CASCADE

Catastrophic shifts in drylands:

How can we prevent ecosystem degradation?

CASCADE REPORT series

Drivers of Change in the Study Sites

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CASCADE

Catastropic shifts in drylands

Deliverable 2.2

Drivers of change in the study sites

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Contents

1 Introduction

As part of the natural environment, soil, much like water, forests, plants and animals is a global renewable resource, as long as it is adequately monitored, protected and conserved. Soil formation rates are so slow (Wilding et al., 1983) that the loss of its biological productivity or economic benefit leading to land degradation can be considered an irreversible process (Tsunekawa, 2000) on human timescales. In most Mediterranean basin drylands, the downward spiral of land productivity that ends up in desertification is driven by population pressure. This pressure is often coupled with aridity that reduces the degree of soil development properties such as soil and infiltration depth, organic content and nutrients (Sombroek, 1990; Wilding et al., 1983), thus hindering primary production and ecosystem resilience. Furthermore, agricultural intensification is leading to contamination of soils with nitrates, pesticides, and even heavy metals (Stoate et al., 2001), while also endangering the quality and quantity of the often scarce water resources. Future conditions will most likely aggravate, as warming trends (Luterbacher et al., 2004) and natural climate variability (Lionello et al., 2006) render the Mediterranean a hotspot of climate change. Finally, a long term shortage of financial resources and institutions, which are critical for arresting or avoiding this spiral (Mazzucato and Niemeijer, 2000), augment the adverse conditions.

The outcomes of CASCADE Deliverable 2.1 “Historical evolution of dryland ecosystems” showed that while at first glance, the main drivers of degradation for each of CASCADE’s Study Sites are different, they are always relevant to the accumulated impact of a pressure: forest fires, marginal agriculture, grazing and wood-gathering activities, and long-term poor land management that is difficult to overturn. Less often, the causes of degradation are related to natural drivers such as climate, which nevertheless serves as a stimulant for all processes. Based on these outcomes, in natural and semi-natural ecosystems a distinction on two centers of influence can be made, either natural or human induced (Figure 1). The former include vegetation (or more broadly ecosystem) behavior and aridity, while the later encompasses land management issues such as abandonment, grazing pressure and offsite water resources management. While there is a large degree of interaction among drivers, forest fires are maybe the most bipolar driver of degradation. More importantly though, all drivers are enhanced or discouraged by climate variability (or change) on one hand, and social issues (i.e. demography, poverty and social inequality, lack of education and policy) on the other.

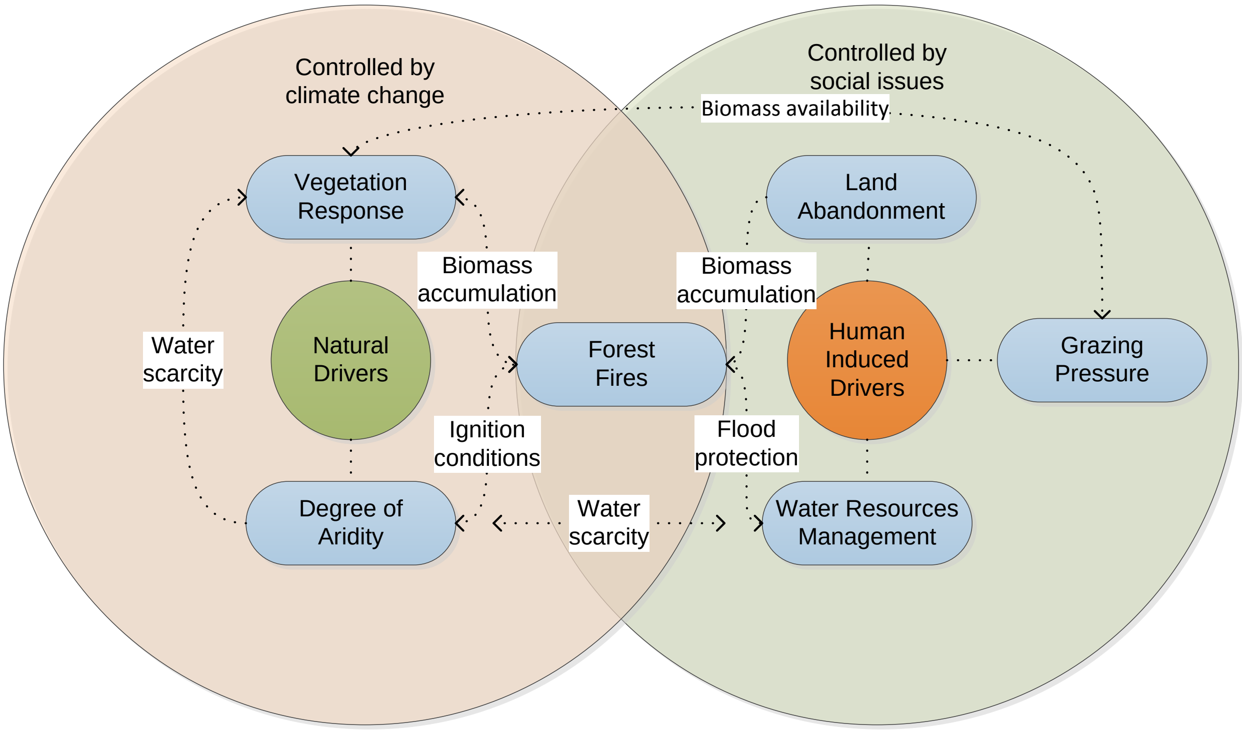

Figure 1: Conceptual framework of natural and human induced drivers affecting land degradation in the Mediterranean.

The drivers depicted in Figure 1 will be further analyzed in the following chapters. The accumulated impact of these drivers can be associated with the concept of sudden ecosystem shifts beyond thresholds that preclude successful restoration of the desirable ecosystem properties and services. These thresholds can either depict a “point of no return” or a “resilience against restoration” owing to constrains due to lack of resources or unfavorable environmental conditions. Whereas for continuous pressure, thresholds may be sought in the combined effect of a secondary pressure, in the case of discontinuous events, thresholds relevant to event frequency and intensity are possible key indicators (Twidwell et al., 2013). Even though these thresholds are often associated with target species physiology, landscape and climatic characteristics and can be expected to have a localized effectiveness, emerging patterns can be organized into a unifying framework. This report strives to further identify and quantify to the extent possible the natural and human induced drivers present in the CASCADE Study Sites.

1.1 Climate pressure

Global Climate Change will seriously affect the hydrological processes and alter the supply of ecosystem services that are vital to human well-being (Schaller et al., 2011). Climate change is expected to affect precipitation and evapotranspiration patterns (Tsanis et al., 2011), and consequently variables such as local water availability, river discharge, and the seasonal availability of water resources (Arnell et al., 2011). The latest state of the art review on climate change research for the Mediterranean region by Ludwig et al. (2011) shows that recently observed trends and projections from climate model ensembles indicate a strong susceptibility to change in hydrological regimes, an increasing general shortage of water resources and consequent threats to water availability and management. Deliverable 2.1 describes in depth the climatic gradient of the CASCADE Study Sites from the overall dryer and warmer Albatera and Randi Forest to the seasonally dry Várzea and Castelsaraceno. Based on the longest available records, all Study Sites display clear trends of moderate temperature increase. These trends are manifested with variable degrees of drought related stresses, from constant to seasonal and from long-established to contemporary. Taking into account the uniformly positive temperature trends that appear to agree with climate change scenarios, there is a high probability that these conditions will aggravate in the future.

1.2 Grazing pressure

The extraction of animal products effectively reduces the net organic carbon and nutrients from the natural ecosystem and, depending on the extraction rate, can potentially deplete the rangeland soil services (Ayal et al., 2005). In overexploited rangelands, vegetation removal and trampling by livestock promotes soil erosion by water through disintegration of biological topsoil (Xue and Dirmeyer, 2004) that is an active site of soil formation and organic matter decomposition (Puigdefábregas et al., 1999). The latter can cause surfaces to seal with a mineral film that reduces infiltration and contributes to soil-eroding flashfloods, thus leading to land degradation. In this mechanism of land degradation, changes in the properties of the land (soil, water, vegetation) do not correspond linearly to changes in vegetation biomass dynamics. Depending on grazing intensity and other human or natural induced pressures, such as the degree of aridity, the system's resilience can be reduced and small changes can lead to transition between stable states (Scheffer et al., 2001) with dramatic difference in ecological value.

Albatera, Castelsaraceno, Messara and Randi Forest face various levels of grazing pressure that contributed to the levels of degradation or conservation in discreet plots of each Study Site. Here, estimates of grazing capacity and grazing pressure from each Study Site will be used with a simple grazing model to quantify long term equilibrium states and the effect of annual climate variations on these states.

1.3 Fire pressure

Fires are part of the natural ecosystem processes in many regions of the world. The relationship between fire frequency and aridity normally shows a bell-shaped curve (Pausas and Bradstock, 2007). The occurrence of fires is highest in medium environmental stress and decreases when stress both increases (due to a lack of biomass and fuel connectivity) and decreases (higher moisture content of fuels). Particularly in the Southern Mediterranean countries, where the abundance of dry vegetation due to seasonal or out-of-season droughts confers an extreme flammability to the plant communities, it is possible that changes in land use in the Mediterranean by recent socio-economic changes may have affected the timing, magnitude, frequency and intensity of these fires. However natural Mediterranean ecosystems have coevolved with fire and are, hence, adapted to it although fire increases the transition rate from forest to shrublands and from shrublands to grasslands (Callaway and Davis, 1993). Nevertheless, the frequency of forest fires has increased in the last 50 years (Pausas et al., 2009, 2004) and these concentrations of fires in small time scales generate a successional loop with little progression of ecosystems towards mature woodlands or forests (Baeza et al., 2005).

The impacts of fire on ecosystem services depend on many factors and any attempt to reduce fire risk and its impacts through good land management practices requires a good understanding of how fire affects the structure and functioning of ecosystems. Vegetation composition and predominant regeneration strategies after disturbances (e.g. seeders or resprouters) will primarily control the retention of resources in situ by determining the start and rate of soil protection and, hence, soil erosion rate. However, the risk of off-site damages by runoff and sediment export is significantly increased after forest fires although it might be site-dependent (Cosandey et al., 2005). The loss of soil resources greatly depends on fire intensity. Fires that generate high soil surface temperatures produce combustion of soil surface organic matter and nitrogen, as well as soil sterilization and sealing (REFs).

1.4 Socioeconomic pressure

Environmental degradation can only be understood within the context of the society that the environment supports. Therefore, the incorporation of socioeconomic variables may improve our understanding of the human causes of land use and ecosystem change and stimulate the creation of relationships among them; information on these variables will support the development of people's motivation and interest to identify and resolve social issues as well as to forecast trends in local policy and economic development. Trends in demographic features such as the urbanization, aging and education level as well as land use and land use intensity changes are often connected to and feedback on the deterioration of what is perceived as the desirable ecological status and ecosystem services. Traditionally, studies across Europe (e.g. Armelagos et al., 1991; Meyer and Turner, 1992) certify that growth in population leads to growth in agricultural production which may also apply to demand or profit opportunity related to expansive pastoralism and grazing. Nevertheless, this relationship may not hold across time scales and regions, due to changes in technology and use of inputs in agricultural production. The direct connection of socioeconomic variables to environmental changes is a difficult task, especially at small spatial scales because of the importance of other variables that affect demand or create spatial impact variation such as global market trends.

Furthermore, socioeconomic feature analysis can often lead to contradictory results. For example, the parameter of poverty is considered to aggravate the resource overuse in developing countries and possibly in the Mediterranean of the current financial crisis. The “poverty trap” or “downward spiral” theory suggests that those not able to invest in resource conserving practices continue to exploit hillsides and overgraze land instead, as they cannot wait for rangelands to recover (FAO, 1995). Similar “inappropriate” management of natural resources under the forced condition of poverty leads to a general land “mistreatment” thus causing negative feedbacks on poverty. However, this hypothesis disregards the fact that poor communities implement land use controls to stabilize their income through increased vegetation cover (e.g. Forsyth et al., 1998) and that poor policy or/and policy implementation can be a more important hurdle against escaping the “poverty trap” (Svenson, 2005). Apart from financial resources, educated farmers are more likely to adopt new technologies and numerous studies show that farmers with education have more benefits from their land and generally higher incomes. Also, higher education may provide off-farm labour opportunities, which can intricately link with agricultural practices.

In the case of wildfires, demographic and socioeconomic factors have been studied less, but they also affect fire occurrence, as changes in behaviour and new lifestyles are the main factors driving the spatial distribution of people in forest areas (Badia-Perpinyá and Pallares-Barbera, 2006; Venevsky et al., 2002). According to literature, three main variables correlate well with fire events: population density, wildland-urban interface (WUI) and landscape accessibility. Song et al. (2006) found power law relation between population density and fire probability in Japan while Syphard et al. (2007) found that fire frequency was well modelled in California (USA) by factors such as population density and distance to the wildland–urban interface. Badia-Perpinyá and Pallares-Barbera (2006), Calef et al. (2008) and Prestemon et al. (2002) found similar results, identifying WUI as a statistically significant wildfire risk factor. Finally, researchers in a number of Mediterranean regions have concluded that spatial patterns of ignition are strongly associated with landscape accessibility (Badia-Perpinyá and Pallares-Barbera, 2006; Calef et al., 2008; de Vasconcelos et al., 2001; Romero-Calcerrada et al., 2010; Yang et al., 2008). In particular, contrast values highlight the importance of distance to roads, recreational areas and trails as an important predictor of ignition risk. These patterns have also been encountered in previous studies (i.e. Badia-Perpinyá and Pallares-Barbera, 2006; Cardille et al., 2001; Chou et al., 1993; Chuvieco and Salas, 1996; de Vasconcelos et al., 2001; Dickson et al., 2006; Yang et al., 2008, 2007).

Several studies relate socioeconomic factors to fire occurrence regarding developed countries, characterized by a higher level of industrialization and quality of life (e.g. low poverty, high educational level, adequate housing conditions) (Syphard et al., 2007). In Northern Mediterranean countries, the decrease in rural population and land abandonment since the mid-20th century have significantly determined the quantity, quality and structure of biomass (fuel) that confer a very high risk of fire for the ecosystem (Baeza et al., 2006). Post-fire regeneration patterns of these landscapes is clearly driven by human activities through fire ignitions and previous land use (Baeza et al., 2007). The increase of recreation versus agricultural activities and the urbanization in forest areas may indicate a potential shift in the nature of Mediterranean wildfire risk (Romero-Calcerrada et al., 2008). Nevertheless, associations detected between socioeconomic conditions and fire occurrence have been found to exist with contrasting demographic characteristics in developing countries (Dondo Bühler et al., 2013). Other contradictory findings include that of lower unemployment correlating with higher fire occurrence (Mercer and Prestemon, 2005; Prestemon and Butry, 2005), fire generated opportunities of a job for firefighters (Martínez et al., 2009), expression of social dissatisfaction through arson (de Torres Curth et al., 2012).

2 Methodology

2.1 Climate datasets

Climate model data were obtained from 9 GCMs (Table 1) of the 5th phase of the Coupled Model Intercomparison Project (CMIP5 - Taylor et al., 2012). The simulations are based on the future emission scenarios as they are defined by specific Representative Concentration Pathways (RCPs - Moss et al., 2010). The RCPs are based on the change in the radiative forcing ($\Delta W/m^2$) due to the application of various Shared Socioeconomic Pathways (SSPs). Here we focus on 3 such RCPs, namely RCP2.6, RCP4.5 and RCP8.5 taking after the projected change in the radiative forcing to be posed in the earth atmosphere (2.6 W/m^2, 4.5 W/m^2, and 8.5 W/m^2 respectively) by the end of the 21st century. The model data have been previously used to assess drought propagation at regional level (Vrochidou et al., 2013) and global scale (van Huijgevoort et al., 2014). Here we focus on a domain that spreads between latitude 34^{o}N to 44^{o}N and longitude 10^{o}W to 35^{o}E that includes the entire European Mediterranean and all the CASCADE Study Sites.

Table 1: List of the CMIP5 GCMs that were used to provide the temperature and precipitation data.

Modeling Center	Model	Institution
CCCma	CanCM4	Canadian Centre for Climate Modelling and Analysis
CNRM-CERFACS	CNRM-CM5	Centre National de Recherches Meteorologiques / Centre Europeen de Recherche et Formation Avancees en Calcul Scientifique
CSIRO-QCCCE	CSIRO-Mk3.6.0	Commonwealth Scientific and Industrial Research Organisation in collaboration with the Queensland Climate Change Centre of Excellence
EC-EARTH	EC-EARTH	EC-EARTH consortium
IPSL	IPSL-CM5A-MR	Institut Pierre-Simon Laplace
MIROC	MIROC-ESM	Japan Agency for Marine-Earth Science and Technology, Atmosphere and Ocean Research Institute (The University of Tokyo), and National Institute for Environmental Studies
MOHC	HadGEM2-ES	Met Office Hadley Centre (additional HadGEM2-ES realizations contributed by Instituto Nacional de Pesquisas Espaciais)
MPI-M	MPI-ESM-MR	Max Planck Institute for Meteorology (MPI-M)
NASA GISS	GISS-E2-R	NASA Goddard Institute for Space Studies

Raw climate model outputs cannot be used in their native form in climate change impact studies, due to the high systematic errors and biases that they feature (Leander and

Buishand, 2007; Sharma et al., 2007; Wood et al., 2004). Methods of statistical bias correction are increasingly being developed and adopted as means to translate climate model projections into actual impacts. These methods involve some form of transfer function (Block et al., 2009; Déqué, 2007; Fowler et al., 2007; Grillakis et al., 2013; Piani et al., 2010), or other techniques for equalization of statistical characteristics between modeled and observed variables (Engen-Skaugen, 2007; Horton et al., 2006; Leander et al., 2008; Leander and Buishand, 2007).

Here we use a widely used bias correction method presented by Haerter et al. (2011) to correct the monthly precipitation and temperature data for biases in mean and variance. The bias in mean is corrected by subtraction of the differences found between observed and modeled values and a correction to the model data is applied to conform to the variability of the historical data. This procedure takes the sequence of anomalies and scales them consistently with the observed historical variability. In the case where data follow the normal distribution the transfer function is linear and is of the form:

$$\chi_{sc}^{cor} = \left(\chi_{mod}^{sc} - \overline{\chi_{mod}^{con}}\right)\left(\frac{\sigma_{obs}^{con}}{\sigma_{mod}^{con}}\right) + \overline{\chi_{obs}^{con}} \quad (1)$$

where χ_{sc}^{cor} is the final adjusted time series, χ_{mod}^{sc} is the raw model predictions for the scenario period, $\overline{\chi_{obs}^{con}}$ and $\overline{\chi_{mod}^{con}}$ are the mean of observed and modeled data for the control period, respectively, and σ_{obs}^{con} and σ_{mod}^{con} are the standard deviations of observed and modeled data for an adjusted baseline mean and standard deviation with respect to the observed data. The methodology is applied for each calendar month in order to cope with the correction in the precipitation seasonality. The E-OBS gridded data set (Hofstra et al., 2009) was used as observational dataset to correct the climate model data for their biases both in temperature and precipitation.

2.2 Vegetation

2.2.1 Vegetation yield

FAO has addressed the relationship between crop yield and water using a simple equation (Doorenbos and Kassam, 1979) where relative yield reduction is related to the corresponding relative reduction in ET:

$$\left(1-\frac{G_a}{G_p}\right)=K_G\left(1-\frac{ET_a}{ET_p}\right) \qquad (2)$$

where G_p and G_a are the potential and actual above ground biomass growth, ET_p and ET_a are the potential and actual evapotranspiration, and K_G is a yield response factor representing the effect of a reduction in evapotranspiration on yield losses. The yield response factor K_G captures the essence of the complex linkages between crop production and water use (Doorenbos and Kassam, 1979). Equation 2 is a water production function which has shown a remarkable validity at basin scale, field scale (Yacoubi et al., 2010), and in decision support systems (Gastélum et al., 2009) and can be applied to all agricultural crops, i.e. herbaceous, trees and vines. It is noteworthy that for particular values of K_G (e.g. alfalfa, bananas and peas with $K_G > 1$), and small values of ET_a/ET_p denoting water stress, the ratio G_a/G_p can become negative, implying that the production of crops is not possible.

Numerous empirical and physically based models have been constructed in order to derive ET_p and ET_a values. Aiming to simplify without much error, the method proposed by Turk (1961) and later verified (e.g. Liang, 1982) can be used to estimate the annual ET_a as $R/\sqrt{c+\left({}^{R}/_{L(T)}\right)^2}$, where R is the annual rainfall (mm), L is a *thermal indicator* given by $L(T)=300+25T+0.05T^3$ with T the mean annual temperature (^oC) and c usually equal to 0.9 (Kaczmarek et al., 1996). Furthermore, the Blaney-Criddle formula (Blaney and Criddle, 1962) requires mean daily temperature T and the mean daily percentage p of total daytime hours, both on a monthly basis, in order to derive daily ET_p as $p(0.46T+8)$. Value of p can be estimated or derived from tables (Doorenbos and Pruitt, 1975) and ET_p can be aggregated to produce long term averages.

2.2.2 Vegetation stress

Despite the considerable uncertainties in the future climate drivers (Koutroulis et al., 2013), temperature increase due to climate change is considered a certainty in the peer-reviewed literature (Parry et al., 2007), with authors questioning not “whether” but rather “when” certain thresholds will be exceeded. Joshi et al. (2011), suggest that while a global average 2°C warming threshold is unlikely to be passed before 2060 for all greenhouse-gas emission scenarios, on regional scales, this threshold will probably be exceeded by 2040 if emissions continue to increase. This projected temperature increase will imply higher evaporation and drier conditions under equal or less rainfall (AWC and WWC, 2009). A recent IPCC report (Bates et al., 2008) predicts that climate change over the next century will also affect rainfall patterns, reducing its frequency and increasing its intensity. While changes differ strongly across models and some areas are projected to experience increased precipitation (Marchal et al., 2011), many parts will experience a decline in precipitation. Concerning many parts of the arid regions, precipitation is expected to decrease by 20% or more over the next century (AWC and WWC, 2009). Agriculture and food security are threatened by these changes, as an increase of T with a parallel decrease or even stationarity of P leads to a decrease of the ET_a/ET_p fraction and therefore to a reduction of the expected G_a. This decrease in the potential crop yields and the productivity of water in agriculture with high temperatures has been previously predicted (Cline, 2007; Kundzewicz et al., 2007) and is expected to cause an increase in the demand of the already reduced supply of agricultural water (Döll, 2002).

Detecting trends and seasonal changes of vegetation related time series may enable the classification of these changes: changes in the trend component often indicate disturbances, while changes in the seasonality may indicate phonological changes (Verbesselt et al., 2010a, 2010b). Several generic change detections approaches have been proposed, e.g. by Lavielle (1999), Adams and MacKay (2007) and (Verbesselt et al., 2010b) after Haywood and Randall (2008). Here we estimate the variance change points of the NDVI timeseries using the method of Lavielle (1999). The method assumes that the data follows a normal distribution and detects simultaneous changes in the mean and variable of the distribution but requires no assumptions about the dependence structure of the time series. After the detection of breaks, the NDVI time series for each of the Study Sites is split into a seasonal and trend component and compared against events and behaviors[1] that may have disturbed the system in some way (e.g. fire, drought, land use change). While trends are easier to detect and attribute to a persistent driver, phonological events are difficult to precisely identify with remote sensing images, so researchers tend to prefer arbitrary classifications (e.g. Eastman et al., 2013). Here we make two main assumptions following Lu et al. (2003) and Prabakaran et al. (2013). The first is that land covers dominated by shrubs and herbaceous vegetation or crops that follow an annual cycle are responsible for the seasonal NDVI signal while land dominated by woody and evergreen vegetation is mainly responsible for the deseasonalised signal. The second assumption is that deciduous forests display an earlier onset of greenness, followed by evergreen forests.

2.3 Grazing

The predator-prey system (Noy-Meir, 1975) typically serves as a basis for models that describe the dynamics of grazing systems. The rate of change of the harvested biomass P (standing crop) is represented by the differential equation:

$$\frac{dP}{dt} = G(P) - C(P) \qquad (3)$$

where $G(P)$ describes the plant growth as a function of the plants biomass and $C(P)$ is the loss rate of biomass due to consumption by herbivores. $G(P)$ is commonly given by the logistic growth equation $G(P) = aP(1 - P/K)$, where a is the plant growth coefficient (the intrinsic growth rate) and K the carrying capacity of the land (the maximum amount of resource stock when there is no harvest activity). From the $G(P)$ equation it is inferred that the relative natural growth $G(P)/P$ is a linear function of the biomass level P and approaches its maximum, equal to a, when P approaches zero. Parameter a is mainly related to the actual species under observation while K depends mainly on the natural environment of the stock, such as size and biological productivity of the habitat.

According to the Gordon Schaefer model (Schaefer, 1957), losses due to grazing can be considered proportional to both biomass (relies on the existing stock of renewable resource at a given period) and herbivore density, giving $C(P) = bHP$, where b is the

[1] Events and behaviors discussed here have been recovered from Deliverable 2.1 of CASCADE.

biomass consumption coefficient. It can be argued that the land's carrying capacity K and plant growth coefficient a are related to the soil quality and the climate conditions, meaning that these parameters are under stress in arid areas. The same is not true for grazing, as the consumption coefficient b possibly increases as nutritional needs may change under pressure (Ayres, 1993; Robbins et al., 1987) and herbivores can be quite effective at grazing.

Following from the above, the rate of change of the harvested biomass P from Equation 1 can be written:

$$\frac{dP}{dt} = aP(1 - P/K) - bHP \tag{4}$$

For $G(P) > C(P)$ the natural growth of the resource is greater than the harvest amount. As a result, the resource stock P will increase. The result is opposite when the harvest amount exceeds the natural growth of the resource. Figure 2 shows vegetation growth and consumption for a system with initial biomass x_0, plotted against time. The effect of increasing herbivore densities on the system is depicted with lighter lines.

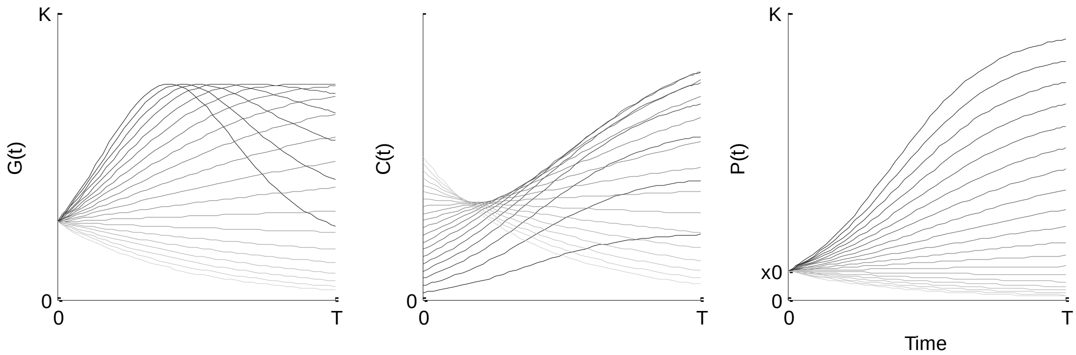

Figure 2: Effect of herbivore density on above ground vegetation biomass, growth of vegetation and its consumption by herbivores. Lighter lines denote higher herbivore density.

Equation 4 can reach two possible states of equilibria $(dP/dt = 0)$. If $P(t) = 0$ then the vegetation resource has been depleted, eventually leading to the reduction of soil services along the mechanisms of the desertification paradigm downward spiral (Adeel et al., 2005; Hassan et al., 2005). On the other hand, sustainable use also implies that the net biomass production is zero but this time the ecosystem can maintain a steady level of biomass stock, such that:

$$0 = aP(1 - P/K) - bHP \tag{5}$$

Solving Equation 5 for P, we get $P = (a - bH)\,K/a$ which means that the biomass and eventually the growth rate comprises a function of H. If the vegetation growth does indeed follow the logistic function for growth then the maximum vegetation biomass yield is equal to $aK/4$, also called Maximum Sustainable Yield (MSY), at which point the herbivore density is equal to $a/2b$. Keeping all other parameters stable along the MSY path, as the herbivores density H increases the biomass production rate can be insufficient to cover it until H reaches the critical value a/b and the system deteriorates.

Substituting G_p with the logistic growth equation, Equation 2 can be transformed to:

$$G_a(P) = \left[1 - K_G\left(1 - \frac{ET_a}{ET_p}\right)\right] aP\left(1 - \frac{P}{K}\right) \tag{6}$$

which infers that G_a is maximized as ET_aapproaches ET_p. Taking into account the effect of climate on vegetation growth and assuming zero net biomass production, Equation 6 is transformed to:

$$0 = \left[1 - K_G\left(1 - \frac{ET_a}{ET_p}\right)\right] aP\left(1 - \frac{P}{K}\right) - bHP \tag{7}$$

As in Equation 5, the maximum vegetation biomass yield is again equal to $aK/4$ but this time it is achieved for $H = \left[1 - K_G\left(1 - ET_a/ET_p\right)\right] a/2b$, so it is also related to climatic conditions (Daliakopoulos and Tsanis, 2014). Based on these empirical relationships, a robust qualitative concept of resilience emerges. While modeling within CASCADE may follow a more quantitative approach, this concept roughly explains the principals of the interaction between vegetation, climate and herbivores. Similar concepts are described by Jeltsch et al. (2014) and Rietkerk et al. (1997, 1996).

2.4 Unifying framework

A generally accepted framework for the description of catastrophic shifts is that of the fold catastrophe (Figure 3). This framework describes the existence of more than one stable state paths of the state of an ecosystem. Assuming that the conserved state of an ecosystem also represents its desirable state, certain disturbances caused by unfavorable external conditions can cause the system to destabilize towards a degraded state. The maximum disturbance that can be absorbed by the system is considered as the system resilience. Knowing the exact magnitude of the system resilience a priory is rather challenging as it usually depends on feedback loops between biotic and abiotic components of the ecosystem.

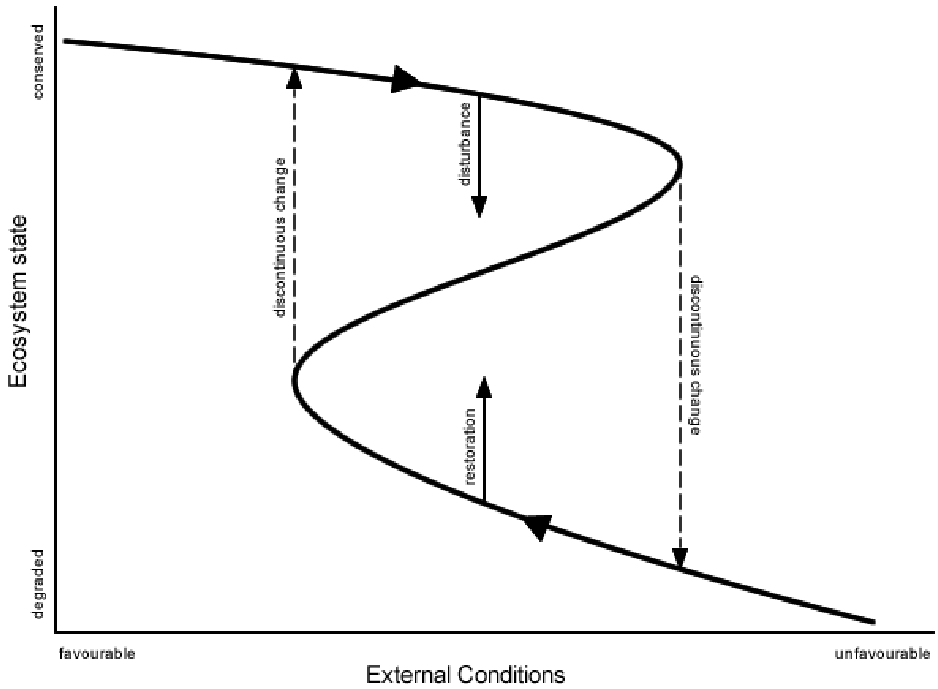

Figure 3: A function that shows a fold catastrophe. The outside branches are stability attractors whereas the inner fold is a metastable repellor.

As long as the disturbance stays within the resilience, the system has the capacity to assume its previous conserved state when external conditions become favorable. When system resilience is exceeded, then the system jumps to a new state through a discontinuous change. This degraded state is also stable in the sense that it also presents a new resilience against external conditions. In many cases, applying extremely favorable conditions (e.g. via a restoration attempt) may not be enough to cause the system to return to the initial desirable state but will instead lead to an alternative situation (not shown). The reason is that the shift from desirable state to alternative state is often accompanied by a loss of resources, and that the restoration attempt might not result in sufficient recovery of these resources to allow a shift back to the original desirable state. This alternative path is a form of system adaptation or an entirely different system which is not always desirable as it may hold inferior ecosystem value than the initial system.

While the above framework explains a one dimensional external input well, ecosystems usually depend on more than one variable and behave in more complex ways. Especially for dryland systems, buffering mechanisms that keep them within certain boundaries are also considered (e.g. Jeltsch et al., 2014, 2000), e.g., the absence or abundance of a resource may prevent transition to a conserved or degraded state respectively. This dependence of an ecosystem to more than one variables or system states can be depicted with the use of the cusp catastrophe (Figure 4).

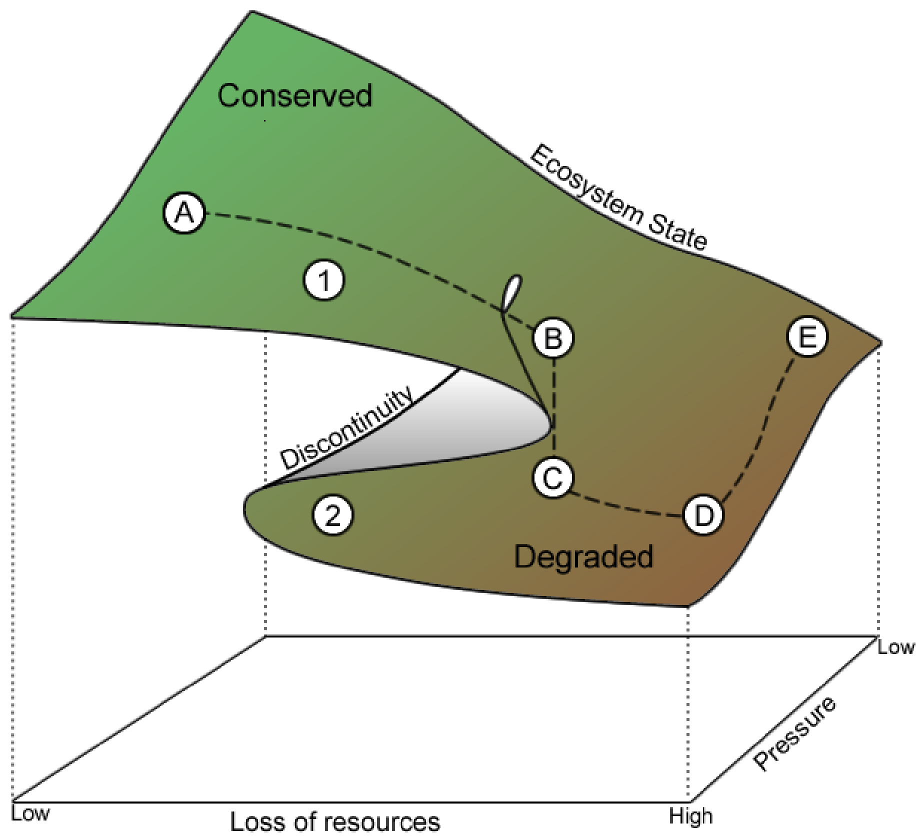

Figure 4: Conceptual framework of catastrophic shifts in land degradation due to an anthropogenic pressure (e.g. grazing, fire frequency) and the variability of resources.

In Figure 4, A represents an area where a driver is causing pressure to the ecosystem which nevertheless retains a good status and high resources. Here the system is in stable state 1, and can maintain this state regardless of the grazing pressure due to its resilience. As resources become depleted the system reaches the tipping point (B) where two alternative stable states (1 and 2) can co-exist. The transition between B to C is very quick as processes that take place are exponential. An example is a grazing system where the rate of consumption becomes significantly higher than the rate of biomass production leading to collapse.. If resources are depleted further (D), transitioning back to C or B may require some kind of effort, especially when drivers and human induced. More importantly, eliminating the exerted pressure will drive the system to an alternative state E rather than back to its pristine condition. Therefore it is possible that the system becomes "trapped" in this alternative state, especially if resources at hand are non-renewable (e.g. soil) and their loss cannot be amended within a reasonable timeframe (e.g. the human lifespan). Recovery from E may be more gradual as resources are recovered provided pressures are controlled.

The cusp catastrophe concept and variations can be adopted for different ecosystems or selected ecosystem health indicators. For example a grazing system that simultaneously faces overgrazing and depletion of water resources can be driven to prolonged periods of no vegetation cover and eventually to desertification. Similar transitions can take place between different states of a forest where the combined high fire frequency and loss of resources (e.g. seed bank or soil) can lead to changes in the phenology of the vegetation and ultimately to a stable shrubland. This framework explains the relevant stability of natural systems when external (e.g. anthropogenic) pressure is absent or minimal and resources are not depleted. Indeed, in the case of a grazing system the cusp catastrophe model predicts continuous changes of vegetation biomass when e.g. water availability decreases. On the other hand, system resilience (i.e. the magnitude of disturbance the system can absorb) decreases as grazing intensity increases towards overgrazing. In a similar manner, water resources and nutrient depletion may cause continuous changes in the vegetation biomass of a forest as long as the fire frequency remains low. At higher fire frequencies the cusp model predicts a hysteresis.

Beyond the cusp catastrophe theory, the CASCADE Project is carrying out microcosm, mesocosm and landscape experiments that can reveal vegetation interactions at finer scales and possibly common processes across the CASCADE Case Studies. In this context, Remote Sensing derived vegetation indices have a high potential for assessing vegetation activity, as a result of both climatic and human induced drivers. Deliverable 2.1 highlighted that seasonal vegetation dynamics of NDVI in each Study Site is highly relevant to the seasonal sequence and the perceived length of winter versus the length and severity of summer drought. Therefore, Várzea and Castelsaraceno show higher vegetation activity around the end of spring (May), whereas the rest of the Study Sites have a preference for early spring, when temperatures begin to rise and water availability is still not an issue. Among the Study Sites, by far the poorest vegetation state can be found in Randi Forest, followed by Albatera and Messara. A general conclusion from the analysis of chlorophyll reflectance from satellite imagery is that all Study Sites show only minor trends in the NDVI index, with the exception of Albatera where a strong correlation seems to exist. Here we will try to extrapolate from this finding and consider finer scales to derive meaningful correlations.

3 Results

3.1 Historical Climate

In the domain investigated herein, most stable European Mediterranean drylands are located in South and East Spain (ES6 & ES5), Insular Italy (ITG), the Aegean Islands and Crete (GR4) and Cyprus (CY0). Regarding the CASCADE Study Sites for the period 1961-2010, Alicante has been Arid, Valencia, Messara and Randi Forest have been stably Semi-arid, Castelsaraceno has been Dry subhumid and Varzea Subhumid (Figure 5). Overall, the CASCADE Study Sites are arid to some degree, with Alicante representing the lowest and Varzea the highest part of the aridity gradient. From the 50 year record, it is unclear whether permanent aridity regime shifts are taking place in the European Mediterranean drylands. Nevertheless, the entire domain under consideration has been undergoing an overall increase in the percent of arid area by almost 15% (from 64% to 78%) in dispense of more humid aridity classes, while arid area has remained more or less stable (Figure 6).

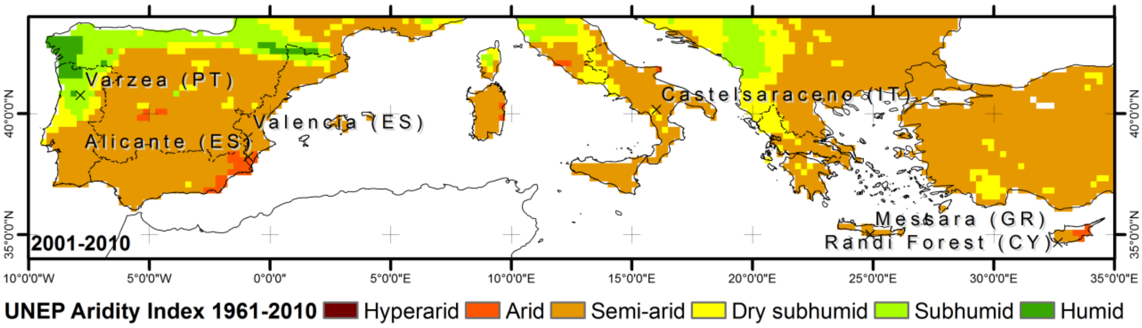

Figure 5: The UNEP Aridity Index averaged over the period 1961 – 2010 for a domain ranging latitudes -10 – 35 and longitudes 34 – 44 excluding areas with poor data availability.

Observing the trend of aridity over the Mediterranean in decadal terms, Varzea has been getting steadily drier, gradually crossing from Humid to Subhumid and Dry-subhumid. This trend can also be observed for the entire northeast Atlantic Ocean coastline within the investigated domain, along the Cantabrian Mountains. A similar aridification can be observed along the axis of the Pindus Mountain range in Greece and the Appennini Mountains in Italy during the past 50 years (Figure 7).

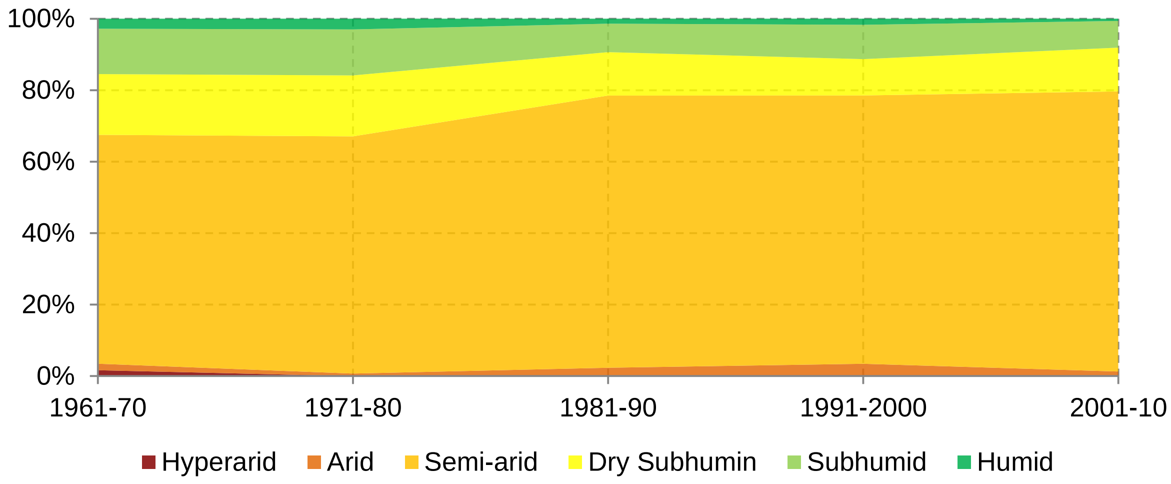

Figure 6: Annual evolution of the area covered by different classes of the UNEP Aridity Index for the period 1961 to 2010 for a domain ranging latitudes -10 – 35 and longitudes 34 – 44.

Comparing today (2010) with the past four decades, the 1960s have been one class less humid for Mainland Portugal (PT1) and Northern Spain (ES1, ES2) and at least two classes less humid for Central Greece (GR2). North West and Central Italy (ITC, ITE) also had some more humid decades during the same period. Alicante, located in one of the most persistently arid areas of the investigated domain, fluctuates between Arid and Semi-arid along with Central Italy (ITE), Cyprus and Crete. Focusing on the CASCADE Study Sites, results indicate that Várzea (representing Mainland Portugal – PT1) may be currently experiencing a regime shift from a predominantly humid area to subhumid or marginally dry-subhumid state. Castelsaraceno (South Italy – ITF) is fluctuating between Dry-subhumid and Semiarid states, whereas Alicante (South Spain – ES6) and Valencia (East Spain – ES5) are fluctuating between Arid and Semiarid states. Finally, Messara (located in Aegean Islands and Crete – GR4) and Randi Forest (Cyprus – CY0) have a stably Semi-arid status.

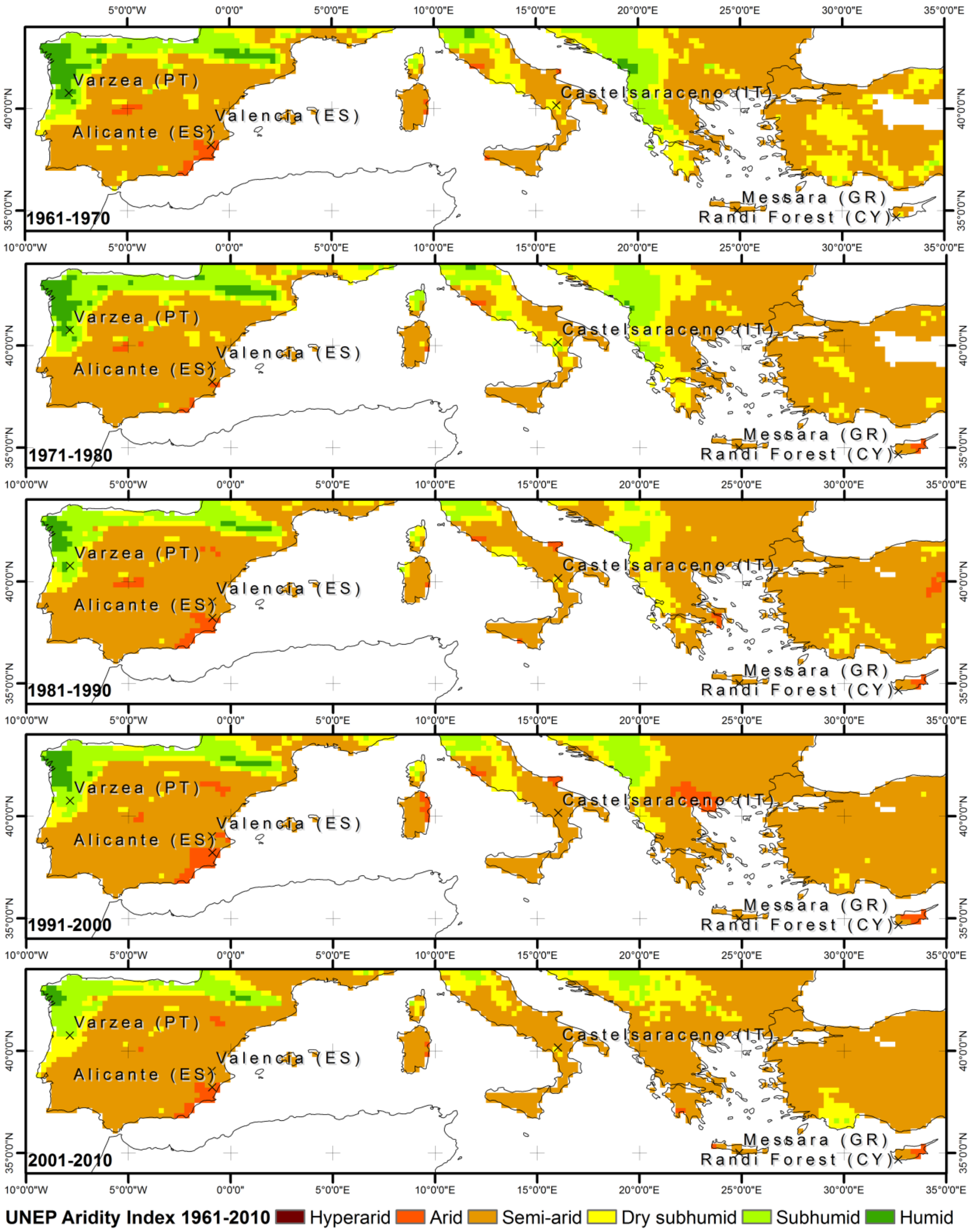

Figure 7: The UNEP Aridity Index estimated over 5 decadal periods from 1961 to 2010 for a domain ranging latitudes -10 – 35 and longitudes 34 – 44.

3.2 Future Climate

For ease of presentation and discussion, climate model results are averaged over the investigated domain for annual, decadal and 50-year time intervals. Here we do not consider uncertainties due to model averaging. Therefore the variability of future climate appears smaller than that of observations presented in the previous chapter. While this is an oversimplification it depicts well projected trends. According to RCP26, the picture of Aridity in Southern Europe does not change significantly (Figure 8). Aridity hotspots such as those in Spain, Italy and Cyprus remain more or less stable for the terms 2011 – 2050 and 2061 – 2100, relative to the reference period 1961 – 2010. Figure 9 depicts this stability, with the semiarid areas remaining at an average of 76% of the total, after the abrupt 14% increase encountered during the reference period. The same applies for arid areas that steadily cover less than 2% of the examined domain.

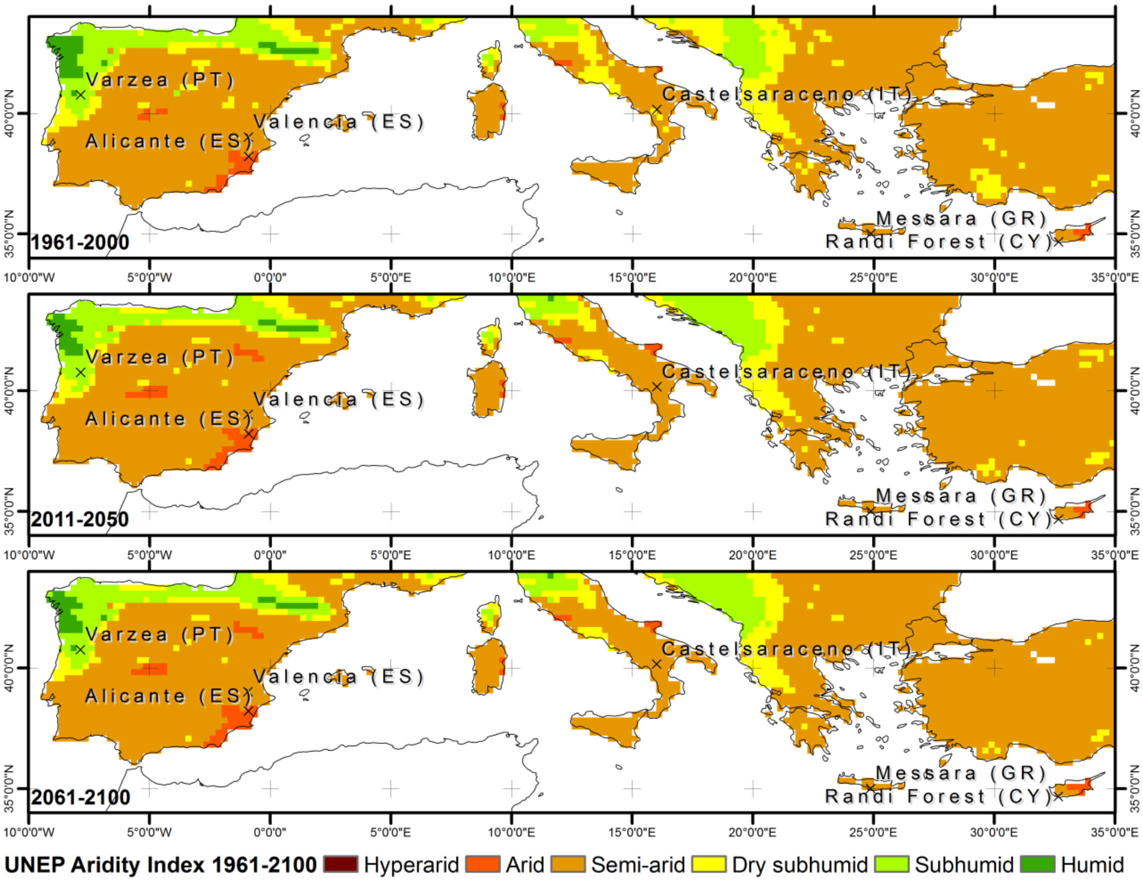

Figure 8: The UNEP Aridity Index averaged over the reference period 1961 – 2010 and periods 2011 – 2050 and 2050 – 2100 under RCP26 for a domain ranging latitudes -10 – 35 and longitudes 34 – 44.

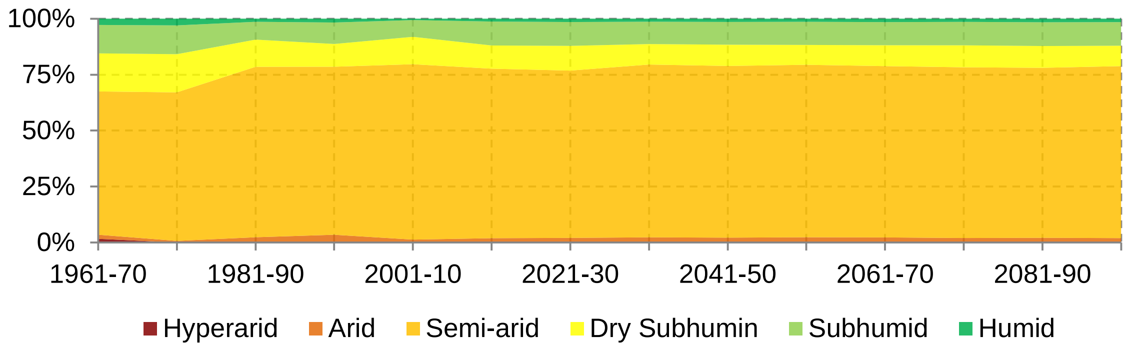

Figure 9: Annual evolution of the area covered by different classes of the UNEP Aridity Index for the period 1961 to 2100 for a domain ranging latitudes -10 – 35 and longitudes 34 – 44 under RCP26.

On the contrary, RCP45 predicts that semi-arid and arid areas gradually increase to reach more than 77% by 2050 and 78% by the end of 2100. Semiarid areas over Italy and Greece gradually expand while arid zones in Spain and Cyprus take alarming extents (Figure 10). During the course of the century, the pathway projects a pronounced decrease of the humid zone from 3% in the 80s to 1% in 2100 (Figure 11), affecting primarily the northeast Atlantic Ocean coastline. The cases of Várzea, Valencia and Castelsaraceno are very characteristic, with all of them steadily becoming one class more arid.

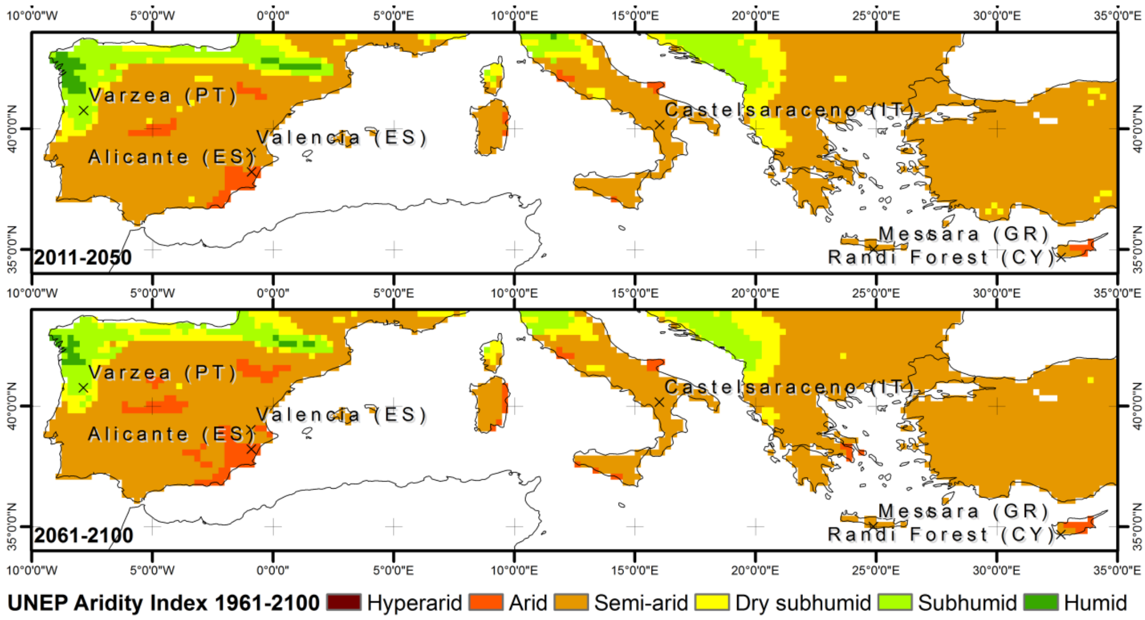

Figure 10: The UNEP Aridity Index averaged over periods 2011 – 2050 and 2050 – 2100 under RCP45 for a domain ranging latitudes -10 – 35 and longitudes 34 – 44.

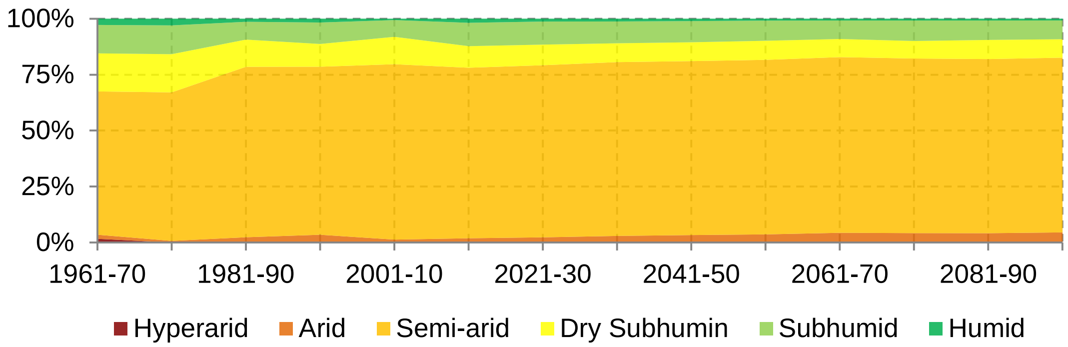

Figure 11: Annual evolution of the area covered by different classes of the UNEP Aridity Index for the period 1961 to 2100 for a domain ranging latitudes -10 – 35 and longitudes 34 – 44 under RCP45.

Finally, RCP85 presents the least optimistic realization with aridification reaching extremes, particularly over Spain, at the end of the century (Figure 12). Overall, Study Site areas that during the reference period belonged to a more humid zone (Várzea, Valencia and Castelsaraceno) appear to be more sensitive to change than those already arid (Alicante, Messara and Randi Forest).

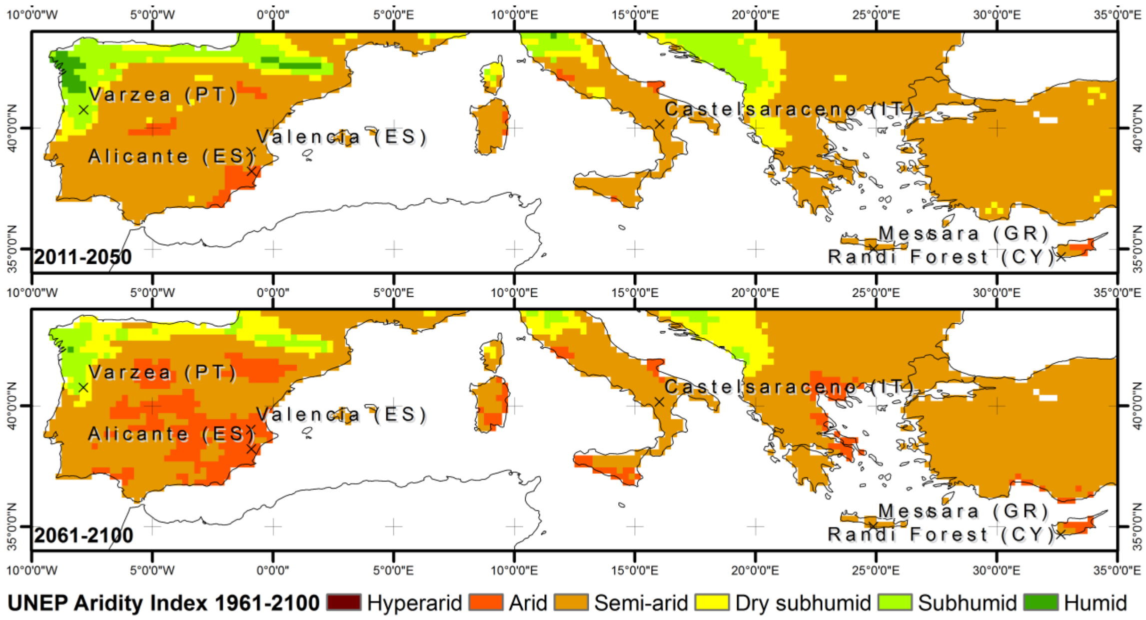

Figure 12: The UNEP Aridity Index averaged over periods 2011 – 2050 and 2050 – 2100 under RCP85 for a domain ranging latitudes -10 – 35 and longitudes 34 – 44.

Humid zones become negligible by 2050, but more importantly, even subhumid zones cover less than 3% by the end of 2100 (Figure 13). After 2050, arid zones gradually cover as much as a devastating 20% of Southern Europe.

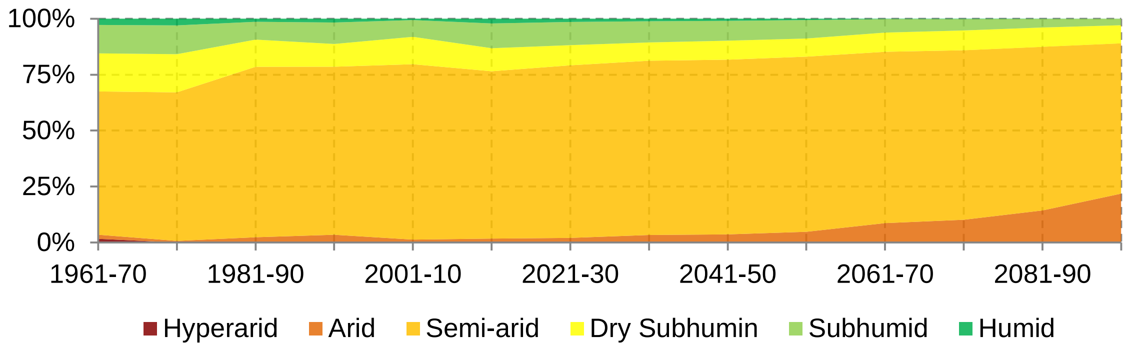

Figure 13: Annual evolution of the area covered by different classes of the UNEP Aridity Index for the period 1961 to 2100 for a domain ranging latitudes -10 – 35 and longitudes 34 – 44 under RCP85.

3.3 Sensitivity of vegetation to climate

The expected change of the ET_a/ET_p ratio for a theoretical range of mean Temperature T of 10 to 30^{o}C and annual precipitation R of 100 to 1500mm is illustrated in Figure 14. The values of the Blaney-Criddle equation are estimated for latitude 32^o. At first glance, low temperatures appear to limit the maximum achievable ET_a/ET_p ratio, which converges to lower values as precipitation increases. Also, for higher annual precipitation values (e.g. over 900mm), the expected temperature causes a sharp increase of ET_a/ET_p, at least up to a certain threshold depending on the respective precipitation value (vector a in Figure 14). Nevertheless, for moderate and lower annual precipitation values (e.g. under 700mm) the fraction of ET_a/ET_p decreases as mean temperature rises (vector c in Figure 14). A notable finding is that this system of equations clearly shows a transition from a moderately unstable to a highly unstable equilibrium as temperature rises. This is highlighted by the observation that for moderate R values (e.g. 500-700mm), a small change in precipitation can cause significantly larger changes in ET_a/ET_p as temperature increases (vector b in Figure 14).

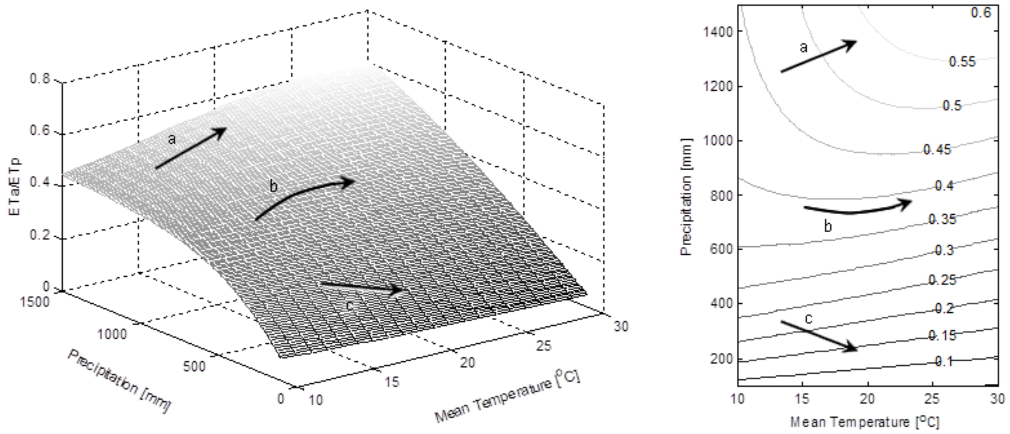

Figure 14: Effect of precipitation and mean temperature change on the fraction of actual over potential evapotranspiration after Turk (Turc, 1961) and Blaney-Criddle (Blaney and Criddle, 1962) at latitude 32°.

Regarding the effect of latitude on the model, the Blaney and Criddle equation predicts that a decrease of the mean daily percentage p of total daytime hours that takes place approaching the equator increases, the value of the ET_a/ET_p. Figure 15 depicts this change, also showing the limits that envelop the ratio ET_a/ET_p for latitudes 0°-50° (contour labels shown only for 50°). For small values of annual R, the effect of latitude can be negligible, showing that rainfall is the most important limiting factor. As precipitation increases, temperature and sunlight have a more decisive role in the estimation of crop yield. In relation to Figure 14 (left) the effect of increasing latitude actually offsets the surface of ET_a/ET_p higher, suggesting that higher latitudes will experience the effects of climate change in a lesser degree.

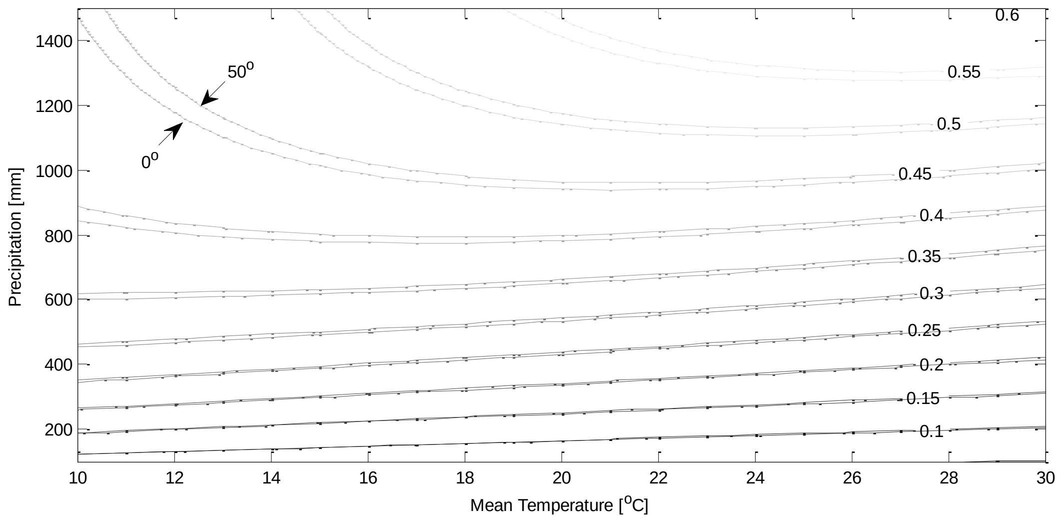

Figure 15: Latitude increase (from 0° to 50°) decreases the value of the fraction of actual over potential evapotranspiration.

3.3.1 Várzea Study Site (PT)

According to the NDIV analysis for the Várzea Study Site, during the periods 1982-1990 and 1996-2003, trends appear moderate (Figure 16). On the other hand, during 1991-1995 greenness shows two consecutive breaks that coincide with the onset and end of a prolonged drought event (SPI48 in Figure 16). Following this event, the trend of the deseasonalised component of the NDVI regains its stability, whereas the seasonal component experiences a reduction in amplitude. A closer look at the NDVI seasonality before and after the 1991-1995 drought period (Figure 17) reveals a pronounced variation at the onset of greenness that also appears about 1 month earlier (February versus March). The variation in the greenness peak is less pronounced but the one-month offset remains (July versus August). Prabakaran et al. (2013) concluded that deciduous forests display an earlier onset of greenness compared to evergreen forests. The early onset could also show signs of annuals that disappear with warmer weather (May to June). Nevertheless, the area appears to be green for a longer period. These observations may signify a phenological change in the Várzea forest. Projected changes in the climatic regime (Figure 10, Figure 12 and Figure 14), combined with the current management approaches will pose an extra risk at the already stressed status of the original Várzea pine forest.

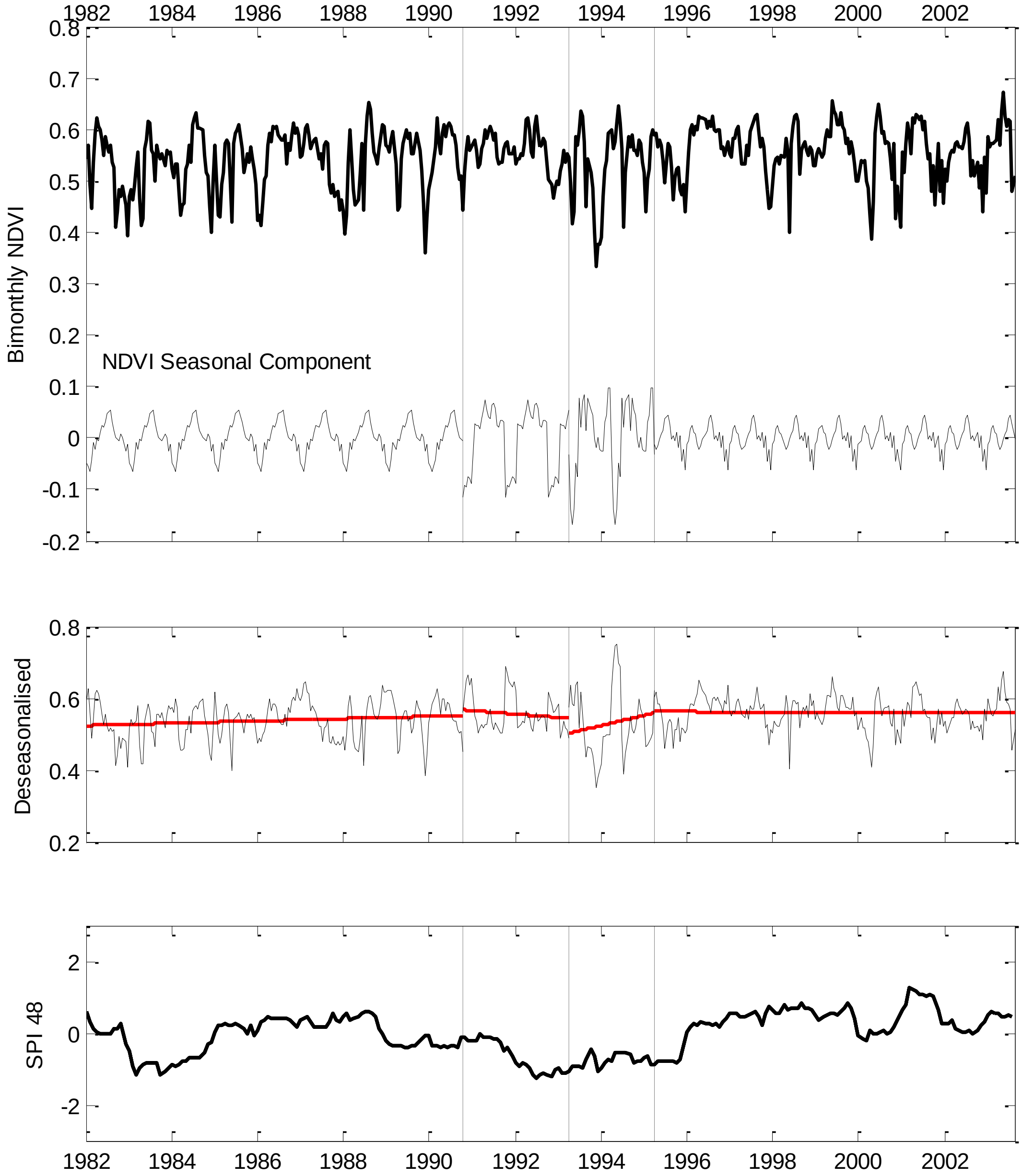

Figure 16: Bimonthly NDVI, broken down into seasonal and deseasonalised components, against SPI48 for the period 1982 – 2003 for the Várzea Study Site.

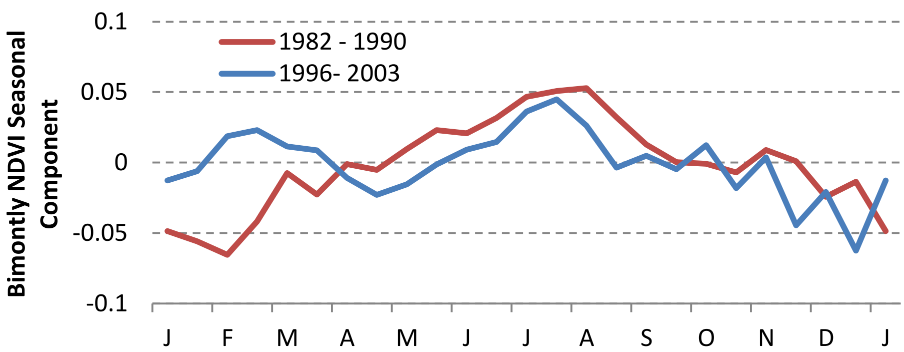

Figure 17: NDVI Seasonal component differences for the periods before and after trend shifts for the Várzea Study Site.

3.3.2 Alicante Study Site[2] (ES)

In Alicante, a brake in the otherwise moderate trend is detected between 1995 and 1997. According to the established timeline of events (see CASCADE D2.1), no significant disturbances appeared during that time. Nevertheless, the period coincides well with the second most prolonged drought in the area after 1982[3]. Therefore it is possible that the driver of change in the area is connected to climate and predominantly precipitation and water deficit. From Figure 19 it is clear that winters and summers have become more pronounced after 1997, following milder transitions before 1994. This change may signify a transition from a system with constant water availability to one that is seasonally stressed.

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[2] In Deliverable 2.1, the Alicante site is referred to as Albatera.
[3] From the available climatic dataset (see D2.1), the most persistent drought in the Valencia site took place between 1963 and 1965, nevertheless, NDVI measurements don't exist for that period.

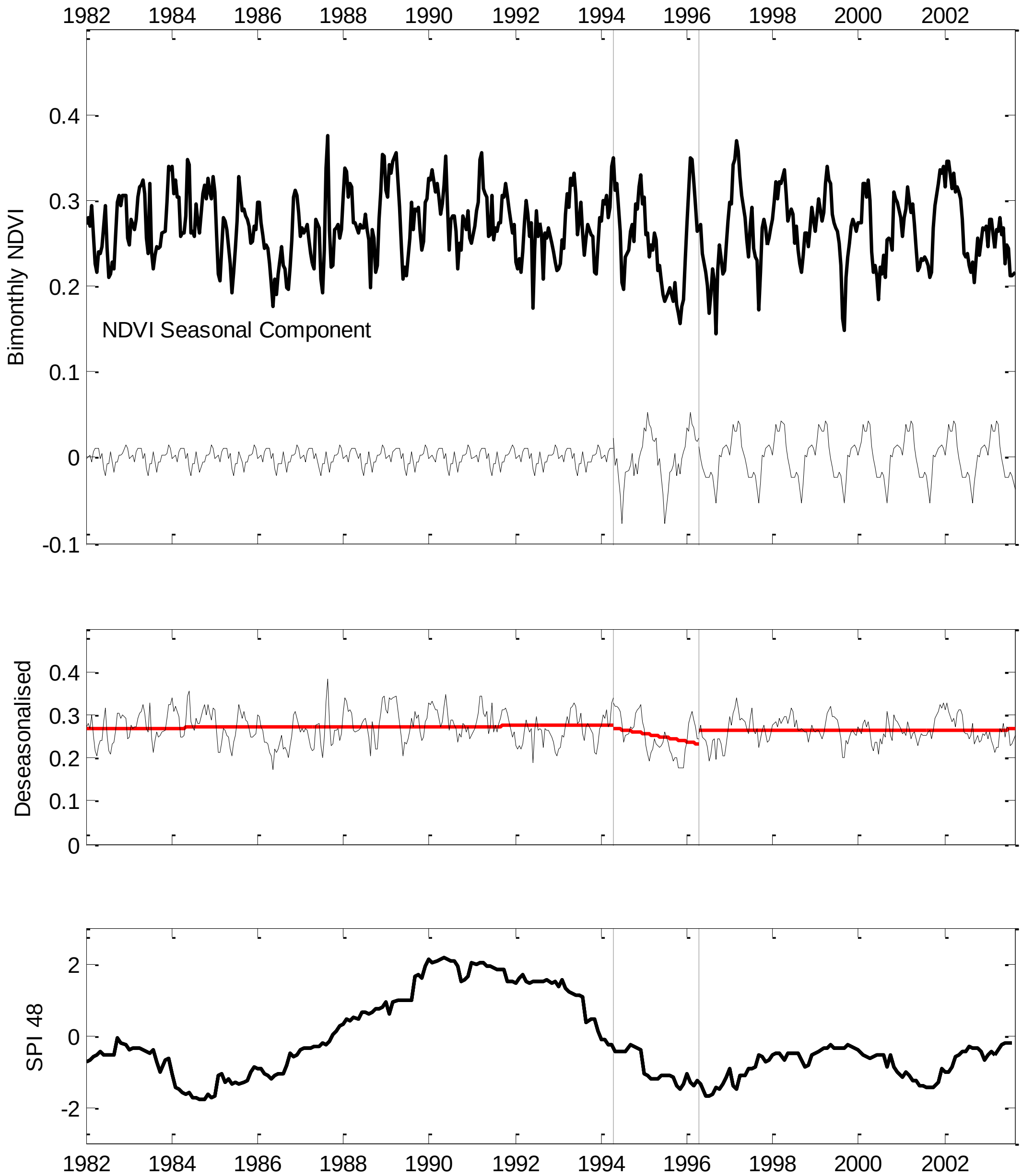

Figure 18: Bimonthly NDVI, broken down into seasonal and deseasonalised components, against SPI48 for the period 1982 – 2003 for the Alicante Study Site.

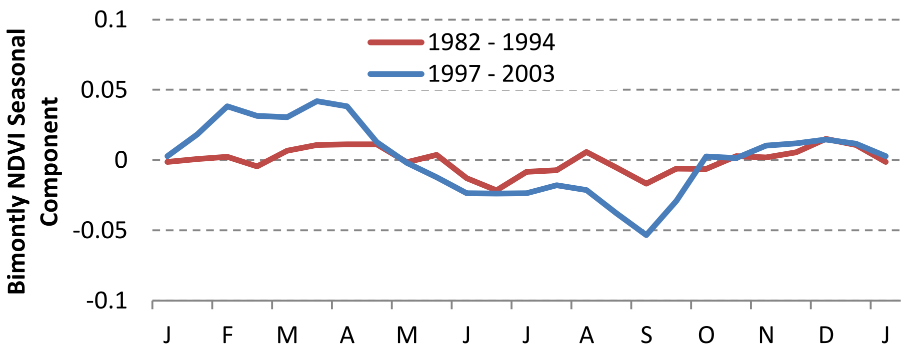

Figure 19: NDVI Seasonal component differences for the periods before and after trend shifts for the Alicante Study Site.

3.3.3 Valencia Study Site[4] (ES)

The NDVI trend analysis for the case of Valencia reveals a break in the increasing trend that is exhibited until 1996, followed by a decrease in the average NDVI, along with a pronounced change in the variability (Figure 20). Figure 21 shows the change of the seasonal component, with a single peak in early summer and minimum values during the winter preceding the break. On the other hand, after the trend break two peak and two minimum greenness seasons appear. According to literature (e.g. Bradley et al., 2007; Maselli, 2004), both these profiles are typical of Mediterranean woody vegetation, but a single peak may signify dependence on water availability and favorable temperatures while shrublands have minimal photosynthetically active vegetation cover and thus change in NDVI within a season is low. The timing of the seasonality change follows the occurrence of several fires (e.g. in 1979, 1984, 1991, 1994 – see CASCADE D2.1), that according to Lauterburg (2014) have resulted to a regime shift from pine forest to seeder shrubland. In addition to this observation, the timing of the seasonality change also coincides with one of the few emerging drought events in the area (SPI48 in Figure 20), highlighting the incremental effect of aridity, loss of resources and fire frequency as illustrated in the cusp catastrophe model. It is therefore possible that the Forest Law 3/1993 (see D2.1) was established too late to control a degradation process that was already under way.

[4] In Deliverable 2.1, the Valencia site is referred to as Ayora.

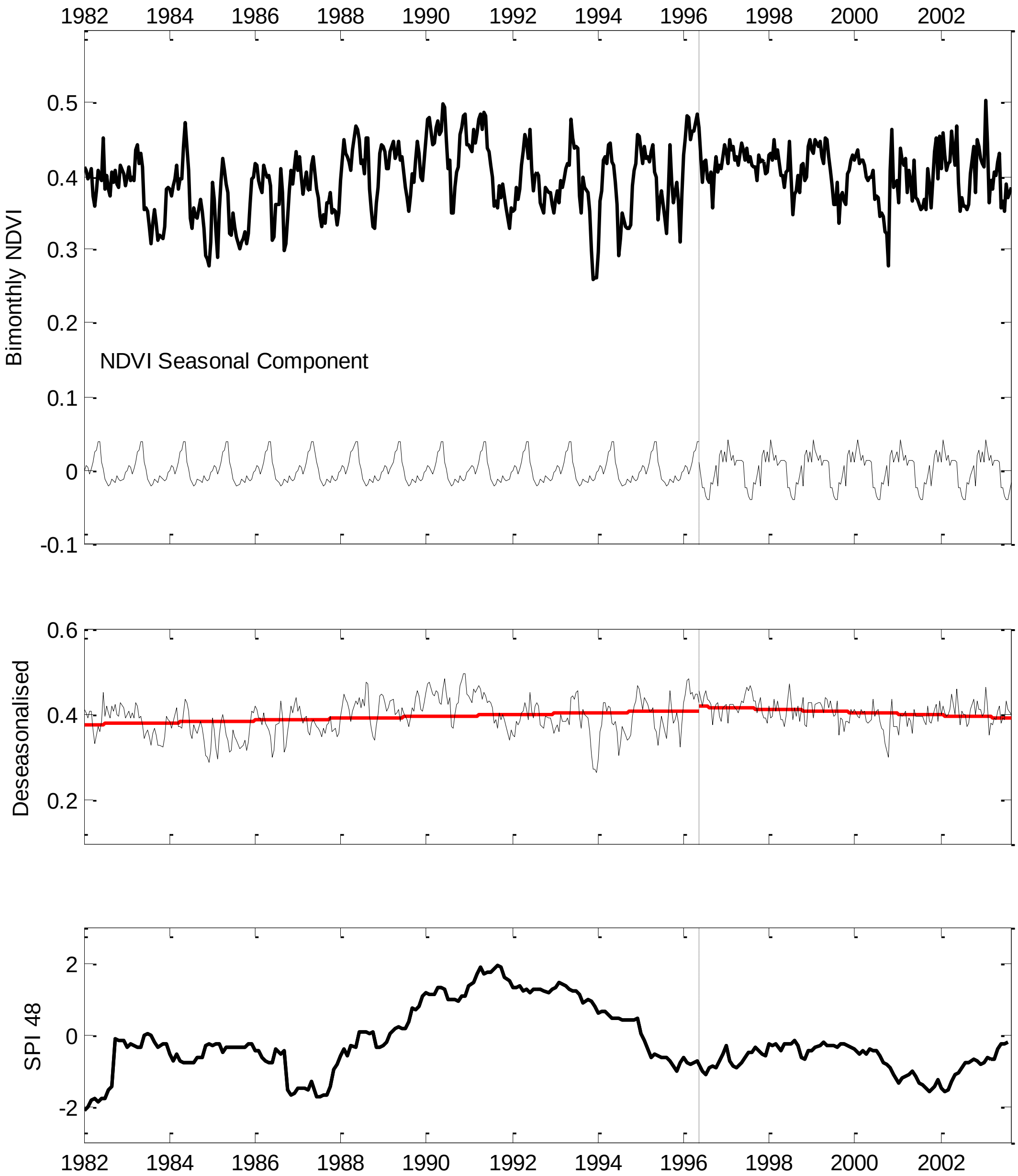

Figure 20: Bimonthly NDVI, broken down into seasonal and deseasonalised components, against SPI48 for the period 1982 – 2003 for the Valencia Study Site.

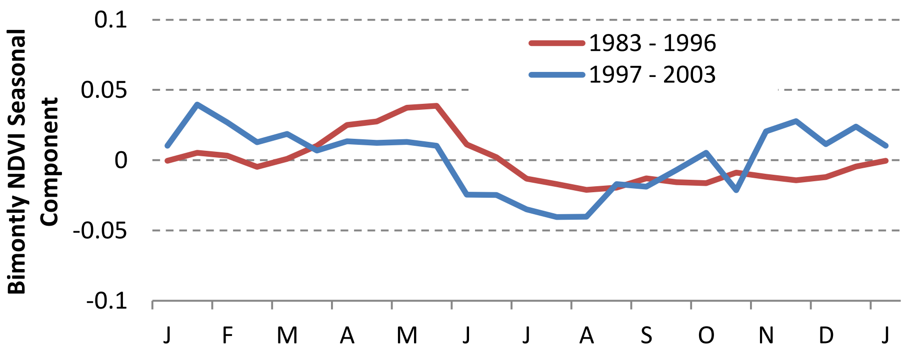

Figure 21: NDVI Seasonal component differences for the periods before and after trend shifts for the Valencia Study Site.

3.3.4 Castelsaraceno Site (IT)

In the case of Castelsaraceno, the stable characteristics of the deseasonalised component on the NDVI (Figure 22) show little relevance with the steep trend of the SPI48 until 2003[5], implying that the climate of the region can sustain healthy vegetation. Nevertheless, a shorter greenness season during the period 1997-2003 versus 1985-1990 may be the result of dryer summers, causing an earlier recede of herbaceous vegetation. Climate change is bound to aggravate these conditions, with higher temperatures and increased aridity stressing vegetation intended for grazing.

[5] The pronounced trend in the SPI 48 values is counterweighed by wetter periods between 2003 and 2010, as presented in Deliverable 2.1. Here this period is not shown for consistency with other Study Sites.

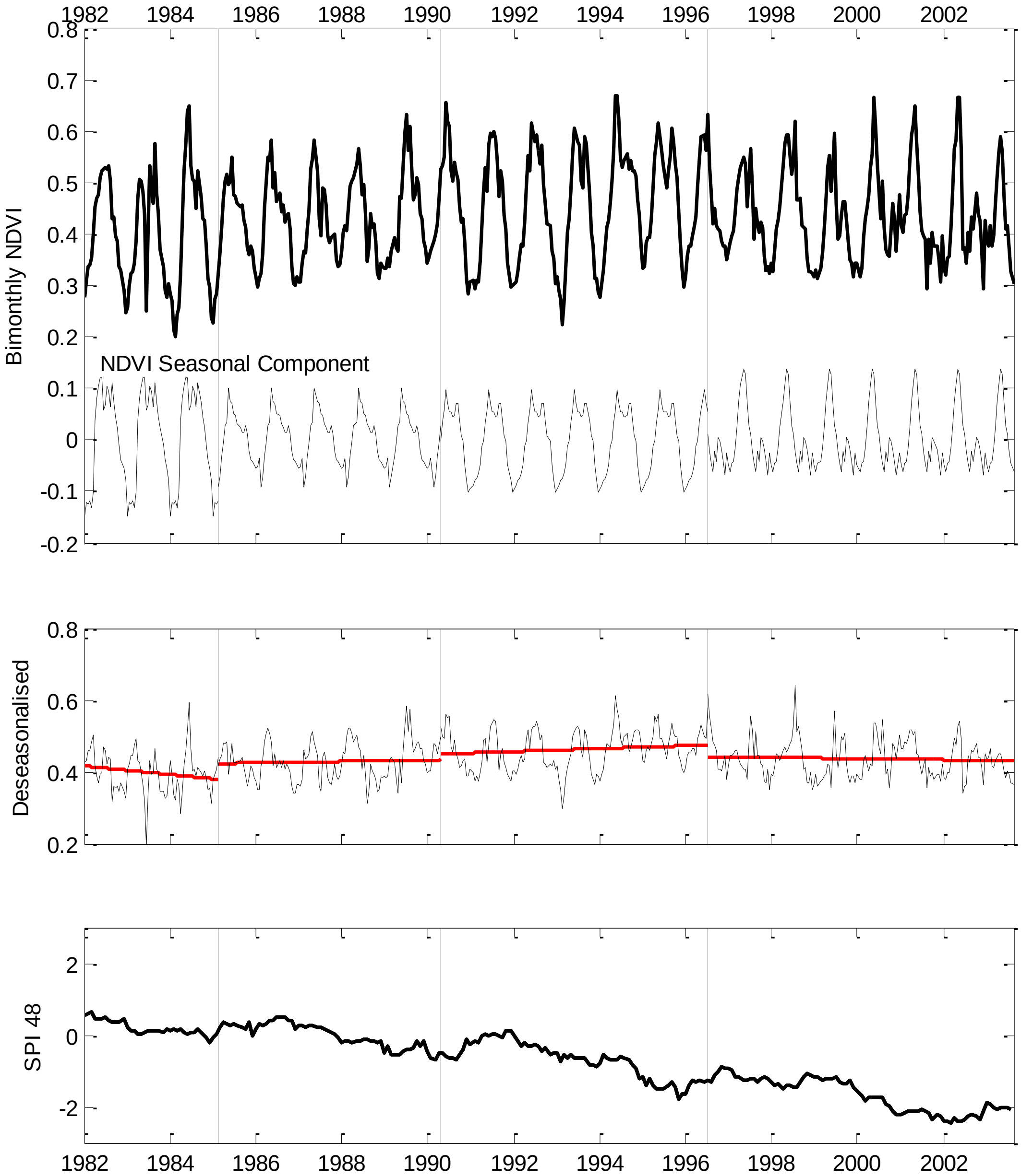

Figure 22: Bimonthly NDVI, broken down into seasonal and deseasonalised components, against SPI48 for the period 1982 – 2003 for the Castelsaraceno Study Site.

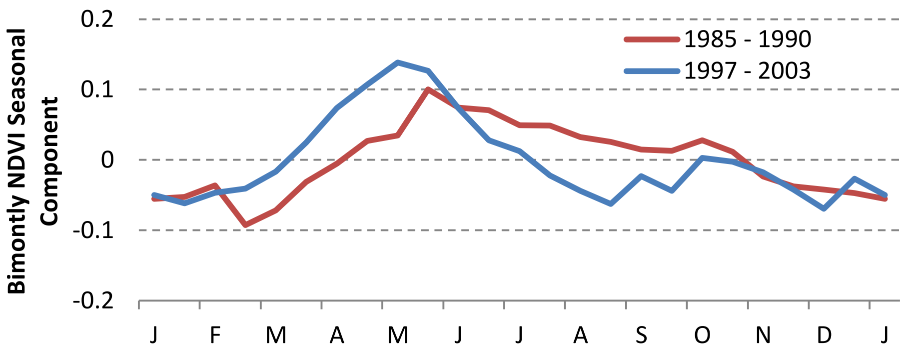

Figure 23: NDVI Seasonal component differences for the periods before and after trend shifts for the Castelsaraceno Study Site.

3.3.5 Messara Study Site (GR)

In the Messara Study Site, the succession of dry years, combined with mismanagement in the resources depletion intensity (e.g. water abstraction, grazing) have caused marked brakes in the greenness trends (Figure 25). A significant disturbance takes place two years after the manifestation of the minimum SPI48 value, in 1995. After the end of this pronounced drought, the deseasonalised NDVI signal increases following a stabilization of the SPI48. Regarding the seasonal component of the NDVI, few changes occur between periods 1985-1991 and 2000-2003. The small dependence on climatic factors can be attributed to the high degree of agricultural activities within the investigated location. Here, water resources and optimal growing conditions for vegetation nearly all year long and can account for the relatively stable NDVI during summer months.

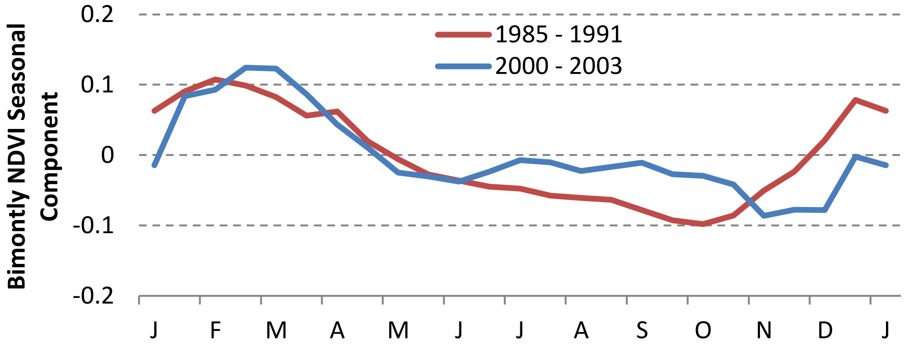

Figure 24: NDVI Seasonal component differences for the periods before and after trend shifts for the Messara Study Site.

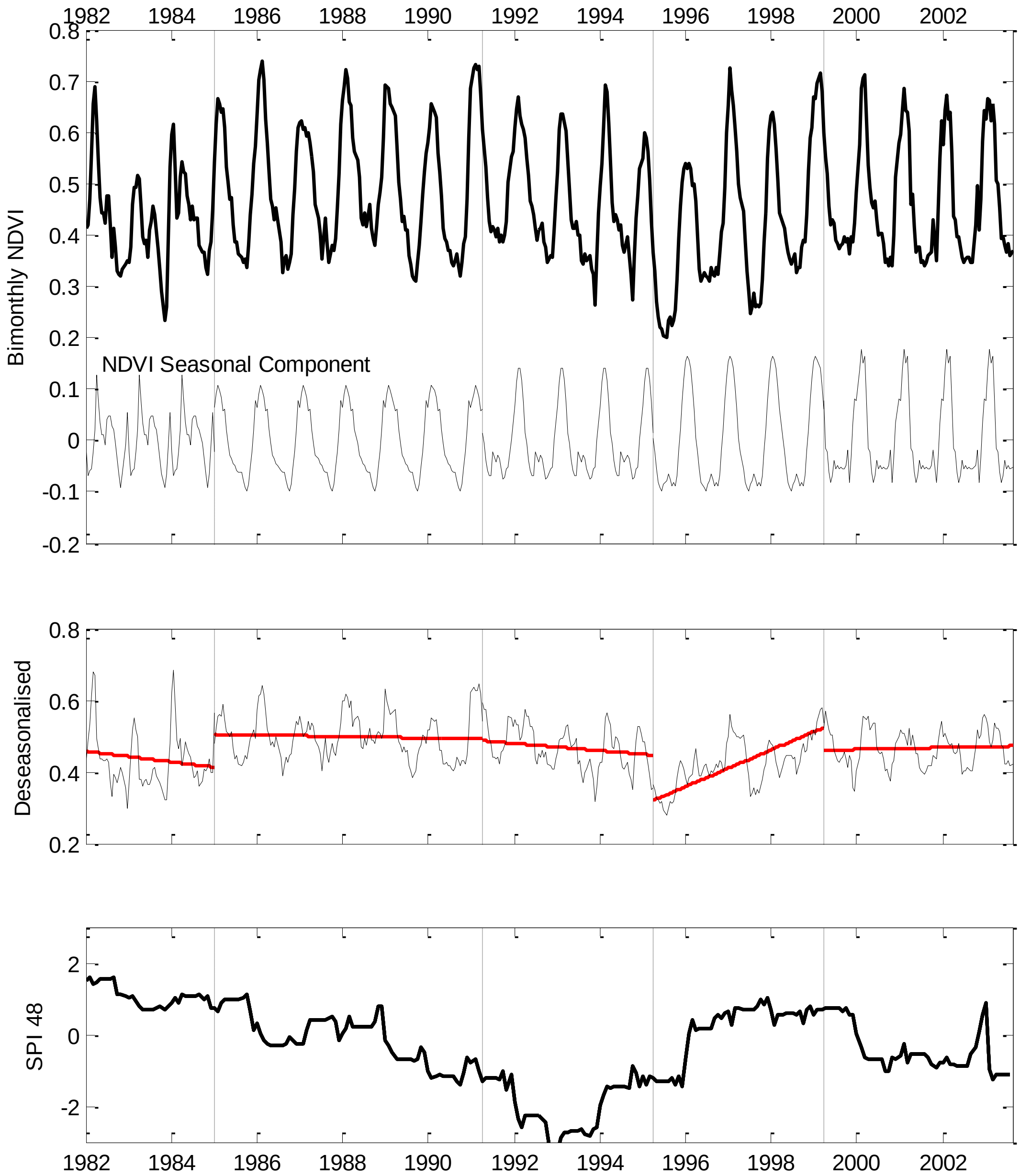

Figure 25: Bimonthly NDVI, broken down into seasonal and deseasonalised components, against SPI48 for the period 1982 – 2003 for the Messara Study Site.

In order to investigate the sensitivity of the method, an alternative area with a smaller proportion of agricultural land versus graze land within the Messara Study Site is also investigated. At this alternative location, vegetation suffered at a greater degree during the 1993 drought (Figure 26). Indeed, this location shows signs of degradation as early as in 1989, possibly due to the lower resilience or the reduced vegetation cover ecosystem. In this case vegetation manages to recover after 1997, however it is possible that the time required for recovery increases as aridity and water stress drive the system to its limits.

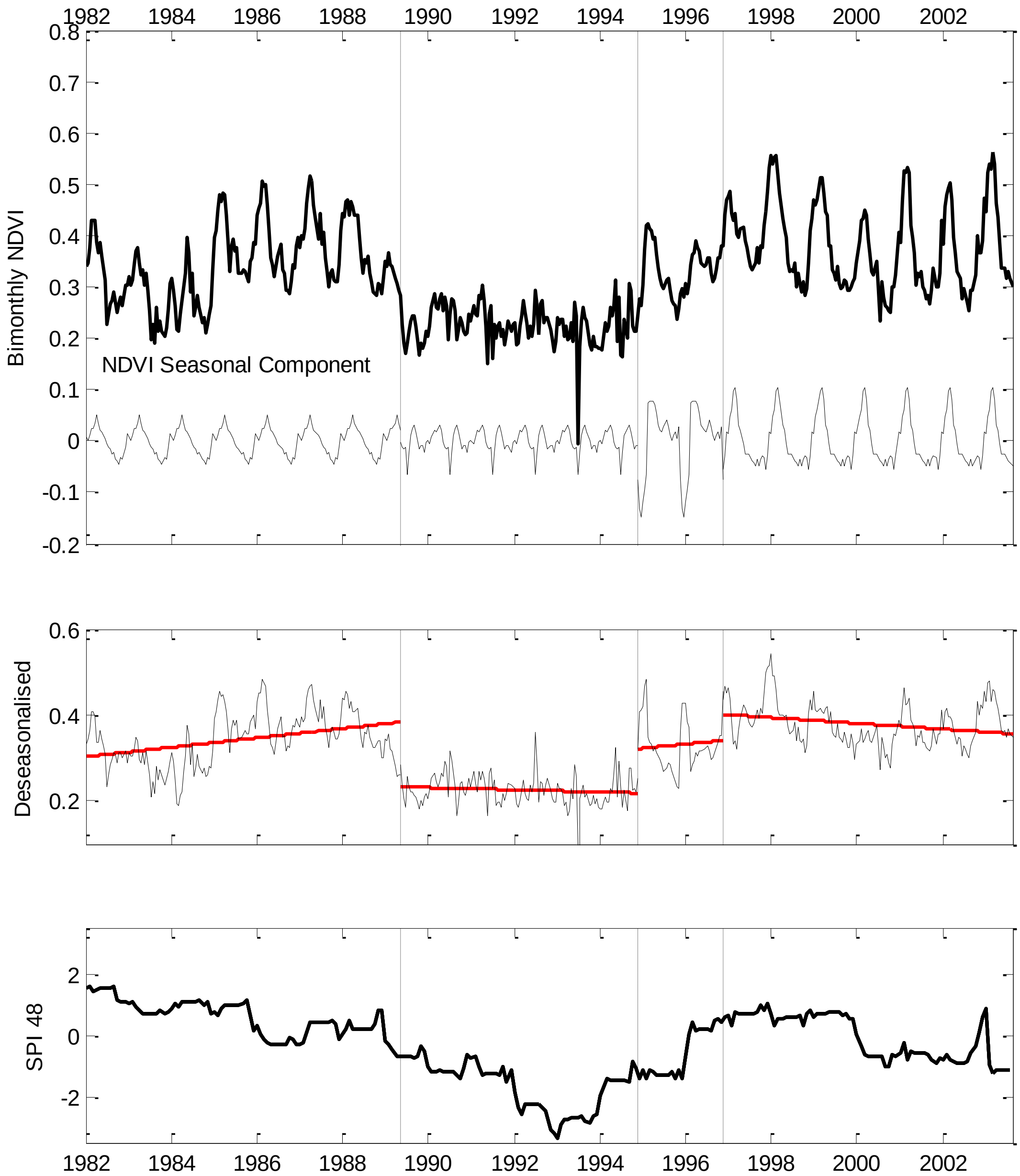

Figure 26: Bimonthly NDVI, broken down into seasonal and deseasonalised components, against SPI48 for the period 1982 – 2003 for an alternative location within the Messara Study Site.

3.3.6 Randi Forest Study Site (CY)

In the Randi Forest Study Site, an NDVI trend brake occurs during 1996-1998, disturbing the otherwise slightly increasing trend of vegetation greenness (Figure 27). This incident comes near the end of an extremely dry period, both as shown in the SPI48 plot in Figure 27 and as perceived by the residents (see Timeline of Events of Randi Forest in D.2.1).

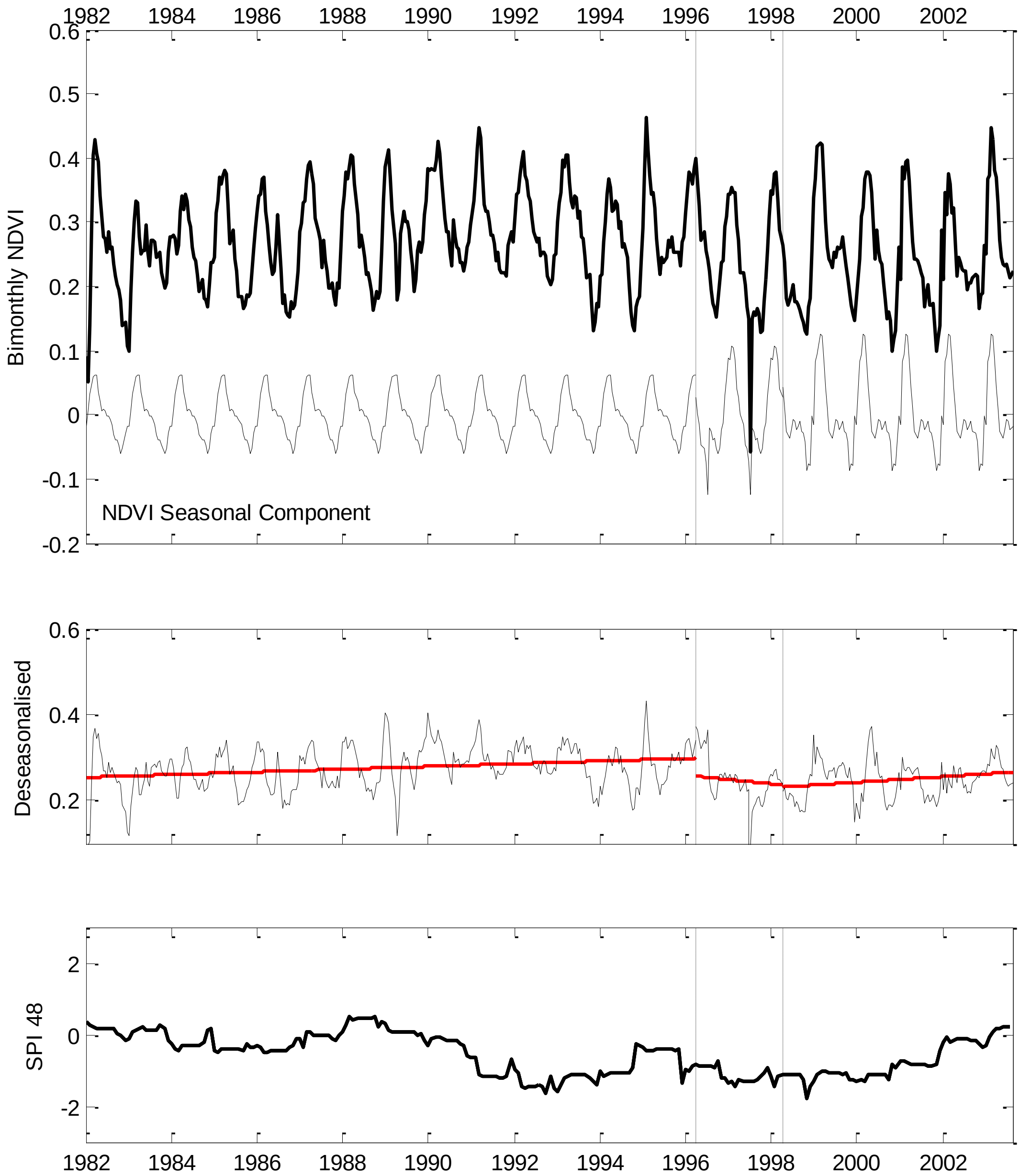

Figure 27: Bimonthly NDVI, broken down into seasonal and deseasonalised components, against SPI48 for the period 1982 – 2003 for the Randi Forest Study Site

The system seems to recover after 1998-2000, with the nearby Asprokerammos dam overflowing in 2004 after record high precipitation (not shown here). Similar to Valencia though, the reduction of grazing licenses that was imposed in 2002 in Randi Forest has come too late to deter the problem. Before and after the NDVI discontinuity the seasonal component of the signal shows moderate variation. It is important to note that in Randi Forest the most significant transition took place when it was transformed from an actual Forest to a graze land.

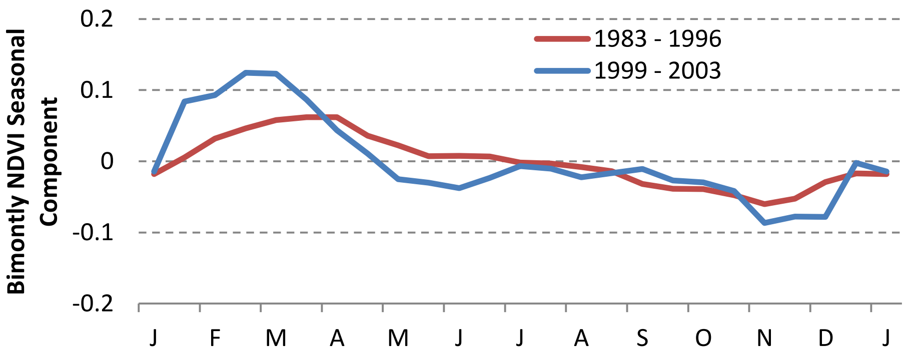

Figure 28: NDVI Seasonal component differences for the periods before and after trend shifts for the Randi Forest Study Site.

3.4 Effects on grazing systems

In this case, as ET_a/ET_p approaches unity H approaches the theoretically predicted $a/2b$. On the other hand, when ET_a/ET_p decreases, the density of herbivores that can be sustained by the vegetation biomass also decreases and is largely controlled by the vegetation characteristics. Figure 29 shows that as ET_a/ET_p decreases, the maximum density of herbivores that can be sustained by the system steadily decreases. More importantly, the system seems to move to a less stable equilibrium where small changes in the herbivore density can cause ever increasing changes in the maximum achievable growth. This characteristic makes the system more prone to abrupt shifts.

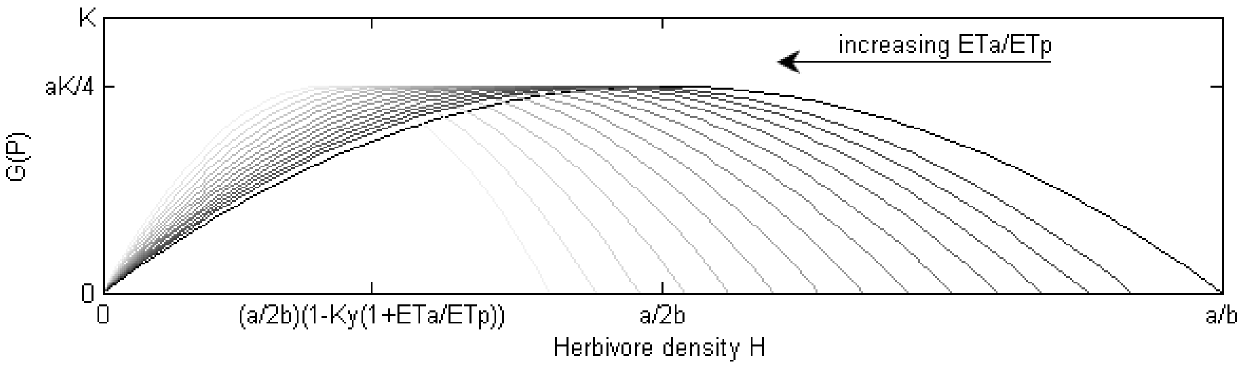

Figure 29: Gordon Schaefer model of biomass dynamic as a function of herbivore density and climate variability.

3.5 Unifying framework

Based on background information collected so far from the CASCADE Study Sites as well as findings on the behavior of ecosystems in response to various pressures, each case is fitted to the cusp catastrophe framework. This step will allow for a common understanding within the CASCADE Project as well as beyond.

In the case of Várzea, pine forest is considered as the desired original state that today exists among eucalyptus plantations. In this Study Site the eucalyptus forest cover is relatively stable as it is actively maintained for logging. On the other hand, pine forests are less actively managed and are exposed to forest fire risk. According to the cusp catastrophe framework, a discontinuous change can occur when the frequency of fires incidents reaches a critical level. The critical frequency depends on the time available for seedlings to reach a mature state and produce seeds for the generation that survives the fire. Beyond critical frequency, the pine forest is lost and a stable state of shrubland is established. Depending on land tenure status, restoration attempts in the area may lead to the establishment of additional eucalyptus plantations in place of the pine forest. This path leads to an alternative stable state that offers different ecosystem services compared to the original state. Loss of soil and aridity also play a role in shaping the state of the ecosystem, as shown in the cusp catastrophe model. Present findings show that in the case of Várzea, aridification has already played a significant role, but more importantly, is projected to be a major driver of change in the future. Also, loss of soil here signifies not only the physical loss of material as a consequence of vegetation cover and subsequent wind and water erosion, but also the loss of nutrients and other resources crucial for the soil services provided at the conserved state.

Similar to Várzea, Valencia is facing a transition of the original oak forest to shrub and bare areas due to fire and failed restoration attempts. Here, land is less actively managed than in the case of Várzea, but restoration attempts have pushed towards the establishment of pine forests, possibly as a cost-effective measure. Similar to Várzea, aridification may play a major role against ecosystem recovery, especially under unfavorable climate change scenarios. In the cases of Alicante and Messara, the original desired state consists of local shrub varieties that contribute various ecosystem services among which fodder for grazing livestock. Grazing intensity, reinforced by loss of soil or water stress gradually lead to a different shrub density or phenology, or even no vegetation cover. In both cases, once system resources have been depleted, restoration attempts lead to an intermediate state that offers ecosystem services inferior to those of the original. Finally, although Castelsaraceno and Randi Forest are located in totally different settings, they both appear to suffer from poor land management and eventually land abandonment that regulate the state of their ecosystems. In the case of Castelsaraceno the system oscillates between different levels of forest encroachment in pastureland, with the desired state being an environment that can sustain grazers. This equilibrium is sensitive, with land abandonment leading to loss of pastures towards shrub and forest and overgrazing leading to bare ground. On the other hand, Randi Forest, now entirely lost, is certainly located in the lower end of the cusp catastrophe model, having lost the required resources to recover to its original state. Land abandonment enables a state of unmanaged grazing that, along with aridity, are highly unfavorable to restoration.

4 Conclusions

Climate is a governing driver for all Study Sites, especially those that do not have significant vegetation cover. Water availability undoubtedly plays an important role, and while it is not addressed directly in this study, we consider that the use of the Aridity Index is a perfectly suited proxy. Results show that for the past 50 years, aridification has been very pronounced along imposing mountain ranges of the Mediterranean; the Cantabrian Mountains, the Pindus Mountain range and the Appennini Mountains. Three main reasons can be offered for this observation: (a) As the meteorological station network becomes denser, areas that were previously harder to access, such as mountain tops, now enjoy more certainty in climatic observations; there is a possibility that earlier aridity values are overestimated thus creating negative trends(although datasets are normally corrected for such biases), (b) The aridification may be part of a greater oscillation that will gradually recover and the current datasets are not long enough to draw consistent conclusions and (c) Due to snow cover, these areas have traditionally been more humid, nevertheless slight temperature rises may affect them earlier than lowlands, rendering them more sensitive to climate change. The latter explanation is the least optimistic but it is also backed by previous observations of a receding snow cover in the mountain-ranges of Southern Europe (i.e. Durand et al., 2009; Morán-Tejeda et al., 2013).

The connection of vegetation with climate and water stress is well established at an annual basis, with widely used relationships estimating yields for one growing season. While this approach is straightforward, the NDVI analysis conducted here has shown that in several cases (e.g. Randi Forest and Messara), drought can have a creeping effect: multiannual periods of stress can cause non-linear breaks in the greenness signal. This behavior implies the existence of thresholds or resources buffers which once crossed lead to a different stable system. In all cases, the seasonal component of the NDVI after a break differs to some degree denoting possible phenological variations. In some cases, the magnitude of the change in the seasonal component can be subtle (e.g. Messara, Randi Forest) and in others more pronounced (e.g. Valencia, Várzea), thus making it easier to attribute it to its driving force.

Besides the simplifications introduced in the grazing model, it is shown that climate change can be expected to undermine the resilience of pastoralism ecosystems, especially under marginal management conditions. The expected consequences comprise a function of system parameters such as the land’s carrying capacity, the plant growth coefficient, the yield response factor and the biomass consumption coefficient. These parameters are strongly related to soil quality, which is under severe stress in arid areas. Results show that arid and semiarid areas will be more prone climate induced pressures in vegetation by the predicted increase in temperature. While the precise effects largely depend on local climatic particularities and specific grazing practices, simple models point to a decrease of the system's stability as the ration of action over potential evapotranspiration decreases.

Until now, the cusp catastrophe framework is seldom used to describe forest composition instability due to fires. The most notable work is that of Frelich and Reich (1999) in the Great Lakes Region which incorporated fire frequency in a general "disturbance severity" concept. Here we show that Mediterranean dryland forests that suffer frequent fires fit well in the cusp catastrophe approach. The effects of fire frequency beyond the limits of the local ecosystem are evident in Valencia where shrubland gradually replaces pine forest, a conclusion also supported by the NDVI analysis. Here, a low fire resistance species was introduced in a fire prone environment, causing additional problems which are bound to escalate with climate change. The situation is less clear in Várzea where management decisions play a significantly more important role in shaping the landscape. Here the transitions to a less humid climate may eventually create a less favorable environment for the water intensive eucalyptus plantations, thus putting the sustainability of the current socioeconomic model of the region at risk.

Land use, land cover, production, consumption and disposal drive the relationship between social and ecological systems, by being embedded in social patterns and processes, such as demography, technology and culture (Redman et al., 2004). Therefore long term relationships are difficult to establish. Population growth, increase of demand, turn to tourism activities have in some areas led to the increase of irrigated land, changes in land use and the banishment of pastoralism activities to marginal lands (e.g. Crete and Randi forest). As discussed in D2.1, the socioeconomic backgrounds of all Study Sites follow patterns similar to those met in the desertification paradigm (Reynolds et al., 2007). Poor management decisions at individual, institutional and policy level can introduce perturbations that often accelerate the downward spiral of degradation and desertification (e.g. Daliakopoulos and Tsanis, 2014). While addressing these interactions with simple trends has been considered, socio-economic factors, scenarios and policies are not taken into account in this report. Such relations are multi-dimensional, certainly in complex landscapes such as the Mediterranean where population, markets, technology of production, infrastructure development and accessibility, policies (and others) all play a role.

Quantification of several features of the human and natural driver interaction with the state and rate of land degradation still poses a challenge as feedbacks and interactions are often ambiguous. Here we have shown that features of the cusp catastrophe model can be identified in all the CASCADE Study Sites. Several of the features identified in these descriptions can be considered as catastrophe flags (Rietkerk et al., 1996). In all cases, two or more stable states (bimodality) are present, possibly in different points in time. In all Study Sites, the transition from the degraded state to the conserved one is hindered to the extent that resources depletion and climate cannot be reversed, thus rendering the conserved state inaccessible. Furthermore, the NDVI analysis has shown that sudden jumps appear when the system is stressed to its limits, especially in the case of arid areas that have a smaller resource buffer. In Messara, an example of divergence is presented, with two neighboring locations beginning from a close starting point of vegetation health and having widely separated final states under the same driving conditions. Finally, in most cases, when degradation is present the transition to a conserved state is virtually impossible without substantial human intervention. This is more pronounced in the cases of Alicante and Valencia where restoration approaches have had limited success.

Concluding, a unifying framework of behavior that can fit the entire range of the Mediterranean drylands instability is an attractive prospective. Nevertheless, it is also critical to recognize that not all drylands are the same, that even within the Mediterranean climate can vary substantially and that the human factor has been significantly altering these parts for millennia. The cases presented here are a good set of examples where the cusp catastrophe framework applies but care must be taken when generalizing.

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