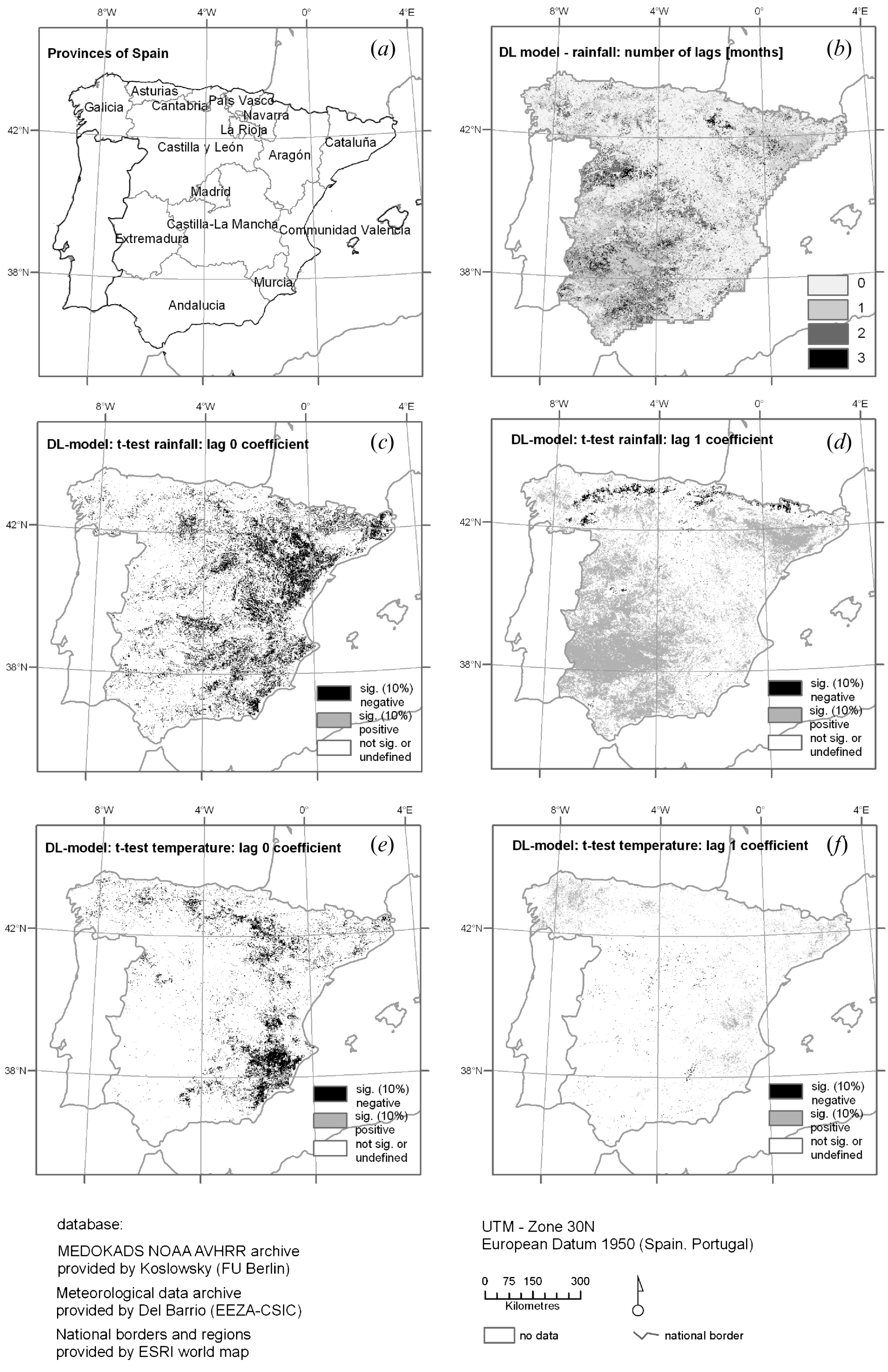

Figure 3. Results from DL models: (*a*) the autonomous communities in mainland Spain; (*b*) the total number of rainfall lags achieved in the monthly models; maps (*c*) and (*d*) illustrate the significance of the rainfall regression coefficients at concurrent (lag zero) and for 1 month shifted (lag one) NDVI–rainfall relationships; (*e*) and (*f*) show the respective significances for the temperature regression coefficients.

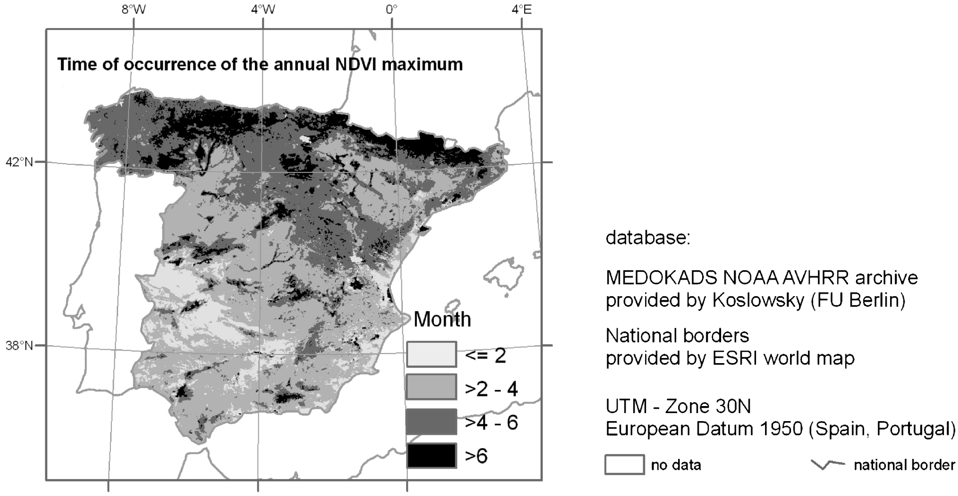

Figure 4. Average peaking times (months) of the NDVI in the observation period.

zero and 0.6. The mountain ranges of the Pyrenees, Cordillera Central and South Nevada, as well as permanently irrigated areas, in which net primary production is decoupled from rainfall, lack significant relationships. The maximum correlation coefficient of 0.6 from our study corresponds well with the results reported by Eklundh (1998), who used a comparable approach to describe relationships between NDVI and rainfall anomalies in East Africa.

Figure 4 illustrates the time of occurrence of the annual green peak. In areas where DL models are significant, the NDVI maxima occur on average between February and April. Comparison with the rainfall data reveals that, with the exception of Castilla y Leon and Catalũna, NDVI peaks and winterly rainfall maxima are not more than 3 months apart. Plants in those regions also benefit from a small secondary rainfall maximum during the spring period. There is a remarkable coincidence between the DL model results in figure 3(*b*) and the spatial patterns in the PET and the seasonal rainfall coefficient of variation (figures 1(*d*) and 1(*e*)).

The results of the LDA, which was carried out for individual land-cover classes to explain differences between sensitive and insensitive areas, are summarized in table 1. The data show that the relationship between the number of pixels with significant and missing rainfall–NDVI correlations is more or less balanced in the land-cover classes arable land and natural grassland. By contrast, the majority of pixels with permanent vegetation (CORINE classes forest, transitional woodland/ scrubland) are insensitive. Agroforestry areas, which are mainly confined to Extremadura, and permanent/annual crops are mostly sensitive towards previous rainfall conditions. In the CORINE nomenclature, agroforestry areas describes a mixed land-cover class comprising annual crops or grazing land under the wooded cover of forestry species. Aggregated to a ground resolution of 1 km^2, this gives rise to spectrally overlapping signatures of the individual components. In figure 1(*f*) this land-cover class has been merged with nonirrigated arable land.

Table 1 also highlights the standardized discriminant function coefficients and the related factor loadings, which describe the correlation of every independent variable with the discriminant functions. In neither case have more than two variables been included in a model by stepwise variable selection. With exception of the model that

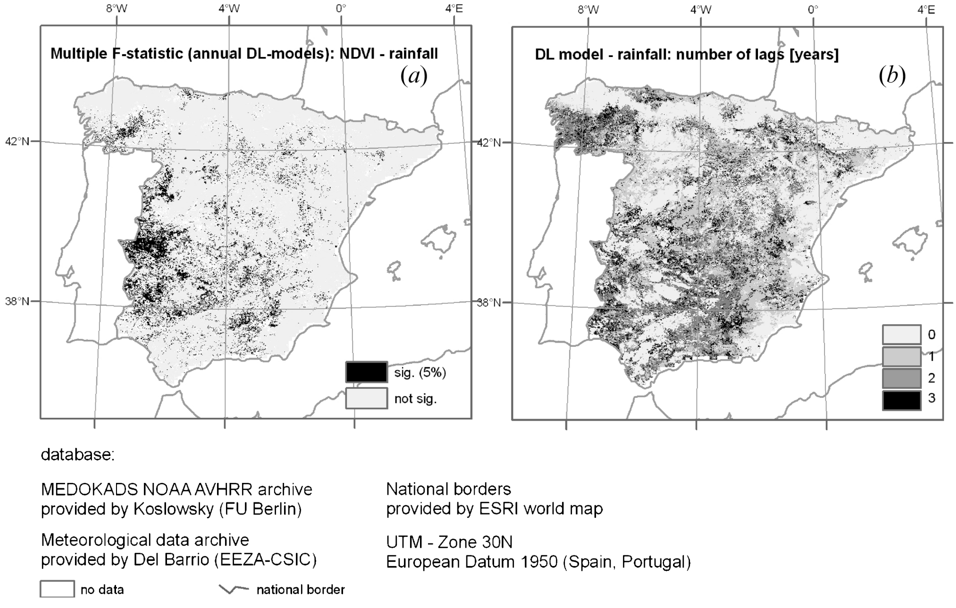

Figure 5. Statistics for the annual DL models: (*a*) the overall significance (*F*-test) of the models and (*b*) number of lags (rainfall) included.

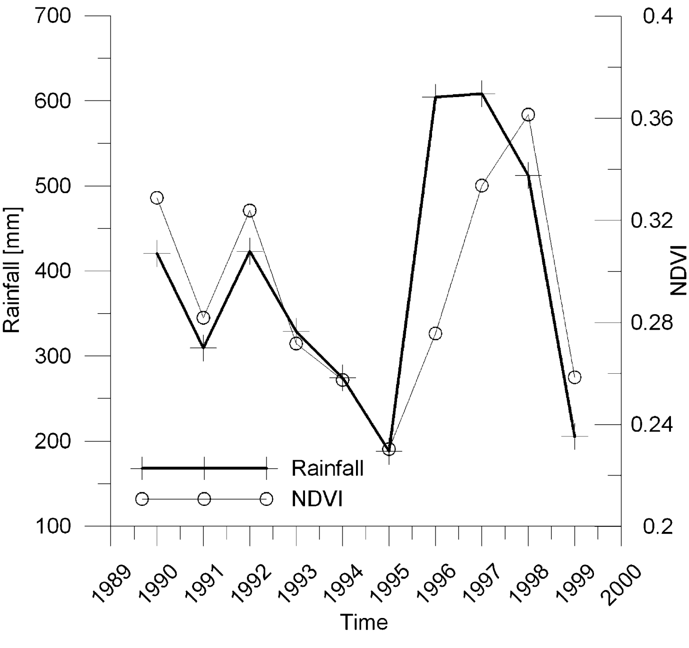

Figure 6. Annual rainfall and mean annual NDVI (hydrological year) from an area in western Spain (Extremadura) covered with semi-natural vegetation (grassland and scrubland) showing a significant annual NDVI–rainfall relationship.

describes the conditions in the class arable land, the rainfall coefficient of variation rather than the average total rainfall amounts explains most of the variability between the two groups. Negative loadings are recorded for the peaking time for the NDVI and rainfall/PET quotients. Positive loadings are common for mean temperatures, PET and the rainfall coefficient of variation. These factors recurrently trigger water deficit, which can be considered as a common background variable to explain the differences between the sensitive and insensitive areas for all land-cover types considered.

Figure 5 shows the results for the annual DL models. Here, the model order varies between zero and three, indicating lagged dependencies of the NDVI from previous accumulated annual rainfall amounts up to 3 years. However, the limited number of data points comprising the series leaves many areas that do not show statistically significant relationships at $\alpha=0.10$. Significant locations mainly correspond to areas with natural grass and scrub vegetation in western Spain and to a minor degree to agroforestry areas.

With regard to this specific region, figure 6 depicts the graphs of the average annual NDVI and total annual rainfall collected from an area in Extremadura covered with semi-natural vegetation. Both graphs demonstrate a close NDVI–rainfall association with a decline during the early drought period until 1995 and a rebound thereafter until 1998. Below-average rainfall amounts in 1999 caused another temporal decline in the annual mean NDVI values.

4. Discussion

The monthly NDVI–rainfall relationship in Spain shows a distinct spatial pattern corresponding to regional variations in aridity, land use and terrain morphology. The majority of the significant DL models comprise monthly rainfall data with a time lag of 1 month. Only occasionally do higher lag orders occur, indicating a longer influence of water deficit on plant growth. In accordance with other studies, regions in Spain showing significant NDVI–rainfall correlations are confined to average rainfall/PET ratios between 0.5 and 2 and annual rainfall amounts less than 700 mm. In South Africa, Richard and Poccard (1998) found the highest NDVI–rainfall correlation at annual rainfall amounts between 300 and 900 mm. In China, Li *et al.* (2002) identified most significant NDVI–rainfall relationships at rainfall amounts between 500 and 700 mm per year. For Botswana, Nicholson and Farrar (1994) reported a threshold of 500 mm per year, below which the NDVI temporal behaviour dynamic was sensitive towards soil moisture.

Precipitation and temperature are the two major climatic factors that govern the absorption of photosynthetic active radiation and the associated gross primary production of the biosphere (Cramer *et al.* 1999). To enable plant growth, a certain minimum temperature threshold has to be exceeded (Schultz and Halpert 1993). However, in our study variability in surface temperatures only explained a negligible amount of the variability in the NDVI in the context of a multivariate DL model, even in the high elevation areas of Spain where plant growth is largely determined by temperature. This result is in agreement with other studies from Syria (Evans and Geerken 2004) and the Great Plains in the USA (Wang *et al.* 2003).

The moderate multiple correlation coefficients of the DL models suggest that additional factors have to be considered to explain the temporal variability in the vitality of a plant. As plant growth in dry areas is largely controlled by local rainfall variation, phenologies vary between years (Wagenseil and Samimi 2006). Supplementary factors that cause nonuniform patterns in the growth cycle include plant nutrition status, diseases, management adaptations and land-cover changes (Udelhoven and Hill 2008).

The results from the LDA suggest that water deficit rather than rainfall amount is the common background variable that recurrently discriminates sensitive from insensitive areas in all land-cover classes considered. Pimentel *et al.* (1999) report that, in the Mediterranean, one hectare of corn (7000 kg/ha yield) is able to transpire about four million litres of water during one growing season while an additional two million litres are concurrently evaporated from the soil. Water deficit, which is caused by a combination of high temperature, PET and rainfall variability accompanying low rainfall amounts, has been identified as the major constraint on plant productivity (Rampino *et al.* 2006). Ciais *et al.* (2005) and Rampino *et al.* (2006) state that environmental stress is amplified if a drought coincides with high summer temperatures accompanying high PET levels and soil water deficits. In general, below a threshold of 0.4 in relative extractable water, stomatal closure and water stress occur, which gradually decrease gross primary production and transpiration (Granier *et al.* 1999). In severe drought years the relationship between biomass production and water deficit is most obvious, such as the European-wide reduction in primary productivity in 2003, where the gross primary production declined in accordance with reduced evapotranspiration and soil drying due to extreme summer heat (Ciais *et al.* 2005).

In the Ebro basin and the eastern parts of Andalusia that experience the highest PET levels in Spain ($>$900 mm), NDVI–rainfall relationships vanish at a monthly level. Both regions represent the driest conditions in Spain, characterized by a rapid depletion of the soil water after winter rainfalls by evapotranspiration. Presumably, time aggregation of 1 month was too coarse to identify significant relationships in those areas because of the irregular rainfall distribution in time. By contrast, Galicia and Asturias in northwest Spain experience abundant rainfall across all seasons; those regions are dominated at low PET levels by insignificant lagged NDVI–rainfall inter-relationships (compare figures 1(*d*) and 3(*d*)) because higher annual precipitation and comparable low PET rates create buffered soil moisture regimes throughout the year. Therefore, the monthly NDVI–rainfall relationship tends to fade out both in arid and humid areas.

Deep-rooting annual and perennial plants, such as perennial shrubs and forest trees, are less sensitive towards interseasonal climatic variability than more shallow-rooted annual crops and grassland (Li *et al.* 2002, Wang *et al.* 2003). This is confirmed by the fractions of pixels belonging to different land-cover classes in the sensitive and insensitive groups. The large spatial corridor with missing NDVI–rainfall relationships, which spans from Asturias in the northwest to Murcia in the southeast of Spain (figures 3(*b*) and 3(*d*)) is covered with wooded plant species, primarily forest and scrublands. These plants are adapted to draw water from deeper soil layers, making them to a lesser degree dependent on previous rainfall amounts. A buffered soil water regime enables woody species to produce biomass even during drought periods. The missing NDVI–rainfall association along the ridges of Spain's high mountains can be explained by the rugged terrain, which generates rapid (sub)surface runoff of incoming rainfall. Furthermore, it should be considered that at low vegetation abundances the NDVI lacks sufficient sensitivity to detect subtle changes in the flora at coarse scales (Schmidt and Karnieli 2001).

An enduring influence of severe drought periods on biomass production is visible from the significant annual NDVI–rainfall relationships especially in the western part of Spain with semi-natural vegetation cover. Positive correlation at delayed time intervals of up to 2 years suggests that abnormal growing seasons might influence vegetation growth of perennial plants in the following season, for instance when aquifers cannot be refilled during a drought year. In figure 6 the common trend in the NDVI and rainfall series is due to the extended drought at the beginning of the observation period and the following recovery period, which both introduce temporal autocorrelation in the data. The graph reflects both the strong relationship between annual mean NDVI and annual total rainfall and memory effects imposed on vegetation by a dry spell. In the period from 1996 to 1998, this region received above-average precipitation after the drought period, but the green biomass reacted with a temporal delay on the more favourable conditions. For Mediterranean macchia-type ecosystems, the response of the vegetation–soil system to differences in water stress might last for 3 to 5 years (Bolle 1999).

5. Conclusions

Significant relationships between lagged NDVI and rainfall anomalies up to 3 months are confined to the subhumid and semi-arid areas in Spain. The common background variable to distinguish sensitive from insensitive areas is water deficit, which is triggered recurrently by high monthly rainfall variability, high PET and surface temperature values and low rainfall amounts. Crops and grassland in the semi-arid regions of Spain are more sensitive towards water stress than perennial woody species. The results further demonstrate that, in the sensitive regions, severe drought periods might have an enduring influence on biomass production in subsequent years. Empirical studies like the present one are useful for discriminating between climatic and human-driven changes in surface condition and provide the basis for predicting changes in productivity that accompany climatic anomalies. It is worth investigating whether the results from this study can be confirmed by replacing the NDVI by an alternative measure of vegetation coverage showing a higher dynamical range at low vegetation abundances in semi-arid and arid areas.

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Chapter IV

Mediterranean desertification and land degradation - Mapping related land use change syndromes based on satellite observations

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Mediterranean desertification and land degradation
Mapping related land use change syndromes based on satellite observations

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ABSTRACT

In past decades, the European Mediterranean has undergone widespread land use transformations. These are largely driven by changes of socio-economic conditions, such as accession to the European Community, and had strong effects on the way the land is being used. Aiming at a systematic description of such change processes on a global level, the syndrome concept was proposed to describe archetypical, co-evolutionary patterns of human–nature interactions, and has been specifically linked to the desertification issue.
In this study, we present an adaptation of the syndrome approach to the Iberian Peninsula. We suggest a data processing and interpretation framework to map the spatial extent of specific syndromes. The mapping approach is based on the time series analysis of satellite data. We have characterized vegetation dynamics using NDVI estimates from the coarse scale, hyper-temporal 1-km MEDOKADS archive, which is based on calibrated NOAA–AVHRR images.
Results indicate that local patches of abrupt disturbance, mainly caused by fire, are contrasted by a widespread increase in vegetation, which is in large parts attributed to the abandonment of rural areas. Although this questions the dominance of classical desertification traits, i.e. decline of productivity after disturbance, it is concluded that the recent greening presents a different sort of environmental risk, as it may negatively impact on fire regimes and the hydrological cycle.

1. Background

1.1. Global change and mediterranean land degradation

Accelerated transformations in the earth system are raising increasing concern (Millenium Ecosystem Assessment, 2005b), and there is compelling evidence that global environmental change is largely due to the activities of people, manifesting in the alteration of almost all terrestrial ecosystems (Steffen et al., 2004; Vitousek et al., 1997). There is a wealth of recent investigations on effects of global change, suggesting for instance that more than 50% of natural ecosystems have been altered by superimposed human use (Kareiva et al., 2007); marine ecosystems are at the brink of collapse due to heavy exploitation (Worm et al., 2006); biodiversity is declining at unprecedented rates (Pimm and Raven, 2000); and greenhouse gas levels are dramatically changing as a result of population growth and economic activities (Steffen et al., 2004). At the same time, there is increasing awareness of the dependence of people on the Earth's ecosystem goods and services (Ayensu et al., 1999). Since the interactions of people and the environment mostly play out at the land surface, land use change has become a major driver and indicator of environmental change (Lambin and Geist, 2006). This might take the form of conversion of natural landscapes for anthropogenic use or by changing management practices in human-dominated landscapes (Foley et al., 2005). Although these changes have increased the provision of particular ecosystem services, such as food, fibre etc., they may also degrade the environment and cause a loss of other important ecosystem services in the long-run. These complex interactions necessitate a coherent approach to understand the coupled human–environment system (Millenium Ecosystem Assessment, 2005a; Rindfuss et al., 2004).

Human population and natural renewable resources are main components of human–environment systems which can be affected by climatic or socio-economic disturbances (Brandt and Thornes, 1996). Large areas of the European Mediterranean are believed to be affected by land degradation processes that lead to desertification mainly emerging from conflicts between past and present land uses, or between economic and ecological priorities. Under steady-state conditions, intensity and duration of disturbances remain within the range of those that have appeared throughout the history of the system (Puigdefábregas and Mendizabal, 1998). Land degradation may then be conceived as a process in which disturbances have

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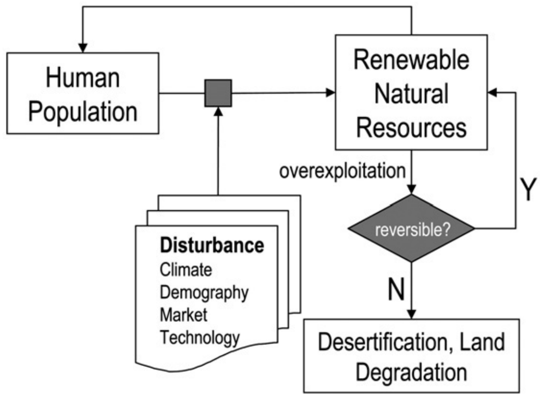

Fig. 1. Impact of disturbance on the feedback loop between natural resource availability and their use/exploitation by human activities (after Puigdéfabregas and Mendizabal, 2004).

gone beyond the resilience of the land and have caused, at human timescales, an irreversible loss of the land's carrying capacity or biological production potential (Fig. 1). It embraces all degradation processes which imply an irreversible loss of, or reduction in, potential productivity of the land in dry climates (e.g. Puigdefábregas, 1995). If the system is endowed with feedback mechanisms to reverse this condition, it can recover and return to the steady state. Otherwise it falls into an overexploitation loop that leads to further decline, thus constituting desertification (Brandt and Thornes, 1996; Millenium Ecosystem Assessment, 2005a).

In the Western Mediterranean the past decades were characterized by significant land use transitions including both intensification and extensification (Hobbs et al., 1995; Millenium Ecosystem Assessment, 2005a). These include such diverse processes as urbanization, industrialization, intensification of agricultural and pastoral uses on the one hand, and the deterioration of rural structures and land abandonment accompanied by shrub encroachment on the other, affecting water budgets and land resources (Brandt and Thornes, 1996; Mazzoleni et al., 2004; Mulligan et al., 2004). In many countries of the Northern Mediterranean, accession to the European Community and its funding schemes has been a major driver of socio-economic change (Dubost, 1998).

1.2. Syndromes of degradation

The various definitions of and perspectives on desertification (e.g., Thomas and Middleton, 1994) will not be repeated here but their sheer existence is indicative for the fact that it is still difficult to identify a unifying concept or explanation for desertification (e.g., Geist, 2005; Geist and Lambin, 2004). Similar complexity exists in the arena of global change, and has been tackled by developing the *syndromes* (Cassel-Gintz and Petschel-Held, 2000; Lüdeke et al., 2004; Petschel-Held et al., 1999; Schellnhuber et al., 1997). The basic idea behind syndromes is not to describe Global Change by regions or sectors, but by archetypical, dynamic, co-evolutionary patterns of human–nature interactions. Syndromes (as a "combination of symptoms") describe bundles of interactive processes and symptoms which appear repeatedly and in many places in typical combinations and patterns. The syndrome concept is considered one of the most promising approaches to place-based integrated assessments of global change. The syndrome framework needs further adjustments and concretisation in order to be applied to drylands. A first attempt in linking the defined syndromes more explicitly to the desertification issue has been proposed by Downing and Lüdeke (2002). They primarily focused on linking vulnerability concepts to degradation processes, and identified those global change syndromes (out of the selection proposed by Schellnhuber et al. (1997)) which are most relevant to desertification. However, what is not yet achieved is taking into account the specific dryland development perspectives and challenges (Reynolds et al., 2007). Taking stock of the empirical work of Geist and Lambin (2004) and Geist (2005) who have identified major causal patterns and underlying forces (human and/or biophysical) associated with desertification, it is of prime importance to strengthen the analytical understanding of these pathways and interaction mechanisms.

In this line of argument two facets of the syndrome approach need to be discerned. First of all, the syndrome perspective is a powerful conceptual framework when it adopts formalisms for combining descriptive variables into functional patterns of interactions. Secondly, it provides a methodological framework for integrating the knowledge about functional relationships in human–environment systems with innovative combinations of dynamic simulation, qualitative differential equations, fuzzy sets and geographic information systems (Downing and Lüdeke, 2002). It is the major objective of this paper to demonstrate that remote sensing data archives can provide a complementary source of information for identifying the existence of syndromes of land use change processes at a landscape level.

1.3. Remote sensing of land degradation

A particularly important issue in assessing and monitoring desertification is to gain an overview about affected areas, and to connect the global dimension with regional and local processes. Remote sensing data are of considerable value in the context of monitoring environmental processes. With the history of operational earth observation sensors reaching back over three decades, they allow retrospective analysis of the state and development of ecosystems at different scales and with different spatial coverage. Remote sensing data adhere to the principles of repetitiveness, objectivity and consistency, which are prerequisites in the frame of monitoring and surveillance (Hill et al., 2004). Consequently, 'ecosystem observatories' based on earth observation satellite data and additional information have frequently been suggested, with the goal of serving requirements of policy-making, planning and land management (Group of Earth Observation, 2005).

Truly quantitative measurements of desertification have been "an elusive goal" since the earliest attempts of UNEP in the "World Atlas of Desertification" (Middleton and Thomas, 1997) as they relied on expert assessments, not on objective measurements. Although remote sensing has substantially contributed to correcting the "myth of the marching desert" (Forse, 1989; Helldén, 1991), and although it helped clarifying that desertification is not occurring across the entire Sahel but may be occurring in specific areas, no systematic surveys have been conducted on global level.

Soil and vegetation properties are tangible items which can be assessed using remote sensing approaches (Hill et al., 2004), but there is no indicator of degradation that is directly inferable from satellite-based data. Suitable indirect indicators need to be defined, which can be related to processes of erosion, degradation, increase of flammable vegetation volume, etc. (Perez-Trejo, 1994; Verstraete, 1994). Secondly, land degradation essentially operates in the time dimension and can be conceptualized as a pathological process of multi-annual land cover dynamics (Prince, 2002). Correspondingly, indicators need to be derived for a sequence of time steps and the time dimension needs to be incorporated into the analysis (Gutman, 1999; Lu et al., 2004).

A wealth of studies exists that assess landscape dynamics in regions affected by desertification and land degradation. Using a variety of data processing and analysis concepts, these provide useful insights on process-patterns on regional to global (Lambin and Ehrlich, 1997; Tucker and Nicholson, 1999; Eklundh and Olsson, 2003; Geerken and Ilaiwi, 2004; Herman et al., 2005; Wessels et al., 2007b) as well as local scales (Hill et al., 1998; Petit et al., 2001; Hostert

Table 1
Characteristics of the Mediterranean Extended Daily One-km AVHRR Data set (Friedrich et al., 2006)

Time period:	1989–present
Resolution:	0.01° (~1.1 km)
Source:	Direct read out High Resolution Picture Transmission (HRPT)
Region:	West-, Middle-, East-, and Southeast Europe, North Africa
Projection:	geographical projection
Satellite:	NOAA-11, -14, -15, -16, -18
Available data:	daily data, 10-day composites and 30-day composites for channel 1–5, satellite/sun zenith angle, satellite/sun azimuth, broadband albedo, bitmap (origin indicator, landmask, cloudmask), NDVI, scattering angle sun satellite, sea surface temperature (SST), land surface temperature (LST), time since equator
Distributor:	Koslowsky, D., Free University of Berlin, Meteorologische Satellitenforschung

et al., 2003; Röder et al., 2007; Röder et al., 2008b). Since required observation periods typically exceed the life-span of individual satellites one needs to establish calibrated archives which include data from several identical or conceptually similar observation systems. Today, several satellite-specific archives for different periods of time exist but connecting them with the intention of covering time spans of several decades with homogeneous data products still remains a major challenge. In particular, a comprehensive framework for an integrated, consistent and widely applicable observation of large areas, that should consider the respective dominant process framework, has so far not been established.

1.4. Objectives: Mapping degradation-related land use change syndromes

Concluding the previous sections, the prime research objective lies in developing a consistent and reproducible analytical approach meeting two major criteria: it should identify the spatial extent of major degradation-related land use change syndromes for a larger section of the Mediterranean Basin; and place them within a coherent interpretation framework. Furthermore, it will be of importance to understand whether the identified processes are compliant with the conceptual framework proposed by Puigdéfabregas & Mendizabal (2004), i.e. whether disturbance effects have caused (irreversible) losses of carrying capacity or biological production potential.

We suggest this can be achieved by pursuing the following goals:

- extend the syndrome approach to accommodate specific degradation-related land use change characteristics;
- derive parameters indicative of landscape dynamics and land use change from suitable remote sensing long-term archives;
- establish a framework for mapping spatio–temporal patterns and interpret them within a syndrome-based framework; and
- validate the achieved results with regard to existing cases studies on land degradation dynamics.

As a first step towards a dryland observatory system, this paper sets a focus on natural and semi-natural vegetation communities of the Iberian Peninsula to address these goals and presents first results from the analysis of coarse-scale hypertemporal NOAA–AVHRR data.

We describe the materials and methods used (§2), followed by a brief presentation of results from the time series analysis (§3). Then, we suggest a rule-based classification approach to map land use change processes associated with specific syndromes (§4.1). The results are shown and discussed in the light of local case studies serving as a source of interpretation and validation (§4.2). We conclude with perspectives on how the approach may be extended spatially and conceptually to eventually form an element of an integrated dryland development strategy (§5).

2. Materials and methods

2.1. Materials

For monitoring the development of vegetation cover at a regional to global scale archives of satellite data with coarse geometric but high temporal resolution have been successfully employed (Tucker and Nicholson, 1999; Goetz et al., 2006). Due to their high repetition rate these data allow a better geographical coverage and temporal monitoring of areas at the expense of spatial detail. At present, the NOAA–AVHRR sensors provide the most comprehensive time series of satellite measured surrogates for surface condition. The NDVI which is derived from the reflectance bands in the red and near infrared domain is a commonly accepted surrogate for vegetation cover, volume and vitality. The present study is based on the "Mediterranean Extended Daily One Km AVHRR Data Set" (MEDOKADS) processed and distributed by the Free University of Berlin (Koslowsky, 1996). The MEDOKADS archive, which covers the time period from 1989 to present, comprises full resolution AVHRR channel data and additional collateral data with a geometric resolution of about 1 km^2 (Table 1).

Calculation of the NDVI requires careful pre-processing of the AVHRR red and near infrared bands in order to allow a meaningful trend analysis. This includes correction for sensor degradation and orbital drift effects that cause non-linear changes in the measured signal, as well as an inter-calibration between the AVHRR/2 (NOAA-11 and NOAA-14) and AVHRR/3 (NOAA-16) sensors to prevent inhomogeneities of the time series. A detailed description of the data pre-processing is given by Koslowsky (1996, 1998) and Friedrich et al. (2006). Despite the applied correction scheme a bias between corresponding NOAA-14 (until 2000) and NOAA-16 (after 2000) data representing areas without long-term trends was found and accounted for by correcting the data by means of an empirically derived equation (Udelhoven, 2007; Udelhoven and Stellmes, 2007). After this adjustment the comparison of the MEDOKADS archive and the SPOT Vegetation time series for the overlapping time period from 1999 to 2004 shows very good agreement of the derived NDVI trends (Fig. 2).

In this study, we used a MEDOKADS subset that covers the Iberian Peninsula to demonstrate the suggested syndrome-based mapping approach. Ten-day NDVI maximum value composites (MVC) from January 1989 to December 2004 were used for the frequency analysis

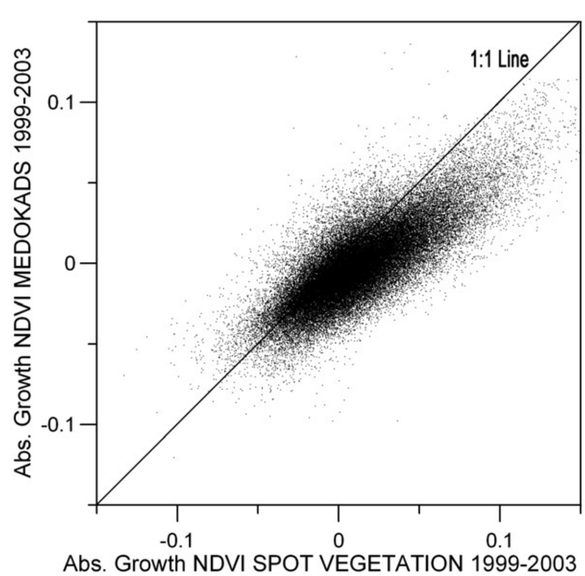

Fig. 2. Comparison of trends in NDVI-based vegetation cover derived from MEDOKADS and SPOT VEGETATION data archives.

of the NDVI signal, whereas 30-day NDVI MVCs served as basis for a trend detection of vegetation cover and the Augmented Dickey–Fuller test. A Savitzky–Golay filter was applied to the 10-day MVCs by means of the method proposed by Chen et al. (2004) to reduce the noise within the time series.

The stratification of the study area was derived from the Corine Land Cover Classification 1990 (CLC1990) raster data with a geometric resolution of 100 m, which is distributed by the European Environmental Agency (EEA, 2007). The utilized raster dataset differentiates 44 land use classes and originates from the vector CLC database provided by the National Teams at a scale of 1:100,000 and a minimum mapping unit of 25 ha (EEA, 2007). The land cover classes were recoded to distinguish between cultivated and semi-natural land and were aggregated to the geometric resolution of the MEDOKADS data by means of a majority filter.

2.2. Methods

The time series calculations were carried out using the TimeStats software package, which was specifically developed for analyzing long-term hypertemporal satellite data archives (Udelhoven, 2006). For extracting information describing the underlying processes several time series parameters were derived from the quasi-continuous MEDOKADS time series. One of the most important descriptors of vegetation dynamics is its long-term trend (increase and decrease) which is determined by a linear regression analysis. Trend analysis of the NDVI series was restricted to yearly aggregated monthly MVC values from March to September to minimize uncertainties due to large sun-zenith angles which occur during the winter period. The statistical reliability of the derived regression coefficient was determined by means of a Student's t-test.

Due to the high temporal resolution provided by the NOAA–AVHRR systems phenological information can be derived from the time series, such as the magnitude of the annual cycle and the time of the occurrence of the annual maximum vegetation cover (phase). For each year the annual and semi-annual magnitude as well as the phase was derived from decadal MVCs by fitting a Fourier polynomial x_t (Shumway and Stoffer, 2006) considering the annual (36 10-day periods) and the semi-annual cycles (18 10-day periods).

$$x_t = a_0 + \phi_1 \cos 2\pi \frac{1}{36} + \phi_2 \sin 2\pi \frac{1}{36} + \phi_3 \cos 2\pi \frac{2}{36} + \phi_4 \sin 2\pi \frac{2}{36} \quad (1)$$

where a_0 is the constant of the Fourier polynomial and Φ_n are the Fourier coefficients.

Subsequently, linear regression was applied to the 16 resulting values of the annual and semi-annual magnitude and phase wherever specific conditions are fulfilled. For example, the determination of a phase shift seems only meaningful if a phenological cycle exists. Therefore the shift of the peaking time was only determined where the associated magnitude is significant and exceeds a certain threshold. Similarly, an observation of the semi-annual magnitude trend is only justified if the semi-annual magnitude exceeds a threshold and simultaneously the semi-annual magnitude is greater than the annual magnitude.

For the detection of abrupt events (e.g. fires) linear trend analysis is not the preferred means, as the magnitude and the direction of the calculated trends are largely dependent on the time of occurrence within the time series. The Augmented-Dickey–Fuller (Dickey and Fuller, 1979) test examines the probability of so-called random walks (sustained disruption) in a time series. The ADF test accounts for integrated processes (I(1)-processes), a special form of auto regressive (AR(1)-) processes (Shumway and Stoffer, 2006). Regarding vegetation cover the Augmented-Dickey–Fuller test determines whether or not the respective time profile is characterized by a sustained event that caused a drastic disruption within the observed time span. Hence, sudden events, for instance caused by wild fires or abrupt changes of land use, can be detected by means of this test.

The decision if a series, say, $\{X\}_{t=0}^{T}$ following an AR(1)-process statistically equals a random walk or if it is (weakly) stationary can be made using following equation (Dickey and Fuller, 1979):

$$\Delta x_t = \alpha + \beta t + \rho x_{t-1} + \sum_{i=1}^{p} d_i \Delta x_{t-i} + \varepsilon_t \quad (2)$$

The term βt describes the influence of an external trend and α is the constant of the model. The purpose of the lags of Δx_{t-1} is to ensure that the error terms are white-noise (ε).

The test is then for $H0$: $\rho=0$. The distribution of ρ is right-skewed and known as Dickey–Fuller (D–F) distribution (Dickey and Fuller, 1979). Before applying the ADF-test to the time series the data have to be de-seasonalized by applying Eq. (1).

3. Results

Fig. 3 illustrates the derived parameters from the time series analysis of the NDVI MEDOKADS archive for the Iberian Peninsula (1989–2004). Three of the individual layers (i.e. absolute growth (trend) of vegetation cover, phase shift and trend of annual magnitude) were combined into one false-color-composite image to illustrate their explanatory power. In a second step the color composite was stratified using the CLC1990 data to distinguish between cropped and semi-natural areas (Fig. 4).

The color combination of 3 layers (NDVI trend, annual phase shift, magnitude trend) clearly emphasizes the combined variance of the extracted time series parameters, where the dominance of reddish tones suggests that the observed 15-year time window is primarily characterized by an increase in vegetation cover and volume in the cultivated and semi-natural strata. Before we concentrate on an in-depth analysis of the semi-natural vegetation systems (the focus of this paper) the explanatory power of the combined data layers is shortly illustrated with reference to selected examples from the cropped land.

Strong positive trends of vegetation cover (reddish tones) in stratum “cultivated” suggest an intensification of agricultural systems during the past 15 years, where positive phase shifts towards later periods of the year without increase of NDVI (bright green tones) or in combination with positive NDVI trends (yellow to white) are in many cases caused by an expansion of irrigated arable land (Fig. 4, [1]). In Spain, irrigated areas are reported to have increased by 80% during the period 1961–1999 (Strosser et al., 1999). This is a general trend throughout the European Union and potentially enhances problems of vulnerability to climate change (Puigdefábregas and Mendizabal, 1998) and soil salinisation (Baldock et al., 2000; Szabolcs, 1990). In contrast, spots with dark blue and violet tones in the central part of Castilla-La Mancha (e.g. Fig. 4, [2]) indicate a strong shift of the phase to earlier times of the year. This region was subject to an action plan launched in 1993 (Fornés et al., 2000; Schmid et al., 2005) which aimed at increasing the proportion of non-irrigated arable land at the expense of extended irrigation systems (which usually have its vegetation maximum during summer) which had been developed since the 1950s (Fornés et al., 2000). The excessive ground water extraction had already severely affected important wetlands of La Mancha (Oliver and Flórin, 1995), and the objective was to reduce water extraction and re-establish and preserve the groundwater aquifers in the region which had been suffering from the enormous expansion of irrigated arable land.

Within the semi-natural vegetation stratum, only a few locations exhibit negative trends in vegetation abundance (dark green tones). Location [3] (Fig. 4) indicates two prominent spots in Catalonia. Closer examination of the data suggests that these areas were affected by

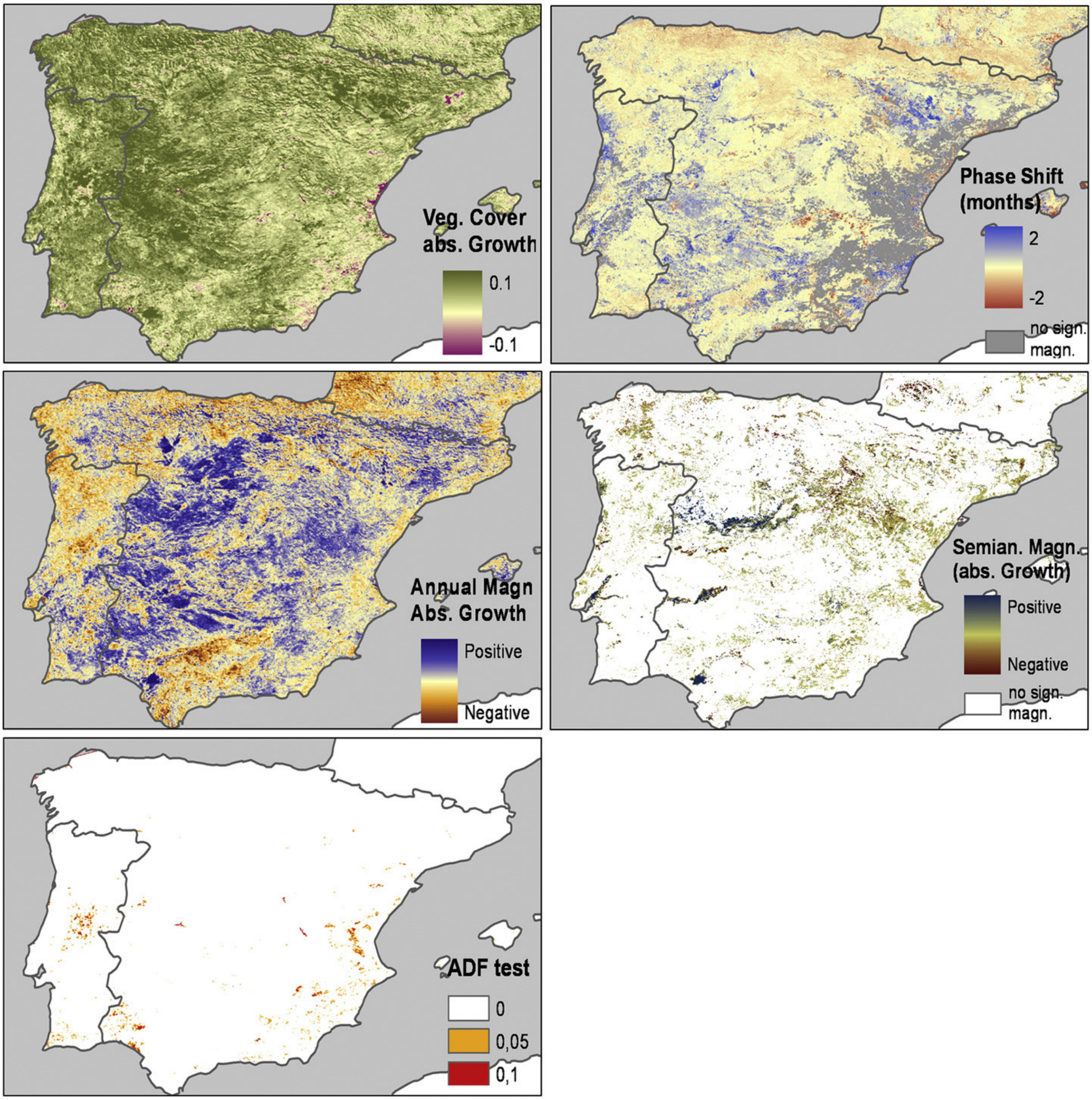

Fig. 3. Parameters derived from time series analysis of the NDVI MEDOKADS archive (1989–2004): absolute growth of NDVI, phase shift, absolute growth of annual and semi-annual magnitude, Augmented Dickey–Fuller test.

wildfires as the NDVI time series is, in accordance with Díaz-Delgado et al. (2004) and Díaz-Delgado & Pons (2001), characterized by abruptly decreasing values. However, large fire sites appear more distinctly on the trend maps when they occur close to the end of the time series, suggesting the use of alternative measures to identify fire dynamics in temporal archives (see §2.2). Such large wildfires are obviously connected to the predominant trend of increasing vegetation cover and volume within the non-agricultural land use systems (Fig. 4). The important question to be further addressed is why this process extends to such large areas of the Iberian Peninsula, especially in mountain ecosystems.

4. Land use change analysis

The findings presented in the previous section suggest that an integrated examination of time series parameters derived from coarse satellite archives can be used to identify the spatial dimension of land use change processes on a regional scale. Preferably, this type of analysis demands a strictly formalized approach; it is further suggested that a semantic interpretation of land use changes within a given period of time (1989–2004 in this case) is skillfully implemented as a variant of the syndrome concept presented in the introduction of this paper. This first attempt to establish an interpretative framework concentrates on

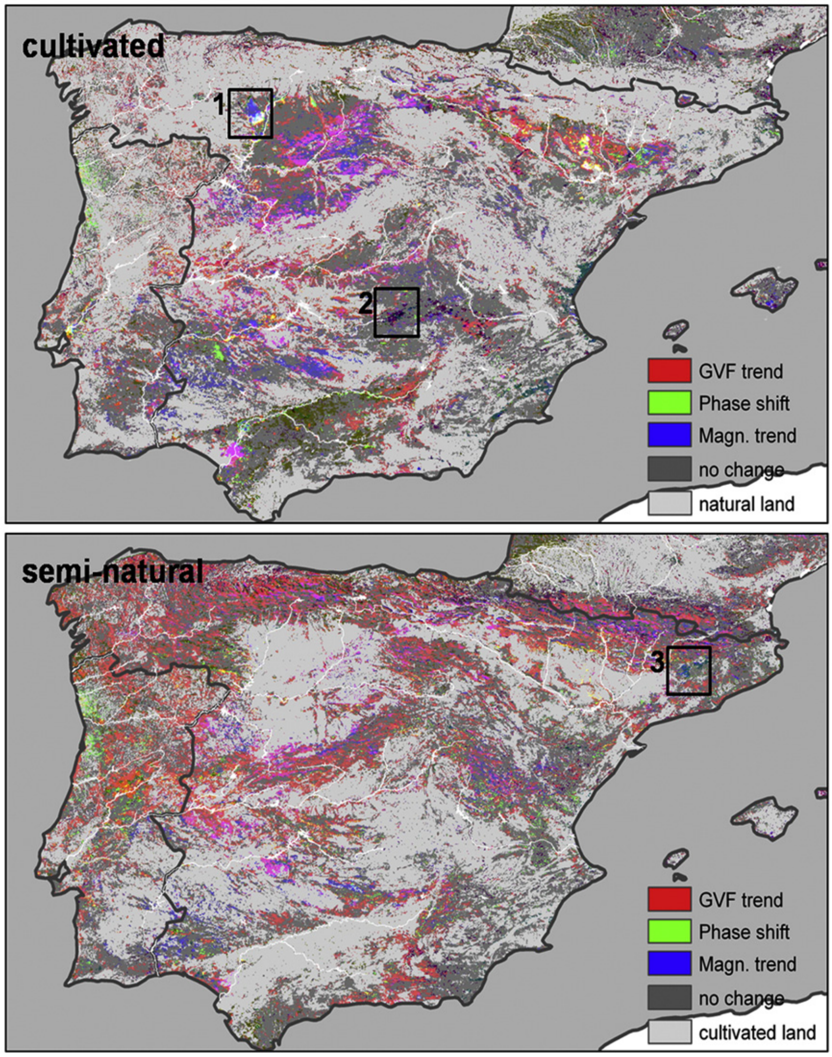

Fig. 4. RGB representation of three layers (NDVI trend, phase shift, trend of magnitude) stratified for cultural land (above) and semi-natural land (below) by means of CLC1990-based stratification.

non-agricultural land but the proposed strategy may equally be adapted to the more complex agricultural land use change processes.

4.1. Mapping of land degradation-related syndromes

As a first step we considered which land degradation-related syndromes are likely to occur in natural areas of the Western Mediterranean, followed by an assessment of how they become manifest in land cover changes and how this could be detected by analyzing satellite time series. Downing and Lüdeke (2002) suggested a subset of global change syndromes (out of the list proposed by Schellnhuber et al., 1997) which appear most relevant to dryland degradation and desertification. Three main syndromes were identified which are also highly significant for the Mediterranean situation: the "Overexploitation", "Rural Exodus" and the "Disaster" syndromes.

The "Overexploitation" syndrome comprises processes which cause use of the land's resources beyond its long-term productivity potential, such as overgrazing, excessive wood extraction, and, eventually, an overexploitation of groundwater aquifers by excessive irrigation. It thus adheres to the concept where human action imposes disturbances which might exceed the resilience of the land and cause an irreversible loss of the land's carrying capacity or biological production potential (Puigdéfabregas and Mendizabal, 1998). Overgrazing, as a typical manifestation, is a long term, gradual process which goes along with a decline of vegetation cover and volume. Wood extraction is generally a rapid disturbance leading to a loss of vegetation followed by gradual processes of reforestation or vegetation recovery.

The "Rural Exodus" syndrome refers to the abandonment of traditional agricultural perimeters. Studies from the Mediterranean (e.g. Mazzoleni et al., 2004) and, in particular, the Iberian Peninsula show that since the middle of the 20th century abandonment of rural areas and the establishment of successional vegetation communities are two of the most important transformation processes, often accompanied by vegetation encroachment (e.g. Moreira et al., 2001; Poyatos et al., 2003; Bonet et al., 2004; Lasanta-Martínez et al., 2005; Vincente-Serrano et al., 2005).

The "Disaster" syndrome in general refers to singular anthropogenic induced environmental disasters with long-term impacts. Although fire through natural and human-induced ignition has been an important factor in ecosystem rejuvenation ever since, and has in fact shaped Mediterranean landscapes for thousands of years (Di Pasquale et al., 2004), it is suggested that large and devastating wildfires may be associated to this syndrome. However, an increasing tendency towards recurrent, large and devastating fires has been reported for most European Mediterranean countries, but in particular for the Iberian Peninsula (Moreno et al., 1998; Moreno, 1998; Moreno, 1999; Moreira et al., 2001) These constitute a substantial threat to human lives and assets.

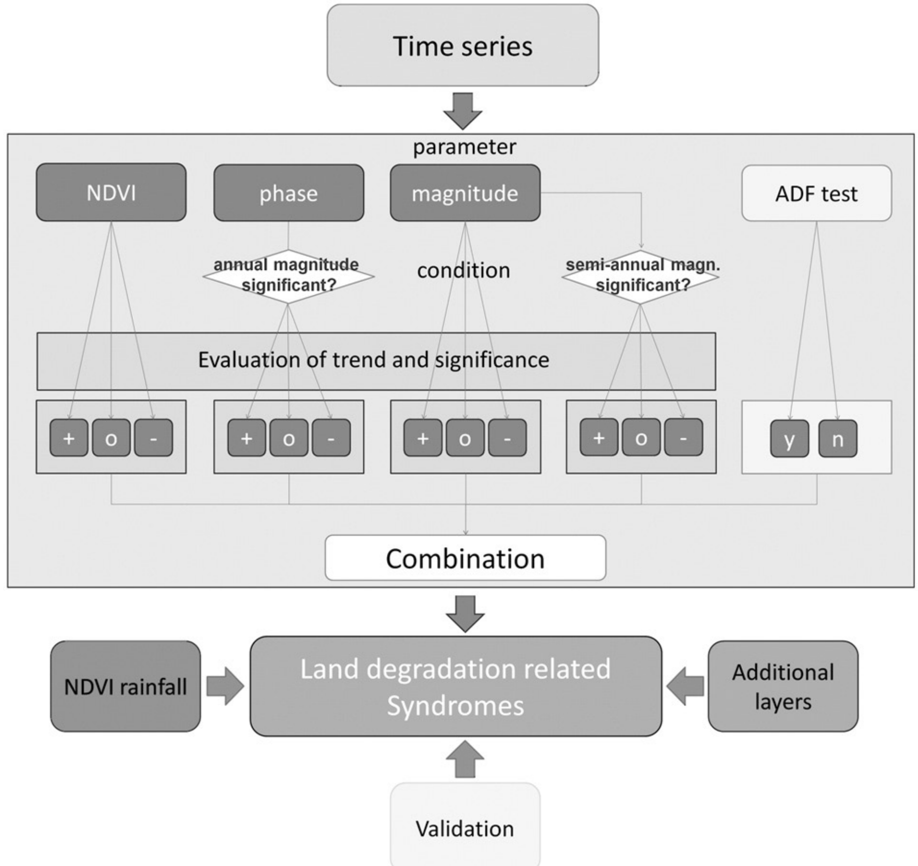

Fig. 5. Rule-based approach used for mapping land degradation-related syndromes from time series parameters.

These transformations and their ecological and socio-economic consequences form the conceptual background of an interpretative framework which was applied in this study (Fig. 5).

In order to 'translate' the combined information layers derived from the time series analysis (NDVI trend, phase shift and magnitude trend) into spatial patterns of areas affected by specific syndromes, these were used to generate discrete classes according to pre-defined criteria. With the exception of the ADF-test parameter, (which can only reach the values random walk (y) or no random walk (n) depending on the significance with α=0.1), the decisive factors are the direction and magnitude of the regression parameter and its significance (α=0.1). Class assignment for the NDVI trend as well as annual and semi-annual magnitude was carried out using threshold factors derived from image statistics. In order to prevent the influence of uncertainties, only the upper and lower 5% of an assumed normal distribution were assigned to the positive and negative classes. For considering the phase factor a shift of at least one month during the full period covered by the time series was required. Moreover, additional significance criteria have to be fulfilled for phase shift and semi-annual magnitude trend if these parameters shall be taken into consideration (compare §2.2).

Following this discretization, specific combinations of time series parameters form the basis for translating the image-derived parameter vector into its local expression of land degradation-related syndromes. A statistical analysis of the parameters and the land use classes of CLC1990 showed that for natural areas the most important parameters are the trend of NDVI and the ADF-test. Changes of the phenological cycle (expressed by phase shift and trend of magnitude) are in general more explanatory for cropped land, although they can also indicate changing proportions of functional vegetation types. Table 2 provides an overview of the parameter combinations employed to deduce specific syndromes.

Accordingly, the "Overexploitation" syndrome is expected to result in a downward development of vegetation with no clear pattern in the behavior of phase and magnitude. Obviously, wood extraction entails a spontaneous change in vegetation cover properties, reflected in corresponding matches of the ADF-test. In this case, the trend direction is determined by the position of the 'disturbance' within the time span considered. The same pattern is observed for large wildfires ("Disaster" syndrome), which is therefore mapped within the same class. The behavior of the 'wood extraction' and 'fire' instances, which thematically belong to different syndromes, hence show that even within one stratum the time series parameters are not always distinct, suggesting a later extension of the approach by including additional information layers (compare §5). The "Rural Exodus" syndrome is clearly associated with positive vegetation cover trends resulting from the increasing vegetation cover and volume after abandoning an active use of semi-natural systems, or converting

Table 2
Land degradation-related syndromes and diagnostic combinations of time series parameters (+/−/o: positive/ negative/insignificant trend, e.g. +oo: positive NDVI trend, no phase shift, no magnitude trend)

Syndrome	Processes	Combinations NDVI trend, phase shift, magnitude trend
No Syndrome	No change	[ooo]
Overexploitation	Overgrazing	[—], [−oo]
	Wood extraction	[—], [−oo], [ADF-test], ([+oo], [+o+], [++o], [+−o], [+o−])
Disaster	Fire	[—], [−oo], [ADF-test], ([+oo], [+o+], [++o], [+−o], [+o−])
Rural exodus	Vegetation encroachment	[+oo], [+o+], [++o], [+−o], [+o−]
Others	Structural change	[o+o], [o−o], [oo+], [oo−]

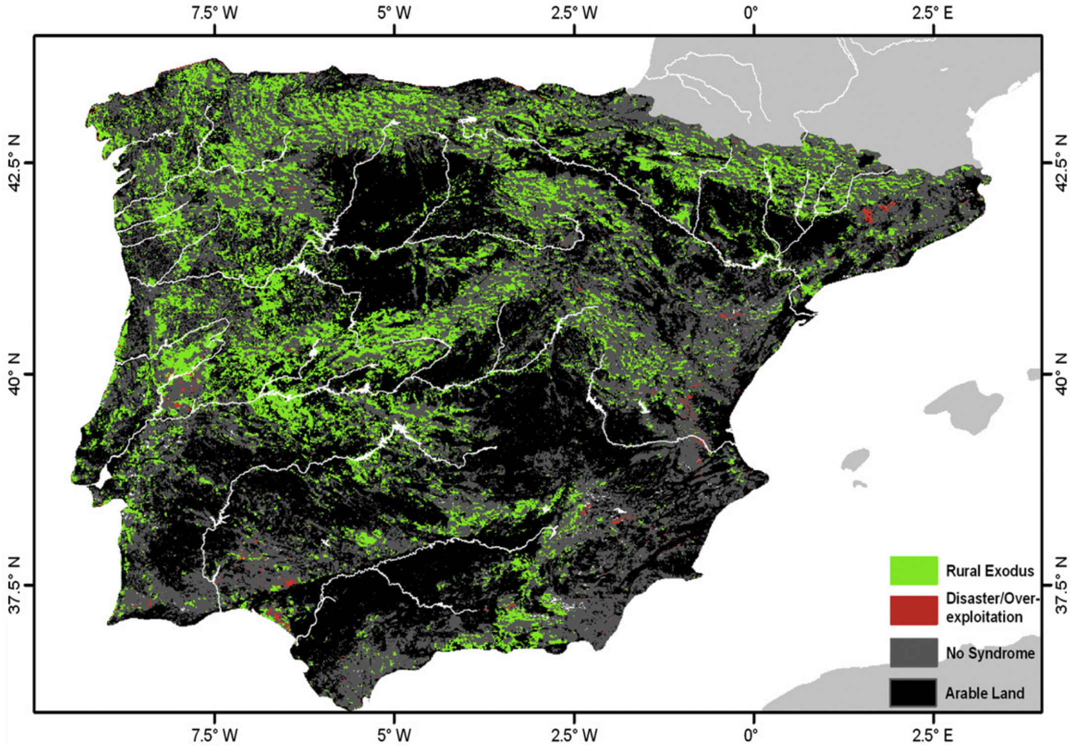

Fig. 6. Areas associated to specific land degradation-related syndromes within the semi- natural vegetation stratum, mapped from combined time series parameters.

formerly agricultural land into extensively used rangelands. Depending on vegetation successional patterns, these might be associated with different combinations of phase/magnitude parameters. Finally, statistically not significant trends suggest ecosystem at stable equilibrium, while their combination with variable phase and magnitude behavior indicates structural changes that cannot be further specified; these are subsumed in the class “others”.

This conceptual framework was employed to spatially map the syndromes for stratum “semi-natural” (Fig. 6). The “Rural Exodus” syndrome (compare Table 2) occupies large areas, and the spatial pattern reflects major demographic trends in Portugal and Spain (e.g. MacDonald et al., 2000). These are characterized by strong urbanization tendencies and concentration of population along the coasts and in major cities, contrasting with decreasing population in the hinterlands. On the other hand, locations associated with the “Disaster/Overexploitation” syndrome appear as distinct spatial patches (Fig. 6). A validation of the emerging patterns is desirable but a direct assessment is rendered difficult by the coarse resolution of 1 km. Nonetheless, an indirect validation of the approach can be conducted by means of an elaborate literature study and comparisons to land use change detection and time series analysis studies of high resolution data (compare §4.2).

4.2. Discussion

The statistical cross comparison of the resulting syndrome map for the Iberian Peninsula (Fig. 6) with CLC land use classes leads to the following conclusions (Table 3):

1. The majority (61%) of the natural vegetation systems in Iberia is not part of any of the relevant syndromes and considered stable over the 15-year time window
2. Areas belonging to the “Rural Exodus” syndrome cover almost one third (29%) of the natural vegetation systems
3. The classical “Disaster/Overexploitation” syndrome is represented by only 1% of the total area

In contrast to the patchy and isolated distribution of areas associated with the combined “Disaster/Overexploitation” syndrome, locations assigned to “Rural Exodus” (primarily characterized by increasing NDVI trends) prevail in all classes of the stratum “non-cultivated”; their spatial distribution appears closely linked to the trend of abandonment of rural areas in favor of urban centers. The decline of economic activities in rural areas, conversion of agricultural land into extensive rangelands, and reduction of livestock grazing and wood extraction is for a long time observed in all countries of the European Mediterranean but particularly affects marginal regions (MacDonald et al., 2000; Taillefumier and Piégay, 2003; Torta, 2004). In the NDVI time series, this process is reflected as widespread greening, i.e. an increase in biological productivity, providing clear spatial evidence of ecosystem recovery after anthropogenic pressure has ceased. A large number of studies describing the encroachment of shrubs and trees following the abandonment of former farmland

Table 3
Aereal statistics [km^2] of land degradation-related syndromes mapped from time series parameters, differentiated according to major classes of semi-natural vegetation (from CLC1990)

Land use	No syndrome	Disaster/ Overexploitation	Rural exodus	Others
Bare rocks	1182	41	811	211
Sparsely vegetated	6819	203	2261	551
Natural grasslands	14,880	102	10,837	2942
Moors and Heathland	7491	38	5655	1525
Scleroph. vegetation	36,365	547	15,234	3827
Trans. Woodland shrubs	31,941	394	16,271	3692
Broadleaved forest	32,531	253	14,421	4912
Coniferous forest	34,096	657	12,462	5368
Mixed forest	13,504	123	5951	2401
LPOA	19,359	195	8243	3827
All classes	198,168	2553	92,146	29,256
Percent	0.61	0.01	0.29	0.09

LPOA = Land principally occupied by agriculture, with significant areas of natural vegetation.

confirm these findings. Vincente-Serrano et al. (2005) used time series of NOAA–AVHRR data to show a significant overall increase of in vegetation cover for the Central Spanish Pyrenees; this is complemented by different studies in north-eastern Spain and the Pyrenees (Dunjó et al., 2003; Pavón et al., 2003; Poyatos et al., 2003; Lasanta-Martínez et al., 2005), north-western Spain (Crecente et al., 2002), central Spain (Romero-Calcerrada and Perry, 2004), eastern Spain (Bonet et al., 2004), as well as Portugal (Moreira et al., 2001). The recovery processes, however, have an ambiguous impact on ecosystem functioning.

For the observation period 1989–2004, the results of this study do obviously not support the wide-spread opinion that large parts of semi-natural Mediterranean ecosystems suffer from reduced biological productivity and desertification (Brandt and Thornes, 1996). However, although some authors emphasize the positive role of vegetation recovery on protecting soil from erosion and on carbon sequestration, there is growing evidence for another sort of environmental risks.

Where formerly utilized land is simply left without rehabilitation strategies, previous mosaics of land use types tend to converge towards homogenized states dominated by dense shrub. This causes a decline in the relative degrees of fragmentation and connectivity that are vital for ecosystem functioning (Wiens, 1995; Forman and Collinge, 1996). For instance, it destroys habitats of species especially adapted to the ecological conditions of semi-natural habitats, living in transitional habitats and needing open spaces (Conti and Fagarazzi, 2005). This is a severe threat to the legacy of long-lasting use of Mediterranean landscapes, where thousands of years of human interventions have created 'cultural landscapes' with high biological diversity and aesthetic value (Farina, 2000; Di Pasquale et al., 2004). In combination with the increase in woody biomass, the loss of gaps and breaks, as well as edges between different fuel types, makes such landscapes much more prone to large and devastating fires compared to disconnected patches of forest, shrub and cultivated fields or grasslands (Puigdefábregas and Mendizabal, 1998; Viedma et al., 2006; Röder et al., 2008a), a risk which is particularly aggravated in periods of drought (Millán et al., 1998; Viegas, 1998; Duguy et al., 2007).

The connected pattern of "Disaster/Overexploitation" areas (Fig. 6) is in accordance with overviews provided by Moreno et al. (1998) and the results of the regular surveys of forest fires in Europe (e.g. European Communities, 2007). Identified perimeters correspond to spatial information supplied for specific fire events in Catalonia (Díaz-Delgado and Pons, 2001; Díaz-Delgado et al., 2003; Díaz-Delgado et al., 2004) the larger Valencia region (Quintano et al., 2005; Röder et al., 2008a) and to fire perimeter maps provided for Portugal by the Centre for Satellite based Crisis Information of the German Aerospace Centre (web resource: http://www.zki.caf.dlr.de). This supports the hypothesis that the combination of time series parameters with the ADF-test is suitable to map major fire perimeters. Furthermore, it is concluded that on the Iberian Peninsula fire has become the major underlying process of the combined "Disaster/Overexploitation" syndrome within semi-natural and natural vegetation areas, while overgrazing and wood extraction are not anymore major factors.

Notwithstanding the resilience of Mediterranean ecosystems to fires (Keeley, 1986), their increasing frequency, severity and spatial extent now create numerous ecological and socio-economic threats. While the typical fire regimes of past centuries supported 'steady-state' ecosystems under auto-succession regimes (Trabaud, 1994; Blondel and Aronson, 1995), modifications and problems may occur as soon as the fire frequency exceeds the time required by plants for regeneration and maturation (Keeley, 1986). This may cause a loss in plant diversity and long-term productivity, as well as a loss of biodiversity due to the homogenization of plant communities (Delitte et al., 2005; Ferran et al., 2005). Frequently, forests and rangelands are replaced by uniform shrublands which represent dangerous fuel types (Viegas, 1998) and favorable areas for ignition and fast propagation of wildfires (Finney and Ryan, 1995; Duguy et al., 2007). Climatic factors, such as the coincidence of hot and dry seasons, or the variability in precipitation may further aggravate the ignition risks (Millán et al., 1998). Large and recurrent wildfires also act as agents of degradation through processes such as the deprivation of soil of its protective vegetation layer, the sterilization of top soil layers or the sealing of the soil surface (Christensen, 1994; Quinn, 1994). This may enhance the threat of accelerated soil erosion, especially in the case of the high-intensity rainfall events typical for Mediterranean areas. Given current climate change scenarios these processes and the related threats to animal populations, as well as human lives and properties are expected to become even more severe in the future (Rambal and Hoff, 1998).

Beside the increasing fire risk, one of the most significant impacts of increasing vegetation cover and volume in the consequence of the "Rural Exodus" is associated with hydrological cycles and groundwater aquifers. For instance, García-Ruiz et al. (1996) showed that some of the most important rivers and alluvial fans in the Pyrenees have already changed their morphodynamic systems. In this respect, homogenisation and greening up of vegetation communities may entail negative impacts, as it is accompanied by higher evapotranspiration rates which tend to decrease the amount of water available for run-off and groundwater recharge. This effect was shown by experimental studies for different watersheds (Llorens et al., 1995; Beguería et al., 2003; Gallart and Llorens, 2004; Gallart et al., 2005; Beguería-Portugués et al., 2006). Notwithstanding their local significance, related effects appear particularly dramatic if their consequences are assessed for larger downstream areas of agricultural production and human settlements that dwell on these resources. According to Gallart and Llorens (2003), these processes already have lead to a reduction of annual streamflow of approximately 25% during the past 50 years. Given existing problems of water scarcity and frequent droughts in Spain and Portugal, as well as likely future climate scenarios, this feedback loop is likely to become a major source of concern in the future (Olesen and Bindi, 2002).

5. Conclusions and perspectives

The rule-based approach to interpreting time series parameters derived from hypertemporal satellite archives (as provided by NOAA–AVHRR, SPOT VEGETATION, MODIS and MERIS) has enabled us to map important land use change syndromes on the Iberian Peninsula at regional scale. The approach is different from mapping land use change classes in the conventional understanding as it is derived from a meta-analysis of socio-economic and biogeophysical processes that lead to the definition of parameter vectors for describing a complex syndromatic behavior of land use systems.

It is important to understand that the approach does not include information from earlier times than the covered observation period. Equally, no additional socio-economic or climatic data have so far been included in the interpretation scheme. It is expected that the assimilation of such spatially explicit data layers may further increase interpretation capacities. For instance, including vegetation cover–rainfall relationships may help to distinguish climatic or anthropogenic influences on vegetation cover (Wessels et al., 2007a). In particular, when extending the approach to agricultural land use systems, incorporating auxiliary information is considered of paramount importance to understand underlying process patterns.

With respect to the observed time window (1989–2004) one can state that the majority of natural vegetation systems of the Iberian Pensinsula have not changed during this period, and that areas of change are primarily characterized by vegetation communities with increasing plant cover and volume. This process happens at the expense of loss of spatial structure and diversity, and negatively impacts on fire regimes and the hydrological cycle. The notion of an ecosystem which is subject to classical desertification processes (decline in productivity after disturbance) is no longer supported. At the same time spatially explicit evidence is provided for adverse ecosystem processes (Mazzoleni et al., 2004) which might become even more critical with regard to future climate change consequences.

Given the global availability of coarse-scale, hyper-temporal data archives, the syndrome-based mapping concept shows great potential for extension to other regions. This spatial extension needs to be complemented conceptually, since other physical and socio-economic settings might result in completely different process frameworks. As the suggested approach explicitly aims at an integrated, place-based assessment, the consideration of underlying process-frameworks is a prerequisite to attain meaningful results.

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Chapter V

Mapping syndromes of land change in Spain using remote sensing time series, demographic and climatic data

Land Use Policy (submitted)

Marion Stellmes, Achim Röder, Thomas Udelhoven and Joachim Hill

Abstract

The country of Spain is representative of land change processes in Mediterranean member states of the European Union, which are often triggered by European, national and sub-national policies. They include widespread land abandonment and urbanisation trends, but also an increase in land use intensities accompanied by strong exploitation of water resources. The Mediterranean is part of the dryland ecoregion which is especially vulnerable to ecosystem degradation. While remote sensing data allow for a characterization of the temporal dimension of land surface processes, the syndrome-based approach aims at integrating this with information on local/regional socio-economic and physical frameworks. In this study we incorporated two major drivers of land change, climatic boundary conditions and population density change, to understand the patterns of assessed land cover changes.

We used the Mediterranean Extended Daily One Km AVHRR Data Set (MEDOKADS), representing the Mediterranean Basin at a resolution of 1 km for the period from 1989 to 2005 at ten-day time intervals. The long-term trend of vegetation could be characterized based on a linear regression analysis of Normalized Difference Vegetation Index (NDVI) values. Using Fourier analysis further descriptors of phenology were calculated, such as the amplitude of annual cycles, or the date of occurrence of the annual maximum of vegetation cover. Subsequently, a linear regression analysis was applied to these parameters to map phenology shifts representative of changes in land use/cover.

Next, a rule-based classification was employed to integrate the individual results and assigned syndromes representing specific land change process patterns. For natural and semi-natural areas, locations characterized by increasing biomass were found to coincide with areas of population loss due to migration to urban centres. On the other hand, intensively used agricultural areas are associated with intensification processes including the expansion of intensive irrigation schemes. Furthermore, areas which are dominated by climatic variability were identified by employing results of an analysis of NDVI-rainfall relationships. The resulting data set identifies major land change syndromes for the whole of Spain and links apparent results of change to driving factors and causes, thus serving as an excellent data source to inform policy makers of effects caused by legislative and management actions.

Keywords: Land cover change, syndromes, Spain, remote sensing, ecosystem services

1 Introduction

At all stages of human existence, people's livelihoods have been sustained by the earth's terrestrial and marine ecosystems. Thus, man has always modified and transformed landscapes according to its then capabilities (Kareiva et al., 2007; Smith, 2007). Yet, in the recent past an acceleration of land use transformations, coupled with an increasing global population has manifested in an alteration of almost all terrestrial ecosystems (Millenium Ecosystem Assessment, 2005b; Steffen et al., 2004). Regardless of the intensity of land use, its goal is always to appropriate primary production for human use, for instance by obtaining food, fibre, timber or other ecosystem goods (DeFries et al., 2004), and about one-third of all terrestrial net primary production is now estimated to be consumed by humans (Imhoff et al., 2004).

The Millenium Ecosystem Assessment characterized dryland degradation as one of the major threats to human well-being in the context of global environmental change (Millenium Ecosystem Assessment, 2005b). Dryland areas cover about 41% of the earth's surface and are home to about one third of the world's population (Safriel et al., 2005). Due to the biophysical boundary conditions, primarily water scarcity, overuse of resources and climate change are a much greater threat for ecosystem functioning in drylands than in non-dryland systems (Millenium Ecosystem Assessment, 2005a).

Therefore, appropriate management of terrestrial resources is a key issue in safeguarding long-term human welfare, and land management is faced with the challenge of trading-off a variety of demands from and pressures on natural resources (Foley et al., 2005), which is particularly important in highly vulnerable regions such as drylands (Reynolds et al., 2007). Adequately informing policy makers and land management about the present state of resources, consequences of land use (change) and alternative options is a key requirement in this context (Millenium Ecosystem Assessment, 2005b).

Obviously, most aspects of global environmental change are directly linked to land use and land cover change, and the effects are most apprehensible through the resulting landscape pattern (Forman, 1995). This has been reflected by an increasing number of studies covering all scales from local to global (Lambin and Geist, 2006). On a conceptual level, the development of land change science has been called for to ensure land change is perceived in the framework of coupled human-environment systems (Rindfuss et al., 2004; Turner II et al., 2007). This holistic concept necessitates consideration of a variety of scientific disciplines to understand socio-economic and physical drivers that are determining the direction and magnitude of change (Lambin et al., 2001). Given current global change scenarios, land management needs to move beyond a mere preservation of a status-quo situation to considering ecosystem resilience and manage change rather than trying to avoid it at all costs (Chapin III et al., 2010; Folke, 2006).

In order to prepare guardrails for land management at regional, national or continental levels, policy makers are in need of suitable spatial and non-spatial information that allows them to assess trade-offs between different land use options (DeFries et al., 2004). The Millenium Ecosystem Assessment promoted the concept of Ecosystem Services and Functions (Millenium Ecosystem Assessment, 2005b), which classifies these into major groups of provisioning, regulating, preserving and cultural services. This may serve as a conceptual framework to assess the state of ecosystems with respect to the individual functions and services, while rating can then be performed according to the respective individual management or development goals (Foley et al., 2005). Map products are particularly illustrative for decision makers, turning spatial information related to land use and ecosystem functions into an important component of policy making (Group of Earth Observation, 2005; Lautenbacher, 2006). In this context, remote sensing data are particularly useful to provide consistent land use/cover information across large areas in a repeatable manner (Hill et al., 2004). Satellite-based systems are able to characterize vegetation phenological cycles and use this information to characterize land use and land cover (DeFries, 2008). Accordingly, a number of continental/global map products exist that have been derived from different sensor systems and using different methods (Bartholomé and Belward, 2005; Friedl et al., 2002; Loveland et al., 2000). Depending on input data and methods used, such maps may largely differ, making the evaluation of map consistency a major issue (Herold et al., 2008a; Herold et al., 2008b).

Besides, integrated global maps delineate major phytogeographical zones (Olson et al., 2001) or combine such information with climatological or socio-economic data to map anthromes (Ellis and Ramankutty, 2008). Such data sets are an excellent source for stratification on continental or global scales; yet, they are often limited to one particular date or period or only available for relatively short time periods, such as the MODIS land cover product. Most importantly, they do not integrate the dynamic nature of processes within socio-ecological systems, hampering their suitability for decision making.

Different conceptual approaches exist to assess dynamic process patterns in a spatially explicit manner. Such an interpretation framework is provided by the syndromes approach that has been developed in the context of global change research (Cassel-Gintz and Petschel-Held, 2000; Lüdeke et al., 2004; Petschel-Held et al., 1999; Schellnhuber et al., 1997). It aims at a place-based, integrated assessment by describing global change by archetypical, dynamic, co-evolutionary patterns of human–nature interactions instead of regional or sectoral analyses. In this framework, syndromes (as a “combination of symptoms”) describe bundles of interactive processes and symptoms which appear repeatedly and in many places in typical combinations and patterns.

Depending on the investigation area, the syndrome framework needs further adjustments and concretisation. Downing and Lüdeke (2002) have attempted to link the defined syndromes to the desertification issue by linking vulnerability concepts to degradation processes and identifying those global change syndromes that are most relevant to desertification (Reynolds et al., 2007). Similarly, Geist & Lambin (2004) and Geist (2005) have identified major causal patterns and underlying forces associated with desertification and employed the syndrome concept to systematically aggregate a large number of case studies. Research undertaken so far suggests that mapping dynamics in coupled human-environmental systems following the syndromes approach may be useful to supply policy and decision makers with information products suitable to review the effects of their decisions (Sterk et al., 2009).

Hill et al. (2008) have suggested a framework to map syndromes related to land degradation in drylands based on time series analysis of coarse-scale remote sensing data with a high temporal resolution. This study was confined to natural and semi-natural areas, and made exclusive use of remote sensing data without considering further spatial information. Our study follows up on this work and extends the syndromes approach to both, natural and anthropogenic strata, and by incorporating socio-economic and climatological information in the interpretation model.

The aim of the study is to map the major areas affected by land cover changes and to link these with underlying causes. This includes the distinction of human driven changes from changes caused by external factors, for instance evoked by climatic variability. For the mainland of Spain, these goals were aimed at by implementing the following steps (compare also figure 1 for concept):

(i) Assess changes of land cover based on time series analysis of remote sensing data with high temporal resolution

(ii) Distinguish human-caused land use changes from climate driven changes

(iii) Identify land use change syndromes and develop rules to describe these syndromes by time series analysis and additional information, e.g. socio-economic factors.

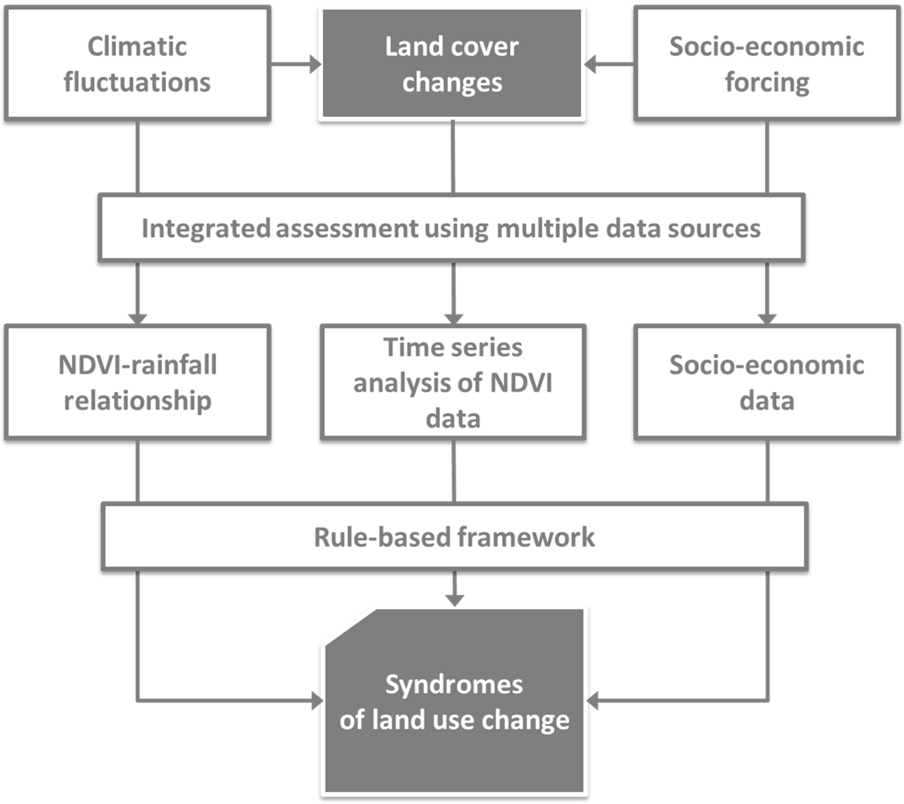

Figure 1: Flowchart of the implemented working scheme.

These objectives are pursued considering both natural/semi-natural and agriculturally utilized areas. Spain has undergone extensive political, economic and social developments within the 20^{th} century, in particular after it joined the European Union in 1986 and benefitted from various European subsidy instruments, like for instance the Common Agricultural Policy (CAP), the European Regional Development Fund, the European Social Fund and the Cohesion Fund. In this respect, it is representative of land change processes in Mediterranean member countries of the European Union, which are often triggered by European, national and sub-national policies. They include widespread land abandonment and urbanisation trends, but also an increase in land use intensities accompanied by strong exploitation of water resources.

2 Data and Methods

2.1 Data

2.1.1 MEDOKADS archive

Changes of land cover are often related to changes of vegetation which can be assessed by remotely sensed data employing suitable spectral indices. In this study we used a time series of the Normalized Difference Vegetation Index (NDVI) (Rouse et al., 1974; Tucker, 1979). Alterations in vegetation cover and composition, which are often accompanied by changes of phenology, change the spectral properties of the earth’s surface and hence, the NDVI.

Monitoring long-term land cover changes at regional to continental scale based on remote sensing data requires consistent data archives with a sufficient geographical coverage rate. Coarse scale remote sensing data as delivered by NOAA AVHRR, SPOT Vegetation and MODIS were employed successfully to map the development of vegetation cover. At present, the NOAA AVHRR sensors provide the most comprehensive time series of satellite measured surrogates for surface condition. This study is based on the „Mediterranean Extended Daily One Km AVHRR Data Set“ (MEDOKADS) processed and distributed by the Freie Universität Berlin (Koslowsky, 1998). The MEDOKADS archive, which covers the time period from 1989 to 2004, comprises full resolution AVHRR channel data and additional collateral data with a geometric resolution of about 1 km^2 (table 1).

Table 1: Characteristics of the Mediterranean Extended Daily One-km AVHRR Data set (Friedrich et al., 2009).

Time period:	1989–2004
Resolution:	0.01° (~1.1 km)
Source:	Direct read out High Resolution Picture Transmission (HRPT)
Region:	West-, Middle-, East-, and Southeast Europe, North Africa
Projection:	Geographical projection
Satellite:	NOAA-11, -14, -15, -16, -18
Available data:	Daily data, 10 day composites and 30-days composites for channel 1–5, satellite/sun zenith angle, satellite/sun azimuth, broadband albedo, bitmap (origin indicator, landmask, cloudmask), NDVI, scattering angle sun satellite, sea surface temperature (SST), land surface temperature (LST), time since equator
Distributor:	Koslowsky, D., Freie Universität Berlin, Meteorologische Satellitenforschung

Calculation of the commonly used Normalised Difference Vegetation Index (NDVI) from the two reflective channels of the MEDOKADS data archive requires careful pre-processing in order to allow a meaningful trend analysis. This includes correction for sensor degradation and orbital drift effects that cause non-linear changes in the measured signal, as well as an inter-calibration between the AVHRR/2 (NOAA 11 and NOAA 14) and AVHRR/3 (NOAA 16) sensors to prevent inhomogeneities of the time series. A detailed description of the data pre-processing is given by Koslowsky (1996; 1998) and Friedrich et al. (2009). Despite the applied correction scheme, a bias between corresponding NOAA-14 (until 2000) and NOAA-16 (after 2000) data representing areas without long-term trends was found and accounted for by correcting the data by employing an empirically derived equation (Udelhoven, 2007; Udelhoven and Stellmes, 2007).

Ten-day NDVI maximum value composites (MVC) from January 1989 to December 2004 were utilized for the frequency analysis of the NDVI signal, whereas 30-day NDVI MVCs served as basis for a trend detection of vegetation cover. A Savitzky-Golay filter was applied to the 10-day MVCs by means of the method proposed by Chen et al. (2004) to reduce the noise within the time series.

2.1.2 Climatic fluctuations

One aim of this study is to distinguish human-induced changes from fluctuations caused by climatic variability.

For this purpose, we used a data set procured by Udelhoven et al. (2009). They addressed the multi-annual influence of rainfall on green biomass production by implementing a regression analysis using accumulated annual precipitation data compiled for the hydrological year in Spain (based on data provided by Del Barrio et al. (2010) and corresponding NDVI values (MEDOKADS archive). Regression analysis was carried out using a distributed lag (DL-) model to assess time-lagged effects of the explanatory variables rainfall amount and mean surface temperature. The lag orders of the rainfall series was determined using a partial F-criterion (Backhaus et al., 2006) by successively increasing the number of lags, starting with a model comprising zero and first order lags using a significance level of $\alpha = 0.05$ significance level.

Model parameter estimation was conducted using the iterative generalized-least-square approach described in Shumway and Stoffer (2006) to enable consistent and efficient parameter estimation in case of autocorrelated data.

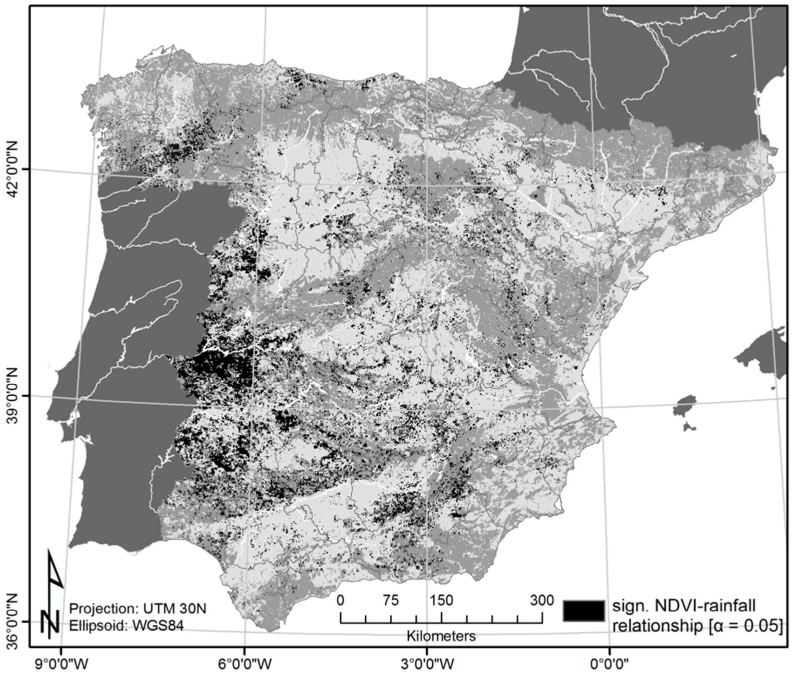

Figure 2: Areas characterized by a significant NDVI-rainfall relationship for the period 1989 to 2000. Within Spain light grey colours indicate non-natural areas and dark grey colours natural areas without significant NDVI-rainfall relationship.

The result we integrated in this study is presented in figure 2. It depicts areas with a significant dependence of annual NDVI on annual rainfall amount which can last up to three years.

2.1.3 Socio-economic data

2.1.3.1 CORINE Land Cover

The stratification of the study area was derived from the 100m CORINE Land Cover Classification 1990 (CLC90) raster data which is distributed by the European Environmental Agency (EEA, 2007). The utilized raster dataset differentiates 44 land use classes and originates from the vector CLC database provided by the national teams at a scale of 1:100.000 and a minimum mapping unit of 25 ha (EEA, 2007). The land cover classes were recoded to distinguish several arable and semi-natural land uses and were aggregated to the geometric resolution of the MEDOKADS data by means of a majority filter (figure 3).

In the preceding study we defined the characteristics of relevant syndromes and their manifestation in hypertemporal remote sensing time series referring to literature addressing land use transformations in Spain and the EU-Mediterranean. In the present study we additionally considered land cover changes identified by CLC between 1990 and 2006. For this purpose, the CLC changes between 1990 and 2000 as well as 2000 and 2006 were spatially degraded from 100 m to 1000 m resolution using a majority criterion. The two layers were combined to derive an information layer comprising the overall changes in the considered period. Areas representing major land use transformations, e.g. from non-irrigated arable land to permanently irrigated arable land, were identified and used to examine the characteristics of the time series parameters for these changes.

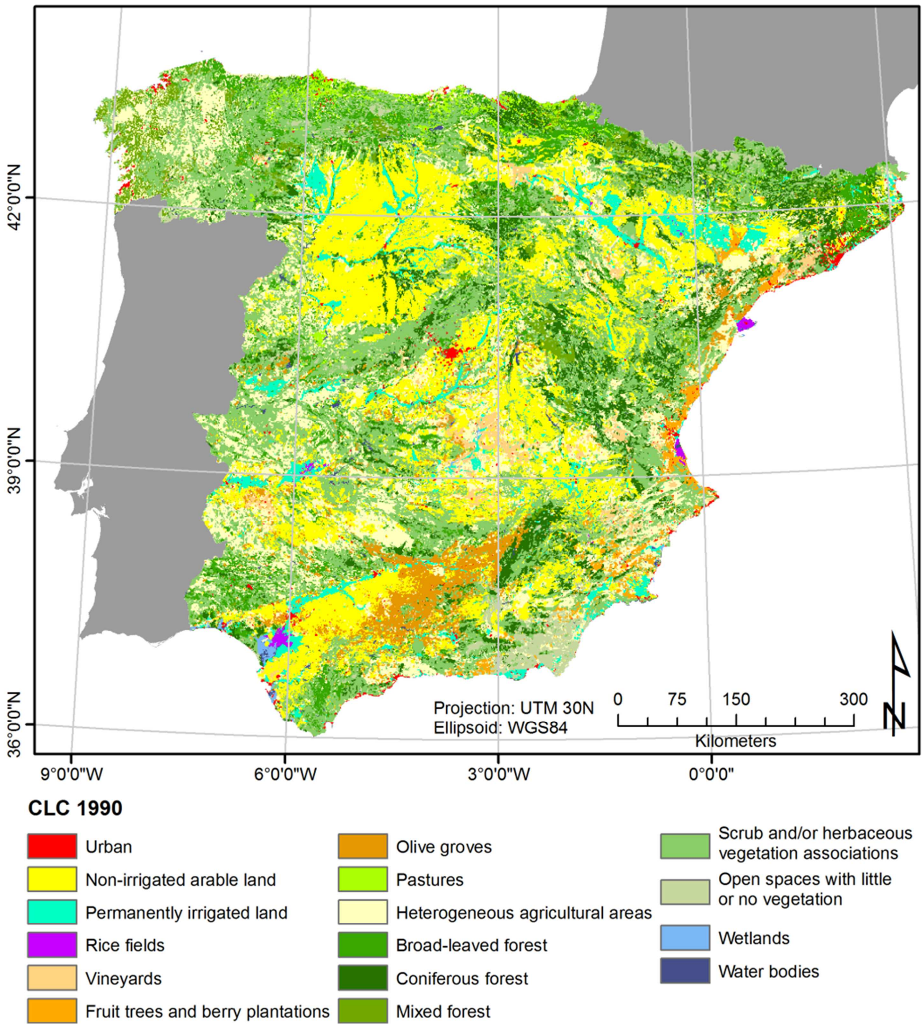

Figure 3: Corine Land Cover 1990, aggregated to 1km resolution (based on data provided by the European Environmental Agency, EEA, 2010).

2.1.3.2 Population density changes

At local scale several studies (Serra et al., 2008) identified major socio-economic drivers of land use change based on a variety of socio-economic statistical data. At coarse scale, the availability of relevant statistical data at an appropriate resolution (here municipalities) is often limited. We followed a similar approach as Ellis and Ramankutty (2008) who employed population density as a main criterion to assess the human influence in the “anthropogenic biomes”. Because we monitor land cover changes we used the demographic development in Spain as a proxy for developments related to socio-economy. Thus, population density and movement indirectly provide information about the attractiveness of different areas, for example with regard to the employment market. For instance, Améztegui et al. (2010) used population change as a variable for farmland abandonment in the Pyrenees. Demographical changes at municipality level (figure 4) were derived from statistics distributed by the national statistical service of Spain (INE, 2010) from 1960 to 2008. Figure 4 shows the population density changes for 1960 to 1981 and 1981 and 2008, where the first period represents the time of the booming Spanish economy (beginning land abandonment) whereas the second time window comprises the accession of Spain to the European Union. The population statistics of Spain reveal major movements of population within Spain due to the socio-economic developments that have taken place in the 20th century. The population change statistics at municipality level were employed to identify areas (i) where depopulation may have caused land abandonment evoking shrub encroachment (rural exodus syndrome) or (ii) where population increase may have caused urbanization (urban sprawl syndrome).

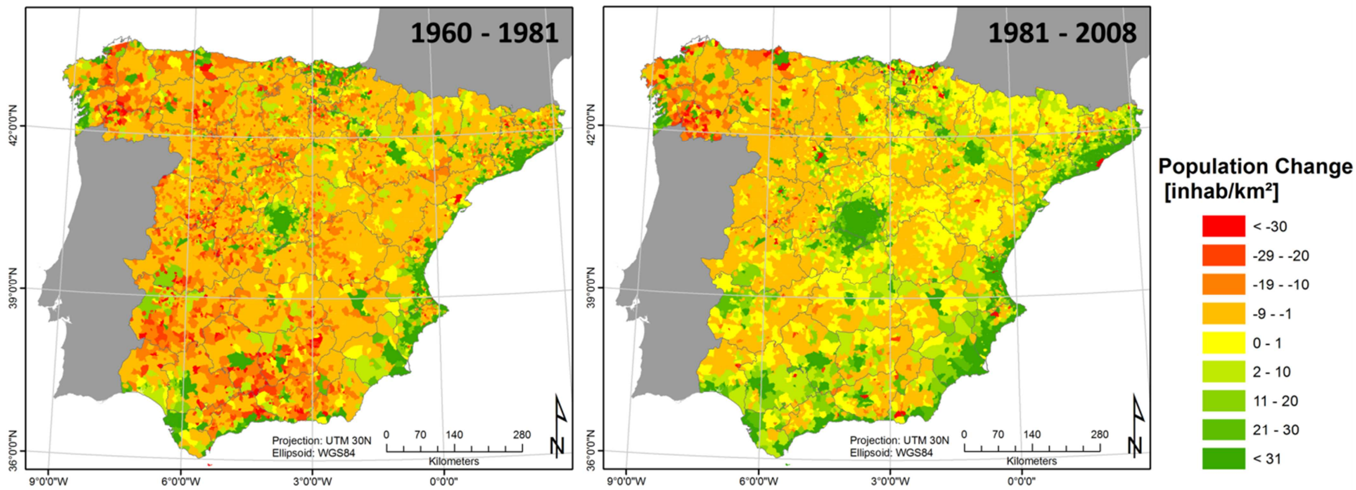

Figure 4: Population changes between 1960 and 1981 (left) and 1981 and 2008 (right); based on data from INE (2010).

2.2 Methods

2.2.1 Time series analysis of the NOAA AVHRR MEDOKADS archive

2.2.1.1 Time series parameters

The time series calculations were carried out using the TimeStats software package, which was specifically developed for analysing long-term hypertemporal satellite data archives (Udelhoven, 2010). For extracting information describing the underlying processes several time series parameters were derived from the monthly MEDOKADS time series. One of the most important descriptors of vegetation dynamics is its long-term trend (increase and decrease) which is determined by a linear regression analysis. Trend analysis of the NDVI series was restricted to yearly aggregated monthly Maxium Value Composites (MVC) values from March to September to minimize uncertainties due to large sun-zenith angles which occur during the winter period and to focus on the vegetation period. The statistical reliability of the derived regression coefficient was determined by means of a Student's t-test.

Due to the high temporal resolution provided by the NOAA-AVHRR systems, phenological information can be derived from the monthly time series, such as the amplitude of the annual cycle and the time of the occurrence of the annual maximum vegetation cover (peaking time). For each year the annual and semi-annual amplitude as well as the peaking time was derived from decadal MVCs by fitting a Fourier polynomial (Shumway and Stoffer, 2006) that considers the annual (36 ten-day periods) and the semi-annual cycles (18 ten-day periods).

Subsequently, linear regression was applied to the 16 resulting values of the annual and semi-annual magnitude and peaking time wherever specific conditions were fulfilled. For example, the determination of a shift of the annual green peak seems only meaningful if a phenological cycle exists. Therefore the shift of the peaking time was only determined where the associated magnitude is significant and exceeds a certain threshold.

To rate the fit of the linear trend model, the root mean squared error (RMSE), the p-value of the two-sided Student's t-test, the correlation (r) and the standard deviation (SD) were calculated.

2.2.1.2 Assessing land cover changes

The different time series parameters derived from the MEDOKADS archive reflect land cover transformations which have taken place in the observed time span. These parameters were jointly analysed to exploit synergistic effects and form the basis of the syndrome approach presented in this study.

On that account, first of all only trends that suggest significant changes ($\alpha = 0.1$) were considered. Secondly, to avoid influences from remaining uncertainties of the time series only regression coefficients exceeding thresholds derived from image statistics were used to identify alterations of land surface. Only the upper and lower 10% of an assumed normal distribution were assigned to the positive and negative classes. To consider shifts in peaking times a shift of at least one month during the full observation period of the time series was required. Moreover, such a shift was only taken into account if the mean amplitude of a NDVI time profile was exceeding a threshold of 0.075 because determining peaking time is only meaningful if a pronounced phenological cycle exists. Considering the aforementioned conditions the trend parameters were discretized into three classes: (i) negative trend, (ii) positive trend and (iii) no significant trend.

Spontaneous events like for instance wild fires often manifest as disturbances of a time series. These disturbances can be identified by incorporating additional parameters in the time series analysis. In this study, we employed the root mean squared error (RMSE) of the linear trend model fitted to the aggregated NDVI time series to evaluate if the model is appropriate to describe the underlying changes. Continuous linear changes are usually characterized by low RMSE values whereas disturbances or non-linear behaviour of the NDVI time profile produce high RMSE values. A threshold was determined to identify areas that show disturbances or non-linear behaviour and areas where the linear trend model is not appropriate.

The resulting maps of the trend classes of NDVI, peaking time, amplitude and the RMSE were intersected to serve as input layers for the syndrome mapping.

In the following step the trend combinations were assigned to land cover change processes, thereby considering the major land use strata of the spatially aggregated CLC90 dataset. The stratification was conducted because the same trend parameters imply different land cover processes, e.g. an increase of vegetation cover is referred to as biomass increase (maybe by shrub encroachment or densification of forests), whereas it indicates increasing productivity in arable land (e.g. use of fertilizers, pesticides etc.). Table 2 provides an overview of the parameter combinations employed to deduce specific land cover changes in natural and arable areas. In the semi-natural stratum the main parameter describing the dominant processes are the change of vegetation cover/biomass described by the NDVI trend and the RMSE. Even though changes of the phenological cycle (expressed by the shift of the peaking time and trend of amplitude) can indicate changing proportions of functional vegetation types, these parameters are in general more explanatory for arable land and were disregarded regarding semi-natural land cover in the present study. In semi-natural areas we distinguished three major processes: biomass increase, biomass decrease and disturbance.

Table 2: Land cover changes and diagnostic combinations of time series parameters (+/-/o: positive/ negative/insignificant trend, e.g. +oo: positive NDVI trend, no phase shift, no magnitude trend).

Process	Trend combination
Semi-natural areas	
No change	[ooo]
Increasing biomass	[+oo], [+o+], [++o], [+-o], [+o-]
Decreasing biomass	[---], [-oo], [-o-], [--o]
Disturbance	RMS exceeds threshold
Arable land	
No change	[ooo]
Increasing productivity	[+oo], [+o+]
Decreasing productivity	[-oo], [-o-]
Expansion of irrigated arable land	[+++], [++o], [++-]
Decline of irrigated arable land	[---], [--o], [o-o], [o--]
Crop Change	[oo-], [oo+], [o+-], [o+o],[o++]
Disturbance	RMS exceeds threshold

For arable land six major land change processes were identified. Besides considering the trend of vegetation cover also the modifications of the peaking time and the amplitude were employed to define these major processes. The shift of the peaking time, for example, is an indicator which can be related to irrigation practices. Whereas non-irrigated arable land is characterized by a peaking time in spring, irrigated arable land usually reaches its maximum vegetation cover in summer to early autumn. Therefore, biomass increase without changing peaking time was termed “increasing productivity”, whereas biomass increase paired with a shift of the peaking time to a later time in the year is interpreted as “Expansion of irrigated arable land”. Analogous, the opposite processes were defined “Decreasing productivity” and “Decrease of irrigated arable land”. Changes in arable land without a change of the aggregated NDVI but changes of phenology were summarized in the class “Crop Change”. Different types of crop changes were summarized to one class but could also be dissociated from each other.

2.2.2 Mapping Syndromes

So far the major land cover change processes have been detected by means of time series analysis of the MEDOKADS NOAA AVHRR archive. The aim is to reveal the major underlying drivers and causes of these changes and assign the detected landscape transformations to potential global change syndromes (compare section 1).

Schellnhuber et al. (1997) defined various global change syndromes encompassing the most important environmental threats and out of these, Downing and Lüdeke (2002) suggested a subset of syndromes which appear most relevant to dryland degradation and desertification. We chose the syndromes which are of high relevance for the EU Mediterranean situation: these comprise "Overexploitation", "Rural Exodus", "Disaster", "Urban sprawl" and "Mass tourism". Table 3 presents an overview of these syndromes and the linked processes.

Table 3: Land change syndromes relevant for Spain (modified after Downing and Lüdeke (2002) and Schellnhuber (1997)).

Syndrome	Underlying Functional Pattern	Syndrome Description	Relevance
OVER-EXPLOITATION	Profit-oriented extraction of renewable resources (mainly from forests) driven by a lucrative market	Vegetation and soil degradation due to capital-intensive, profit-oriented overuse of renewable resources	Wood extraction Over-grazing Irrigation
RURAL EXODUS	Abandonment of traditional agricultural practices and migration to urban centres	Increased biological productivity and vegetation cover, bush encroachment and changing species composition	Increasing wildfire risk Inaccessible resources Hydrological change
DISASTER	Single anthropogenic environment-al disasters with long-term impacts	Various	Wildfires
MASS TOURISM	Large-scale touristic developments	Environmental degradation through development and destruction of nature for recreation	Surface sealing water consumption
URBAN SPRAWL	Planned expansion of urban infrastructure	Destruction of landscapes	Surface sealing Urban heat islands

The study of Hill et al. (2008) showed the potential to map global change syndromes by implementing a rule-based approach which is based upon coarse-scale hypertemporal satellite data. In this study, which was focussed on the semi-natural areas of the Iberian Peninsula, two major processes were identified. On the one hand, large areas of the marginal and mountainous areas are characterized by increase of the NDVI caused by land abandonment (rural exodus syndrome). On the other hand, disturbances were detected that were mainly caused by wildfires (disaster syndrome).

In the present study we expanded the approach to non-natural landscapes. Furthermore, additional data were used to support the translation of the land cover changes derived from the MEDOKADS archive into spatial patterns of areas affected by specific syndromes.

Table 4: Assignment of relevant syndromes related to major land cover transformations.

Syndromes	Land Use Change	Constraints
"Rural exodus"		
Natural	Shrub and forest encroachment	Increasing biomass , Depopulation (< -5 inhabitants per km²)
Arable	-	
"Overexploitation"		
Natural	Biomass decrease (e.g. overgrazing, logging)	Decreasing biomass
Arable (irrigated)	Intensification	Increasing Productivity , CLC90 – irrigated arable land or expansion of irrigated arable land
Arable (non-irrigated)	Intensification	Increasing Productivity , CLC90 – arable land (without irrigated)
Arable (degradation)	Biomass decrease (e.g. extensification, soil degradation)	Reduction irrigated arable land or decreasing Productivity
"Urban sprawl"	Decreasing Productivity	Population increase (> 5 inhabitants per km²)
"Mass tourism"	Often connected to urban sprawl	
"Climatic fluctuation"	Rainfall-driven	Sign. NDVI-rainfall relationship
"Disaster"	Disturbance-driven (e.g. fire and droughts)	Disturbance

Table 4 summarizes the assignment of the land cover changes to the relevant land use processes and syndromes as well as additional constraints that have to be fulfilled. Not all land cover changes, e.g. crop change were considered for the syndrome mapping. Some of the syndromes were separated into different subclasses. For example, processes that could contribute to or result from "overexploitation" of ecosystem resources can occur in natural (wood extraction, overgrazing) and arable land (overuse of resources). Within arable land there is a further discrimination. Hence, we distinguished between the risks of overexploitation due to intensification of non-irrigated arable land and irrigated arable land, respectively. We decided to designate irrigated areas separately from other arable land due to the exceptional importance of water resources and availability in semi-arid areas. Furthermore, we designated areas where overexploitation might have already caused decreasing productivity. Extensification in terms of reduction of irrigated arable land was also added to this syndrome because this process is often driven by water scarcity caused by overexploitation of aquifers (compare section 4.1.2). "Urban sprawl" and "Mass tourism" were combined to one syndrome because they effect landscape changes in many cases in comparable ways. Thus, mass tourism is frequently accompanied by extensive construction of buildings and furthermore, it generates jobs and attracts the migration of people which is reflected in population density change rates.

Additionally, the result of the NDVI-rainfall analysis was incorporated in the syndrome approach to map areas that are significantly sensitive to climatic fluctuations. Pixels that showed a significant relationship ($p < 0.05$) were classified as areas where rainfall is the major driver of NDVI behaviour.

3 Results

3.1 Time series analysis

Figure 5 shows the results of the time series analysis of the MEDOKADS archive, figure 6 and figure 7 illustrate the major land use changes that have taken place from 1989 to 2004. Characteristic examples of NDVI time profiles representing the identified changes are presented in figure 8.

Major parts of the semi-natural areas are characterized by an increase of NDVI in the time period considered, whereas only few areas are affected by a negative development of NDVI (compare figure 6). Patches of disturbance are spread all over Spain. Figure 8b shows a NDVI time series for an area affected by a fire event which caused a sudden drop of NDVI in 1998.

Figure 7 illustrates that arable land is characterized by processes of increasing productivity and the expansion of irrigated arable land. Corresponding to the semi-natural areas only few areas are affected by processes of decreasing productivity or decrease of irrigated arable land. A hot spot of decreasing productivity is located along the Mediterranean coastline around Valencia. Decrease of irrigated arable land (characterized by a forward shift of the peaking time) can be found in the province of Ciudad-Real (Castilla-La Mancha) where the fraction of irrigated arable land was reduced and replaced by non-irrigated arable land. The NDVI time profile (figure 8g) reflects this development. The annual cycle of the time series is a mixture of the signal of irrigated and non-irrigated arable land and therefore, is characterized by two peaks. At the beginning of the time series the peak in late summer is dominating whereas the peak in spring is secondary, indicating that the proportion of irrigated arable land is dominant. Around 1993/1994 the proportions begin to change, and instead the spring peak is becoming dominant; the fraction of non-irrigated arable land has increased at the expense of irrigated areas. In the autonomous communities Castilla y León, Andalusia and Extremadura large patches of “Disturbance” are located.

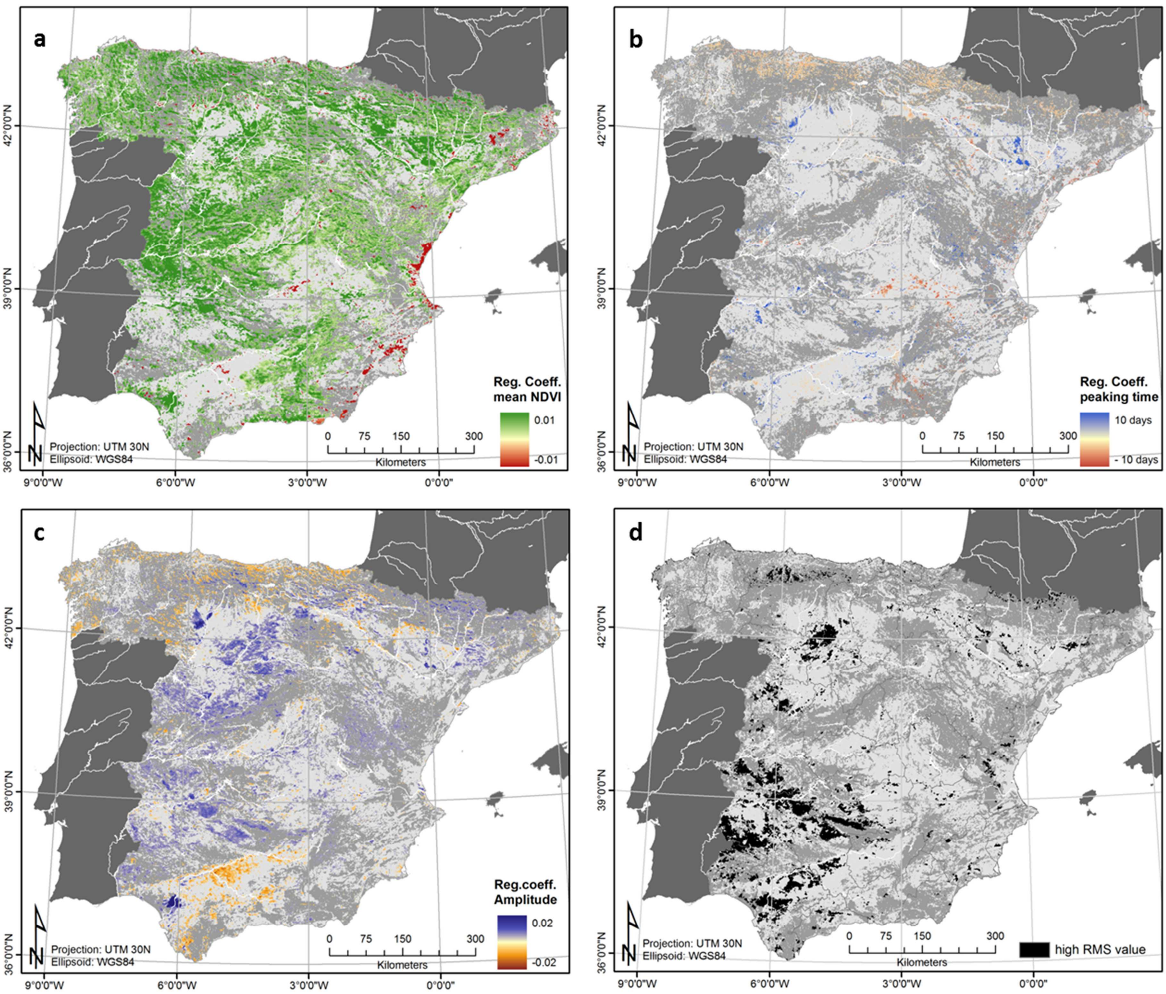

Figure 5: a) Regression coefficient of the linear trend model for the aggregated mean NDVI (Mar-Sep) derived from the MEDOKADS archive for the period 1989 to 2004. b) Regression coefficient of the linear trend model for the annual peaking time derived from the MEDOKADS archive for the period 1989 to 2004. c) Regression coefficient of the linear trend model for the annual amplitude derived from the MEDOKADS archive for the period 1989 to 2004. d) Areas likely to be affected by disturbance, e.g. a wildfire due to high RMSE value (> 0.04) of the linear trend model fitted to the aggregated annual NDVI (Mar-Sep) data series.

Within Spain light grey colours indicate non-natural areas and dark grey colours natural areas without significant trend ($\alpha = 0.1$).

Also areas dominated by crop changes were identified. Figure 8h shows, for instance, the NDVI profile of an area in Andalusia where an expansion of olive groves took place at the expense of non-irrigated arable land. The example shows that this kind of change does not necessarily involve relevant changes of aggregated NDVI or the peaking time but that a decrease of the amplitude is characteristic. Nevertheless, it is remarkable that there is still a pronounced phenological cycle which is generated by the remaining fraction of non-irrigated arable land within the pixel.

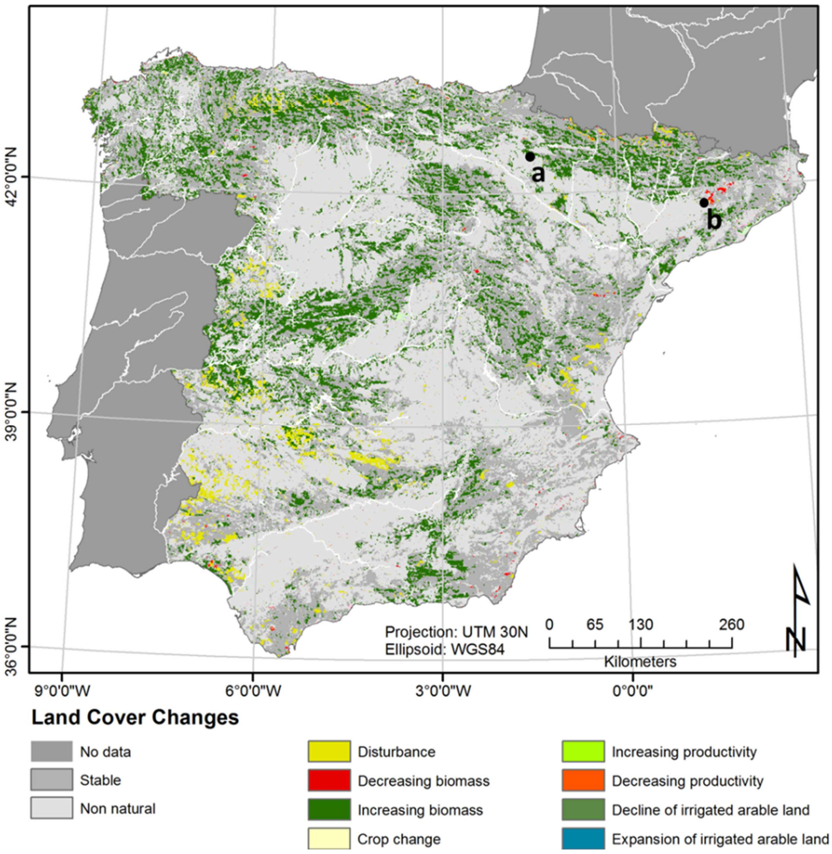

Figure 6: Land cover changes derived from the MEDOKADS archive in semi-natural areas (1989-2004).

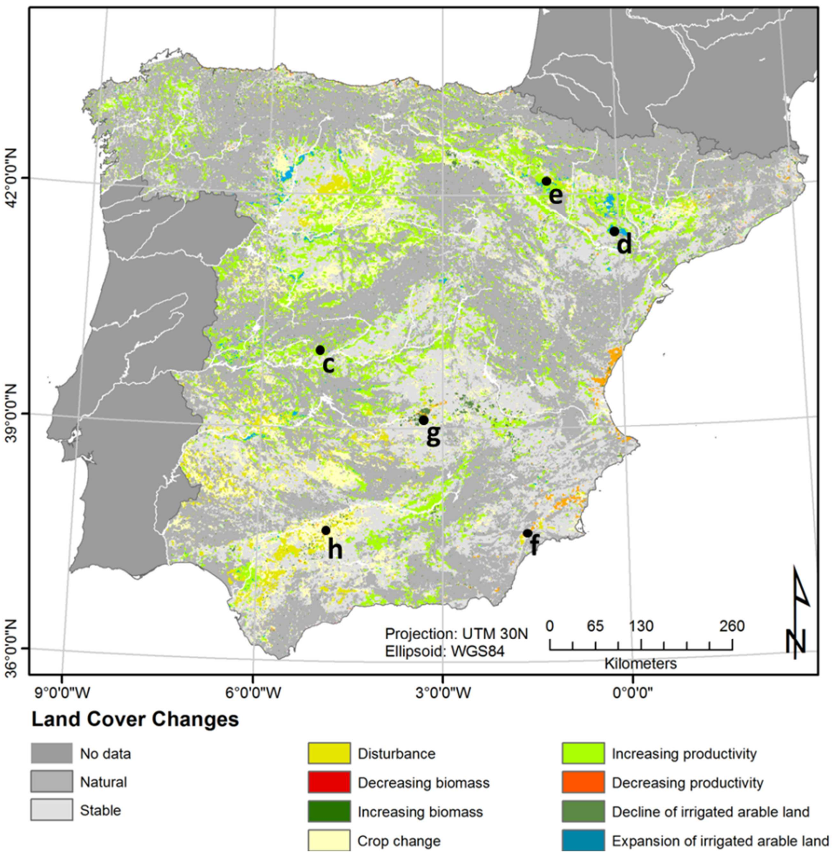

Figure 7: Land cover changes derived from the MEDOKADS archive in arable areas (1989-2004).

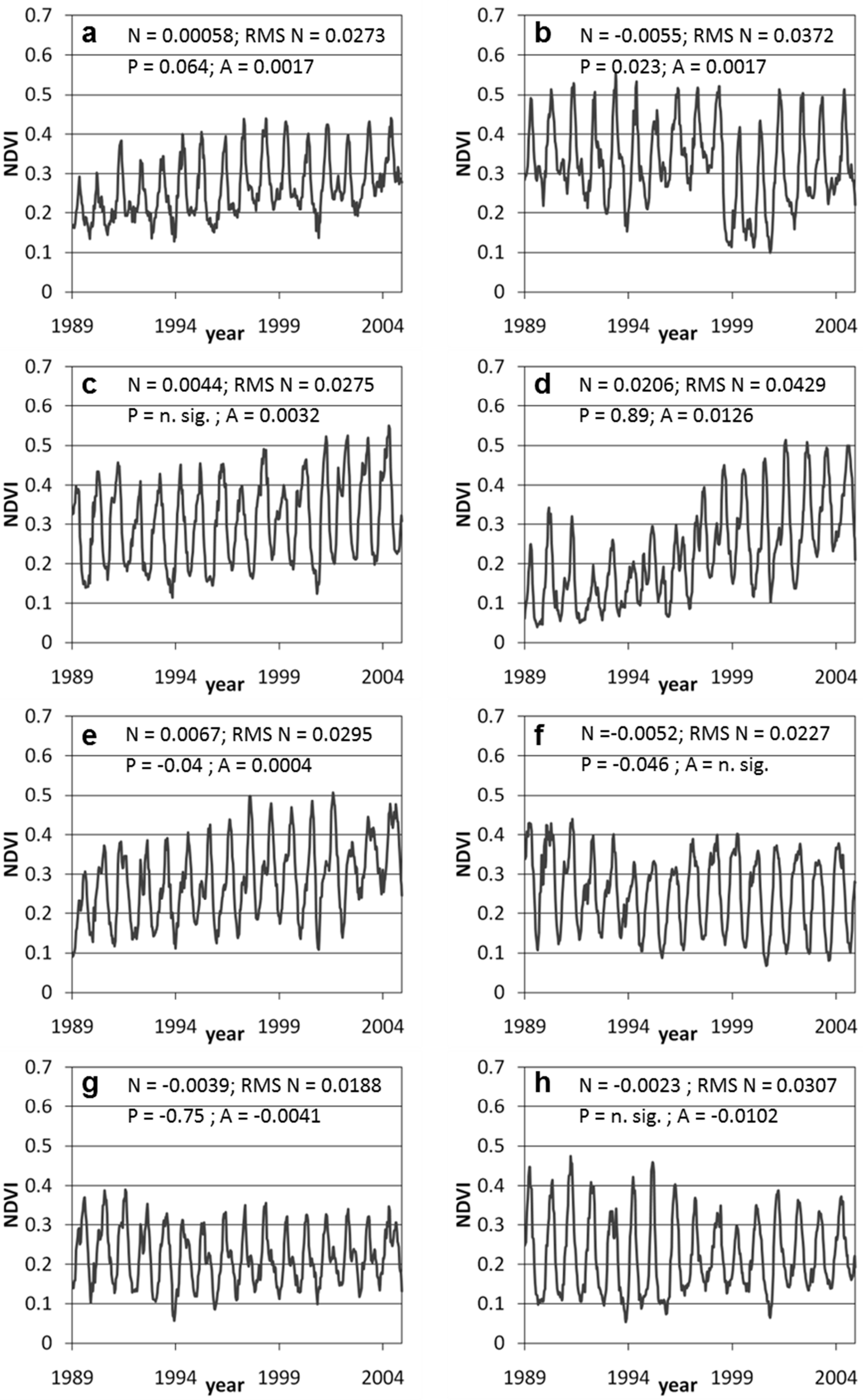

Figure 8: NDVI time profiles and corresponding time series parameters of several land cover changes derived from the MEDOKADS NOAA AVHRR archive (1989-2004): a) increasing biomass; b) Disaster (fire); c) Increasing productivity non irrigated arable land; d) Expansion irrigated arable land; e) Intensification irrigated arable land; f) Decreasing productivity; g) Extensification; h) Crop change (increasing fraction of olive cultivations); N: regression coefficient aggregated NDVI; RMS N: RMS of linear regression of aggregated NDVI; P = regression coefficient peaking time, A; regression coefficient amplitude; n.sig.: regression coefficient not significant (α = 10%).

With regard to the two examples outlined in this section, it is important to call to mind that pixels of 1 km² are often composed of a mixture of several land use types, especially in the heterogeneous landscapes of the Mediterranean. Hence, a change of a pixel's seasonal response does not necessarily mean that the entire pixel has undergone a change but that only fractions may have altered.

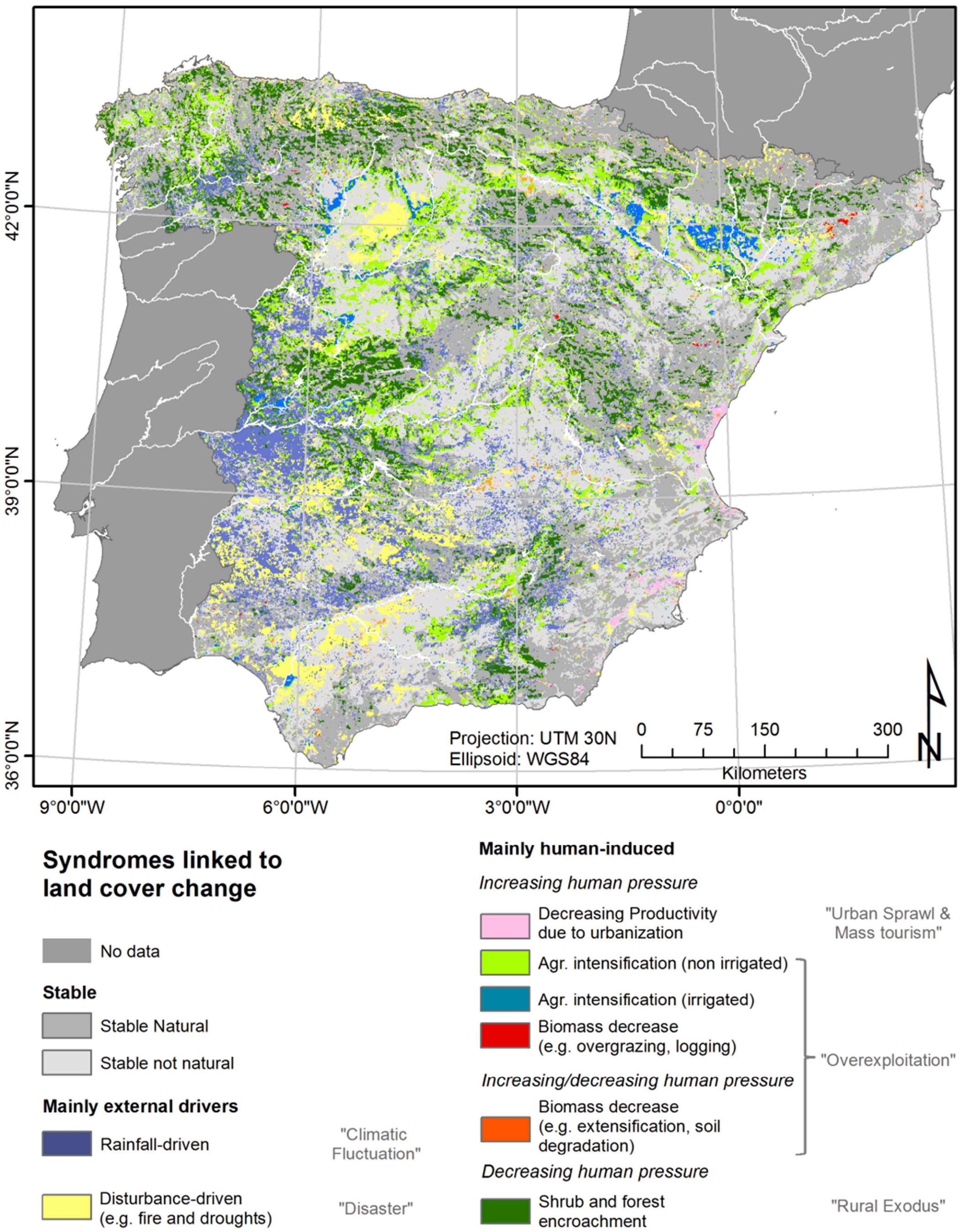

Figure 9: Syndromes and main drivers of the identified land cover changes in Spain (1989-2004).

3.2 Land use change syndromes

Figure 9 shows the resulting syndrome map of the implemented rules described in section 2.2 c. The map indicates stable areas (65% [of total area]), areas of decreasing or increasing human pressure as well as regions where mainly external drivers prevail. Dominant syndromes of land cover change are linked to an increase of vegetation or productivity: on the one hand, the rural exodus syndrome in the marginal mountainous areas of Spain (9.5%) and, in contrast, the intensification of arable land in already intensively used agricultural zones of Spain (9% rainfed agriculture, 1.5% irrigated arable land) including further expansion of irrigated arable land.

Syndromes related to a decrease of vegetation are restricted to small regions. "Urban sprawl / Mass tourism" stretches along the Eastern coastal zone (0.4%). Decreasing biomass (0.1%) and productivity (0.4%), respectively, solely emerge as small patches.

Areas affected by disturbance (5.4 %) are spread all over the country in semi-natural as well as arable land, whereas areas significantly influenced by precipitation (9%) are concentrated in the semi-arid western part of Spain. These estimates are partially determined by the thresholds defined to delineate only areas of substantial changes.

4 Discussion

4.1 Syndromes of land cover change

The results of the implemented framework suggest that the dominant human-induced land change processes are biomass increase in marginal areas due to land abandonment (decreasing human pressure) and on the other hand the intensified use of fertile agricultural zones of Spain (increased human pressure).

Owing to the underlying image resolution and the conceptual setup of this approach, it is not possible to directly validate results in a spatially explicit manner. Yet, our findings (figure 9) support and reflect the outcomes of various studies addressing different aspects of land use change in Spain during the last decades. Repeatedly the same major processes were identified as driving forces of the dominant land use transformations observed at local and regional scale all over the country:

- Starting with the upsurge of Spanish economy in the middle of the 20th century agriculture was industrialized in favourable areas with the aid of machinery, fertilizers, pesticides and improved irrigation techniques, making less people necessary to produce more agricultural products.

- In contrast, marginal areas, mainly mountainous areas and areas in inner Spain were abandoned because traditional farming was not profitable anymore and therefore, people migrated to bigger villages or cities to work in the industry or service sector.

Urbanization proceeded due to the migration of rural population to urban centres and especially along the Mediterranean coastline tourism and residencies for foreign pensioners have become one of the major economic factors (Breuer, 2008).The demographic development clearly reflects the socio-economic development of Spain towards a modern industrialized and service-oriented society. Marginal areas in inner Spain and within the mountainous areas as well as areas of intensively used agricultural zones show trends of depopulation (figure 4). In contrast the urban centres and settlements along the coastline of the Mediterranean Sea show a strong population increase. Whereas the depopulation of marginal areas is linked to the abandonment of arable land because cultivation is not profitable, the migration from the intensively used fertile zones of Spain results from the decline of required workforce due to increased productivity. The results of the syndrome approach reveal the major land use transformations that took and still take place due to these socio-economic changes. The consequences for ecosystem goods and services which are caused by the dominant land cover changes are discussed in detail in the following.

4.1.1 Natural areas

Figure 9 illustrates that “rural exodus” associated with biomass encroachment is the dominant change syndrome in semi-natural areas all over Spain, especially in the mountain ranges of northern Spain. The depopulation of marginal areas includes the abandonment of extensively used agricultural areas, the discontinuation of traditional forms of land uses, e.g. *Dehesas,* and the decrease of livestock grazing (Delgado et al., 2010; Rescia et al., 2010) and was identified as the main land cover change driver in the Pyrenees in the second half of the 20th century, leading to expansion and encroachment of shrub and forest cover (Bonet et al., 2004; Crecente et al., 2002; Delgado et al., 2010; Poyatos et al., 2003). Besides the natural succession on abandoned fields by shrub and forest cover also extensive active afforestation measures led to an increase of forested areas. Under the dictatorship of Franco almost 3 million ha forest, mainly autochthonous pine species, had been planted (Valbuena-Carabaña et al., 2010). Also in the last decades reforestation measures were promoted, between 1994 and 2006 for instance more than 650,000ha have been reforested financed by “Common Agriculture Policy” (CAP) subsidies of the European (Valbuena-Carabaña et al., 2010).

Secondly, patches of “Disturbance” characterize the semi-natural areas. Some of these patches are caused by extreme years that disturbed the time series, a major part of the “Disturbance” syndrome comprises fire events, which are in accordance with overviews provided by Moreno et al. (1998) and the results of the regular surveys of forest fires in Europe (e.g. European Communities, 2007). In the period between 1961 and 2005 around 2.7 million ha of forested areas were destroyed by fires (MMA, 2006). Only few, almost negligible patches are allocated to “overexploitation”, these areas were mainly subject to fires that could not be captured by the RMSE.

Various aspects of these landscape transformations in marginal areas of Spain, their positive and negative consequences for ecosystems’ functioning were investigated in numerous studies. Main consequences of the rural exodus syndrome combined with the disaster syndrome are the following:

- Vegetation cover recovery is reported to reduce soil erosion, especially at steep slopes as it reduces run-off and stabilizes soils (Garcia-Ruiz et al., 1996; Thomas and Middleton, 1994).
- Forests serve as carbon sinks (Padilla et al., 2010) because their sequestration rates are estimated higher than for example for arable land (Rudel et al., 2005).
- Forest and shrub expansion tends to decrease the water available for run-off (as mentioned above) and groundwater recharge because of the increasing evaporation rate compared to other land uses and thus, water yield is reduced (Beguería et al., 2003; Gallart and Llorens, 2003).
- Legacy of Mediterranean cultural landscapes frequently develop from initial mosaics of different land use types towards homogenized states with dense shrub. This causes a decline in the relative degrees of fragmentation and connectivity that are vital for ecosystem functioning and biodiversity (Forman and Collinge, 1996; Wiens, 1995). Moreover, the homogenization of the landscape involves a change of the aesthetic value (Di Pasquale et al., 2004).
- Increase in woody biomass, the loss of gaps and breaks, as well as edges between different fuel types, make such landscapes more vulnerable to fires compared to disconnected patches of forest, shrub and cultivated fields or grasslands (Duguy et al., 2007; Puigdefábregas and Mendizabal, 1998; Röder et al., 2008a; Viedma et al., 2006). This is particularly aggravated in periods of drought (Duguy et al., 2007; Viegas, 1998).

4.1.2 Non-natural areas

Major areas of arable land are affected by intensification of agricultural management or even the expansion of irrigation practices, whereas the reduction of productivity affected only few patches. As pointed out in section 2.2.2, the label “Overexploitation” does not necessarily mean that there actually is a situation of overexploitation of resources. Rather, it indicates that the present combination of factors signifies a risk warranting closer inspection and possibly management or legislative action. Irrigated areas are reported to have increased by 80% during the period 1961-1999 (Strosser et al., 1999). At the end of the 20th century water demand from agriculture made up for 68% of the overall water use in Spain (MMA, 1998) and this type of water demand is likely to increase further. The following consequences are attributed to the intensification of agricultural practices:

- Intensification of arable land increases productivity, and thus, the profit of farmers; moreover, irrigation practices reduce crop-yield risks (Garrido et al., 2006).
- Agricultural biodiversity depends to a great extent on the level of intensity (Reidsma et al., 2006). Intensification of agricultural practices includes increasing farm sizes and mechanization, less crop rotation, monocultures, and decrease of non-crop areas like trees and field boundaries which reduce landscape diversity (Stoate et al., 2001).
- These changing agricultural practices as well as the use of fertilizers and pesticides also affect soil structure composition. Moreover, all these factors lead to higher rates of soil erosion (Stoate et al., 2001).
- The additional input of fertilizers and pesticides carries the risk to contaminate soils and water resources (Peña et al., 2007; Stoate et al., 2001). Furthermore, irrigation practices can cause salinization of soils and water (Baldock et al., 2000; Szabolcs, 1990).
- Increasing water extraction from surface water and ground water irrigation leads to ground water depletion (lowering ground water tables); together with changing discharge behaviour of rivers due to extensive river control this can be a severe threat to dependant ecosystems (Garrido et al., 2006).

The expansion of greenhouse cultivation in Almeria and other parts of Southern Spain represents a special case. This process was only identified in some cases and in these cases the areas were classified as “Decreasing productivity”. This results from the conversion of mostly sparsely vegetated areas to greenhouse cultivations, which involves only a subtle change of the NDVI time profile. Adapting the employed thresholds could partly solve this problem.

“Disturbances” in the arable stratum are often bound to annual drops of the NDVI signal which could be caused by single drought events. For instance, the area affected by “Disturbance” located in Andalusia which is dominated by non-irrigated arable land is characterized by a decline of the NDVI signal in 1999. For the year 1998/1999 García-Vila et al. (2008) reported that the entire rainfall amounted to only 150mm in the province of Cordoba. A similar decline of NDVI can be observed in Castilla y Léon in 1992. Rainfall anomalies of the hydrological year 1991/1992 (Del Barrio et al., 2010) illustrate that the specific year was especially dry in this region.

“Overexploitation - Loss of Productivity” is restricted to few areas. As mentioned in section 3.2, there is a patch of extensification in the region of Castilla-La Mancha. This region was subject to an income compensation programme which was launched in 1992/1993 (Fornés et al., 2000) and by means of compensatory payments to farmers aimed at increasing the proportion of non-irrigated arable land at the expense of extended irrigation systems. The action plan was implemented because due to excessive ground water extraction the aquifers of the region, and linked to this important wetlands like the Tablas de Daimiel, suffered from the depletion of the ground water table (Oliver and Flórin, 1995).

The combined “Urban Sprawl / Mass tourism” syndrome stretches along the eastern coast around Valencia and occurs in the province of La Mancha. Besides the increasing population density statistics of tourism development (e.g. number of hotel beds 1995-2008) indicate that tourism has increased dramatically within the last decade along the Mediterranean Coast (Eurostat, 2010). Important effects of urbanization comprise the following:

- The growth of urban agglomerations is often accompanied by decreasing vegetation cover, the sealing of surfaces and thus, alters energy fluxes and causes the “urban heat island effect” which alters the regional climate (Foley et al., 2005). The increase of impervious surfaces may also increase the risk of floods, even though in arid areas urbanization may reduce the tendency of flash floods because the establishment of green spaces within the urban areas has stabilizing effects (DeFries et al., 2004).
- Water demand increases with the growth of the touristic sector and often competes with other uses like agriculture (Peña et al., 2007).
- The expansion of urban areas into semi-natural areas leads to the fragmentation of landscapes and can be a threat to the functioning of semi-natural ecosystems (Peña et al., 2007).

- Urbanization is linked to increased emission of pollutants that alter air and water quality (Foley et al., 2005).

The population change map (compare figure 4) indicates that many urban centres experienced a severe increase of population but where not captured by the implemented approach. There are several reasons that could hamper the detection of the “Urban sprawl / Mass tourism” syndrome. Figure 5a shows that a significant NDVI decrease has occurred in many places along the coast, but the changes are partially this small that the magnitude of the regression coefficient falls below the threshold of changes that are considered in the study (analogous to the case of the greenhouse cultivations, see preceding section. Secondly, urbanization may only have subtle influence on the NDVI time profile because of the pixel size of 1 km². Additionally, urbanization often comprises the sealing of surfaces but also the generation of green spaces, for instance gardens or lawns for recreational purposes and therefore, the mixture of both alterations may not have a significant influence on the NDVI signal.

4.1.3 Climatic fluctuations

Areas that are significantly driven by variations of annual rainfall amount are present throughout Spain, but are particularly concentrated in its semi-arid western part (Extremadura) and in the mountainous areas of Andalusia in southern Spain. The fact that in the most arid region in the Southeast no significant relationships occur might be explained by the fact that the NDVI is not a sensitive biomass indicator in case of low vegetation cover (Price, 1993). Figure 10 illustrates this dependence of the mean annual NDVI from rainfall for three areas in Spain. Compared to the Corine Land Cover map, mainly shrublands (ca. 35%) and heterogeneous extensively cultivated areas (ca. 22%), e.g. Dehesas, are associated with this class.

At this point it is important to mention that in addition to the dependence on rainfall these areas can also be affected by land use changes and disturbances that could be accounted for in the syndromes’ map. Owing to reasons of presentability we did not consider this.

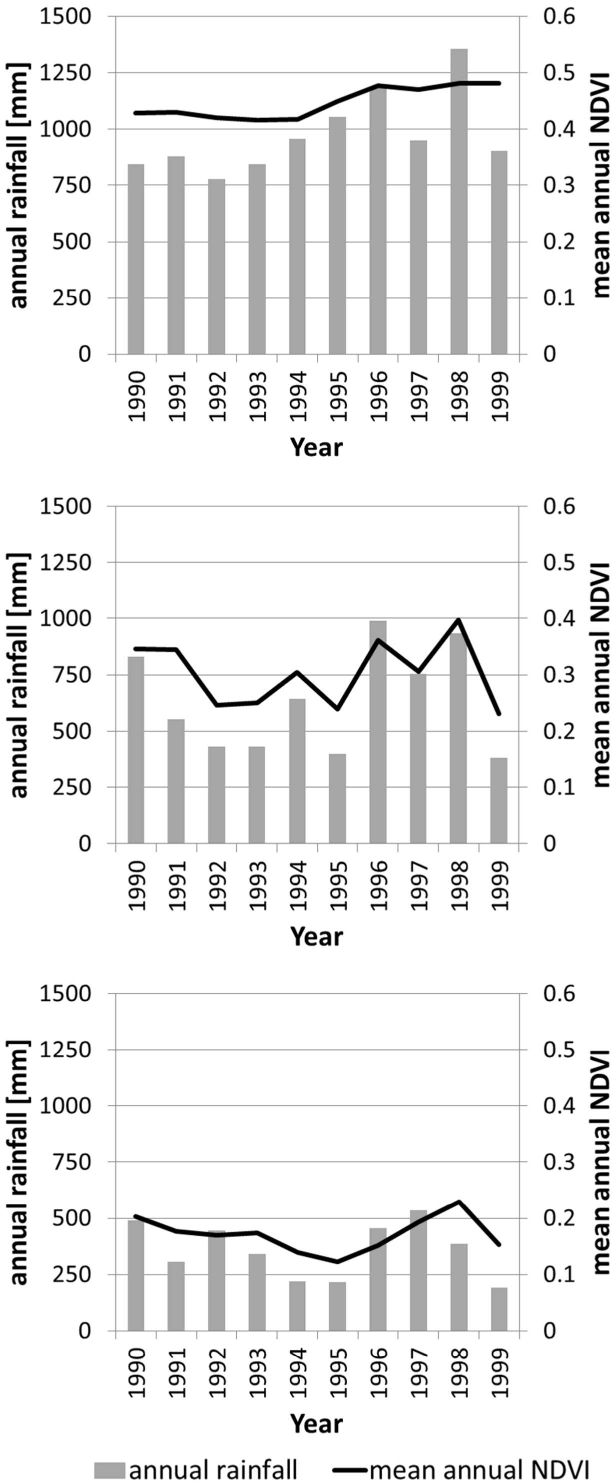

Figure 10: Examples of areas characterized by a significant relationship between annual rainfall and the mean annual NDVI (hydrological years 1990-1999), north-western Spain (above), western Spain (middle), south-eastern Spain (below).

4.1.4 Interactions between identified syndromes and implications for decision makers

Depending on the desired functions of an ecosystem the perception of landscape change may be different and frequently competing demands lead to trade-offs that need to be considered (Chapin III et al., 2010; DeFries et al., 2004).

For instance, regarding climate change carbon sequestration is regarded as an important ecosystem service and the expansion of forested areas is experienced as a positive process. Padilla et al. (2010) estimated for a study area in South-Eastern Spain that the replacement of dryland farming and pastures by holm oak and pine forests did double the annual carbon sequestration rate within the 20th century. In this context they stress that to preserve these rates the conservation of forests in this region is desirable. On the other hand, shrub encroachment and forest expansion tend to decrease the water available for run-off and groundwater recharge because of the increasing evapotranspiration rate compared to other land uses. According to Gallart and Llorens (2003), these processes have led to a reduction of annual streamflow of the Ebro of approximately 25% during the past 50 years. Begueria et al. (2003) reported similar decrease of the annual discharge of a basin in the Central Pyrenees due to land cover change. Many mountainous areas serve as headwaters of rivers and provide water resources that are essential for irrigated agriculture and the supply of the population and tourism. In this context, Otero et al. (2011) suggest that with regard to water availability the management of abandoned mountainous areas should aim at an appropriate distribution of arable and wooded areas.

The preceding example demonstrates that changes of terrestrial ecosystems do not only have implications for the altered ecosystem itself but also for functions and services of adjacent ecosystems. Olesen and Bindi (2002) consider problems of water availability due to climate change as a major concern in the future. Climate conditions in Spain have already altered within in the last decades. Brunet et al. (2007) estimated an increase for minimum and maximum temperature of 0.10°C and 0.12°C per decade, respectively. The studies related to precipitation show that the trends are varying spatially and seasonally (Gonzalez-Hidalgo et al., 2009; Río et al., 2010; Ruiz Sinoga et al., 2010). However, several studies indicate that water deficit / aridity increased for many areas. For instance, in north-western Spain the mean temperature has been reported to have risen by 0.10°C and estimated evapotranspiration rates to have increased by 13 mm per decade in the period 1910 to 1994. Within the observation period of the same study precipitation did not change significantly and in consequence, water deficit did increase (Piñol et al., 1998). Ruiz Sinoga et al. (2010) found significant negative trends for precipitation amount in the inner area of Andalusia (southern Spain) and indicate that the reported decrease might impact negatively on water resources because these areas are important to build and recharge water resources. Projections expect that aridity will further increase in southern Europe (Bates et al., 2008). For the Iberian Peninsula mean temperature is expected to increase further, summer precipitation is likely to decline, whereas winter precipitation is forecasted to slightly increase (Schröter et al., 2005). Furthermore, inter-annual precipitation variability is likely to increase.

In addition, the aforementioned reduction of water yield in important head waters due to shrub and forest expansion after land abandonment could aggravate the reduction of water reserves. With regard to expected increasing water demands of agriculture, population and tourism these trends could be extraordinary volatile (Schröter et al., 2005).

These developments will put further pressure on water management in Spain. Lorenzo-Lacruz et al. (2010) investigated the impact of droughts and water management on the hydrological consequences for the headwaters of the Tagus River in central Spain and discussed the implications of climate change on the management of the Tajo-Segura-Transfer in the face of expected increasing severity and frequency of droughts in southern Europe. In recent years, water resources severely declined due to drought events and led to a situation where managers had to reduce both the transfer to Segura and the flow to the Tagus River (Lorenzo-Lacruz et al., 2010). This example underpins that – also with regard to the discussed water transfers to the Mediterranean region – the handling and management of water resources is an issue with an ever-growing political relevance. In this context, Peña et al. (2007) explain that policy makers often tend to solve water demand by the implementation of technological progress, e.g. water transfer and desalinisation plants, but that this will unlikely be a solution as long as unsustainable development of agriculture and tourism further increases water demand. Since the accession of Spain to the European Union one of the major driving forces of Spanish agriculture is the subsidy policy of the Common Agricultural Policy (CAP). The CAP reform of 2005 started to decouple subsidies from productivity and supports sustainable farming practices. However, it is too early to assess the effects of these recent policy shifts. Similarly, the Water Framework Directive of the EU (WFD) will have consequences on land use in Spain. Adopted in 2000, it aims at sustainable water use and moved to full-price water recovery in 2010 which will have impacts on the agricultural irrigation management as water prices will rise (Downward and Taylor, 2007). Using the example of the Segura river basin, Grindlay et al. (2010) show that the implementation of the WFD is accompanied by many difficulties due to the intended environmental aims of the directive and the needs of farmers.

The preceding explanations illustrate that the consequences of land use changes on ecosystem goods and services are both complex within an ecosystem and between them; especially bearing in mind that the full range of ecosystem functions should be regarded and evaluated in ecosystem management. The syndrome approach provides a possibility to review the major land use transformations with regard to their driving forces and take them into account for future planning.

4.2 Uncertainties

The results of this study demonstrate that the syndromes approach is capable of monitoring the dominant land use changes and their driving forces at national scale. Nevertheless, the approach bears some uncertainties and leaves room for improvements as discussed in the following.

Dryland monitoring requires long term data archives because the processes prevailing are often of gradual nature and moreover, are often superimposed by climatic variations (Mulligan et al., 2004). Therefore, monitoring of drylands demands long term data archives. The remote sensing time series employed in this study comprises 16 years, which is quite long for remote sensing data but might be short in terms of the observed processes. Moreover, the results are only valid for the observation period and do not allow for an evaluation beyond this time window.

In the present study we focused primarily on land use changes within the major land use strata and did not consider land use transformations between for instance natural and agricultural land use. A comparison of our results with changes assessed with Corine Land Cover between 1990 and 2006 showed that these “inter-strata” changes concern only a relative small part of the study area. These changes are partially classified as “disturbance” by the syndromes’ approach, which is acceptable, but some are also misinterpreted as changes within the observed strata. The spatial extent of the syndromes obviously depends on the thresholds employed to delineate areas of substantial change (compare section 2.2.1.1). These have been set to conservative value ranges.

Another uncertainty, which was already mentioned in the discussion, is caused by the geometric resolution of 1 km². Especially in heterogeneous areas like the Mediterranean, pixels of this size are always a mixture of several land cover types. Changes within a pixel will only be detected if the magnitude and the proportion of certain land use changes are big enough to alter the NDVI signal in a significant way. Thus, small scale changes can remain undetected (Stellmes et al., 2010) even though they might be of high relevance, especially for local/regional land management and for understanding individual actor's decisions (Lambin and Geist, 2006; Lorent et al., 2008). Therefore it is important to underline that the objective of this approach is to reveal the main processes of land use change at coarse scale. Another problem associated with the pixel size is that direct validation of the results, e.g. using ground truth data, is not feasible. Therefore, plausibility of results was reconsidered by a thorough literature review and cross-checking with land cover changes detected by Corine Land Cover.

One major problem of including socio-economic data is the level of detail that often differs considerably from the scale of the remote sensing data and puts problems on the integration. Additionally, the availability of suitable statistical data sets at all or for different dates is often not given. In this case, assumptions of driving forces have to be made. In this study, we followed a simple approach to include the population change figures. More sophisticated approaches including fuzzy and neighbourhood techniques could be used to improve the method.

5 Conclusion

The present study aimed at setting up a framework to identify major land cover changes and their driving forces for the mainland of Spain, and therewith assesses areas with increasing or decreasing human influence on land cover. Further approaches were included in the framework to distinguish between areas where vegetation cover and its development is mainly caused by human actions or where it is probably driven by external factors including climatic fluctuations and disturbances (e.g. wildfires, drought years). The results for Spain show that the presented approach is meaningful to assess major mechanisms of land cover processes at coarse scale. Major land cover changes identified were (i) biomass increase in semi-natural areas due to land abandonment, (ii) intensification of already intensively used arable areas including further expansion of irrigated arable land and (iii) urbanization effects due to migration and mass tourism. Further land cover processes related to a decrease of greenness did not play an important role. Thus, only few patches were identified, suggesting that no large-scale land degradation processes are taking place in the sense of decline of primary productivity.

Nevertheless, the detected land cover processes may impact ecosystem functioning and bear risks for the provision of goods and services. This risk is not only confined to the affected ecosystem itself but can also impact adjacent ecosystems due to inter-linkages. This was elucidated with the example of the ecosystem service “water provision” which will be an increasingly important issue in many drylands facing climate change.

The approach can be transferred to other regions and countries. Depending on the underlying drivers of change and on available data, i.e. remote sensing time series, socio-economic data sets and climate data archives, the approach has to be adapted to the corresponding context.

Syndrome based maps may support evaluating trade-offs between certain land use decisions with respect to the ecosystem goods and services concept. Such an assessment might be based on known consequences of land use change for specific goods and services. It could also be of interest to implement a thematic mapping of consequences for single ecosystem goods and services based on known consequences (e.g. from scientific studies) of land use change for the issue of interest. Regarding for instance the consequences of the syndromes assessed in Spain, a map illustrating implications for water resources (compare section 4.1.4) could be derived based on scientific results concerning this topic.

There is a growing need of basing policy decisions on credible, spatially explicit data that represent well-defined classes. Syndrome maps comply with this demand by highlighting the spatial extent and pattern of the dominant land changes and their causes. They may thus serve as a source of information for policy makers to assess main consequences of socio-economic changes on ecosystems and, moreover, to understand interactions between the different processes and their implications for the ecosystem’s functioning (and the functioning of dependent ecosystems). For example, the consequences of shrub encroachment in marginal areas should not be rated without considering their effects on water provision for irrigated agriculture and urban centres often located hundreds of kilometres away. In conjunction with other information layers providing, for instance, information about land condition (Del Barrio et al., 2010) or information about the intensity level of agricultural practices (Weissteiner et al., 2011), end-users could gain a comprehensive picture of the present state conditions. This could be complemented by scenarios on future land use developments that are based on policy and societal analyses or follow model approaches (Rounsevell et al., 2003; Verburg et al., 2006). An integration of such information appears well suited to inform policy makers and land managers of potential results of their actions.

Concluding, our method can indeed provide a synopsis of major change processes and bears the potential to serve as a framework to identify major land use transformations and their underlying causes.

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Chapter VI

Synthesis

1 Summary

The aim of this thesis was the implementation of an integrated assessment methodology that should be able to reveal major land use transformations for large areas under consideration of human-induced changes and climatic variability. Four objectives were formulated to pursue this aim. Each of these objectives was treated in a journal publication. In the following a short summary of the results is given with special consideration of the overall research intention.

Objective 1: Possibilities and limits of monitoring land system change processes with remotely sensed data at different spatial and temporal scales

Chapter II evaluates the possibilities and limitations of time series analysis of two different types of earth observation data operating at different temporal and spatial scale. For a study site in Greece the overlapping time period of an annual Landsat TM/ETM+ time series operating at medium scale and coarse scale NOAA AVHRR data was taken advantage of. The comparison was performed to identify strengths and limits of both time series for the detection of land cover changes by means of time series analysis.

The results suggest that hypertemporal coarse scale data as provided by the MEDOKADS archive are suitable to monitor large scale land cover changes. Thereby, it was demonstrated that the incorporation of parameters describing the phenological cycle like the amplitude and the peaking time significantly enhances the ability to detect different types of land cover changes compared to change detection solely based on trends of greenness. The comparison furthermore revealed that the ability of coarse scale earth observation systems to detect changes is largely dependent on magnitude and structure of land cover changes, for instance altered rangeland management in the study area generated small scale patterns of vegetation cover decrease and increase. This could be monitored by means of the Landsat TM/ETM+ time series but were not detectable by the MEDOKADS archive. Hence, the study indicates that major land cover changes are likely to be identified employing coarse scale data but especially local scale land cover changes might be unrevealed. Yet, these changes may be of high relevance for local management strategies and the understanding of decisions taken by local actors (Lambin et al. 2006; Lorent et al. 2008). It is important to take this constraint into account when setting up a framework to assess land cover changes and when interpreting results based on coarse scale remote sensing time series. A further implication of the results suggest that a multi-scale observation system cannot be a simple top-down hierarchical system in the sense that bright and hot spots of land cover transformations are solely identified by an analysis at coarse scale which are then further analysed by data of finer resolution.

Objective 2: Distinction of climatic fluctuations and socio-economic forcing

Objective 2 addressed the evaluation of the dependency of NDVI rainfall on climatic variability for monthly and annual data. Significant monthly NDVI rainfall relationships up to three months were found for semi-arid and sub-humid areas whereas significant influence of temperature was confined to few regions. Rainfed crops and grassland in the semi-arid regions of Spain were demonstrated to be more sensitive towards rainfall anomalies compared to woody species. The driest areas of Spain (South Eastern Spain and the Ebro basin) did not show a significant relationship, indicating that the interval of the monthly data set seems to be too broad to capture the anomalies of vegetation caused by precipitation.

For the annual lag analysis the model order varied between zero and three. This result indicated lagged dependencies of the NDVI from previous accumulated annual rainfall amounts up to 3 years. Significant locations mainly corresponded to areas with natural grass and scrub vegetation in western Spain and to a minor degree to agroforestry areas. The implemented method proved to be useful to assess the variability of NDVI that is explained by rainfall comprising duration, explained variability and significance of the relationship. Thus, the approach permits to distinguish human-caused and climatic changes of vegetation cover. This option was included in the study presented in chapter V. Beyond that, the approach could also be employed to predict changes in productivity caused by climatic anomalies, where the effects of droughts might be of particular interest.

However, the limited observation period of 10 years leaves many areas that do not show statistically significant relationships. Moreover, with low vegetation abundances the sensitivity of NDVI is limited to capture subtle changes at coarse scales (Schmidt and Karnieli 2001). The latter reason could explain the non-significant relationship observed in south-eastern Spain where sparse vegetation is prevailing.

Objectives 3 and 4: Implementation of a regional monitoring framework that is capable to identify land cover change based on remote sensing time series with a high temporal resolution and enhancement of the remote sensing based syndromes approach by incorporating other data sources

The main research question of this thesis was to set up a framework to allow for the identification of land use changes in drylands and reveal their underlying drivers (objectives 3 and 4). Whereas the study presented in chapter IV aimed at establishing a rationale based on coarse scale hypertemporal earth observation data, chapter V expanded and enhanced the implemented method. The concept of describing land cover change processes in a framework of global change syndrome was introduced by Schellnhuber et al. (1997) and is considered, as outlined before, as one of the most promising approaches to describe human-nature interactions. Instead of describing single processes (symptoms), syndromes (a combination of symptoms) describe bundles of interactive processes which appear repeatedly and in many places in typical combinations and patterns. In the frame of chapter IV the syndrome approach was adapted for the EU-Mediterranean drylands, i.e. in a first step relevant syndromes were identified. In the following a syndrome approach was implemented for semi-natural areas of the Iberian Peninsula based on time series analysis of the MEDOKADS archive. In the subsequent study (chapter V) the syndrome approach was expanded and adapted to other land cover strata. Furthermore, results of the analysis of the relationship of annual NDVI and rainfall data presented in chapter III were incorporated to designate areas that show a significant relationship indicating that at least a part of the variability found in NDVI time series was caused by precipitation. Additionally, a first step was taken towards the integration of socio-economic data into the analysis; population density changes between 1961 and 2008 were utilized to support the identification of processes related to land abandonment accompanied by cessation of agricultural practices on the one hand and urbanization on the other.

The main findings of the two studies comprise three major land cover change processes caused by human interaction: (i) shrub and woody vegetation encroachment in the wake of land abandonment of marginal areas, (ii) intensification of non-irrigated and irrigated, intensively used fertile regions and (iii) urbanization trends along the coastline caused by migration and the increase of mass tourism. Even though these results cannot be directly validated by ground measurements these outcomes are supported by the results of numerous case studies performed in Spain.

Land abandonment of cultivated fields and the give-up of grazing areas in marginal mountainous areas often lead to the encroachment of shrubs and woody vegetation in the course of succession or reforestation (Bonet et al. 2004; Poyatos et al. 2003). Whereas this cover change has positive effects concerning soil stabilization (Garcia-Ruiz et al. 1996; Thomas and Middleton 1994) and carbon sequestration (Padilla et al. 2010) the increase of biomass involves also negative consequences for ecosystem goods and services; these include decreased water yield as a result of increased evapotranspiration (Beguería et al. 2003; Gallart and Llorens 2003), increasing fire risk (Duguy et al. 2007; Viedma et al. 2006), decreasing biodiversity due to landscape homogenization (Forman 1995; Wiens 1995) and loss of aesthetic value (Di Pasquale et al. 2004).

Arable land in intensively used fertile zones of Spain was further intensified including the expansion of irrigated arable land. The intensification of agriculture has also generated land abandonment in these areas because less people are needed in the agricultural labour sector due to mechanization. Urbanization effects due to migration and the growth of the tourism sector were mapped along the eastern Mediterranean coast. Urban sprawl was only partly detectable by means of the MEDOKADS archive as the changes of urbanization are often too subtle to be detected by data with a spatial resolution of 1 km².

Further land cover processes related to a decrease of greenness did not play an important role. Thus, only few patches were identified, suggesting that no large-scale land degradation processes are taking place in the sense of decline of primary productivity after disturbances.

Nevertheless, the land cover processes detected impact ecosystem functioning and using the example of shrub encroachment, bear risks for the provision of goods and services which can be valued as land degradation in the sense of a decline of important ecosystem goods and services. This risk is not only confined to the affected ecosystem itself but can also impact adjacent ecosystems due to inter-linkages. In drylands water availability is of major importance (Olesen and Bindi 2002) and the management of water resources is an important political issue. In view of climate change this topic will become even more important because aridity in Spain did increase within the last decades (Brunet et al. 2007; Piñol et al. 1998) and is likely to further do so (Bates et al. 2008). In addition, the land cover changes detected by the syndrome approach could even augment water scarcity problems. Whereas the water yield of marginal areas, which often serve as headwaters of rivers, decreases with increasing biomass, water demand of agriculture and tourism is not expected to decline (Schröter et al. 2005).

In this context it will be of major importance to evaluate the trade-offs between different land uses and to take decisions that maintain the future functioning of the ecosystems for human well-being.

2 Conclusions and Outlook

The developed syndrome-based mapping concept proofed capable to reveal major land cover change processes that have taken place in Spain between 1989 and 2004 at 1 km². The land cover changes identified by time series analysis of coarse scale hypertemporal earth observation data were linked to the underlying climatic and socio-economic drivers to allow for distinction between climatic fluctuations and human-induced changes and moreover, to reveal the underlying socio-economic drivers that involved land use changes.

The observation period of the study ends in 2004, one year before the new Common Agricultural Policy (CAP) reform was coming into force. This and other policies and directives like for instance the EU Rural Development Policy, EU Water Frame Directive or the National Water Plan of Spain will have impacts on the socio-economic background (proximate causes) and consequently, probably also lead to further land use change. Such alterations may require adaptation of the process framework of the syndrome approach when applied to other observation periods.

The syndrome approach shows great potential to be transferred to other regions of the world but beforehand, it is essential to adapt the approach to the regional background. The presented approach was developed based on a conceptual process framework that was especially adapted to the situation in the northern EU-Mediterranean countries. Other physical and socio-economic settings might result in completely different processes. Udelhoven and Hill (2009), for instance, showed that the abandonment of cropped areas in marginal areas of Syria is characterized by a long-term decline of vegetation cover, although it can be rated as a positive land use/cover change process in these marginal areas. Thus, a global assessment would require not only a stratification based on major land-use strata but also on socio-economic conditions.

Furthermore, the combined analysis of remotely sensed data and socio-economic data is often hampered by differences in the spatial domain. Whereas remote sensing images are raster data, socio-economic data are often ascertained at administrative units such as municipalities and regions or result from household surveys. In the concluding study presented in chapter V a simple intersection of both datasets was performed to include population density changes into the analysis. More sophisticated approaches like for instance fuzzy logic and neighbourhood relationships could be utilised to enhance the rationale. Another problem often arises from the fact that relevant socio-economic data are not available at all, not at the appropriate scale or not at the required time intervals. In this case it will be necessary to either base the assignment of syndromes to land use changes on expert knowledge or to use alternative datasets. Night time satellite imagery as derived from Defense Meteorological Satellite Program's Operational Linescan System (DMSP OLS) could, for instance, serve as surrogate for demographical data such as population size, distribution and rates of urbanization (Sutton et al. 2010).

Another issue of importance concerns data availability and continuity of remote sensing data. The 1km MEDOKADS archive, for example, only provides data for the Mediterranean area and parts of Western Europe. An alternative global NOAA AVHRR archive is the GIMMS data set which comprises the period from 1981 to 2006. This archive consists of data with a resolution of 8x8 km² which aggravates the detection of land use changes. Sensors like SPOT VEGETATION, MODIS and MERIS provide excellent alternative data archives, however these data sets are only available since the end of the 20th and beginning of the 21th century, respectively. At this point it is also important to consider the relevant observation period to capture relevant land change processes. A further problem regarding enhanced sensor systems like MODIS concerns the restricted time-span covered by individual sensors because only few systems like the NOAA AVHRR, Landsat series and SPOT Vegetation are continuously resumed and thus, provide the possibility to cover long observation periods with one sensor. Connecting data from different sensors with the intention of covering time spans of several decades with homogeneous data products still remains a major challenge. Recalling the necessity of adapting the syndrome approach in consequence of changing process frameworks for instance triggered by changing subsidy schemes, it might rather be of advantage to split the observation period than considering the entire period in one analysis. This would also permit to employ several remote sensing data archives. For example the MEDOKADS archive could serve as data basis until 2003, whereas subsequently, the MODIS data could be employed.

Even though the presented approach reveals dominant land use transformations, it cannot serve as a stand-alone methodology. It is of major relevance to understand that this methodology is one important component that supports the understanding of coupled human-environmental systems at large scales. However, a comprehensive picture of the status and development of dryland areas can only be achieved if complementary information of land condition as provided by the 2dRUE approach (Del Barrio et al. 2010) or the intensity level of land use (Weissteiner et al. 2011) is integrated. The results of the implemented approach may as well be coupled with scenarios on future land use developments that are based on policy and societal analyses or follow model approaches (Rounsevell et al. 2003; Verburg et al. 2006).

The comparison of the Landsat TM time series and the NOAA AVHRR archive showed that small scale changes cannot be detected based on this approach, even though they might be of high relevance for local management of resources (Cohen and Goward 2004; Lambin et al. 2006). This underlines the fact that land degradation processes are multi-scale problems and that data of several spatial and temporal scales are mandatory to build a comprehensive dryland observation system. Sommer et al. (2011) suggested that the syndrome approach could serve as a stratification or framework for local studies, e.g. to support the choice of suitable indicators or to look at the results of local to regional studies in a broader perspective. In the context of scale issues it will be also interesting to evaluate to which extent synthetically hypertemporal Landsat time series derived from a spatial and temporal fusion between Landsat and MODIS (Gao et al. 2006) can contribute to an improved monitoring of large areas.

The outcomes of the observed land change syndromes as related to the provision of ecosystem goods and services suggest that the term land degradation should not be restricted to the actual (or potential future) loss of productivity of ecosystems in terms of primary productivity. Rather, it should be perceived in a wider sense, in which degradation of an ecosystem can also be defined as the decline of important ecosystem services, also suggested by the Millennium Ecosystem Assessment (2005a). Even though this definition further increases the complexity of dryland assessment, it might be more compliant with the needs of policy makers and land management to develop and establish sustainable land use practices.

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Appendix A

Assessment and monitoring of land condition in the Iberian Peninsula, 1989–2000

Remote Sensing of Environment (2010), 114, p. 1817–1832

Gabriel del Barrio, Juan Puigdefabregas, Maria E. Sanjuan,
Marion Stellmes and Alberto Ruiz

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ABSTRACT

Diagnosis of land condition is a basic prerequisite for finding the degradation of a territory under climatic and human pressures leading to desertification. Ecosystemic approaches, such as the one presented here, address ecosystem maturity or resilience. They are low cost, not very prone to error propagation and well-suited to implementation on remotely sensed time–series data covering large areas. The purposes of this work were to develop a land condition surveillance methodology based on the amount of biomass produced per unit rainfall, and to test it on the Iberian Peninsula.

In this article, we propose parallel and complementary synchronic assessment and diachronic monitoring procedures to overcome the paradox of monitoring as a sequence of assessments. This is intrinsically contradictory when dealing with complex landscape mosaics, as relative estimators commonly produced for assessment are often difficult to set in a meaningful time sequence. Our approach is built on monthly time–series of two types of data, a vegetation density estimator (Green Vegetation Fraction-GVF) derived from Global Environmental Monitoring satellite archives, and corresponding interpolated climate fields. Rain Use Efficiency (RUE) is computed on two time scales to generate assessment classes. This enables detrended comparisons across different climate zones and provides automatic detection of reference areas to obtain relative RUE. The monitoring procedure uses raw GVF change rates over time and aridity in a stepwise regression to generate subclasses of discriminated trends for those drivers. The results of assessment and monitoring are then combined to yield the land condition diagnostics through explicit rules that associate their respective categories.

The approach was tested in the Iberian Peninsula for the period 1989 to 2000 using monthly GVF images derived from the 1-km MEDOKADS archive based on the NOAA-AVHRR sensors, and a corresponding archive of climate variables. The resulting land condition was validated against independent data from the Natura 2000 network of conservation reserves. In very general terms, land was found to be healthier than expected, with localised spots of ongoing degradation that were associated with current or recent intensive land use. Static or positive vegetation growth rates were detected almost everywhere, including Mediterranean areas that had undergone increased aridification during the study period. Interestingly, degrading or static trends prevailed in degraded or unusually degraded land, whereas trends to improve were most represented in land in good or unusually good condition.

1. Introduction

The land degradation concept aims at covering a range of climate and human-induced processes leading to a decline in soil potential to sustain plant productivity. The first attempt to produce a global assessment occurred at the end of the last century, resulting in the Global Assessment of Soil Degradation (GLASOD) (Oldeman et al., 1991). GLASOD was a qualitative assessment, largely based on expert judgement that distinguished the main processes leading to soil degradation, such as water and wind erosion–sedimentation, soil and water salinisation, loss of soil organic carbon and nutrients, loss of soil structure, etc. The GLASOD database was used in the preparation of the World Atlas of Desertification (UNEP, 1992).

Later on, the GLASOD approach was upgraded in a new worldwide project entitled Land Degradation Assessment in Drylands (LADA, 2006) sponsored by the United Nations Environment Program, the Global Environmental Facility and the Food and Agricultural Organization. Whilst retaining the original GLASOD soil degradation categories, LADA took a step forward by aiming at quantitative deliverables. This was achieved by including socio-economic drivers and by enlarging its scope to carbon balance and biodiversity as components of the functional land system and its degradation.

A third global initiative with implications for land degradation assessment was the Millennium Ecosystem Assessment (MA) developed from 2001 through 2005. Its desertification synthesis (Adeel et al., 2005) evaluates the status of desertification in drylands by

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asking key questions and providing answers based exclusively on the reports generated for the MA. It yields a consistent picture of links between land degradation, global change and biodiversity loss. It also includes guidelines for improving assessment and monitoring approaches by accounting for the role of human action and climate variability.

The three aforementioned projects reveal a historical trend of increasing complexity in approaches to land degradation assessment, which go from considering effects on 'soil' to explicitly including drivers on 'land', to being more concerned with global interaction of desertification with the atmospheric system through changes in land surface properties, and wide-scale effects on downstream delivery of water and sediments due to changes in hillslope to channel connectivity. This trend has largely been propelled by the United Nations Convention to Combat Desertification (UNCCD) 1994 definition of desertification as 'The land degradation in arid and semi-arid and dry-sub-humid areas resulting from various factors, including climatic variations and human activities'. This definition, in spite of its generality and simplicity, has the advantage of providing a benchmark for designing assessment and diagnostic methods. The outcome or symptom of desertification is land degradation, and its driving forces are climate variation and human activities. Furthermore, land degradation is defined by the UNCCD as 'the loss of land's biological and economic productivity and complexity'. This is a holistic definition that focuses on the overall impact rather than on particular causes like soil erosion, salinisation, etc.

Whilst retaining that holistic character of the land degradation concept, we propose an ecosystemic approach evolved from the original UNCCD definition, in which, land condition becomes the key to interaction between biophysical and human systems in the desertification process. There are several ecosystem attributes that can be associated with ecosystem maturity or complexity. In this context, a landscape's long-term capacity to retain, utilise and recycle local resources, and to buffer environmental changes, provides an objective basis for assessing its ecological functionality or condition.

Procedures based on this approach are relatively unsophisticated. They provide process-based indices that work by measuring the deviation in land condition status between any particular location and a reference one. They therefore demand few data, are low cost, not very prone to error propagation and well-suited to implementation on remotely sensed time–series data for application to large areas. One of such procedures is described here.

The main purpose of this project was to develop a land condition diagnostics methodology for large territories during a given time period. The target user profile was specified as a national or international institution building an information support system for a national desertification plan, or the UNCCD. The requirements were the following: i) an operational definition of land condition to be based on ecologically interpretable functions; ii) input data from already existing, generally available sources; iii) objective and repeatable procedures, leading to consistent results, even if found by different teams; and iv) results connecting explicit technical elements of land condition with lay understanding of desertification. The Iberian Peninsula was an ideal benchmark for this kind of approach because it combines a suite of land degradation syndromes that spread over wide climate gradients. Therefore, a second purpose was to apply and validate the proposed diagnostic method in that territory.

The Compact Oxford English Dictionary (Soanes & Stevenson, 2005) defines assessment as "evaluation or estimation", and monitoring as "to keep under observation, especially as to regulate, record or control". It is a common assumption in environmental science that monitoring can be built on assessments repeated over time. However, this is true only if the boundary conditions under which assessments are made remain constant. Climate is a boundary condition for land degradation. A changing climate may lead to different assessments even if land condition remains constant. In this work, the term assessment therefore refers to the synchronic estimation of land condition made over a relatively long time period, and the term monitoring refers to the diachronic observation of vegetation trends during the same period. This period should be long enough to collect a representative sample of vegetation performance. The combination of assessment and monitoring results is thus expected to yield meaningful land condition diagnostics.

2. Data

2.1. Study area, period and resolutions

The Iberian Peninsula is occupied by Portugal and Spain and extends over ca. 581,000 km^2 (Fig. 1). It is a rugged country criss-crossed by mountain ranges, many of their divides exceeding 2000 m.a.s.l. Enclosed by them, two large central plateaus and some tectonic basins define the main drainage network, which includes several rivers over 500 km long. Synoptic air masses create a NW to SE precipitation gradient, ranging from Atlantic humid climate zones on the north and west coasts, to pure Mediterranean on the east coast, the SE corner becoming the most arid zone of Europe. Indicative total annual precipitation in those extremes is 1600 and 140 mm/year, respectively. In addition, altitudinal temperature and precipitation gradients are associated with the main mountain ranges, creating a genuinely alpine climate in the Pyrenees, and well distributed extra-zonal belts of humidity elsewhere.

The Iberian Peninsula has a mosaic of land cover that includes significant areas of traditional and newly developed agriculture (49% of land), embedded in a matrix of natural and semi-natural vegetation (47% of land) (EEA, 2007). The former includes most of the present-day active desertification hot spots, whilst somewhat stabilized areas of inherited desertification associated with historical processes since the 15th century are well represented in the latter (Puigdefabregas & Mendizabal, 1998).

The period of analysis was intended to reflect a fair range of mid-term vegetation performance. Two practical conditions were formulated for its selection: i) only complete years to be used because the methodology is partly based on yearly summaries; and ii) only complete fields within the period (*i.e.* no gaps to be filled using statistical techniques) to meet both the applied and the methodological goals. Continuity of the time–series input was not strictly required as long as some yearly sequences were included and eventual gaps were grouped discretely in between. Years were defined in a manner similar to the meteorological convention reflecting the progression of hydrological and ecological seasons. Hydrological years encompass whole annual pulses that include the season of maximum soil moisture recharge and conclude with the season of maximum evapotranspiration (Glickman, 2000). Summer is relatively dry in most of the Iberian Peninsula, and first precipitations after the warm period usually fall in September. That reason is why the definition used for hydrological year throughout this study is from 1 September to 31 August.

The period of analysis was set as September 1989 through August 2000 to maximize the continuity of a vegetation and climate input data time–series whilst containing full hydrological years. A failure of the National Oceanic and Atmospheric Administration-Advanced Very High Resolution Radiometer (NOAA-AVHRR) at the end of 1994 and consequently missing input data for the derivation of Green Vegetation Fraction estimates resulted in a gap from August 1994 to January 1995 in the Green Vegetation Fraction time–series (see the next section). Because hydrological years are from September to August, data from both 1993–94 and 1994–95 had to be excluded. Therefore the number of whole years available for calculation was limited to 9. The climatic representativeness of these years is discussed below in the description of the climate archive. Nevertheless,

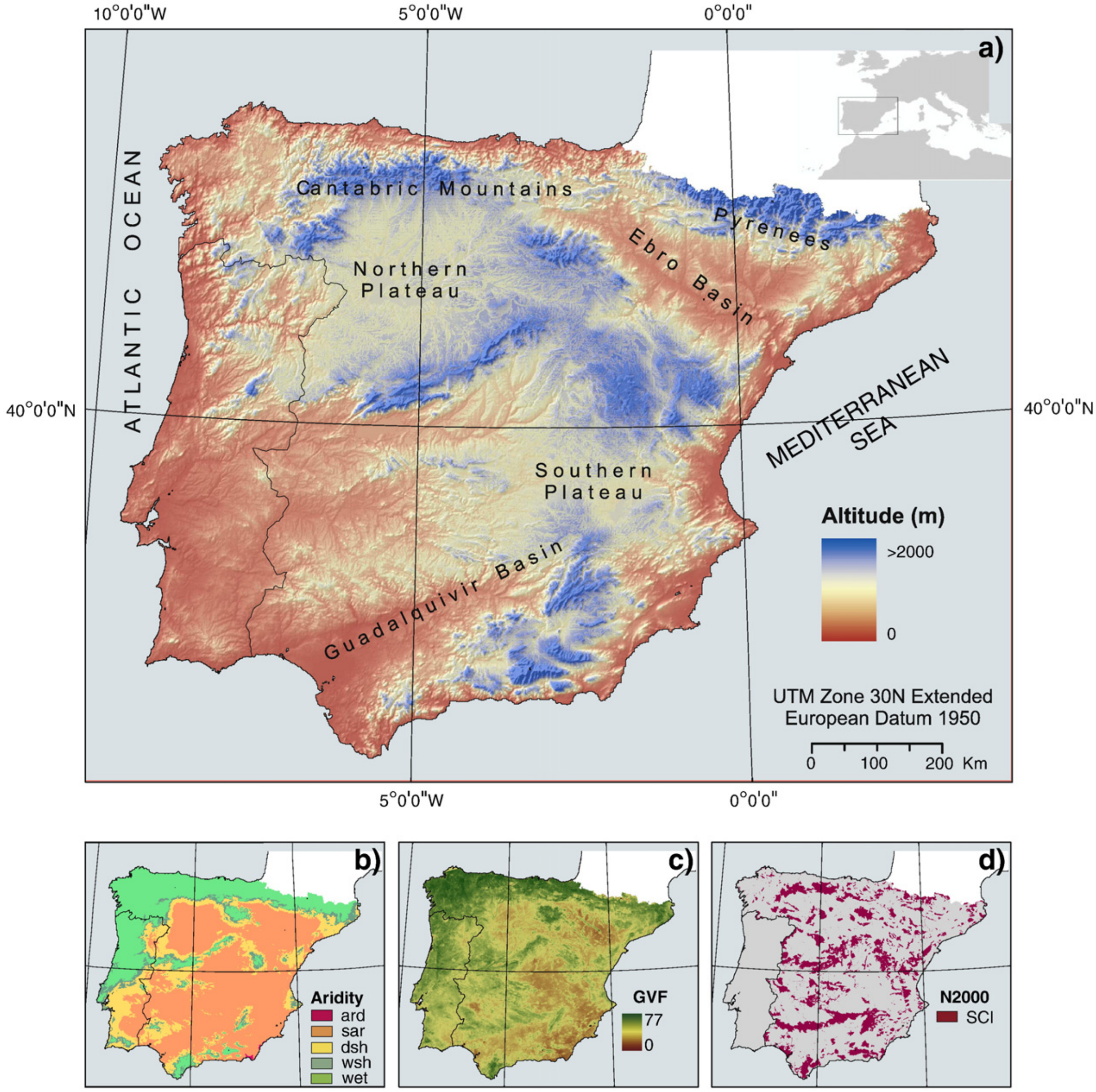

Fig. 1. Location and main geographic features of the Iberian Peninsula. *a)* Dominant relief patterns. *b)* Mean annual aridity, 1970–2000. UNEP classes are shown: arid (ARD), semi-arid (SAR), dry subhumid (DSH), wet subhumid (WSH), and wet (WET). *c)* Mean monthly vegetation density, 1989–2000. Green Vegetation Fraction shown is for a reference range of 1 to 100. *d)* Natura 2000 network in continental Spain (N2000), showing the distribution of Sites of Community Interest (SCI).

that period immediately follows the accession of Portugal and Spain to the European Economic Community in 1986, and their ratification of the Treaty of the European Union in 1992. The important changes required in both countries for their adaptation were reflected in their respective landscapes and therefore make a suitable framework for this study. The spatial and temporal resolutions necessary to capture those changes were 1000 m and 1 month respectively.

2.2. Green vegetation fraction

The approach presented needs a consistent and reliable time–series archive of a proxy for net primary productivity, such as vegetation surrogates from remote sensing data. Satellite data files with coarse geometric, but high temporal resolution, have been successfully employed for monitoring vegetation at a regional to global scale (Goetz et al., 2006; Tucker & Nicholson, 1999). Due to their high repetition rate these data provide better geographical coverage and temporal monitoring at the expense of spatial detail. At present, NOAA-AVHRR sensors provide the most comprehensive time–series of satellite measured surrogates for regional-scale surface conditions. The Normalized Difference Vegetation Index (NDVI) which is derived from the reflectance bands in the red and near infrared domain is a commonly accepted surrogate for vegetation cover, volume and vitality (e.g. Hielkema et al., 1987; Myneni et al., 1997; Tucker, 1979).

However, the NDVI is known to be influenced by soil and rock background (Price, 1993). Furthermore, it is sensitive to parameters such as atmospheric conditions due to aerosol properties and concentration and gaseous components, illumination, and the observation geometry, although this is supposedly partly eliminated by temporal maximum-value compositing of the data (Holben, 1986). Moreover, NDVI values are platform-dependant due to different spectral properties as well as the observation geometry, which complicates direct comparison of different sensors. The time–series archive used in this study is based on an unmixing approach making use of the well-known relationship of vegetation cover and surface temperature (T_s) and overcomes some of the limitations of the NDVI. It is based on the observation that under dry conditions the land surface temperature is inversely proportional to the amount of vegetation canopy cover and thus, the NDVI. This is due to a variety of factors including latent heat transfer through evapotranspiration, lower heat capacity and thermal inertia of vegetation compared to soil (Choudhury, 1989; Goward & Hope, 1989). Whilst on a small spatial scale the variation in vegetation species and soil classes may show high variability (Choudhury, 1989), a coarse geometric resolution shows the variation in surface temperature to be mainly caused by the vegetation fraction (Nemani et al., 1993). As there are local gradients related to altitude and exposure, temperature variation due to soil moisture, variable evaporation and evapotranspiration respectively, and remaining cloud artefacts (e.g. Lambin & Ehrlich, 1996; Sandholt et al., 2002), the feature space covers a triangle or trapezoid. This feature space, amongst others, has been used for deriving parameters like soil moisture content (e.g. Chuvieco et al., 2004; Nemani et al., 1993; Sandholt et al., 2002) and improving land cover classification (Lambin & Ehrlich, 1997). The linear unmixing approach derives Green Vegetation Fraction (GVF) from the triangle described above. The three vertices were derived and served as endmembers (full vegetation cover, dry soil and the 'cold' endmember representing the wet edge of the triangle) to unmix the feature space, thereby deriving GVF estimates (Stellmes et al., 2005; Weissteiner et al., 2008). The relationship between NDVI and T_s as described above develops only under stable atmospheric conditions, which are often present in semi-arid to arid areas, especially from spring to autumn. This approach, if such conditions were not given, would cause a bias in the estimated GVF. This constraint was accounted for by labeling data which did not fulfill the prerequisite as 'no data'. A detailed description of the technical implementation of the NDVI-Ts unmixing approach and derivation of GVF can be found in Stellmes et al. (2005) and Weissteiner et al. (2008) .

The NOAA AVHRR Archive used to derive the GVF is the "Mediterranean Extended Daily One Km AVHRR Data Set" (MEDOKADS) processed and distributed by the Free University of Berlin (Koslowsky, 1996). The MEDOKADS archive, which covers the period from 1989 to 2005 and is delivered as 10-day composites, comprises full resolution AVHRR channel data, NDVI, T_s and additional auxiliary data with a geometric resolution of about 1 km^2. This includes correction for sensor degradation and orbital drift effects that cause non-linear changes in the signal measured, as well as inter-calibration between the AVHRR/2 (NOAA 11 and NOAA 14) and AVHRR/3 (NOAA 16) sensors to prevent inhomogeneities in the time–series. A detailed description of data pre-processing may be found in Koslowsky (1996, 1998) and Friedrich and Koslowsky (2009). T_s is derived by the split window approach (Coll & Caselles, 1997; Coll et al., 1994) and normalized to the time of the local sun zenith plus 1 h and 42 min (Billing, 2007).

2.3. Climate archive

The climate time–series was extracted from an archive of monthly fields of mean maximum, mean and mean minimum air temperatures, and of total precipitation, generated for 1970–2000 for the Iberian Peninsula. This archive was interpolated from monthly summaries of georeferenced meteorological stations throughout the territory. Data for Portugal were downloaded from the Portuguese Water Resources Information System (http://snirh.pt, accessed January 2010) and complemented with series from AgriBase (Instituto Superior de Agronomia). Data for Spain were received from the Spanish State Agency of Meteorology. Gaps in the data fields were not filled in using statistical techniques. Station monthly summaries were computed only for stations with less than 5 days of missing data for the corresponding month, and interpolation for any given month was done using only stations with complete monthly summaries for the corresponding year. This resulted in variable networks of input stations for each surface, averaging 390 and 1877 points for temperature and precipitation surfaces respectively. In addition to these, approximately 10% of stations were reserved to enable cross-validation of the resulting surface.

Interpolation was done using thin-plate smoothing as implemented in ANUSPLIN (Hutchinson, 1995), an accepted technique to interpolate noisy multivariate data such as climatic variables, which has performed well in comparisons with other spatial interpolation methods (Hutchinson & Gessler, 1994; Jarvis & Stuart, 2001; Price et al., 2000). Latitude, longitude and altitude were specified as covariates for surface fitting. Temperature data were input raw, but precipitation was transformed to its square root during the interpolation to reduce skewness. The finest resolution of the thin plate smoothing surfaces was 3 arc-min, which was then registered to a 1000 m grid.

ANUSPLIN generates internal statistics that can be used to assess the quality of any fitted surface. The Square Root of the Mean Square Error (RTMSE) is a true estimator of the overall interpolation error. It is an absolute error in the same units as the original variable, but its ratio to the mean produces a relative error that is related to the predictive error of the interpolated surface. The degrees of freedom of the fitted surface are estimated through the signal. This parameter ranges from zero to the number of data points. Extremes are interpreted as failed spatial optimisation, from either under or overfitting, whilst around half the number of data points are considered appropriate (Price et al., 2000). Smooth transitions of signal from one month to the next are an indicator of the absence of systematic errors.

The signal to data point ratio was below 50% for temperature surfaces, with many values between 25% and 35%. This suggests limitations in the number of input data points which may have resulted in a loss of detail on the interpolated surfaces. This is not necessarily bad in terms of broad spatial patterns, but it does indicate that absolute predictions should be used with caution for fine microclimatic patterns. Mean maximum temperature signals are higher in summer, and mean minimum temperatures in winter, suggesting that the corresponding surfaces have at least partially succeeded in reflecting the complex patterns associated with the main relief mesoforms of the Iberian Peninsula. For precipitation, the signal to data point ratio was around 50%, which can be considered appropriate, although again, slightly on the low side because the admissible range only extends to 80% for this variable. Both temperature and precipitation series show uniform signals, which we interpret as a combination of the regional influences of climate. Smooth signal transitions between months reflect absence of bias.

The RTMSE was lower than 1 °C for all temperatures. The highest error in the mean maximum temperatures was 0.85 °C in the peak summer months, and mean minimum temperatures tended to concentrate the highest errors up to 0.73 °C in late summer and autumn. This probably adds to the previous interpretation concerning excessive smoothing of the respective surfaces. Absolute errors in precipitation did range from 4.41 mm in July to 12.82 mm in December, but the seasonal pattern of prediction errors was inverse and corresponded to 24% and 13% respectively.

Those results can be compared to others using the same interpolation technique. Yan et al. (2005) found RTMSEs of 0.42 to 0.83 °C for temperature surfaces, and 2 to 15 mm for precipitation surfaces interpolated for China, the latter with predictive errors ranging from 8 to 15%. And the findings of McKenney et al. (2006), working on Canada and United States, in general less than 1.5 °C for temperature, and 20 to 40% for precipitation, were also comparable.

Precipitation is a critical factor in this study, and the climate archive described above can provide some insight on the representativeness of the hydrological years in the 1989–2000 study period compared to the 1970–2000 reference period. The reference mean annual precipitation for the whole Iberian Peninsula is 705 mm. Compared to this, the driest years in the study period were 1998–99 (−28%), 1994–95 (−22%) and 1992–93 (−16%). The wettest years were 1995–96 (+28%), 1996–97 (+19%) and 1997–98 (+18%). These figures are only indicative, but in addition to the temporal variability they suggest, wide spatial variation is also associated with geographic gradients in this large, complex study area. This can be checked using individual maps of standardised residuals that show how much each location departs from its own mean during the reference period in any given year (Fig. 2). Two important conclusions may be drawn from those maps. First, a year that is not especially anomalous when precipitation is averaged over the whole area can still contain extreme deviations for individual locations. This occurred in 1989–90, which although it was only 7% wetter overall, still shows wide contrasts between locations that received much more or much less precipitation than their respective local average. And second, local extreme deviations do not necessarily occur in the overall driest years. For example, the driest year in a zone in the Northern Plateau was 1991–92 with precipitation of approximately 2.3 standard deviations below the local mean, and this is well below the precipitations received in 1998–99 and 1994–95, which were −1.2 and −0.6 standard deviations respectively.

The methodology applied in this work relies on the detection of vegetation performance under spatial and temporal ranges of climatic conditions. From this point of view, the main requirement for input data is the availability of a wide spectrum of variability in local precipitation rather than over the whole study area. The 9 years used in this work encompass a fair representation of such variability given the elements discussed in the preceding paragraph. Whilst it obviously would have been preferable to include 1993–94 and 1994–95, these years do not contain unique or extreme features that would prevent safe application of this approach.

2.4. Other complementary data

Three additional data sets were used in this work. The Global 30 Arc-Second Elevation Dataset (GTOPO30) (EROS, 1996) was used as an indirect source of topographic data for generation of the GVF and the climate archives. COoRdinate INformation on the Environment (CORINE) is a programme managed by the European Environment Agency (EEA, 2007) to provide consistent information on land cover and land cover changes in Europe at a spatial scale of 1:100,000 with a decadal temporal resolution. The editions for the reference years 1990 (abbreviated CLC1990) and 2000 (abbreviated CLC2000) were used in the selection of input data and as an external control to assist in interpreting the results. Finally, the subset for continental Spain (MARM, 2006) of the Natura 2000 network, established in 1992 (EEC, 1992) to provide a coherent structure for nature protection areas across the European Union, was used for validating the results. These data sets are described as necessary in the corresponding sections below.

3. Methods

The diagnostic procedure involves independent assessment and monitoring components that operate on the same database. On the one hand, the assessment component aims at quantifying the relative performance of each landscape location with respect to its reference potential conditions. Therefore, each cell is synchronically compared to all others over the period of analysis. On the other hand, the monitoring component aims at detecting evolution of every location over time, both because of its response to changing climate drivers

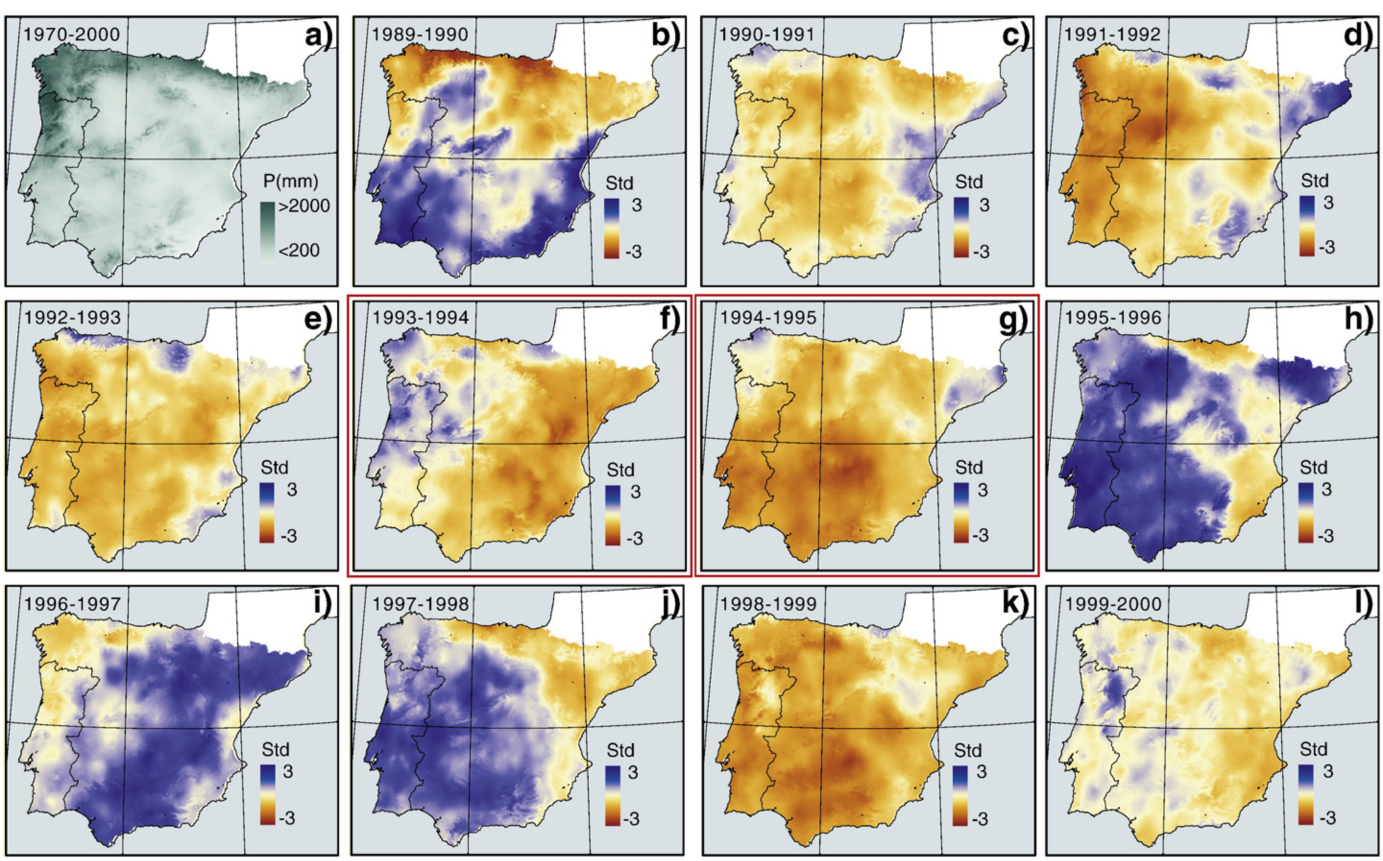

Fig. 2. Representativeness of the period of analysis (1989–2000) in the precipitation reference period (1970–2000) for the Iberian Peninsula. Hydrological years are shown. *a)* Mean annual precipitation in the reference period. *b)* through *l)* Yearly departures from the local reference mean in standard deviation units. Years 1993–1994 and 1994–1995 were excluded from the analysis and are highlighted in red.

and because of its internal ecological dynamics, so each cell is diachronically compared to itself over the same period. The results of assessment and monitoring then converge in the land condition diagnostics component through explicit rules that associate their respective categories.

The methods described in this article have been coded in R and are available in a software package called *r2dRue* which can be downloaded from any Comprehensive R Archive Network (CRAN) mirror at http://www.r-project.org/ (accessed January 2010).

3.1. Assessment

The reduction of plant biomass and Net Primary Productivity (NPP) below those of land not desertified under equivalent environmental conditions is proportional to land degradation undergone. This is an accepted perception of land condition and the most closely related to the UNCCD definition of desertification. Both biomass and NPP can be reasonably estimated using satellite-derived information (Tucker et al., 1986). Two ecosystem attributes are relevant for this assessment, annual average biomass and seasonal or inter-annual growth peaks. The first shows the ecosystem's long-term capacity to sustain biomass, whilst the second concerns its resilience for recovering from disturbances, in particular, rainfall fluctuation (Pickup, 1996). Annual average biomass and NPP may be expected to decrease along land degradation gradients whilst peak NPP (resilience) is at its maximum at intermediate degradation states (Pickup et al., 1994).

Rain Use Efficiency (RUE) was originally defined as the ratio of NPP to precipitation (P) over a given time period (LeHouerou, 1984), in which may be interpreted as proportional to the fraction of P released to the atmosphere through the vegetation cover. This ratio is a suitable descriptor of ecosystem condition because it can only be higher if the soil remains fully functional, and in this context it has been used to describe the sustainable carrying capacity in rangelands (Guevara et al., 1996). The NDVI has been used before as a suitable NPP proxy in the computation of RUE using satellite imagery as input data (Prince et al., 1998). Similarly, an appropriate integration over time of the Green Vegetation Fraction (GVF) described in the preceding section was used here as a surrogate NPP.

The formulation of RUE as a ratio implies that it can be expected to be higher for drier conditions if vegetation remains constant. If the RUE is computed over a large area with strong climatic gradients, as it is the case of the Iberian Peninsula, drylands often account for the highest values because of their very low P, which impedes direct comparison between locations under different climates. To avoid this, RUEs were plotted against an Aridity Index (AI) that was computed for the corresponding period as the ratio of potential evapotranspiration (PET) to P. This formulation is as simple as that of UNEP (1992), but the inverted terms expand the numerical scale of drylands without resorting to too many decimals, and also make it slightly more intuitive. The method of Hargreaves and Samani (1982) was used to compute PET using temperatures from the climate archive and extra-terrestrial solar radiation.

Only areas of rainfed natural and seminatural vegetation were used for the RUE vs. AI scatterplot, thereby ensuring that only climate driving forces were considered. A selection mask was therefore constructed to extract all the locations that maintain the same land cover allocation in CLC1990 and in CLC2000, and belong to one of the following categories: coniferous forest, mixed forest, natural grassland, moors and heathland, schlerophyllous vegetation, transitional woodland-shrub, beaches, dunes and sand plains, bare rock, sparsely vegetated areas, burnt areas, and glaciers and perpetual snow.

The upper and lower boundaries of this scatterplot are interpreted to convey the maximal and minimal vegetation performance, respectively, for a given aridity, which is a first step in detrending climate. These boundaries were computed by first identifying 10 classes of AI in the scatterplot using regular percentiles of the AI frequency distribution over the study area, then extracting the 95th (for the upper boundary) and 5th (for the lower boundary) percentiles of RUE for each of these classes, and finally, empirically fitting a function to each of the series of 10 RUE percentile and AI class median pairs. The boundary functions were selected by optimising statistical significance whilst maintaining the model as simple as possible under visual inspection. The resulting functions could then be spatially modelled using the AI layer as the independent variable, yielding two layers showing the expected maximum and minimum RUE, respectively, for every map location. Similar rescaling was done by Boer and Puigdefabregas (2003) for potential vegetation index prediction, and by Wessels et al. (2008) for generating relative NPP on a land capability gradient.

A relative RUE (rRUE) was then computed to yield a new layer showing the position of the RUE observed in each location within the range of its maximum and minimum potential. Values of rRUE should therefore range from 0 to 1, and it is assumed to reflect vegetation condition in terms of observed performance with respect to the minimum and maximum performances found empirically for that climate. Because these boundaries were derived from percentiles, and because they correspond to natural vegetation, some locations were expected to exceed that range. This was useful for detecting zonal anomalies. For example, the relatively large plant biomass in irrigated areas receiving water from topographical or technical sources not accounted for by this method, was properly detected as 'overperforming' (*i.e.* rRUE greater than 1) when processed by the procedure described.

The rRUE has two important properties. First, because the RUE boundary functions were fitted from RUE percentiles within each AI class, the expected maximum and minimum RUE for each location was already corrected for the local AI, and therefore the rRUE map was climatically detrended. And second, the RUE boundary functions are a natural benchmark for potential vegetation performance for a given aridity, against which any location can be compared whatever its land cover. In this sense the rRUE map was also detrended with respect to land uses under the assumption that undisturbed natural vegetation shows optimum performance in terms of RUE. This means that direct comparisons could be made between any two locations irrespective of their dominant climate or land use.

The detailed operation of the RUE concept used in this approach has been omitted from the above description for the sake of simplicity. Beyond that, the definition of the period over which observed RUE is computed targets different responses of the vegetation cover. Long-term RUE computed over full years may reflect average biomass and can be compared to ecological maturity in the framework of the ecosystemic approach described in the Introduction. However, short-term RUE computed using only antecedent precipitation would rather reflect immediate response capacity and can be interpreted in terms of productivity or resilience. This is why two implementations were used here. In the first one, an overall mean observed RUE (RUE_{OBS_me}) was computed for the full period ($n = 9$ years) by averaging annual observed RUE. Each of these was computed over a hydrological year as the mean of 12 monthly *GVFs* divided by the sum of 12 *P*. For month *i* in year *j*:

$$RUE_{OBS\_me} = \frac{1}{n} \cdot \sum_{j=1}^{n} \left(\frac{\frac{1}{12} \cdot \left(\sum_{i=1}^{12} GVF_{j,i} \right)}{\sum_{i=1}^{12} P_{j,i}} \right). \tag{1}$$

The corresponding overall mean aridity index (AI_{OBS_me}) is computed accordingly:

$$AI_{OBS\_me} = \frac{1}{n} \cdot \sum_{j=1}^{n} \left(\frac{\sum_{i=1}^{12} PET_{j,i}}{\sum_{i=1}^{12} P_{j,i}} \right). \tag{2}$$

In the second implementation, an extreme observed RUE (RUE_{OBS_ex}) was computed again for the full period. In this case, the maximum GVF in each cell (GVF_t) was selected from among the monthly data available in the time–series, and this was then divided by the sum of P over the six months preceding the month (t) when that maximum was detected. The result was a composite layer, as the maximum GVF appeared in each cell at a different time:

$$RUE_{OBS\_ex} = \frac{GVF_t}{\sum_{i=1}^{6} P_{t-i}}. \tag{3}$$

An associated aridity index (AI_{OBS_ex}) was also computed for this particular six month period:

$$AI_{OBS\_ex} = \frac{\sum_{i=1}^{6} PET_{t-i}}{\sum_{i=1}^{6} P_{t-i}}. \tag{4}$$

Both implementations of observed RUE (RUE_{OBS_me} and RUE_{OBS_ex}) were processed following the steps described above to find their respective boundary functions and relative values ($rRUE_{\mathrm{me}}$ and $rRUE_{\mathrm{ex}}$ respectively; computation is shown for the first):

$$rRUE_{me} = \frac{RUE_{OBS\_me} - RUE_{EXP\_me\_P05}}{RUE_{EXP\_me\_P95} - RUE_{EXP\_me\_P05}} \tag{5}$$

where the subscripts $_{P05}$ and $_{P95}$ refer respectively to the expected minimum and maximum RUE corresponding to the aridity in each location, given by the boundary functions (subscript $_{EXP}$) as explained above in the scatterplot procedure.

The distinction between mean and extreme RUE is considered potentially useful, as they may be different for vegetation types depending on the type of cover (e.g. annual plants, irrigated areas, forests, etc.).

The use of the mean of 12 monthly GVFs to compute annual observed RUE (Eq. 1) departs from the convention of approaching NPP through a summatory of individual values of the selected vegetation index. It was done this way to facilitate frequent follow-up of the process by querying input and output layers in search of spatial and seasonal errors, as all of them would share the order of magnitude. This consideration also applies to the monitoring component of our approach (see the following Section). The division by a constant term of 12 has no effect on the relative spatial or temporal variations in cumulative GVF, as it is computed anyway. The study by Bai et al. (2005), which can be taken as an example of experimental confirmation of this, reports two relevant findings. They compared NDVI-based indicators of land degradation derived from the Global Inventory Modelling and Mapping Studies (GIMMS) dataset, both between them and with RUE estimates. On one hand, they found a very high correlation ($r^2 = 1$) between the mean and sum of NDVI values, and also similar temporal trends, and were therefore considered alternates. On the other, highly significant correlations ($p < 0.001$) were detected between RUE estimates computed using NPP data derived from a carbon exchange model and those derived from the NDVI, again considering them alternates. In our assessment component, computation of relative RUE from the boundary functions (Eq. 5) further transforms the numerical scale of observed RUE into a common reference range of 0 to 1, whatever the scale of the original estimate.

3.2. Monitoring

Changes in plant biomass over time make an accepted indicator of trends in land condition. A gradual depletion of biomass is generally interpreted as ongoing degradation, and reciprocally, an increase is interpreted as improvement in a responsive ecosystem. It is important to remark that such a basic understanding refers only to rates of change, and is therefore independent of the bulk ecosystem biomass, which was an issue in the assessment explained above.

In this case, RUE could not be used to monitor vegetation trends during the period for two reasons: the bias associated with the use of precipitation both as for the computation of RUE and as a tested predictor (Hein & de Ridder, 2006), and the relative nature of detrending in the computation of rRUE. Raw GVF was therefore used instead.

When using low resolution Earth Observation in a large territory, trends in plant biomass can be simplified as driven by two factors: climate and internal ecological dynamics. Only the second is relevant to land condition, as it can be related to a secondary succession. Therefore, monitoring of plant biomass along relatively long periods requires accounting for possible effects of a drifting climate. Failing to do so might result, for example, in erroneously interpreting a piece of healthy land as degrading, when biomass is declining simply because of decreasing precipitation.

Multiple stepwise regression was carried out to isolate the effects of time and climate on the green biomass in each cell. As resolution was yearly, each cell entered a regression with 9 points. In each of these, the dependent variable was the yearly mean of 12 monthly GVF values, and the two predictors were the year sequence number and the yearly AI computed using the corresponding 12 months. The purpose of this analysis was to detect partial contributions of the two predictors, creating significant trends in the dependent variable (not to make predictions of the latter for given values of the former). Accordingly, the regressions were used in a standardised form, in which the partial regression coefficients are expressed in standard deviation units, rather than in the original units of each variable (such coefficients are also known as beta coefficients). This enabled direct comparison of the relative strength of time and aridity for imprinting a change of one standard deviation on the GVF. All the procedures followed the formulations in Sokal and Rohlf (1995).

When there is a correlation between the two predictors, as might be the case with time and aridity, the overall significance of a multiple regression may not be indicative of their individual effects. Hence, a second independent variable must be incorporated in the regression model only as long as the additional increment of determination it produces is significant. We proceeded to deal with this requirement in the following way. First, the coefficient of multiple determination was computed using both time and aridity as predictors ($R^2_{GVF\_1,2}$), and its overall significance was tested to a threshold of $\alpha = 0.10$. For significant cases, the second variable in the multiple regression was the one with the lowest simple correlation coefficient with GVF (r_{GVF_2}). Then the increment in the determination due to the second variable over the determination using the first variable alone was tested by comparing the observed statistic F_s:

$$F_s = \frac{R^2_{GVF\_1,2} - r^2_{GVF\_2}}{\left(1 - R^2_{GVF\_1,2}\right) / (n-3)} \tag{6}$$

with the expected F-distribution at a threshold of $\alpha = 0.10$ for 1 and $n-3$ degrees of freedom ($F\alpha_{=0.10[1,n-3]}$), where n is the number of data points. Eq. (6) shows a simplified formulation of this step using only two independent variables. The parameters used are commonly provided by statistical software packages and are not described here.

If this was also significant, a multiple regression would be accepted and the respective standard partial regression coefficients would be used to quantify the effects of both time and aridity. Otherwise, simple regressions were explored for each variable through the significance of its respective correlation coefficient. For significance at $\alpha = 0.10$, the corresponding correlation coefficient was used as a standard single regression coefficient because it is by definition the slope of a regression in standard deviation units. If no significance was found in any of the above tests, GVF was assumed not to be affected by either time or aridity.

3.3. Land condition

The assessment and monitoring tasks described in the sections above aimed at yielding four basic maps quantitatively reflecting four different land condition components, overall relative biomass (in terms of $rRUE_{me}$), overall relative productivity (in terms of $rRUE_{ex}$), biomass response to aridity and biomass response to time (in terms of their respective standard partial or simple regression coefficients) as primary products to be used as elementary information in land management or additional modelling. Further to that, a higher level understanding of land condition in the study area was gained by simplifying their respective individual information to binary attributes, which were subsequently combined to produce the legend categories on the land condition map.

In the assessment task, both implementations of relative RUE were ranked and defined in turn by boundary functions associated with percentiles. Hence, by definition, there will always be locations outside these ranks. A broad summary of land performance in reaching and maintaining an optimal biomass was therefore achieved by classifying each of the relative RUEs into three basic categories: below range (lower than 0), within range (0 to 1), and above range (greater than 1).

The obvious criterion for reclassifying monitoring results was whether biomass had a positive, negative, or non significant trend over time or aridity.

Each of the four basic maps was thus reclassified into three categories. Hence a merely combinatorial approach could yield up to 81 classes of land condition. That undesirable complexity was overcome in a meaningful manner by composing the legend on the final map on two hierarchical levels. The higher one corresponded to assessment classes and indicated the state of land, whilst the lower level was found using monitoring attributes to indicate land trends over time. The final definition of this legend depends somewhat upon the final results and required some adaptation in consistency with the concept of degradation. Therefore, legend classes and subclasses are precisely described in Section 4 of this article.

3.4. Validation

Land condition is a rather abstract concept that was applied in this work on a decadal time span. This, and the relatively coarse spatial resolution, prevented directly testing the results against field observations. In practical terms, the validation of an approach like the one presented here should meet two basic conditions, the spatial scale of validation data should be comparable to that of the final land condition map, and the information conveyed by this validation dataset should be interpretable at a level of abstraction equivalent to that of the land condition concept.

For validation, we employed a set of spatially distributed data for which a landscape condition could be assumed, the Natura 2000 network Sites of Community Interest (SCI). This European conservation network was set up in 1992, and SCI land condition can be taken as representative of the study period. The SCI network in continental Spain includes 853 designated sites, which accounts for 22% of the territory (Fig. 1*d*). Because their designation aimed at representing relevant natural species and habitats, it contains a variety of landscapes. And because the conservation of those habitats is the network's main purpose, conditions in the portion of territory included in it can be assumed to be favourable for vegetation to thrive. The Natura 2000 network includes natural and semi-natural landscapes where traditional management is allowed, but land uses leading to the over-exploitation of natural resources are normally excluded. Whilst traditional management does not necessarily mean sustainable management, in general terms, land condition within the Natura 2000 space can be expected to be good, and our validation is based on this working hypothesis.

Validation was done using mainland Spain (about 85% of the Iberian Peninsula) as the test area, at a working resolution of 1 km. The concrete purpose was to find out whether there is an association between landscape conservation, in terms of membership in the Natura 2000 network, and land condition, in terms of the final output map classes. A sample of 45,731 cells, roughly representing 10% of the whole test area, was extracted from the original maps using a stratified-random design. This dataset was assumed to be made up of independent samples, including absence of spatial autocorrelation.

A chi-square test was done for the null hypothesis that there is no association between conservation status and land condition. The sample dataset described above was divided in two groups defined by membership to Natura 2000, and the proportions of cases belonging to the land condition classes were compared. The test evaluates whether or not the differences observed in these proportions significantly exceed those that could be expected by mere chance (Siegel & Castellan, 1988). To do this, an observed statistic (χ^2) was computed from the sample data and its probability was then found in the chi-square distribution. If the alternative hypothesis was accepted, the residual (*i.e.* observed minus expected) frequencies would be used to interpret the sense and meaning of that association.

It could be argued that the use of conservation reserves to validate land condition is incomplete because only better condition classes are tested. This is only partly true. In fact, these classes are expected to be around the upper boundary of observed RUE over aridity. It is therefore this boundary that is really validated, that is, the maximum reference condition against which all the locations of a given degree of aridity will be assessed.

4. Results

4.1. Assessment

Scatterplots and boundary functions of observed RUE over aridity are shown in Fig. 3*a* and *b*. In the absence of a theoretical criterion for a particular model for those functions, several models were tested and the most significant fit was selected in each case (Table 1). It is worthy of note that the increase in mean observed RUE becomes gentler with increasing aridity, which is confirmed by the inverse model selected for both boundary functions. On the contrary, extreme observed RUE increases with aridity confined within an upper power and a lower quadratic boundary function.

The resulting relative RUE maps are shown in Fig. 3*c* and *d*. The distribution of values does not reflect any apparent climatic bias in either map. Patches with high mean relative RUE are scattered around the study area, some of them associated with the banks of major rivers. Extreme relative RUE is somewhat spotty because it is a composite map. In spite of this, zones with high values are consistently detected surrounding the Ebro basin and in the Northern Plateau.

As expected, the geographic patterns of mean and extreme relative RUE do not generally match. This is clearer in the contingency table shown in Table 2. The dominant combination is by far the one with both estimators of relative RUE within the 0 to 1 range. This is a predictable outcome of the method, as the respective boundary functions were fitted to percentiles that enclose the majority of the data points. That middle combination should be taken as a baseline condition where acceptable oscillation is associated with land management and cover.

Wherever any of the relative RUE estimators exceeds the stated range, it should generally be interpreted as a deviation associated

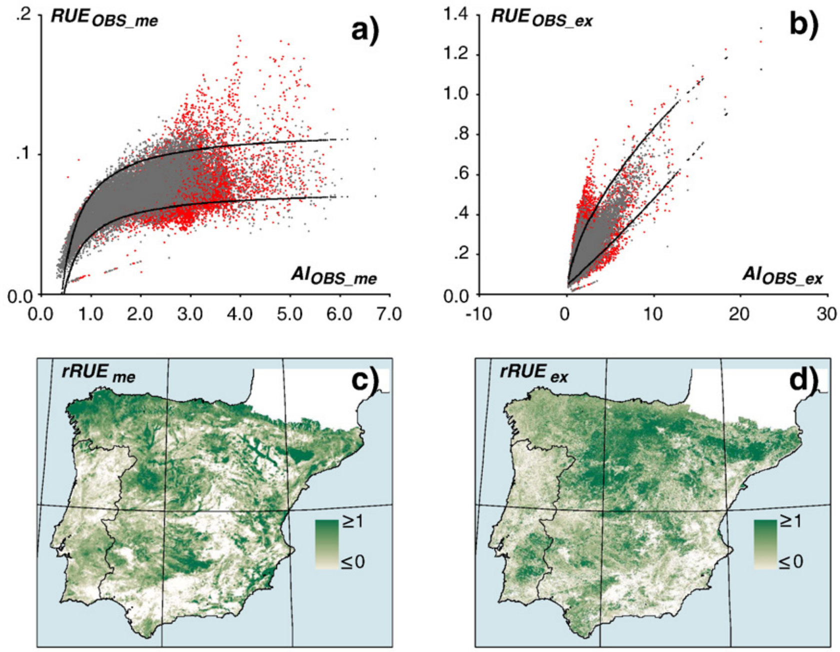

Fig. 3. Rain Use Efficiency (RUE) in the Iberian Peninsula (1989–2000), computed as the inter-annual mean of each location over the full period (*a* and *c*), and for the six-month period preceding the time when maximum vegetation density was detected at each location (*b* and *d*). Empirical boundary functions fitted to the scatterplots of observed RUE over aridity (*a:* RUE_{OBS_me} vs. AI_{OBS_me} ; *b:* RUE_{OBS_ex} vs. AI_{OBS_ex}) define the potential limits of expected RUE for any aridity level. Only locations of rainfed natural and seminatural vegetation (grey dots) were used to fit the boundaries, whilst irrigated crops and other surfaces not responding to climate (red dots) are shown for information. Relative RUE (*c*: $rRUE_{me}$; *d*: $rRUE_{ex}$) is then computed for each location as the position of its observed RUE within the referred limits.

with under or overperforming vegetation in the respective component. Apart from the baseline combination, mean and extreme relative RUEs assess land condition similarly in 3.41% of the locations. However, the sum of combinations where both estimators differ in their results accounts for 24.19% of the study area, which suggests that they convey different information. In two of these combinations, mean and extreme relative RUE yield completely opposite results. They represent 0.18% of the study area and are considered as uninterpretable anomalies.

The assessment map legend (Fig. 4) was made using the combinations in Table 2 as follows. Locations where the mean and extreme relative RUE are below range were considered to be consistently underperforming in terms of the basic interpretation of RUE, and were reclassified to unusually degraded land. If one of the estimators was below range whilst the other was within range, the corresponding locations were reclassified to degraded land. The baseline combination where both mean and extreme relative RUE were within range was considered to represent land in good condition. And any other combination where either of the estimators was above range (except the two anomalies) was considered to be overperforming and was therefore reclassified as land in unusually good condition.

4.2. Monitoring

The significance of effects of time and aridity on the GVF is shown in Fig. 5*a*. Approximately 5% of the Iberian Peninsula showed a significant response to both time and aridity in the regression procedure. This is comparatively low with respect to the proportion of individual responses to any single predictor (40%), and is likely to be associated with the fact that a stepwise approach is more conservative than a simultaneous multiple one. 55% of the study area did not show any linear response to either of the predictors.

Fig. 5*b* and *c* shows the effect of aridity and of time, respectively, on GVF over the study period. Correlation coefficients and standard partial regression coefficients have been combined on the same map for each of the predictors depending on whether a significant single or multiple regression was fitted at every location. They are mere reclassifications by the coefficients sign, however they are useful in displaying the geographic distribution of effects. Aridity produces large patches of negative effects (*i.e.* depletion of GVF with increasing aridity) which are well distributed over the areas of Mediterranean influence, whilst positive effects are confined to northern mountain ranges such as the Pyrenees and Cantabric mountains. The effect of time on GVF is positive in the vast majority of locations where a significant relationship was detected, and only a few small spots scattered over the Mediterranean area show GVF depletion over time.

The monitoring map was then made by using significance categories (Fig. 5*a*) as a mask, and the sign of effects (Fig. 5*b* and *c*) to identify trends. Its legend includes all the possible combinations of single or multiple and positive or negative effects, and is further detailed in Table 3. In general, effects of aridity should be attributed to climatic variation within the study period, and effects of time to the overall trend

Table 1
Parameters of the boundary functions fitted to the 5th and 95th percentiles of observed RUE vs. aridity. $RUE_{EXP\_me\_P05}$: mean expected RUE at 5th percentile; $RUE_{EXP\_me\_P95}$: mean expected RUE at 95th percentile; $RUE_{EXP\_ex\_P05}$: extreme expected RUE at 5th percentile; $RUE_{EXP\_ex\_P95}$: extreme expected RUE at 95th percentile; AI_{OBS_me}: observed mean aridity index; AI_{OBS_ex} observed aridity index computed for the six-month period preceding the month when maximum vegetation density was detected.

Y	*X*	Model	b_0	b_1	b_2	*p*
$RUE_{EXP\_me\_P05}$	AI_{OBS_me}	$Y=b_0+(b_1/X)$	0.076	−0.033		<10E−4
$RUE_{EXP\_me\_P95}$	AI_{OBS_me}	$Y=b_0+(b_1/X)$	0.119	−0.047		<10E−4
$RUE_{EXP\_ex\_P05}$	AI_{OBS_ex}	$Y=b_0+b_1\cdot X+b_2\cdot X^2$	0.047	0.040	4E−4	<10E−4
$RUE_{EXP\_ex\_P95}$	AI_{OBS_ex}	$Y=b_0\,X^{b1}$	0.218	0.584		<10E−4

Table 2
Contingency table of mean vs. extreme relative RUE ($rRUE_{me}$ and $rRUE_{ex}$ respectively) after their classification in three basic intervals. Table entries are: assessment class allocation and total area [km^2 (%)].

	$rRUE_{me}<0$	$0\leq rRUE_{me}\leq 1$	$1<rRUE_{me}$	Total
$rRUE_{ex}<0$	*Unus. Degr.* 14,261(2.45)	*Degr.* 24,914 (4.28)	*Anomaly* 340 (0.06)	39,515 (6.79)
$0\leq rRUE_{ex}\leq 1$	*Degr.* 391,47 (6.73)	*Good* 421,094 (72.40)	*Unus. Good* 28,612 (4.92)	488,853 (84.05)
$1<rRUE_{ex}$	*Anomaly* 688 (0.12)	*Unus. Good* 47,004 (8.08)	*Unus. Good* 5566 (0.96)	53,258 (9.16)
Total	54,096 (9.30)	493,012 (84.76)	34,518 (5.93)	581,626 (100.00)

of GVF excluding climatic responses. This, in turn, is associated with land management or with intrinsic ecological processes occurring in the vegetation cover. Single effects interpretation is straightforward. Both explanations are still true for multiple effects, but their importance as drivers must be in the order of the local standard partial regression coefficient strength. Table 3 reports averages, but some general interpretations are possible. For example, where GVF increases in time and decreases with aridity, the mean of the latter coefficient is stronger. It follows that arid spells or a sustained increase in aridity over the whole study period have played a dominant role in GVF depletion, but this is still accumulating over time, albeit at a slow positive rate. This is the only case in Table 3 where the mean of an aridity coefficient is higher than its equivalent time coefficient.

4.3. Land condition

The assessment map reflects bulk GVF whilst the monitoring map conveys rates of change in GVF. Land condition categories were therefore derived from a hierarchical combination of assessment classes and monitoring subclasses. In this exercise, subclass labels were adapted to the contents of the main classes in the following ways:

- Land with no significant trends in either time or aridity was called *static* whatever its condition.
- Land with negative trends over time was called *degrading* whatever its condition.
- Land with no trends over time but with a significant response to aridity in any sense was considered to react to variation in climate. It was called *fluctuating* if the condition was degraded, and *resilient* if condition was good.
- Land with positive trends in time was called *recovering* if it was degraded, and *improving* if it was already in good condition.

Table 4 shows the details and extent of the classes and subclasses so defined, and the corresponding land condition map is shown in Fig. 6. It is important to stress that such names and definitions aim

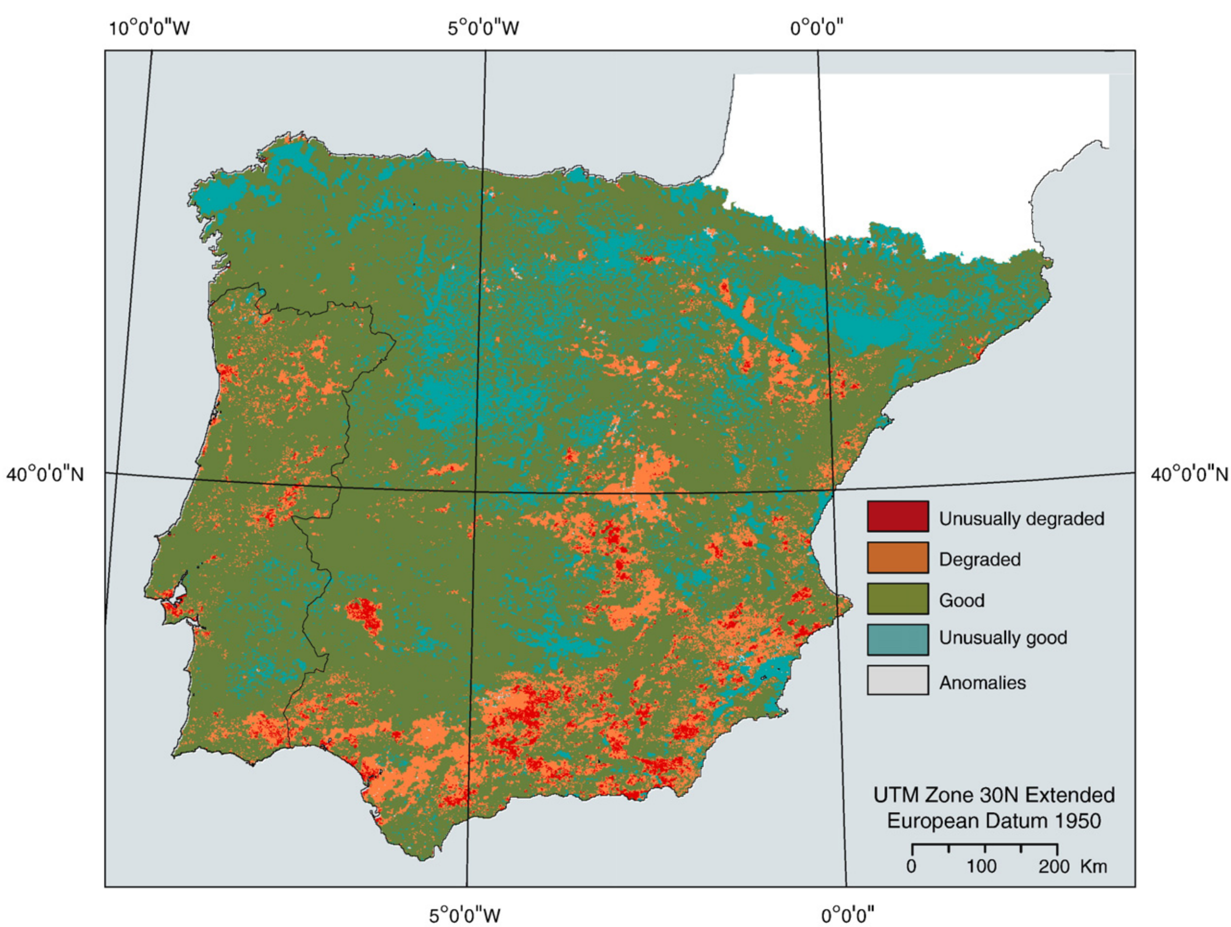

Fig. 4. Assessment of land condition in the Iberian Peninsula (1989–2000). Legend is consistent with Table 2.

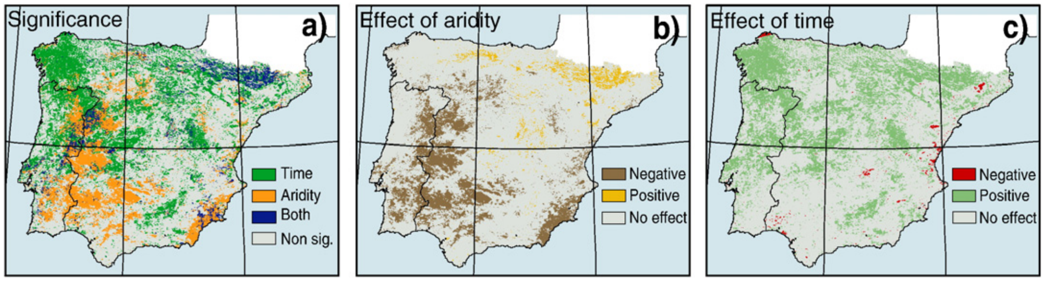

Fig. 5. Linear effects of time and aridity on Green Vegetation Fraction in the Iberian Peninsula (1989–2000). *a*) Significance at the threshold of $\alpha = 0.1$. Response to both predictors refers to the coefficient of multiple determination and to the increment of determination associated with the second predictor. Response to a single predictor refers to the coefficient of correlation. *b*) Effect of aridity in terms of the sign of the standard partial or single regression coefficient. *c*) Effect of time in terms of the sign of the standard partial or single regression coefficient.

only at using a consistent nomenclature at a level sufficiently generalized to deal with a large territory like the Iberian Peninsula. Hence the land condition map should be taken as a kind of broad stratification, and real diagnostics of particular locations should be made using the numerical values in the assessment and monitoring procedures.

Nevertheless, some conclusions can be drawn at that general level. First, the amount of land in any condition that is degrading is relatively small and accounts for only 0.85% of the Iberian Peninsula. This degrading trend is relatively more frequent in already degraded land, although it is still marginal even in that class. Truly static land showing no trend during the period is absolutely dominant whatever the condition (54.67% of the area), but its relative importance in assessment classes increases as condition deteriorates. It amounts to 75% of unusually degraded land, but to only 55% of land in unusually good condition. A contrary pattern is detected in land with a positive trend in time as its relative importance increases as condition improves, from unusually degraded land where recovering trends account for 12% of extension, to land in unusually good condition, where 33% of the locations are improving. On a whole, positive trends in time are detected in 29.13% of the territory. Fluctuating or resilient trends have a maximum share in intermediate condition classes, where they reach 17% of degraded land and 16% of land in good condition respectively.

The spatial distribution of the land condition categories follows some apparent geographic patterns too. Land in good condition with clearly identifiable resilient or improving patches prevails south of the Pyrenees and in the western third of Iberia, whilst static patches form a matrix in the rest of the territory in which other categories are embedded. Unusually degraded, or degraded, static land is frequent in continental areas with Mediterranean influence and in the south, whilst fluctuating trends can be found in a large compact zone in the southeast. Finally, static and resilient land in unusually good condition are found on both central plateaus and in the northeast, whilst improving patches can be seen tracing the banks of large rivers (notably the Ebro in the northeast) and in a more compact area in the northwest corner of the Peninsula.

4.4. Validation

The chi-square test has two requirements if any of the variables has more than 2 classes: less than 20% of expected frequencies lower than 5, and no expected frequency lower than 1. As reported in the preceding Section, the land condition categories *Unusually degraded – Degrading*, and *Degraded – Degrading* are comparatively small and their further subdivision by conservation status would not meet the latter requirement. Therefore, and for validation purposes only, they were grouped with the *static* trend in their respective assessment class. Table 5 shows the results of the chi-square test of the regrouped land condition categories and conservation status. The test was significant, and therefore the alternative hypothesis that there is an association between them could be accepted. Residuals express the difference between observed and expected counts when there is no relationship between variables. Their interpretation enables further details to be inferred on the meaning of that association.

Unusually degraded, or degraded land, are negatively associated with SCI (*i.e.* classes underrepresented in conservation reserves) whatever the trends. Land in good, or unusually good condition, is also negatively associated if it is degrading. On the contrary, SCI are positively associated with land in good condition for most of the remaining trend subclasses. SCI are also positively associated with static land in unusually good condition. These basic facts are consistent with

Table 3
Relative strength of effects of time (t) and aridity (AI) on GVF (rate of change in standard deviations of GVF per one standard deviation of each predictor). Combinations correspond to areas of significant effects shown in Fig. 5. Table entries are mean and 95% confidence interval of either coefficient of correlation (r) for single effects, or standard partial regression coefficient (b') for multiple effects; and area [km^2 (%)].

Effect of time	Effect of aridity			Total
	−	0	+	
−	*Degrading* $b'_t = -0.628 \pm 0.013$ $b'_{AI} = -0.597 \pm 0.013$ 415 (0.07)	*Degrading* $r_t = -0.716 \pm 0.003$ n.a. 4397 (0.76)	*Degrading* $b'_t = -0.817 \pm 0.025$ $b'_{AI} = 0.500 \pm 0.028$ 201 (0.03)	5013 (0.86)
0	*Fluct. or Resilient* n.a. $r_{AI} = -0.700 \pm 0.001$ 82,139 (14.12)	*Static* n.a. n.a. 318,553 (54.77)	*Fluct. or Resilient* n.a. $r_{AI} = 0.667 \pm 0.002$ 6256 (1.08)	406,948 (69.97)
+	*Recover. or Improving* $b'_t = 0.633 \pm 0.003$ $b'_{AI} = -0.670 \pm 0.004$ 14,447 (2.48)	*Recover. or Improving* $r_t = 0.721 \pm 0.000$ n.a. 140,971 (24.24)	*Recover. or Improving* $b'_t = 0.837 \pm 0.003$ $b'_{AI} = 0.562 \pm 0.002$ 14,247 (2.45)	169,665 (29.17)
Total	97,001 (16.68)	463,921 (79.76)	20,704 (3.56)	581,626

Table 4
Land condition categories in terms of the assessment and monitoring estimators. Area in the Iberian Peninsula (1989–2000) for each class and subclass.

Assessment	Monitoring: Effect of time	Monitoring: Effect of aridity	Land condition	Area [km^2 (%)]
			Unusually degraded	**14261 (2.45)**
$rRUE_{me}<0$ AND $rRUE_{ex}<0$	−	−/0/+	*Degrading*	119 (0.02)
	0	0	*Static*	10651 (1.83)
	0	−/+	*Fluctuating*	1717 (0.30)
	+	−/0/+	*Recovering*	1774 (0.31)
			Degraded	**64061 (11.01)**
($rRUE_{me}<0$ AND $0\leq rRUE_{ex}\leq 1$	−	−/0/+	*Degrading*	832 (0.14)
OR	0	0	*Static*	42336 (7.28)
($0\leq rRUE_{me}\leq 1$ AND $rRUE_{ex}<0$)	0	−/+	*Fluctuating*	10910 (1.88)
	+	−/0/+	*Recovering*	9983 (1.72)
			Good	**421094 (72.40)**
$0\leq rRUE_{me}\leq 1$ AND $0\leq rRUE_{ex}\leq 1$	−	−/0/+	*Degrading*	3211 (0.55)
	0	0	*Static*	220019 (37.83)
	0	−/+	*Resilient*	66627 (11.46)
	+	−/0/+	*Improving*	131237 (22.56)
			Unusually good	**81182 (13.96)**
($1<rRUE_{me}$ AND $0\leq rRUE_{ex}\leq 1$)	−	−/0/+	*Degrading*	841 (0.14)
OR	0	0	*Static*	44953 (7.73)
($0\leq rRUE_{me}\leq 1$ AND $1<rRUE_{ex}$)	0	−/+	*Resilient*	8969 (1.54)
OR	+	−/0/+	*Improving*	26419 (4.54)
($1<rRUE_{me}$ AND $1<rRUE_{ex}$)				
			Anomaly	**1028 (0.18)**
($rRUE_{me}<0$ AND $1<rRUE_{ex}$) OR ($1<rRUE_{me}$ AND $rRUE_{ex}<0$)				1028 (0.18)

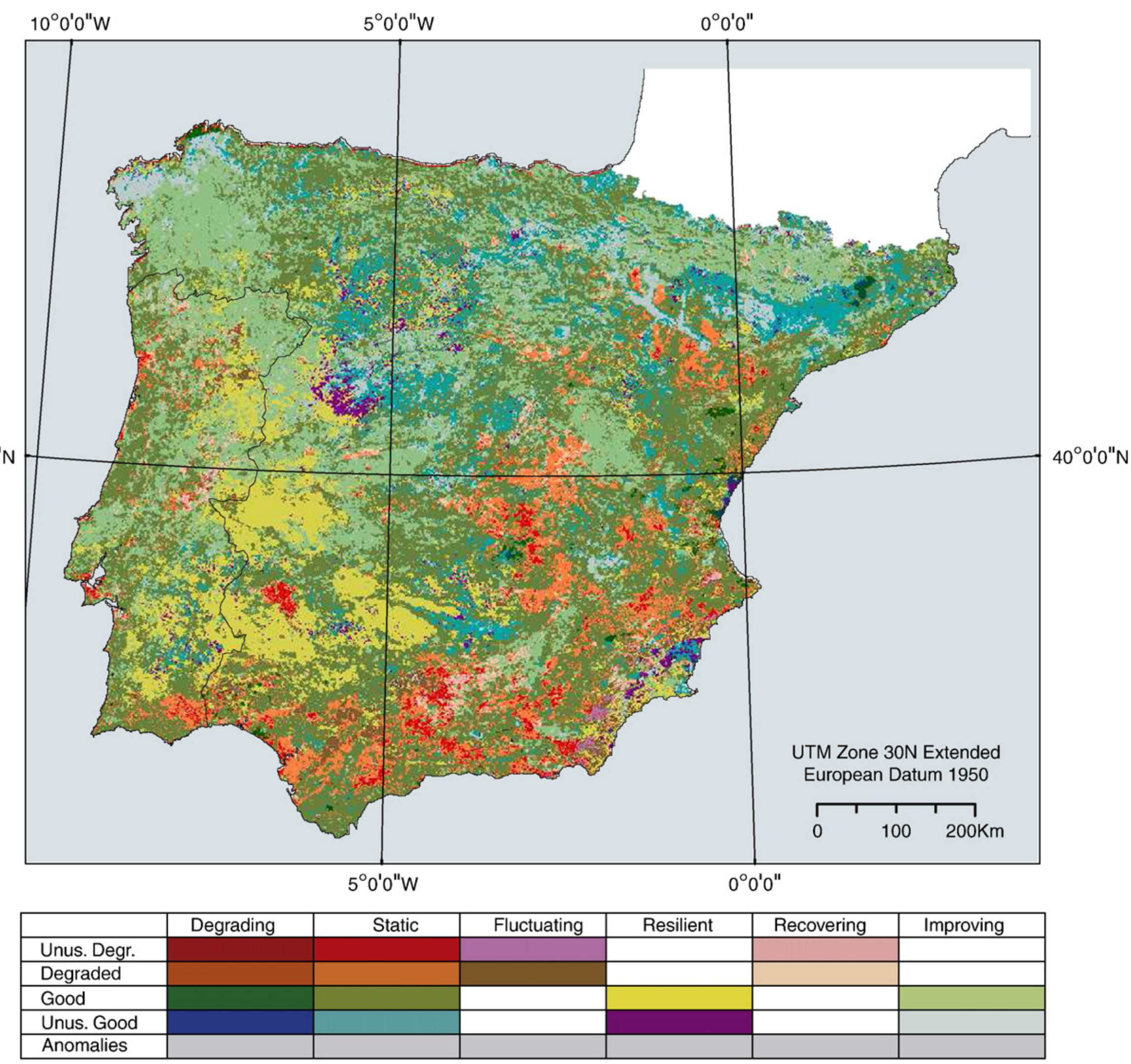

Fig. 6. Land condition map for the Iberian Peninsula (1989–2000). Legend is consistent with Table 4.

Table 5
Results of a chi-square test of regrouped land condition categories and membership to the Natura 2000 network Sites of Community Interest (SCI) in continental Spain. Table entries are residual (observed minus expected) counts ($\chi^2=427$, *df*: 13, $n=45731$, $p<10\text{E}-4$). See explanation in the text.

Land condition		Conservation status		Total
		Non-SCI	*SCI*	
Unusually degraded	Degrading or Static	45	−45	710
	Fluctuating	20	−20	127
	Recovering	11	−11	140
Degraded	Degrading or Static	252	−252	3558
	Fluctuating	112	−112	847
	Recovering	28	−28	622
Good Condition	Degrading	8	−8	233
	Static	−392	392	17,599
	Resilient	89	−89	4713
	Improving	−304	304	9892
Unusually good condition	Degrading	14	−14	66
	Static	−39	39	4065
	Resilient	56	56	734
	Improving	101	−101	2425
Total		35,810	9921	45,731

the conservation goals of the Natura 2000 network and back the initial working hypothesis that landscapes belonging to it are in favourable condition for maintenance and succession of the vegetation cover.

Land in unusually good condition that is improving, and resilient good condition land, are negatively associated with SCI in apparent contradiction to the general results. However, a query on the land cover of these locations did show that uses are mainly agricultural uses, often with intensive irrigation as discussed below. Therefore this exception actually confirms the type of association between conservation status and land condition and, indirectly, contributes to the validation of the latter.

5. Discussion

The section above reports on the main findings derived from the analysis performed, and interpretation is restricted to a minimum. However, there are several points in both the Methods and Results that require further discussion to better understand the results of our work.

The core of the assessment procedure is the fit of boundary functions to the scatterplot of observed RUE vs. observed aridity. The fact that a least squares model could be fitted with a high level of significance suggests that the scatterplot is compact and its shape can be approached using simple equations. This in turn supports the idea that reference areas could be statistically detected along the full gradient of aridity in the study area.

Both mean (RUE_{OBS_me}) and extreme (RUE_{OBS_ex}) observed RUE increase quickly with aridity from low values corresponding to humid locations. That is the predictable behaviour that makes RUE unsuitable for assessing vegetation cover in large regions containing different climates. Such increase tapers with aridity in the case of RUE_{OBS_me}, showing that less precipitation is compensated in the long-term by a proportional loss of density in the vegetation cover. This is at least true for the range of arid zones in the Iberian Peninsula, but it should not be extrapolated outside that range.

In contrast, extreme observed RUE increases indefinitely with aridity at a rather constant rate. Whilst that pattern is clearly observed, its explanation exceeds the scope of this work and refers to a variety of processes related to the response of vegetation to rainfall events. It has especially intriguing relationships with the findings reported by Huxman et al. (2004), by which very different biomes show similar and maximum RUE under water-limiting conditions. Notwithstanding, the use of a common six-month, antecedent period for all locations is an oversimplification, and the refinement of models to detect a significant number of time lags, for example as incorporated in Udelhoven et al. (2009) would probably improve this, but at the expense of computation simplicity.

Nevertheless, the empirical model presented here seems to have overcome two main limitations of assessment approaches, values for zones with different climates are comparable, and reference areas can be found (Prince et al., 2007; Veron et al., 2006). The second issue has been successfully approached by identifying natural reserves (Garbulsky & Paruelo, 2004), but it is difficult to maintain a strict control of climate variability and there is also a subjective component. We have done this statistically by specifying percentiles for calculating expected RUE. The Iberian Peninsula contains landscapes covering the full range of land condition in every climate zone. Therefore, the 5th and 95th percentiles were used to allow a wide symmetrical range of relative RUE results to be included in the middle assessment class. But these percentiles could be changed depending on the purpose or the characteristics of the study area, which could require some preliminary work. For example, if an area is known to be generally degraded and reference vegetation is scarce or lacking, an asymmetrical interval of percentiles in the upper ranges of observed RUE (say 40% and 95%) could still yield an unbiased assessment of land condition.

The observed RUE found within the specified percentiles define land in good condition and the reason for aiming at as wide an interval as possible is to obtain enough numerical resolution in the computation of relative RUE. Obviously, that step controls the number of locations that are considered out of range. In our study, the classes ‘unusually degraded’ and ‘degraded’ together are equivalent to ‘unusually good’ in terms of their definitions with respect to the range defined by ‘good’. ‘Good’ accounts for 72.40% of the locations (Table 2), and the marginal classes together account for 27.42% almost evenly divided between below and above (the remaining 0.18% corresponds to uninterpreted anomalies). This proportion is slightly higher than the 10% that might be expected strictly from the percentiles, and the difference should be attributed to the fit of the boundary functions. In spite of that built-in concentration of observations in a middle range, the use of two observed RUE implementations leads to a meaningful discrimination of types of landscapes, as discussed below.

The stepwise multiple regression applied in the monitoring procedure proved to be useful to separate the effects of time and aridity on the GVF change rates. The climate in the Iberian Peninsula did not remain constant during the second half of the 20th century but was subject to both spatial and temporal variability (de Luis et al., 2008; Gonzalez-Hidalgo et al., 2008), with an overall trend to aridification in the Mediterranean zone (Gonzalez-Hidalgo et al., 2001). Our correlation analysis (between GVF, aridity and time) was consistent with those results and showed that such climate drift was not homogeneous. Most of the Iberian Peninsula did not show significant trends, but important areas in the northwest did evolve towards greater wetness, whilst most of the Mediterranean zone experienced increased aridity.

The effects of aridity on vegetation density are known to be negative, and have been reported for large areas of the Ebro Valley in NE Spain for an equivalent period (Vicente-Serrano et al., 2006). Interestingly, Fig. 5*b* shows negative effects of aridity on GVF in coincident Mediterranean areas, and Table 3 reveals that the magnitudes of such effects are comparable to those of time. It follows that areas where aridity and time have respectively negative and positive effects on GVF, could be misclassified as static if only time were used as a predictor. In fact, a cross tabulation between the effects of time as in Fig. 5*c* and an equivalent map of effects constructed using straight-forward significant correlation coefficients between time and GVF detected such a situation in 10,941 km^2, most of them in Mediterranean areas. A better interpretation for those areas would be that a decrease in rainfall has had a negative effect on the rates of growth of vegetation, and once this effect is removed, it is observed to still be accumulating over time.

Increased aridity shows positive GVF effects in certain areas (Fig. 5*b*), mainly associated with northern mountain ranges such as the Cantabric or the Pyrenees Mountains. This occurs in wet or wet sub-humid zones and is likely to be more associated with higher temperatures in environments where this factor is limiting than to a decrease in precipitation (Vicente-Serrano et al., 2004).

Isolating the effects of aridity on vegetation trends is more than a mere methodological refinement. Rainfall anomalies cause vegetation anomalies, which allow detection of overall greening at a global level (Hellden & Tottrup, 2008). But land degradation addresses precisely what is left after such effects have been removed (Herrmann et al., 2005). It is generally accepted that a depletion of biomass is an indicator of land degradation. This has been confirmed by several studies (e.g. Geerken & Ilaiwi, 2004; Herrmann et al., 2005; Lambin & Ehrlich, 1997). Several of these studies have also taken climatic conditions directly or indirectly into account. Other studies (Olsson et al., 2005; Wessels et al., 2007) have reported on the influence of variability in precipitation on biomass directly in the Sahel zone and in southern Africa, respectively. Thus, Mulligan et al. (2004) stated that climate variability "is king" in semi-arid to arid environments and may thus camouflage human-induced changes. The stepwise regression approach implemented in this study makes it possible to assess the magnitude of the influence of the two factors, climate and time, separately. As the time factor is connected to human-induced changes, overestimation of degradation trends can be prevented.

Some additional interpretation is required to identify the types of landscapes that may be associated with the classes and subclasses on the final land condition map. We employed Level 3 of the CORINE Land Cover 2000 database (CLC2000) for two reasons, it is a comprehensive and hierarchical classification of prevailing land cover classes in Europe, and it is updated regularly, therefore enabling future repetitions of this work either in other countries or in the Iberian Peninsula.

We found a significant association between dominant CLC2000 classes and land condition categories using the same sampling network that was used for validation ($\chi^2=14{,}548, df\,208, n=45731, p<10\mathrm{E}-4$). Its complete interpretation is beyond the scope of this work, but some key relationships provide useful insights into the approach used here. For example permanently irrigated land and, to a lesser extent, fruit trees, are strongly associated with land in unusually good condition, both static and improving. Such land uses are based on water brought in from outside the system to increase vegetation density beyond the zonal standards of aridity. Therefore when such exuberant cover is evaluated relative to its local rainfall, as in RUE, it scores as overgrown. That is further accentuated by the fact that irrigation often involves intensive management to maximize production in semi-arid zones. Irrigated land then goes beyond the 95% percentile of observed RUE for its degree of aridity, and is appropriately considered as unusually good by the assessment procedure. This also explains the poor affinity between this land condition category and membership to the Natura 2000 network.

In a less extreme situation, there is a high frequency of agro-forestry and natural grasslands in resilient land in good condition, and of broad-leaved forest, coniferous forest and transitional woodland-shrubs on improving land in good condition. This mirrors the landscape in mountain ranges and abandonment of much cultivated land after Spain and Portugal joined the European Union. This trend has been reported using NOAA-AVHRR time–series data for both NE Spain (Lasanta & Vicente-Serrano, 2006; Vicente-Serrano et al., 2003) and for the whole Iberian Peninsula, where a rural exodus syndrome was identified (Hill et al., 2008). Those locations are commonly located close to the upper boundary function in our analysis.

Sparsely vegetated areas show avoidance for land in good or unusually good condition except if it is degrading, and has a high affinity for all categories of degraded or unusually degraded land. This land cover is the bottom line in all cases and is around the lower boundary function. Areas truly limited in their vegetation performance by inherent properties of their habitat would be relatively rare, especially in the drier regions of the aridity gradient which are one of our main targets. As aridity increases, the role of water as a limiting factor increases proportionally over the importance of other physical factors such as soil nutrients. This is the basis of convergence to a common RUE across biomes in dry seasons (Huxman et al., 2004), and becomes relevant at the spatial and temporal scales at which this study was done.

The interpretations above used natural or semi-natural vegetation as much as possible, because it represents rather stable and predictable responses to the effects of the drivers used. Proper agricultural land uses show mixed affinities and avoidances across the full spectrum of land condition categories, and do not allow patterns as regular as those commented above to be found. This is the case of non-irrigated arable land, for example, as it contains a broad variety of crops under many management practices. We believe that such moderate dispersion is in fact a subject for the approach reported in this work, rather than an element for interpretation.

6. Conclusion

The approach for the assessment and monitoring of land condition described in this article has demonstrated consistent performance in its application to the Iberian Peninsula. The use of a long period for developing parallel procedures of synchronic assessment and diachronic monitoring was a milestone in overcoming the paradox of monitoring as a sequence of assessments. This is intrinsically contradictory when dealing with complex landscape mosaics, as assessments commonly require relative estimators, the results of which are often difficult to set in a meaningful time sequence. We have based the whole approach on an estimator of vegetation density (GVF) derived from Earth Observation. A relative ratio (rRUE) was used to generate assessment classes, which enabled detrended comparisons across space. However, rates of change in raw GVF over time and aridity were used to generate subclasses of monitoring in terms of trends with respect to those drivers. We believe that, beyond the particular methods used for every procedure, this concept is general and may be applied whenever assessment and monitoring must be performed jointly to detect land condition.

The concrete methodologies developed here for the assessment and monitoring procedures have succeeded in confronting some important challenges. In the first, boundary functions were effective in two ways, statistical detection of reference areas, which is probably a better alternative than their designation by expert, but subjective criteria, and comparable ranges for the calculation of relative RUE across different climate zones. Making the RUE of any given location relative to its most probable range means that locations can be directly compared. As a corollary, different land covers or uses can also be directly compared on that basis, a common limitation in assessment procedures. This was further reinforced by the two time scales used for the computation of mean and extreme RUE, which were useful for improved discrimination of landscape types. With respect to the monitoring procedure, the use of a stepwise multiple regression was demonstrated to be effective in discriminating effects of a drifting climate from other internal trends in vegetation, which can then be attributed to internal dynamics or to land management. Nonetheless, an advantage of this assessment and monitoring approach is the use of a relatively modest input dataset including time–series of an indicator of vegetation density and climate data. Whilst this is not yet readily available for many areas, generic databases are evolving quickly and this or similar models will see increased applications in the near future.

Our approach is not free of problems, however. First of all, it is strictly empirical. Whilst every effort has been made to make vegetation condition values consistent within a given study area and period, there is no explicit link made between statistical procedures and underlying ecological functions. The boundary functions are selected by best fit, and their shape remains largely unexplained in terms of plant ecophysiology. As a result, absolute references of vegetation performance are lacking in this model and land condition must be assessed by its relative position in an interval. The same is true for the length of the period. So far the only condition is that it should be long enough to account for representative mean and extreme RUE in all locations, and to detect any trend in vegetation density. However, a theoretical framework for the dependency of vegetation growth on climate variability would considerably improve this requirement.

The above limitations are explained in terms of the approach described, but in general terms they are shared by many empirical approaches. The main problem is that existing functional models are difficult if not impossible to parameterise for practical purposes, such as land degradation surveillance. Therefore, for the time being, at least, we will have to live with empirical models. We foresee that in some immediate respects, our model may require attention. The use of a significant number of time lags for extreme RUE instead of a fixed interval of six months would surely enhance its discriminatory capacity. And the representativeness of the study period could possibly be approached by a GVF frequency analysis.

Because of their relevance, two aspects of the methodology reported should be emphasized. First, it should be considered as a language rather than as a self-contained model with fixed steps. The whole is more important here than its individual parts, which may be replaced or upgraded depending on the intended application. For example, detrending observed RUE from aridity is more important than the use of the mean instead of summatory values of the selected vegetation index as an approach to the NPP. Another aspect that could easily be adapted is the specification of assessment categories. For example, lack of knowledge of the study area might make it advisable to use the within-range combination as a relative reference, and call it simply 'normal' rather than 'good condition' as we have done. And second, the nature of the data is more important than any concrete data product. For example, MEDOKADS-GVF was used for this application for the reasons explained in Section 2, but GIMMS-NDVI might be used for a different area if it were considered appropriate. The important point here is that consistent and reliable archived time–series are used according to the general requirements formulated in Section 2.

The Iberian Peninsula was a challenging area to use as a benchmark. As described in the data section, it is a large and complex territory with many sources of land condition variation that operate in many cases at a finer spatial resolution than the one used here. Nevertheless, the model succeeded in detecting interpretable patterns. In very general terms, it shows land to be healthier than expected, with focalised spots of ongoing degradation associated with current or recent intensive land use. Static or positive vegetation growth rates were detected almost everywhere, including Mediterranean areas that were undergoing increased aridification during the study period. The model would therefore depict a landscape matrix in which natural vegetation and areas of inherited desertification are interwoven in a mosaic of static, resilient or thriving patches. Comparatively smaller areas of intense economic development embedded in it would appear, grow, over-grow and decay at a much faster pace than their surrounding landscape. Perhaps the most striking fact detected in this work was that degrading or static trends prevail in degraded or unusually degraded land, whilst trends to improve are represented most in land in good or unusually good condition. This suggests an irreversible divergency in landscape evolution that supports the perceived drama of desertification and is a warning not to overlook degradation hot spots. This will no doubt be a priority target for future work.

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Appendix B

Integrating MODIS-EVI and Gridded Rainfall/temperature Fields for Assessing Land Degradation Dynamics in Horqin Sandy Lands, Inner Mongolia (China)

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Integrating MODIS-EVI and Gridded Rainfall/temperature Fields for Assessing Land Degradation Dynamics in Horqin Sandy Lands, Inner Mongolia (China)

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Abstract. Horqin Sandy Lands in NE-China represent a widely discussed case of ongoing land degradation which receives much attention in China and the outside world. Located in an area of strongly continental climate and extended sandy substrates, the region is undergoing substantial land use change dynamics, largely determined by political decisions and regulations issued by the central and regional governments. While climate data suggest for some stations an increase in annual precipitation during the past 10 years it appears mandatory to relate satellite measurements of biological productivity to corresponding changes in plant-available water over time. In the course of EU-funded research projects (LADAMER; DeSurvey) an approach based on ecological responses of vegetation to physical drivers (2dRUE) has been designed which is suited to more comprehensively characterize areas affected by desertification and transitional land use change processes. The concept departs from the assumption that Rain Use Efficiency, i.e. the ratio of net primary productivity to precipitation, decreases with land degradation. The approach is based on combining satellite-derived indicators of biological productivity (e.g. EVI, FAPAR) and spatially interpolated climate data; an important asset is that RUE is climatically de-trended for varying levels of aridity. The concept is currently applied to various study areas of the DeSurvey Project, one of which covers the Horqin Sandy Lands in Inner Mongolia (China). Results for this site are produced by integrating gridded climate station data and EVI time series acquired by MODIS for the time span 2001-2002, and are compared to a calibrated 20-year-time-series of Landsat-TM/ETM+ data (1987-2007).

Keywords. Land degradation, MODIS-EVI, rain use efficiency, DeSurvey Project

Introduction

Land degradation has long been recognized as a major threat to the global environment which directly impacts on human well-being and social welfare, particularly in in drylands, i.e. the arid, semiarid and sub-humid regions that cover more than 40 % of the Earth's terrestrial surface and are home of more than 30 % of the earth's population [1]. The driving processes are complex, not always completely understood, and diverse views exist on the relationship between climatic and anthropogenic causal factors of desertification. Climate surely sets important boundary conditions, but is not necessarily the major driver of dryland degradation because pulse-/event-driven dryland ecosystems share considerable resilience and recovery capacities. Degradation processes in drylands rather emerge from the human impact

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on the delicate balance between the demand for, and the supply of, ecosystem services (population pressure, socioeconomic and policy factors, land use practices, globalisation phenomena) [2, 3].

Designing and implementing sustainable land management strategies is a substantial endeavour – intrinsically tied to objective, repeatable and spatially distributed information on the state of ecosystems and available resources. The development of integrated assessment and monitoring concepts therefore remains an important challenge, as much as the improvement of environmental data bases and an efficient linkage of terrestrial and space-borne observation systems.

Starting from the perspective that land degradation may be defined as a long-term loss of ecosystem functions (i.e. its ability to provide goods and services) a range of geospatial approaches has emerged which conceptualize measuring degradation through assessing changes of net primary productivity. Earth observation satellite data were already successfully used for reconstructing green vegetation cover (as surrogate for NPP) over time spans of several decades which led to the identification of degraded vegetation resources on local scales [4]. These approaches have been complemented with additional elements of time series analysis and applied to regional scales based on continuous multi-annual time series from global environmental monitoring satellites (such as NOAA-AVHRR, VEGETATION and MODIS) [5].

Vegetation abundance itself interferes more or less directly with all water distribution processes at a site in order to maximise to a certain extent the local water availability for their own benefit (an optimisation process which involves several sub-processes and feedback mechanisms). Boer [6] has proposed a theoretical framework for land degradation assessments which relies on these vegetation functions for estimating the local water balance in terms of rainfall to evapotranspiration ratios. His approach has been modified with the objective to accommodate its use also under operational constraints in terms of limited spatial data availability [7]. Recently a similar but more simplified approach designed by [8] has recently been used to propose an updated global assessment of land degradation. Their results were substantially criticized, particularly because rainfall is not a limiting factor to net primary production NPP in more humid areas, and because RUE ratios largely correlate with rainfall and RUE trends thus merely reflect trends in precipitation [9].

In comparison, the approach of Del Barrio et al. [7], being developed and tested in the context of EU-funded research projects (LADAMER; DeSurvey[2]), relies on more elaborate expressions of 'Rain Use Efficiency' termed "2dRUE". The objective of this paper is to test and evaluate the 2dRUE concept in the context of a desertification-affected area in Horqin Sandy Lands, a dryland ecosystem with summer rainfalls in Inner Mongolia, China.

1. The Study Site

Desertification is an important issue in parts of Northern China [10]. While the region is less densely populated than Southern and Central China, densities are nevertheless very high, relative to the inherent low productivity of these arid and semi-arid zones. Horqin Sandy Lands (i.e. the Naiman Banner) are located in the agro-pastoral zone between the Inner Mongolian Plateau and the Northeast Plains in China (42°41'-45°45'N, 118°35'-123°30'E). Covering more than 42,000 km² it is one of the major sand areas in Northern China and an important part of Inner Mongolian grasslands. The vegetation consists predominantly of shrubs and perennial grasses, trees are found in places where water conditions are favourable. Climatically, the region is part of continental drylands with hot and short summers and very cold winters. Mean annual temperature minima and maxima range from -8.8°C in January to 30.4°C in July; mean annual precipitation is 375 mm with nearly 80 % concentrating in the months from June to September. An important characteristic is its irregularity: rainfall varied from 205 to 679 mm year-1 in the observation period from 2000 to 2008.

[2] http://www.ladamer.org/ladamer/ and http://www.desurvey.net/

The main aspect of degradation in this area is what Chinese authors term “sandy desertification” (including shifting sand dunes, sand dune reactivation, shifting sands spreading into grasslands, and wind erosion in dry farmland) which develops on sand sheets and fans of quaternary origin [11]. Although the role of drought is not exhaustively clarified yet most authors suggest that human activities and cultural change are key determinants of degradation processes [12]. Main drivers resulted from the transformation of the traditional Mongolian grazing society. Chinese immigrants brought with them a cultural tradition that is rooted in farming, a practice further promoted by the socialist regime [12, 13]. Reclaiming rangelands in areas of marginal rainfall and strong winds for agricultural production has caused the destruction and loss of topsoil, while the concentration of cattle in the remaining rangeland areas increased stocking rates above sustainability thresholds. Ever since, national policies, such as Deng Xiaoping’s reform policies, but also decisions and regulations issued by the regional governments have encouraged both, new and intensified land use practises (expansive groundwater-based irrigation) as well as restoration and protection measures (enclosures of pastureland, grazing regulation, tree planting campaigns).

2. Material and Methods

The approach presented in this paper builds on consistent and reliable time series of satellite observations with high revisit capacity and coarse spatial resolution which can be used to produce proxies for net primary productivity (e.g. vegetation indices, green vegetation cover derived from spectral unmixing, or biophysical variables such as FAPAR). The dimensionality of the NPP equivalent is not important for the 2dRUE concept as quantities will be expressed as proportional values.

2.1. Satellite Data

In this study we used the MODIS Enhanced Vegetation Index (EVI) product (MOD13Q1) for the biennial period January 2001 – December 2002. EVI is a composite of daily bidirectional reflectance, available as 16-day-composite. Compared to other vegetation indices, EVI is less sensitive to soil atmosphere effects, minimizes brightness-related soil effects and is more responsive to canopy structural effects. The EVI time series resampled to monthly time steps for handling them equivalently to monthly temperature and rainfall averages.

In addition to the 2-year MODIS data series, a second time series of Landsat-TM and -ETM+ images over the core study area has been established which includes 12 satellite observations (1987-2007) from the same phenology period (August/September). This series of images has been corrected for atmospheric effects [14, 4] based on the latest calibration data [15]; the resulting reflectance images were then subject to spectral mixture analysis for generating abundance images for the endmembers “Photosynthetic Vegetation” (PV), “Active Sands” (AS) and “Water” (W). The PV time series was then used for analysing linear trends in NPP over the past 20 years.

2.2. Spatially Interpolated Meteorological Data

Meteorological data in tabulated form was received from the National Climatic Data Centre (NCDC) which holds the complete National Oceanic and Atmospheric Administration (NOAA) data and is the World’s largest archive of weather recordings. For the Horqin area data are available from the 1950s onward; we used daily mean, minimum and maximum temperatures as well as precipitation from six stations inside and from additional seven stations outside the study area. The meteorological data were pre-processed by replacing missing values and outliers with the seasonal average of the period 1973-2008.

The 2dRUE-concept requires spatially explicit data layers. To generate equivalent temperature and rainfall fields the station data for the time interval 2001-2002 (condensed to monthly averages) were interpolated with a spherical Kriging procedure (where the spatial interpolation of temperature recordings was performed by using SRTM-derived altitude values as covariate).

2.3. The 2dRUE Land Degradation Assessment Concept

The reduction of plant biomass and Net Primary Productivity (NPP) below those of land not desertified under equivalent environmental conditions is proportional to land degradation impact. This is an accepted perception of land condition and the most closely related to the UNCCD definition of desertification [7]. Two ecosystem attributes are relevant for this assessment, annual average biomass and seasonal or inter-annual growth peaks. The first demonstrates the ecosystem's long-term capacity to sustain biomass, whilst the second concerns its resilience for recovering from disturbances, in particular, rainfall fluctuation. Both attributes are derived from a climatically de-trended expression of rain use efficiency (RUE).

Rainfall Use Efficiency (RUE) was originally defined as the ratio of NPP to precipitation (P) over a given time period [17], which can be interpreted as the fraction of rainfall released to the atmosphere through the vegetation cover; in this study monthly averages of MODIS EVI were used (as surrogate to NPP) for computing an "annual average RUE" layer. The interpolated rainfall and temperature fields together with extra-terrestrial solar radiation inputs provided the basis to generate a potential evapotranspiration (PET) map [16] which permits to establish an annual average aridity map based on the relationship PET/P (Figure 1). In parallel, the concept was used to generate maps of an "extreme observed RUE" (by the dividing the maximum monthly EVI by the accumulated rainfall over the three preceding months); the associated aridity index is derived by computing the ratio between the accumulated PET values by the accumulated rainfall figures from the same 3-month-period.

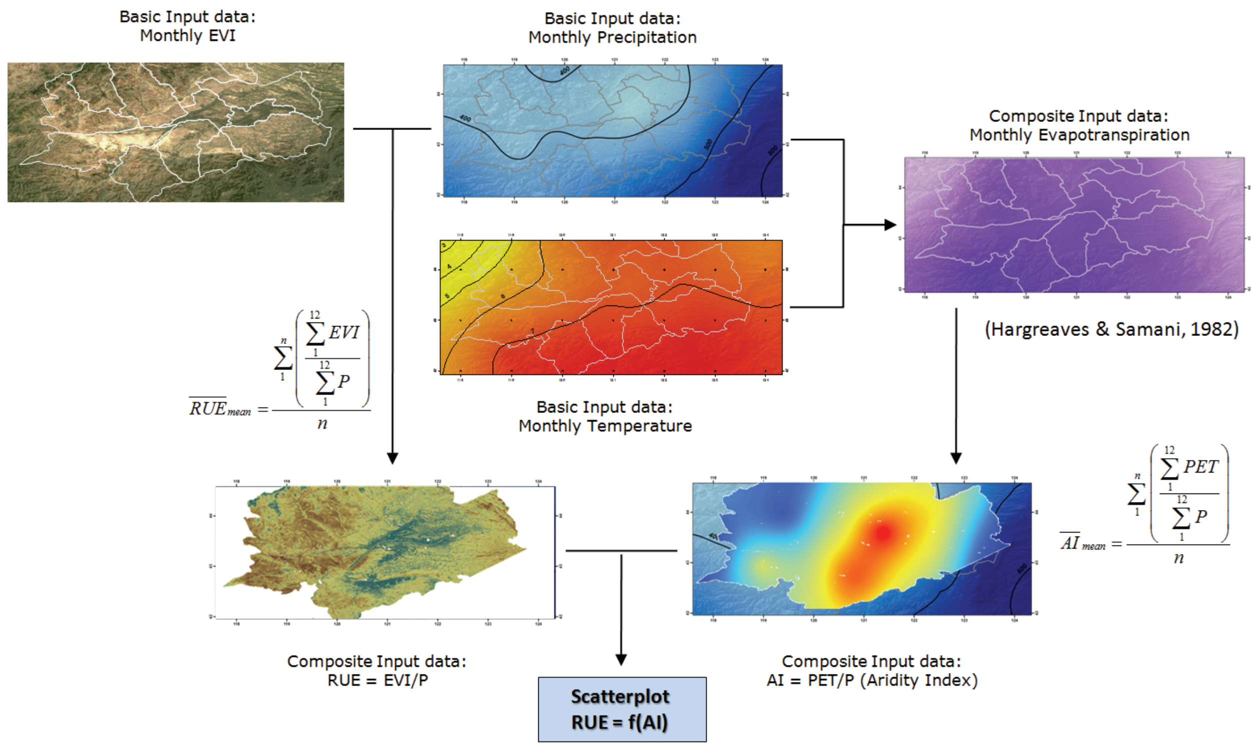

Figure 1. Input layers for the 2dRUE-based land condition assessment representative for the period 2001-2002

If RUE layers are computed over large areas with climatic gradients (as they exist between lowland and upland ecosystems in Horqin Sandy Lands) dry areas tend to account for the largest scores because of their very low P values. This effect renders a direct comparison between locations under different climates difficult, if not impossible (see also [9]). Del Barrio et al. [7] proposed a detrending concept where annual RUE values are plotted against the Aridity Index (AI) (Figure 2). The upper and lower boundaries of these scatter plots (here identified by fitting linear functions to the 5- and 95-percentiles) are considered representative for the maximum and minimum vegetation performance within a given aridity level. The final step was to compute a layer showing the relative position of the observed RUE within the range formed by the maximum and the minimum expected RUE. This relative RUE (rRUE) is assumed to reflect the vegetation condition as the observed performance (in terms of the satellite-derived vegetation index integrated over time) with respect to the minimum and maximum performance that can be expected for the particular climatic situation (defined in terms of AI). Pixels from irrigated vegetation (marked in blue) were not used for defining these upper and lower bounds (Figure 2).

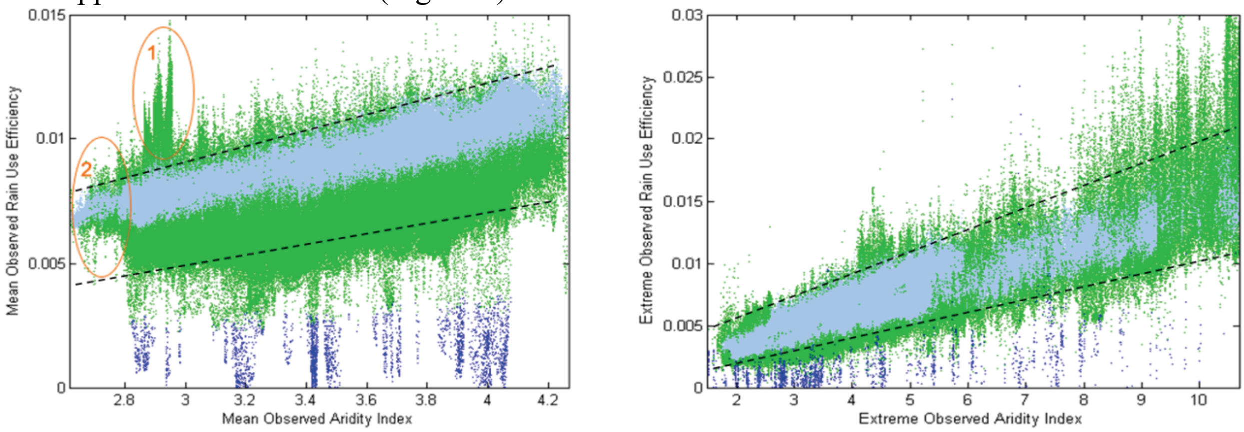

Figure 2. RUE vs. AI scatter plots used for defining the lower and upper vegetation performance range (5 - 95 percentiles) for long-term (left) and short-term response to rainfall

It is important to note that the definition of the **relative** (mean or observed extreme) RUE for each aridity level transforms the numerical scale into a common reference range from 0 to 1, where values close to one indicate high productivity, values around or below 0 more or less degraded areas. This relative assessment scale is a major justification for selecting a surrogate for NPP (here MODIS EVI) instead of quantified biomass model outputs.

3. Results and Discussion

The resulting relative RUE maps are shown in figure 3. Wherever any of the relative RUE estimators exceeds the stated range, it may be interpreted as a deviation associated with under- or over-performing vegetation in the respective component [7]. It is interesting to note that in the Horqin study area the highest relative mean RUE values are found in mountainous forests (Figure 3-top: 1) and agricultural areas which mainly extend along the river system (Figure 3-top: 2) or occur as isolated patches within the grasslands (Figure 3-top: 3); many of these agricultural areas are irrigated.

The short-term relative RUE maxima exhibit a modified distribution: in comparison to the vegetation communities in the mountainous areas (Figure 3-bottom: 1) and extended rangelands in the northeast (Figure 3-bottom: 2) only few agricultural areas (Figure 3-bottom: 3) arrive at high relative short-term RUE scores. This demonstrates that most agricultural areas are decoupled from rainfall and thrive on surplus water from irrigation systems, while it are primarily rangeland and forest ecosystems which are capable of responding quickly to rainfall events.

Combining both maps provides an attractive option to generate a map of ecosystem performance which reflects long-term productivity and the ability to react on surplus water for increasing productivity in the short run (2dRUE). This is achieved by dividing the both RUE-ranges for each aridity level into 5 discrete intervals (<0, 0-0.25, 0.25-0.75, 0.75-1, >1) and cross-tabulating the resulting RUE classes. This yields a 25-class table (figure 4), where the upper left quadrant represents situations where both, long- and short-term RUE responses are on high performance level (class 55, 45, 54, and 44); the lower right quadrant indicates adverse conditions, i.e. where short- and long-term ecosystem performance is in critical state (class 11, 21, 12, and 22). The remaining classes (36, 34, 53, 43, 31, 32, 23, and 13) represent intermediate (i.e. neutral) states; blue-coloured classes (25, 15, 14, 52, 51, 41) do not occur in the study area. Closer inspection reveals that the red areas (Figure 4: 1) represent active dunes and sand sheets with only occasional plant growth. The dark brown tones indicate unusually degraded land; contiguous areas of this type occur in the vicinity of active sand sheets, on extended rangelands north of the Xar Moron River (Figure 4: 3), and on erosion affected mountainous terrain (Figure 4: 2). Additional degraded areas occur in isolated patches.

This spatial pattern is in good agreement with field observations collected during several field trips. However, a systematic validation of such map products is an extremely difficult issue. Del Barrio et al. [7] related their RUE-based results to the Natura 2000 network Sites of Community Interest (SCI) in continental Spain and found that a majority of SCIs (i.e. protected areas with no or only limited disturbance) were positively associated with land in good condition and unusually good condition.

Since similar system of conservation sites or spatially distributed reference areas does not exist in Horqin Sandy Lands an alternative validation strategy is to consider the 20-year Landsat-TM/ETM+ time series as completely independent source of information and to compare how specific land condition classes are characterised in this data set. The underlying hypothesis is that areas in unusually degraded condition might already be under pressure for longer periods of time, and as such be characterised by decreasing trends in productivity. First results seem to support this assumption: rangeland areas with below average productivity (no. 3 in Figures 4 and 5) tend to coincide with decreasing vegetation abundance over the past 20 years, highly productive agricultural areas (no. 2 in Figures 4 and 5) typically exhibit inverse trends (increasing productivity over time), while some of the active dune fields and sand sheets (no. 1 in figures 4 and 5) appear invariant over this extended time span.

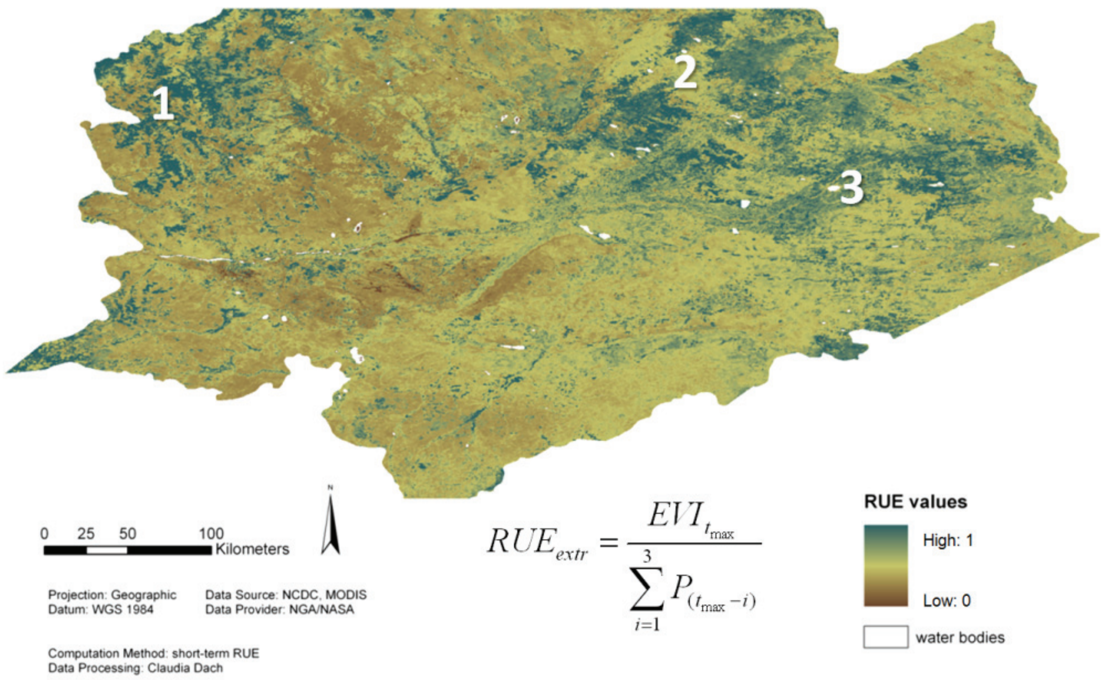

Figure 3. Maps of relative mean RUE (long-term ecosystem response, top) and relative extreme RUE (short-term ecosystem response, bottom) in Horqin Sandy Lands (2001-2002).

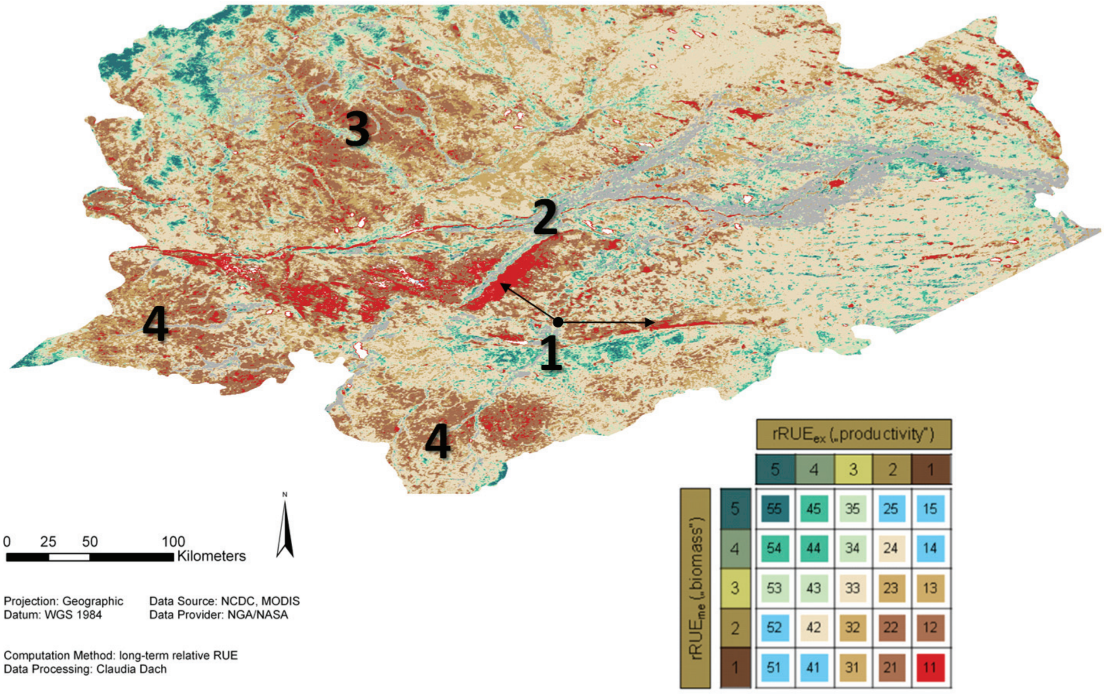

Figure 4. Integrated land condition assessment map (2dRUE) for Horqin Sandy Lands. The map is derived from cross-tabulating classified relative mean RUE and relative extreme RUE outputs; grey colours indicate irrigated agricultural areas

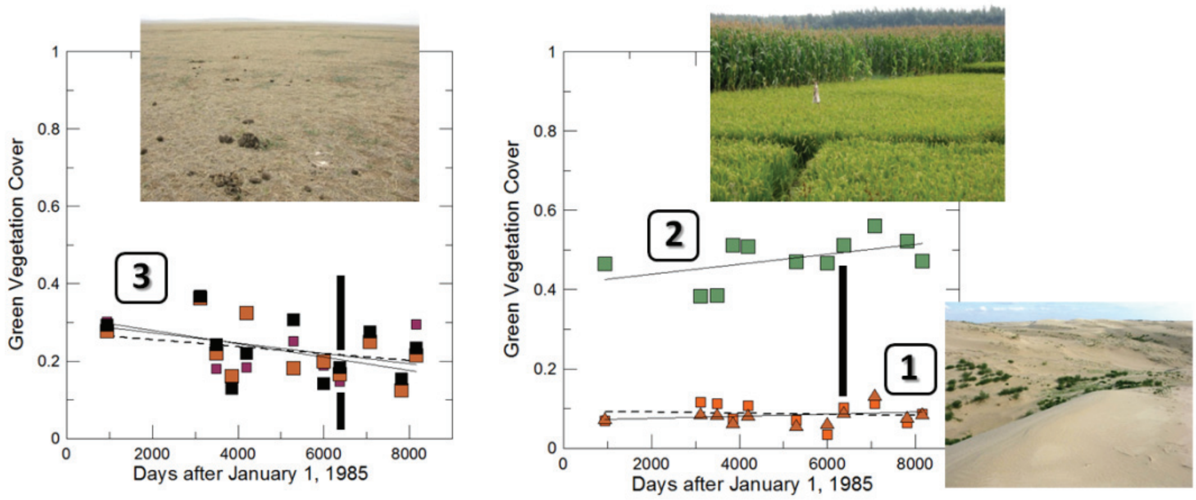

Figure 5. 20-year trends of SMA-derived photosynthetic vegetation abundance for active dune systems (1), an intensely farmed agricultural system (2) and degraded rangelands (3). The vertical bars represent the MODIS observation period (2001/2002) used in this study; numbers relate to Figure 4.

4. Conclusions

First results of applying the rRUE-based land degradation assessment concept to a marginal and desertification-affected area in Northeast China (low and irregular summer rainfall which coincides to the vegetation period) demonstrate that the method holds the potential to be applicable under a wider range of rainfall-temperature regimes under which it was originally developed. Making the RUE of any given location relative to its most probable range means that locations can be directly compared. The relatively modest data requirements (publicly available long-term climate recordings and multi-annual satellite observations of vegetation cover and biomass) constitute an important advantage of this concept; in the absence of sufficiently dense station networks also meteorological satellites (e.g. Meteosat/MSG-1, TRMM) might be considered as alternative data sources.

However, care must be taken until a more systematic validation of the assessment results is available. Until now, only field observations and long-term cover/biomass trends derived from 20 years of calibrated Landsat-TM/ETM+ observations (1987-2007) have been evaluated for selected sites. Forthcoming work will focus on a systematic comparison between the Landsat-based trend analysis and the 2dRUE assessment maps at a common pixel size of 250 m. Additionally, the rRUE-based land condition assessment will be established for a second time step (2008-2009); comparison of both maps will not only reveal recent land condition changes but provide additional hints about the consistency of the approach.

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