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

CASCADE REPORT series

Report on the restoration potential for preventing and reversing regime shifts

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Paterna, Spain.

CEAM
FUNDACIÓN CENTRO DE ESTUDIOS AMBIENTALES DEL MEDITERRÁNEO

May 21st, 2016
Version 1.0
Report number 13
Series: Scientific reports

Deliverable 5.2

This report was written in the context of the CASCADE project
www.cascade-project.eu

DOCUMENT SUMMARY	
Project Information	
Project Title:	Catastrophic Shifts in drylands: how can we prevent ecosystem degradation?
Project Acronym:	CASCADE
Call Identifier:	FP7 - ENV.2011.2.1.4-2 - Behaviour of ecosystems, thresholds and tipping points
Grant agreement no.:	283068
Starting Date:	01.01.2012
End Date:	30.09.2015
Project duration	66 months
Web-Site address:	www.cascade-project.eu
Project coordinator:	Prof. Dr. C.J. Ritsema - (coen.ritsema@wur.nl)- +31 317 486517
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Project manager:	Dr. Rudi Hessel - (rudi.hessel@wur.nl) - +31 317 486530
Deliverable Information	
Deliverable Title:	Report on the restoration potential for preventing and reversing regime shifts
Deliverable Number:	D.5.2
Work Package:	WP5
WP Leader	*Centro de Estudios Ambientales del Mediterráneo (CEAM)*
Nature:	Public
Author(s):	Alejandro Valdecantos (CEAM),V. Ramón Vallejo (UB), Jaime Baeza (CEAM), Susana Bautista (UA), Matthijs Boeschoten (UU), Michalakis Christoforou (CUT), Ioannis N. Daliakopoulos (TUC), Oscar González-Pelayo (UAVR), Lorena Guixot (UA), J. Jacob Keizer (UAVR), Ioanna Panagea (TUC), Gianni Quaranta (UNIBAS), Rosana Salvia (UNIBAS), Víctor Santana (UAVR), Dimitris Tsaltas (CUT), Ioannis K. Tsanis (TUC)
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Date of Delivery	May 21st 2016.

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13	WAGENINGEN UNIVERSITEIT	WU	Netherlands
14	FUNDACION CENTRO DE ESTUDIOS AMBIENTALES DEL MEDITERRANEO	CEAM	Spain
15	University of Barcelona	UB	Spain

CASCADE
Catastropic shifts in drylands

Deliverable 5.2

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Deliverable number:	*5.2*
Authors:	*Alejandro Valdecantos (CEAM),V. Ramón Vallejo (UB), Jaime Baeza (CEAM), Susana Bautista (UA), Matthijs Boeschoten (UU), Michalakis Christoforou (CUT), Ioannis N. Daliakopoulos (TUC), Oscar González-Pelayo (UAVR), Lorena Guixot (UA), J. Jacob Keizer (UAVR), Ioanna Panagea (TUC), Gianni Quaranta (UNIBAS), Rosana Salvia (UNIBAS), Víctor Santana (UAVR), Dimitris Tsaltas (CUT), Ioannis K. Tsanis (TUC)*

The work leading to this publication has received funding from the European Union's Seventh Framework Programme (FP7/2007-2013) under grant agreement n° 283068.

Contents

1 SUMMARY

The intensity in time and pressure, and the frequency of the degradation drivers severely impact ecosystem properties and services moving the ecological system to states with different potentials to recover functionality and structure. In CASCADE's field sites the loss of ecosystem services is positively related with the aridity of the site and certain degradation thresholds might have been already passed. Different restoration approaches are needed depending on the degradation degree of the site but little is known about the relationship between the restoration potential and the accumulated loss of ecosystem services. This deliverable focuses on the assessment of some important ecosystem services in degraded and restored states of target ecosystems in all six CASCADE study sites to determine the restoration potential. We have also included data of the Reference undisturbed ecosystems as the starting point that should be desirable to achieve.

The restoration actions included different treatments to remove the burned trees early after the fire in Várzea, selective clearing of fire-prone shrublands combined with planting resprouter seedlings in Ayora, clearing undergrazed areas in Castelsaraceno, restrict grazing in overgrazed areas in Castelsaraceno, Randi and, combined with planting carob trees in Messara, and planting tree and shrub species with different ecotechnologies in Albatera. In order to do an across-site analysis, we applied the same field methodology to evaluate five common ecosystem services in Reference, Degraded and Restored states: water and soil conservation, carbon sequestration, nutrient cycling and biodiversity.

In Várzea, restoration actions improved ecosystem properties and services at the very short term (< 2 years) after their implementation although the dynamics of the plant communities were slowdown, probably due to the impact of the heavy machinery on the earliest regenerated plants. On the contrary to our expectations, traditional (salvage) logging was more effective recovering ecosystem function than the conservation logging. Nevertheless, more time is needed to assess whether the traditional and conservation logging treatments affect differently to the recovery of ecosystem properties in Várzea.

In Ayora, the assessment was conducted more than ten years after the application of restoration actions and they had positive impacts on most ecosystem properties and services, especially on biodiversity and fire risk reduction. Only C sequestration

was negatively affected by restoration as actions included the removal of seeder fire-prone vegetation and hence the aboveground biomass. The reduction of fire hazard, together with increasing the resilience of the plant community, was the main objective of the restoration carried out.

In Randi and in the overgrazed state in Castelsaraceno we observed a general improvement of ecosystem properties and services by grazing exclusion, especially in Randi where plant cover, litter accumulation and aboveground biomass recovered to similar levels found in the undisturbed reference areas. As a consequence, the five ecosystem services calculated did not show differences between the restored and the reference areas and were significantly improved from the overgrazed degraded lands. In Castelsaraceno, the degradation due to overgrazing seems more severe than that due to undergrazing and biodiversity is the most improved service associated to the two restoration approaches. Fencing overgrazed areas did not achieve the overall balance of services provided by the references while in the undergrazed areas the restoration through clearing showed the highest balance of services. We discuss about the interest of defining intermediate stocking rates that might optimize ecosystem services in these grazed Mediterranean areas, including provisioning services.

Restoration in Messara aimed to transform land use from grazing to carob tree orchards as a silvopastoral or agroforestry system rather to recover the pre-disturbance state of the ecosystem. Under these circumstances, the conducted evaluation based on LFA assessment should be complemented with plant data (cover, biomass, diversity).

In Albatera, the two restoration approaches based on planting conifers on terraces (traditional) and on the implementation of different ecotechnologies (water harvesting, compost addition) in multispecific plantations improved most ecosystem services in relation to the degraded areas, especially the ecotechnological restoration approach. However, the extremely harsh conditions in Albatera (highest aridity index) determined low recovery rates of ecosystem structure and function after restoration and it is expected that the positive effects of this management option will increase over time as ecological processes act at slow rate in these extremely stressed sites.

Despite of the diversity of degradation drivers, reference ecosystems, restoration approaches and time since the restoration actions, we observed a positive relationship between the degree of degradation and the recovery of ecosystem services after restoration. But this positive linear trend drops sharply in the highest degraded field site, Albatera, where the restoration actions are considered as successful, our results suggest that the relationship between restoration potential and degradation level matches a non-linear model, being positive until certain threshold in the loss of services, beyond which the benefits of restoration drop sharply. From the management perspective, the implications of these results are of paramount importance for prioritizing restoration efforts and assessing the cost-benefit of restoration as a function of degradation.

2 INTRODUCTION

Degradation in drylands, especially when the pressure exceeded critical thresholds, implies losses of ecosystem functioning and diversity, and the capacity of the system to recover the original values of these altered properties determines the resilience of the system. In CASCADE's WP5, we have observed that the degradation drivers considered in the project severely impacted, in occasions beyond recovery thresholds, ecosystem properties and services in CASCADE field sites with higher losses along the gradient of aridity represented by the field sites (Valdecantos and Vallejo 2015). There are ecosystem properties such as the spatial distribution of vegetation that, when changed, may indicate overpass or proximity to this eventual threshold. The intensity, both in terms of pressure and time, of degradation can affect the resilience of an ecosystem hampering or even impeding the reversal. Whisenant (1999) proposed the existence of two degradation thresholds beyond which the natural recovery of ecosystem is extremely difficult or impossible. At the lower degree of pressure of the degradation driver, the first one is controlled by biotic interactions and the system still maintains the capacity to capture and retain resources and can be considered as a functional system. In these cases, it is only required an appropriate manipulation of the biotic component (mostly vegetation) to increase ecosystem function. If the pressure or degradation increases, a second threshold controlled by abiotic interactions can be exceeded, primary processes are not functional any longer, and the recovery of ecosystem functions requires the manipulation of the physical environment. In all these cases restoration actions, acting as accelerated succession (Hilderbrand et al., 2005), should be envisaged to recover the integrity of the site although a complete restoration is not always possible without perpetual management (Lindig-Cisneros et al., 2003). However, some studies suggested that there is no evidence that the lower the functionality of a given ecosystem, the lower the restoration success or the higher the economic input needed (Cortina et al. 2006; Maestre et al. 2006).

In addition to these considerations, ecosystem management for restoration has to include the expected climate change scenarios as successful approaches in the past might not be effective in the future. Global climatic change represents an additional factor of uncertainty not only in the outcomes of forest plantations, but also on the very subsistence of current dryland landscapes. Alkemade et al. (2011)

predicted that up 25% of the species currently present in natural landscapes of the Mediterranean Basin will disappear by 2100, being the Mediterranean shrublands one of the ecosystems in Europe most threatened by climate change projections.

Within the framework of CASCADE, the two fire-affected ecosystems of the project considered restoration actions at two different time scales: within the first year after the fire when vegetation reestablishment is still very low (tree trunk removal or logging in Várzea) or several decades after the fire when the forest did not recover but a continuous shrubland was established (selective clearing and planting in Ayora). Salvage logging after fire in pine forests consists in the removal of all burned tree trunks and is one of the most common emergency actions carried out in the Mediterranean in the very early months after forest fires (de las Heras et al. 2012; Moreira et al. 2013). The main objectives of this practice are, especially, to have some return with the market value of the wood, but also to reduce fuel, to avoid erosion once the trees fall down some years after the fire, to reduce aesthetic impact, and, in case of weaken but still alive trees, to avoid pests spread (Vallejo et al. 2012). Potential negative impacts of this practice include the reductions of growth of regenerating seedlings, reductions of deadwood associated fauna, elimination of perches for birds dispersing seeds from neighbor undisturbed habitats, and reduction of microclimatic heterogeneity (Vallejo et al. 2012). There is also a risk to increase erosion associated to wood removal after fire but this impact is highly dependent on the soil properties of the area (Bautista et al. 2004).

Selective clearing of vegetation is one of the preferred management options aimed at sharply reducing fire hazard in Mediterranean fire-prone communities (Baeza et al., 2003). As compared to prescribed or controlled burning, also proposed and accepted as fuel control technique, vegetation clearing offers more positive effects especially related both to the protection of soil surface to erosion and resource export off-site and to the lag in the build-up of large fuel loads in the community (Baeza and Vallejo, 2008). The combination of this fuel control technique with the plantation of seedlings of late-successional species and with the ability to rapidly resprout after further disturbances (Valdecantos et al., 2009) may increase, at the same time, the resistance and the resilience to forest fires.

In the previous assessment of ecosystem services as a function of fire as degradation driver, we observed marked reductions in most ecosystem properties and services at the short term after fire (Valdecantos and Vallejo 2015). But at the long-term, burned areas recovered functionality to values similar to the Reference pine forest, with a spatial arrangement of vegetation that better conserve the resources.

Grazing has deep impacts on ecosystem structure, composition and functioning (Milchunas and Lauenroth, 1993). Grazing exclusion is a worldwide extended practice to recover important ecosystem properties affected by overgrazing such as plant cover, vegetation and litter biomass, diversity, infiltration rate, soil fertility and soil biological properties (see Rong et al. 2014). For instance, it has been proposed as an effective management action to promote services such as soil C sequestration in areas severely affected by desertification (Li et al., 2012; Wang et al., 2016). The time elapsed since the avoidance of animals to graze as well as the ecosystem properties assessed determine the magnitude and significance of the effects of grazing exclusion. Under areas that were transformed from forest to grazed lands, fencing results in heavy and rapid forest encroachment by an increase of woody vegetation (Su et al., 2015).

On the contrary, areas where the stocking rates are very low are susceptible to woody vegetation encroachment compromising grassland ecosystem types and threaten the biotic component, both plants and animals (Archer and Predick, 2014). However, there are no conclusive evidences that ecosystem services are compromised by woody vegetation encroachment while the recovery of the targeted ecosystem service after shrub management is only ephemeral and may depend on other factors. For instance, Alberti et al. (2011) observed that soil C pool reduces with clearing encroached pasturelands in moist areas but increases under dry environments.

We have observed that the CASCADE field sites affected by grazing showed a generalized decrease in diversity as compared to the reference states of the ecosystems but differences between the three grazed field sites were observed (Valdecantos and Vallejo 2015). Plant pattern in the grazed states was markedly different than in the ungrazed ones modifying the resource sink capacity of the system. LFA derived indices were lower in all Degraded sites than in their respective References suggesting a worsening of soil surface conditions and, hence, soil, water and nutrient conservation. Ecosystem services have shown important losses due to grazing in the order Randi>Messara>Castelsaraceno following a decreasing order of aridity.

Albatera, the most stressed site with an aridity index of 0.16 and affected by multiple stressors, showed the highest relative losses of all individual and combined ecosystem services of all CASCADE field sites. The main ecosystem properties affected by degradation were those related to the sink/source spatial pattern and biodiversity. The assessment and quantification of the spatial distribution and arrangement of vegetation and, in general, of sink and source areas is especially relevant to address the restoration potential of drylands as this features have been described to determine seedling survival and growth of planted seedlings in restored semiarid sites (Urgeghe and Bautista, 2015).

Biodiversity represents a structural feature of ecosystems with direct influence in all other services (MA 2005). Monitoring biodiversity in different states of the ecosystem, identifying the local extinction of keystone species and the appearance of exotics, is extremely important as its changes may have irreversible consequences in ecosystem goods and services (Hooper et al., 2005). Restoring biodiversity and maximizing ecosystem services are priorities in the EU Biodiversity Strategy (Lammerant et al., 2013). The ecosystem services we have included in the assessment include: i) water cycle regulation, that is a central ecosystem service for maintaining fresh water resources, controlling floods and, hence, protecting people living downstream (Vörösmarty et al., 2005), ii) nutrient cycling, regulated by a great variety of organisms and its alterations have deep impacts on ecosystem functioning (Lavelle et al., 2005), iii) soil conservation as its loss could be an irreversible process at the human and ecological scale, and its retention contribute to maintain primary productivity and to prevent harmful effects because of soil erosion (de Groot et al., 2002), and iv) C sequestration in different compartments of the ecosystem.

This report follows the structure and methodology used for *D5.1 Report on structural and functional changes associated to regime shifts in Mediterranean dryland ecosystems*, applied to determine the restoration potential of the field sites affected by degradation. However, and with the aim of facilitating the independent reading of this report, we have included the description of the sites and the methods with lower degree of detail.

3 SITE DESCRIPTION

The organization of this deliverable is the same as in D5.1, with two general blocks and a particular case. The two blocks respond to the main degradation driver acting in the study sites: grazing or fire. The Albatera study site is described separately as the current landscape is not a result of a single dominant agent but of many of them. Within the six CASCADE field sites (Fig. 1), there is a clear climatic gradient (Table 1). Two of the sites fall within the humid climate, three belong to the dry sub-humid climate, and one is classified as semi-arid. The average annual rainfall ranges from 267 mm yr^{-1} in Albatera to 1289 mm yr^{-1} in Castelsaraceno. There are also large differences in temperatures along the field sites. Castelsaraceno is again the coldest station with average annual mean temperature below 10ºC, while the hottest field site is Randi forest in Cyprus with mean annual temperatures close to 20ºC. The two Spanish sites show the lowest aridity indices (0.16 and 0.26 in Albatera and Ayora, respectively) while Castelsaraceno and, in a lesser extent, Várzea showed the highest aridity indices (1.05 and 0.84, respectively). Therefore, in addition to types and levels of degradation pressures, the CASCADE project includes a great variety of climates, soils, land uses and land use history (Table 2) that may eventually condition the loss of ecosystem services as described in Daliakopoulos and Tsanis (2013). Similarly, the ecosystems that have been selected as references or undisturbed states of the ecosystem also show much contrasted values of key ecosystem structure and function properties (Figs. 2 and 3).

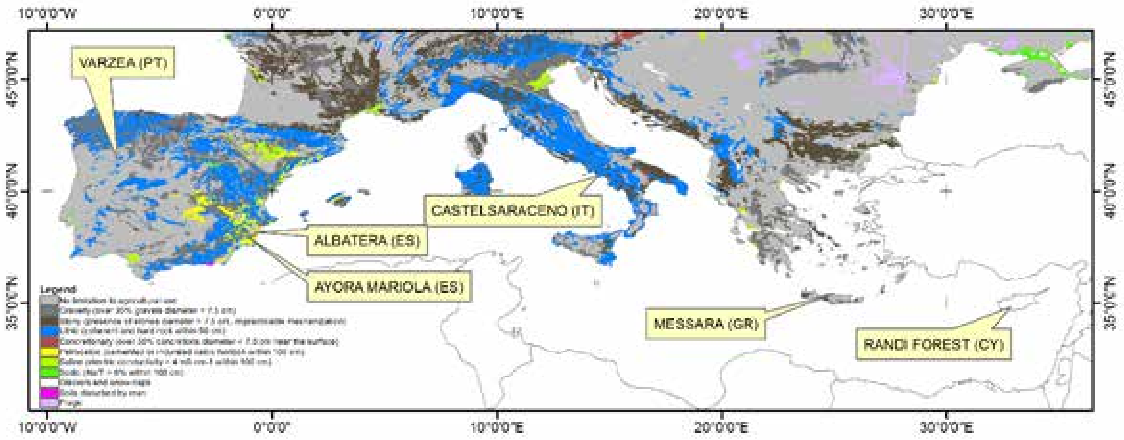

Figure 1. Location of the six CASCADE field sites (taken from D2.1, Daliakopoulos and Tsanis 2013).

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

	Várzea	Albatera	Ayora	Castelsaraceno	Messara	Randi
Climate	Humid	Semi-arid	Dry sub-humid	Humid	Dry sub-humid	Dry sub-humid
Average annual rainfall (mm)	1170	267	385	1289	503	489
Average mean temperature (ºC)	13.0	18.0	14.6	9.1	17.9	19.5
Aridity Index (mm/mm)	0.84	0.16	0.26	1.05	0.31	0.29
PET (monthly)	118.6	136.0	123.4	102.5	136.0	141.5

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

	Várzea	Albatera	Ayora	Castelsaraceno	Messara	Randi
Elevation	450-600 m	225-310 m	830-1030 m	972-1284 m	100-230 m	90-230 m
Bedrock	Schists	Dolomites, conglomerates and sandstones	Marl and limestone colluvium, limestones	Limestones and dolomites	Limestones and marls	Marls
Soils	Cambisols	Calcisols, Cambisols and Fluvisols	Regosols, Cambisols and Leptosols	Regosols	Cambisols and Luvisols	Calcaric regosols
Land use	Forests and shrublands (and agriculture in lesser extent)	Agriculture (52%) and shrublands (24%)	Forests and Shrublands	Cropland, pasturelands and forests	Croplands and shrublands	Croplands and shrublands
History	Recurrent fires (1978, 1985, 2005, 2012)	Abandonment of rainfed croplands, alpha grass harvesting and wood gathering. Afforestations	Fire (1979) and abandonment of wood harvesting	Land abandonment (especially after 1990s)	Overgrazing and overexploitation of water resources	Agriculture and grazing

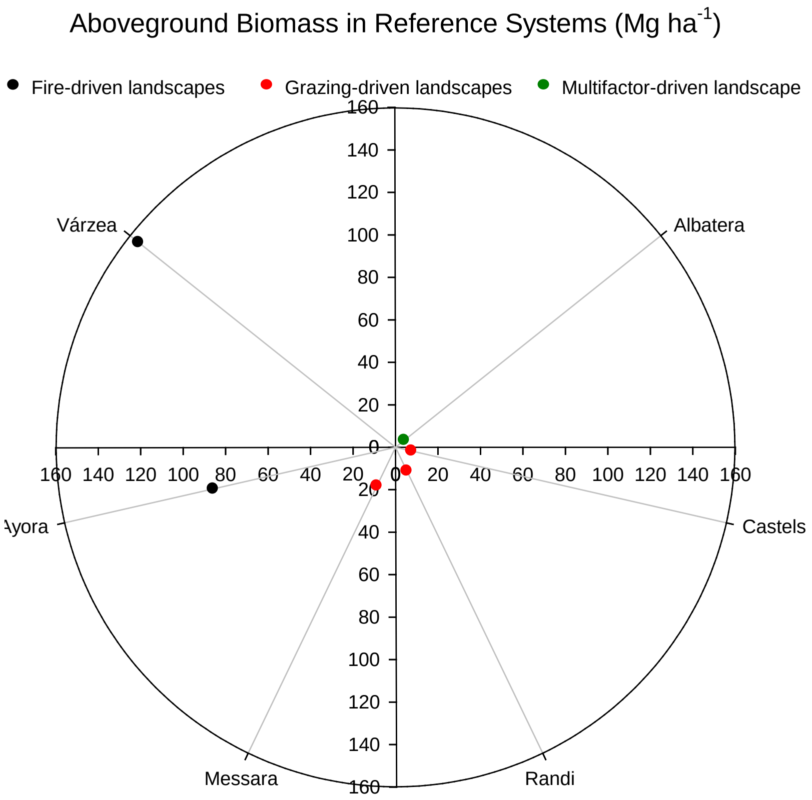

Figure 2. Total aboveground biomass (Mg ha^{-1}) in the Reference state of the ecosystem in all CASCADE field sites. The position of the field sites is random.

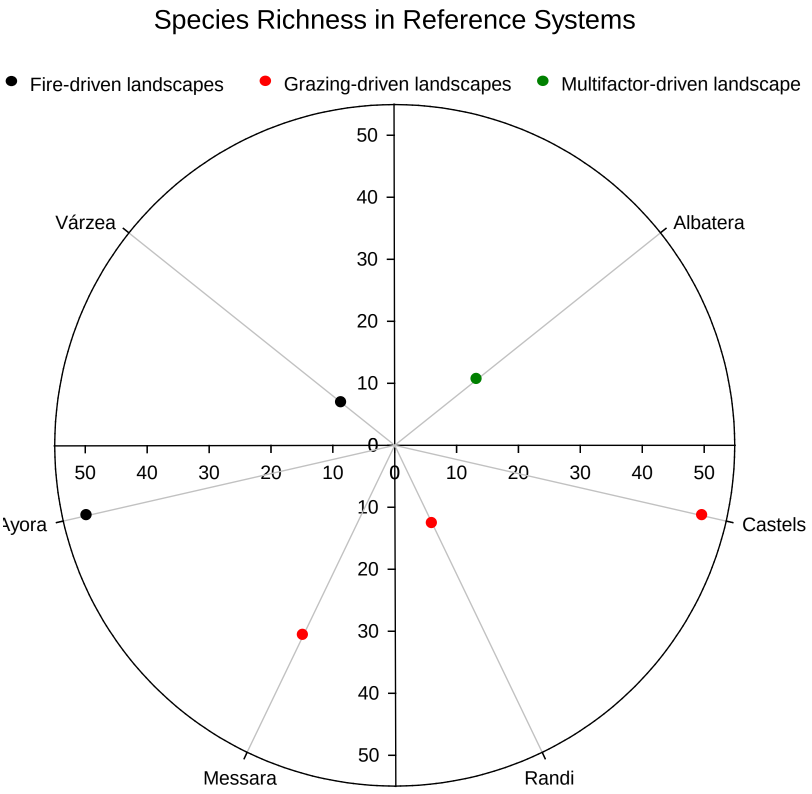

Figure 3. Species richness of vascular plants (number of species/100 m^2) in the Reference state of the ecosystem in all CASCADE field sites. The position of the field sites is random.

3.1 Fire-Driven Landscapes

The two field sites affected by wildfires share the mature forests of maritime pine (*Pinus pinaster*) as the Reference state of the ecosystem. But the assessment of losses of ecosystem properties and services due to degradation has been conducted at two contrasted time scales: at the very short term on a repeatedly burned site (Várzea) and at the long term on a community without significant recovery of the overstory layer (Ayora). The restoration potential has also been assessed though rather different approaches: by actions carried out within the first year (Várzea) or 23 years after the fire (Ayora).

3.1.1 Várzea

The reference state in Várzea is represented by a forest of maritime pine (*Pinus pinaster*) where no wildfire has occurred since 1975. At the opposite extreme, the degraded areas suffered four wildfires occurred since 1975, the last one in 2012. A third situation was represented by areas that burned twice (1985 and 2012) and where burned trees were removed after the last fire by two contrasting methods: standard or traditional logging in which all wood was removed from the site, and conservation logging, in which logging residues were left on the ground organized in piles. Both logging activities were conducted during the first year after the fire and implied the cuttingof all burned trees.

The experimental setup was conditioned by land availability of the burned area. Three spatially replicated plots of ca. 1000 m^2 were established in the reference mature pine forest (> 40 years old), in the 4-times burned (last fire in 2012) and in the standard logging areas, while three smaller plots were selected under the conservation logging treatment.

As the number of blocks was different in the reference and conservation logging sites (3) in relation to the degraded and traditional logging sites (1), we randonmly selected transects (plant cover and LFA – Landscape Function Analysis) and subplots for biomass assessment (aboveground and litter; see *Materials and Methods* section) to balance the data. In the case of the conservation logging, plots were differentiated by piles (accumulation of woody residues), inter-piles (lines between piles) and roads (paths or tacks for logging machinery). Sampling was proportionally conducted on these three contrasted spatial situations.

3.1.2 Ayora

The Reference ecosystem is a mature pine forest of *Pinus pinaster* and *P. halepensis* that was traditionally managed for different uses. The degraded ecosystem is an old and dense shrubland where pines did not recover after a wildfire in 1979. This >30 years shrubland bears a very high risk of fire as it accumulates large amounts of standing and ground fine, dead fuel. In 2003, restoration actions were carried out with the main objective of reducing fire risk. These actions included selective clearing of fire-prone shrub species and planting seedlings of more resilient resprouter species.

Three spatially replicated plots were established under three states of the ecosystem and the assessment strictly followed the evaluation protocol described below.

3.2 Grazing Driven Landscapes

Grazing is the major degradation pressure in three out of six CASCADE field sites. From those, Messara and Randi share many landscape characteristics, physical features and land use histories while Castelsaraceno shows clear specificities. The three sites represent a good example of the most important environmental and socio-economic features of their respective regions.

3.2.1 Castelsaraceno

The vegetation cover for the study site shows that broad-leaved forest is the most representative land cover and only a small part of the land is devoted to agriculture. After 2000, and due to rural exodus, a large part of the territory is covered by natural grassland and broad-leaved forest. Land cover under transition is noteworthy and there has been a progressive woods and shrublands encroachment on former pastures. The target Reference ecosystem is a productive pastureland with a sustainable grazing pressure composed by annuals and, in a lesser extent, perennial grasses, and where shrubs disappeared because livestock farming is widespread. Since 1991, the land was unevenly grazed resulting in over- and undergrazed zones depending on the stocking rate supported.

Two different restoration approaches have been considered in Castelsaraceno in relation to the different grazing pressures. For the undergrazed situation, where shrubs were colonizing, the restoration action was a selective clearing of vegetation ca 10 years ago. When overgrazing was the degradation driver, fencing (8-15 years before the evaluation) to avoid animals was the restoration measure considered.

The experimental setup in Castelsaraceno included three spatially replicated blocks, Monte Alpi, Favino and Piano del Campi. We have identified Reference, Overgrazed, Undergrazed, Fenced and Cleared ecosystems in all of them and three replicated plots were established for each block x pressure combination (15 plots). The assessment strictly followed the protocol described in Section 4 *Materials and Methods*

3.2.2 Messara

The natural landscape in Messara is dominated by the evergreen maquis/phrygana and the main driver of pressure to these reference ecosystems is grazing. Many marginal areas under natural vegetation were cleared in the past and planted with olives. Widespread olive production in steep hilly areas in combination with grazing has triggered desertification processes. In addition, further land abandonment led to less productive lands susceptible to degradation and at the same time grazing pressure significantly increased (more than 200% increases in sheep and goats between 1980 and 1990).

In addition to the Reference and Degraded ecosystems, we selected an intermediate state of pressure defined as Semi-Degraded. It was difficult to find areas subjected to any restoration action in the past in Messara. However, we found two areas where carob trees orchards were established on overgrazed areas: Melidochori and Odigitria. In Melidochori (Fig. 4), restoration works started in 1998 and two years old carob tree (*Ceratonia siliqua*) seedlings were planted in 2000 in a 6 x 6 m grid with maintenance actions (irrigation, fertilization and replanting dead individuals) for the first three years after planting. Grazing was excluded for ten years. LFA assessment was conducted 14 years after the establishment of the actions. Carob trees in Odigitria (Fig. 5) were established by the homonymous monastery about 7 years before the assessment and irrigation was conducted during the first two years after planting. No other maintenance actions were considered. In contrast to the Melidochori site, grazing is not controlled in Odigitria.

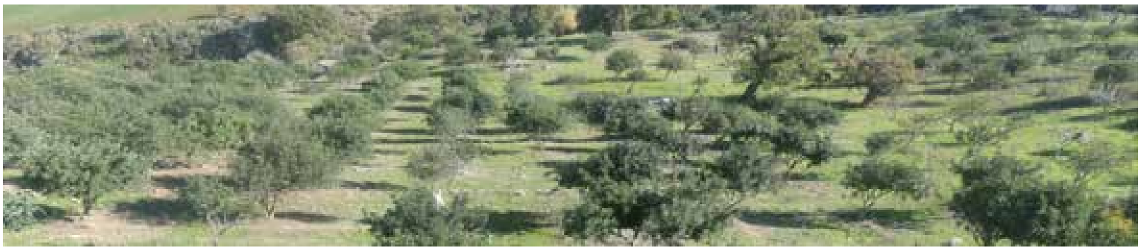

Figure 4. Restored area in Melidochori site.

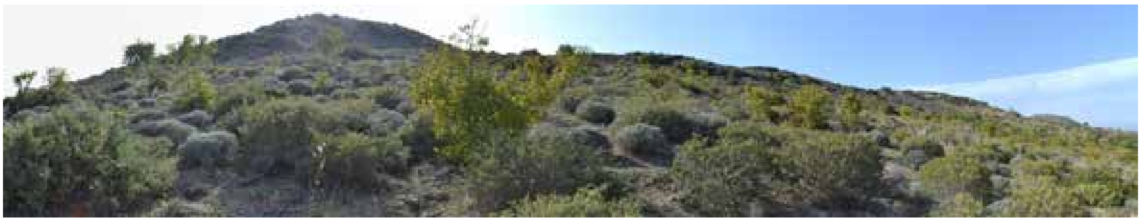

Figure 5. Restored area in Odigitria site.

Three replicated plots were established in the Reference, Degraded and Semi-Degraded states but one of the Semi-Degraded plots was completely affected by a fire in summer 2013 before WP5 field assessment and only two plots were left. Only one plot was established in the two Restored areas.

3.2.3 Randi

The natural landscape is the result of human activities and is dominated by shrublands, the typical Mediterranean phrygana, with open areas with shrubs and sparse carob and olive trees. The three studied states of the ecosystem in Randi, Degraded, Reference and Restored areas, used to be pine forest 100 years ago. After the allowance to local people to cut the pine forest and use them for firewood, only shrubs and olive trees were grown in the area but the land is not suitable for agriculture anymore and it is used for grazing, in particular goats and sheep. In the decade of 1950 goat and sheep farms were established in the area and started grazing the areas around the farms. The Restored areas (Fig. 6) are far from the farms but were grazed at different intensities depending on the distance to the shelters. Animals were excluded 20 years ago from these areas but continued to graze in the degraded and on the borders of the restored areas.

Figure 6. Restored area in Randi field site.

Three replicated plots were established in the Restored areas and the WP5 assessment protocol was completely and strictly applied in all of them.

3.3 Multifactor Driven Landscapes

3.3.1 Albatera

In this site, degradation of natural shrubland areas has resulted from a complex interplay of multiple drivers (some of them are no longer active), especially past over-exploitation of resources (overgrazing, mining, multiple cycles of marginal agriculture and land abandonment, and fire-wood gathering), in combination with harsh climate conditions. However, there are some scattered healthy shrubland areas that have been subjected to low past pressures and remain in a reasonably good shape. These areas represent the Reference state of the ecosystem considered in CASCADE WP5.

This site holds two different scenarios for the assessment of restoration actions, differing in both the implementation time and in the technologies and species used:

- Old (traditional) Restoration. Implemented over the 1970s and 1980s, and consisting on a plantation of only one tree species, *Pinus halepensis* (Aleppo pine), on large afforestation bench terraces (Fig. 7). A number of pine forest patches scattered on terraced slopes with varying degradation degree have resulted from this action.

Figure 7. Old reforestation in degraded terraces.

- New (ecotechnological) Restoration. In 2003 – 2004, a demonstration restoration project was performed by the Regional Forest Administration on one small catchment (24 ha) in the Albatera range area. The project counted on the scientific advice of CEAM and the Department of Ecology of the University of Alicante and it was designed to specifically combat degradation of drylands. The restoration action was performed combining several field techniques and plant species through spatially heterogeneous plantations, to better address the characteristic high heterogeneity of dryland landscapes (Chirino et al., 2009; Fig. 8).

Figure 8. Degraded water pipe channel (left) and several years after the New restoration (right).

Three replicated plots were established in the two alternative restoration approaches and the WP5 assessment protocol was completely applied in all them except litter accumulation and root biomass.

4 MATERIALS AND METHODS

A common methodology has been set up to be applied in all six CASCADE field sites to assess changes in ecosystem properties due to degradation and the potential to restore them. However, the protocol has been adapted locally to fit singularities, constraints and possibilities of the different field sites (see section *3 Site description* for details). The general framework includes the identification of representative Reference and Degraded ecosystems, according to the main pressure acting in each specific site, and Restored areas where any corrective measure has been conducted in the past (Table 3).

In general terms, we established three spatially replicated plots for every level of ecosystem state (reference, degraded and restored) in every field site to conduct the assessment of different variables of ecosystem structure and functioning. Replicated plots in every specific field site shared most physiographic, climatic, and edaphic variables as well as land use history. From these variables, we calculated a balanced set of ecosystem services. The potential for restoration was derived through the comparisons of the ecosystem structure and function as well as of ecosystem services in the Degraded and Restored states.

The three aspects of the evaluation process carried out are: 1) the determination of plant composition and diversity, 2) quantification of stand plant biomass, litter and belowground biomass, and 3) the application of the methodology of Landscape Function Analysis.

Table 3.Summary of pressures, reference, degraded and restored ecosystems in the six CASCADE field sites.

Field Site	Pressure	Reference Ecosystem	Degraded Ecosystem	Restored Ecosystem
Várzea, PT	Fire	*Pinus pinaster* forest	4-times burned areas (2-years after last fire)	Traditional & conservation logging
Albatera, SP	Multifactor	Semi-steppe dry shrubland	Dwarf shrubland	Traditional & ecotechnological reforestations
Ayora, SP	Fire	*Pinus pinaster, P. halepensis* forests	Shrublands. Areas burned in 1979	Selective clearing & planting
Castelsaraceno, IT	Grazing	Productive pastureland	1.Overgrazed 2. Undergrazed	1. Fencing 2. Clearing
Randi, CY	Grazing	Shrubland	Unpalatable community	Grazing exclusion
Messara, GR	Grazing	Shrubland	Unpalatable community	Carob tree plantation

4.1 Plant composition

Three 33-m linear transects were deployed following the maximum slope and the line intercept method was applied. Plant contacts and soil surface characteristics (bare soil, litter, stone, biological crust) were recorded every 50 cm along the tape (66 points per transect).

Transects were deployed avoiding ‘strange’ or artificial features of the plot such as pathways, stone accumulation points, gullies… In case that the size of the plot did not allow 33-m long transects, more shorter transects were established but always totalling 100 m per plot.

4.2 Plant biomass

Three 1-m^2 quadrats (subplots) were defined in every single transect. The placement of the quadrats was predefined to avoid subjective selection of microsites. In these subplots we evaluated biomass of shrubs by two alternative approaches:

- By clipping, drying and weighing. When possible, we cut all the individuals whose stems were within the quadrat limits and took them separately to the lab. We dried the plant samples at 60ºC for 48h in an oven and weighted them. Grasses were not separated by species.
- By allometric relations. We applied available allometric equations for some shrubs species. By knowing a morphological variable (basal diameter, total height or biovolume of the plant), we calculated the biomass of the individuals. Alternatively, as was the case of some shrub species in Messara and Randi field sites, we built up our own allometric equations by harvesting, drying and weighing a number of individuals outside the plots covering the range of plant sizes present within the plot.

4.3 Litter and belowground biomass

After harvesting grasses and shrubs, we collected the litter layer in a 25 x 25 cm sub-subplot. We avoided taking mineral soil particles in the samples as they are much heavier than the litter fractions and would produce significant error. Samples were taken to the lab to dry them at 60ºC for 48h. In the same sub-subplot, and once the organic layer was removed, we took a soil core of the uppermost soil (0-10, 0-15 or 0-20 cm depending on the site). Roots were separated from the soil in the lab by sieving and washing gently with water before drying at 60ºC for 48h.

4.4 Landscape Function Analysis (LFA)

This method was used for the assessment of ecosystem functioning in WP5.

Following is a much resumed procedure of the method (extracted from Tongway and Hindley 2004).

- Transects set-up: Transects started at the downslope edge of a patch following the maximum slope and as taut as possible.
- Patch and inter-patch identification: By definition, patch accumulates or diverts resources by restricting flow of water, soil and organic particles. They act as a sink of resources. Different types of patches may have different behavior and therefore should be discriminated when possible. Inter-patches represent areas where resources do not accumulate and even act as net export of resources (source areas).We measured three parameters along the transects: the number of patches, the width of every single patch (at the soil level, not the canopy, and up to a maximum of 10 m), and the distance between patches (inter-patch length).
- Soil Surface Assessment: This assessment was conducted in five 50 x 50 cm areas per type of identified patch and inter-patch in each plot. These five replications were distributed throughout the plot. The soil surface assessment is rapidly made by the use of simple visual indicators:
    - Rainsplash protection: ephemeral grasses and foliage at heights above 50 cm and litter were excluded.
    - Perennial vegetation cover
    - Litter: amount, origin and degree of decomposition. It included annual grasses and ephemeral herbage (both standing and detached) as well as detached leaves, stems, twigs, fruit, dung, etc. Three properties of litter were assessed: Cover (% and thickness of the litter layer), Origin (whether it was local or transported) and Degree of Decomposition/Incorporation.
    - Cryptogam cover
    - Crust brokenness
    - Soil erosion type and severity: Five major forms of erosion were assessed: Sheet erosion (progressive removal of very thin layers of soil across extensive areas with few, if any, sharp discontinuities to demarcate them), Pedestal (is the result of removing soil by erosion of an area to a depth of at least several cm, leaving the butts of surviving plants on a column of soil above the new general level of the landscape), Terracette (abrupt walls from 1 to 10 cm or so high, aligned with the local contour), Rill

(channels cut by the flowing water), and Scalding (the result of massive loss of A-horizon material in texture-contrast soils which exposes the A2 or B horizon).

- Deposited materials: presence of soil or litter materials transported from upslope.
- Soil surface roughness: due to soil surface micro-topography or to high grass density.
- Surface nature: resistance to disturbance.
- Slake test: The test was performed by gently immersing dry soil fragments of about 1-cm cube size in distilled water and observing the response over a period of a minute or so. If the soil floats in water (high organic matter), then it is stable, and if it cannot be picked (loose soils) was scored as not applicable.
- Texture

Spreadsheets were prepared and were filled out with the collected information and Stability, Infiltration and Nutrient Cycling indices were automatically calculated (Table 4). These indices varied between 0 and 100% depending on ecosystem functionality (100% represents fully functional systems).

Table 4.List of the soil functional indicators and their contribution to the indices of stability, infiltration and nutrient cycling (following Tongway and Hindley 2004). Shadowed cells mean that the indicator is scored in the calculation of the index given above.

Indicator	Indices		
	Stability	Infiltration	Nutrient Cycling
Rainsplash protection	■		
Perennial vegetation cover		■	■
Litter cover	■	■	
Litter origin and decomposition			■
Cryptogam cover	■		■
Crust brokenness	■		
Soil erosion type	■		

and severity	▨		
Deposited materials	▨		
Soil surface roughness		▨	▨
Surface nature	▨	▨	
Slake test	▨	▨	

4.5 Data analysis

In every CASCADE field site we conducted one-way ANOVA followed by post-hoc analysis (where three or more ecosystem states were identified) to assess if observed differences in all composition, functional, diversity and service variables were statistically significant. We calculated the relative changes of all measures and ecosystem variables in the Restored areas in relation to the Degraded ones to highlight which are the ecosystem properties more and less sensitive to be improved through the restoration actions considered. We conducted Principal Component Analysis (PCA) on specific plant cover data to assess general changes in vegetation composition and cover between Degraded, Reference and Restored sites.

Acquired data of structural and functional ecosystem properties were then grouped into related ecosystem services through standardization. We have selected regulating and supporting services as well as biodiversity, which underpins all services (Table 5). In addition, in Ayora we have included in the assessment the reduction of fire risk as it was the main objective of the restoration actions. Each variable was standardized using

$$ZPlot=(XPlot-AvgTot)/SDTot,$$

where *ZPlot* is the standardized variable, *XPlot* the original variable, *AvgTot* the average of the variable of all plots within a field site, and *SDTot* the standard deviation of all the plots within a field site. Variables were assigned to services as they were derived from validated methodologies selected on the basis of being appropriate indicators for this service (Table 5). When several variables were combined into one service, each variable was weighted equally, as all of them are considered to be good indicators for the respective service and no available information points to a better performance of any of them. The five (six the case of Ayora) selected ecosystem services were also weighted equally and averaged for Reference, Degraded and Restored plots in each field site as a global result of ecosystem service changes. This way, the assessment provides a baseline integrated and global evaluation based on the simplest assumption. However, it is worth mentioning that stakeholders' preferences regarding ecosystem services could be incorporated in the assessment in the form of different weights for each service, which could yield different global outcomes.

The selection of the key common indicators and assessment methods has been based on the work developed by the EU-funded PRACTICE project on ground-based assessment indicators (Bautista and Mayor, 2010). They represent few essential indicators that could characterize ecosystem function for a majority of drylands worldwide, mostly focusing on water and soil conservation, nutrient cycling, carbon sequestration, and biological diversity. Most provisioning and cultural services are considered to be very much context dependent (Rojo et al. 2012). Furthermore, half of the sites included in CASCADE are natural areas that are not expected to directly deliver goods. Therefore, our across-site comparative assessment of ecosystem services provision has been only based on supporting-regulating services, which together with biodiversity, are considered to be baseline services and properties that underpin other types of services (Bautista and Lamb, 2013).

Table 5. List of ecosystem services measured, variables from which their relative states were estimated through standardization, and the methodology used to obtain the data of the variables.

Ecosystem Service	Variables	Methodology
Water Conservation	Infiltration Index	LFA + Point-intersect
	Interpatch Cover	
	Plant Cover	
Soil Conservation	Stability Index	LFA + Point-intersect
	Interpatch Cover	
	Plant Cover	
Nutrient Cycling	Nutrient Index	LFA
	Litter	
Carbon Sequestration	Plant biomass	Allometries + direct quantification
	Root biomass	
	Litter	

Fire Risk Reduction*	Interpatch Cover	LFA + Point-intersect
	Dead/Green Cover	
	Seeder/Resprouter Ratio	
	Plant biomass	Allometries + direct quantification
Biodiversity	Richness	Point-intersect
	Diversity	
	Evenness	

*. Only in the Ayora field site.

5 RESULTS

5.1 Fire Driven Landscapes

5.1.1 Várzea

The two post-fire approaches considered showed significant differences in total plant cover assessed three years after the fire (Fig. 9 left). Traditional logging (79.6%) improved plant cover in relation to conservation logging (45.8%) that left wood remains piled on the ground in lines. The degraded area, where no action was conducted after the last fire, and the reference unburned forest showed values of plant cover similar to the traditional logging. Most of the total cover of the Degraded and Restored areas was due to species of the understory while pine cover was above 70% in the Reference forest. As a consequence, the cover of the understory in the Reference fell below 50%, significantly lower than the four-time burned area (Fig. 9 right).

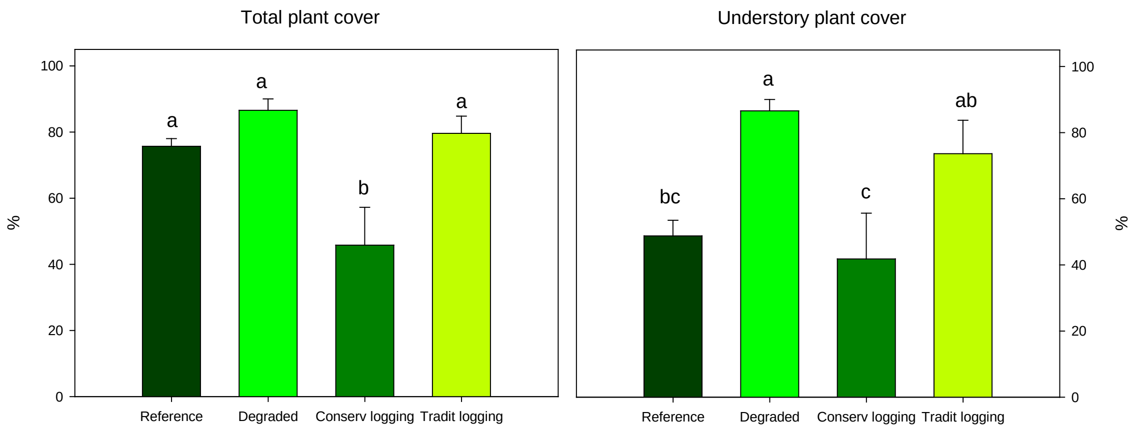

Figure 9. Total (left) and understory (right) plant cover in the Reference, Degraded and the two Restored states in Várzea field site. Mean and standard errors are shown. Different letters indicate significant differences.

Although there were not many differences in total plant cover between the Reference and the other states of the ecosystem, the composition of species was rather contrasted. *Pinus pinaster* is the most abundant species in the unburned forest (68.7%) followed by *Agrostis curtisii* and *Ulex minor* (18.4 and 15.9%, respectively). The three most represented species in the Degraded areas three years after the fire were *A. curtisii* (56.2%), *Pterospartum tridentatum* (38.8%) and *Erica umbellata* (27.4%). Two species showed cover values above 10% both in the Conservation and Traditional Logging areas but with contrasted percentages. *Agrostis curtisii* was much more abundant in the Traditional than in the Conservation site (54.2 vs 21.9%) while *P. tridentatum* showed similar percentages in both areas (17.4 and 15.9%, respectively). Pine regeneration was also higher in the Taditional than in the Conservation treatment (8.0 vs 4.5%). These contrasted specific plant covers resulted in a clear separation of the plots regarding their state. The first and second axis of the PCA conducted on plant specific composition explained 28.1 and 21.9% of the total variance, respectively (50.0% of accumulated explained variance). The first component clearly separated the Reference from the rest of the plots along the first component (Fig. 10). The species with highest positive weight in PC1 were *Ulex minor* (eigenvalue 0.911), *Pteridium aquilinum* (0.881), and the overstory species *Quercus robur* (0.859) and *P. pinaster* (0.819), while *A. curtisii* was negatively extracted on this axis (-0.712). The Degraded and Conservation Restoration areas were separated along both the first and the second axis. The species with highest positive weight on the second axis are *Halimium laisanthum* (0.948), *Erica cinerea* (0.925) and *Agrostis delicatula* (0.925). The areas subjected to Traditional Restoration showed high variability within the group and were plotted in a wide range of values of the second axis but in a very narrow range of the first axis.

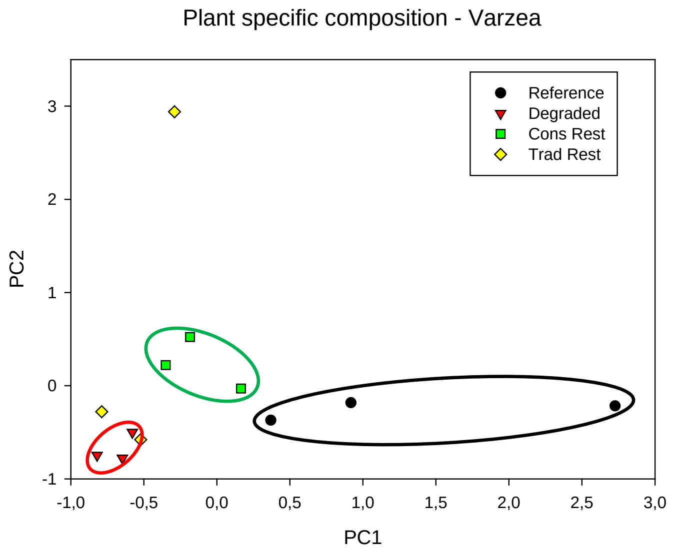

Figure 10. Plot distribution in Várzea according to the two first axis of PCA conducted on plant cover. Plots are marked and grouped by the ecosystem state.

None of the diversity indexes assessed showed significant differences between the four states of the ecosystem (Fig. 11). The total number of plant species recorded was very low with a slight trend to increase in the Restored areas in relation to both the Degraded and the Reference plots. Conversely, evenness was slightly lower in the Restored states than in the Reference and Degraded plots, falling from 0.78 to 0.59. The Shannon-Wiener index was quite similar in all four states of the ecosystem.

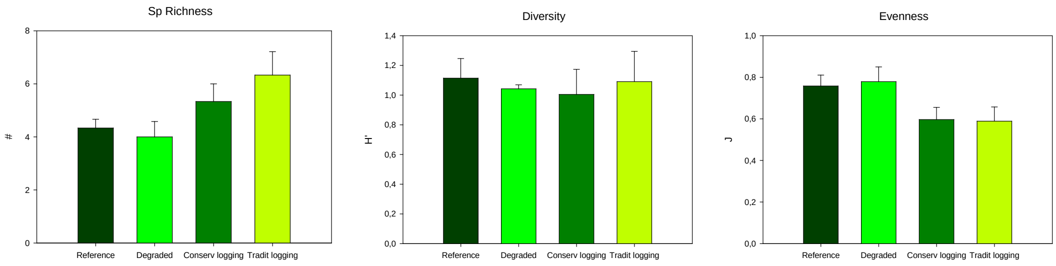

Figure 11. Number of plant species (left), Shannon-Wiener Index of diversity (center) and evenness (right) in the Reference and Degraded states in Várzea field site. Mean and standard errors are shown.

The two restoration approaches significantly reduced the cover of interpatches from 38.0 to 16.8 and 8.6% in the Conservation and Traditional Logging, respectively (Fig. 12). In addition, the quality of these interpatches also differed in the four ecosystem states; pine needles in the unburned, plant remains in the logged, and bare soil and ashes in the degraded. The size of the patches was also increasedin the two Restored areas in relation to the Degraded area, increasing significantly the width 4.7 and 5.3 times in the Conservation and Traditional Logging, respectively. The patches showed also a trend to be longer in the two Restored than in the Degraded areas but differences were not statistically significant. In addition, the typology of the patches was also contrasted. In the Degraded state patches consisted mainly in plants while litter and remains of wood extraction after the fire were the main patches in the two Restored areas. All four variables related to patch and interpatch characteristics in the Reference forest were not different than in the Restored areas.

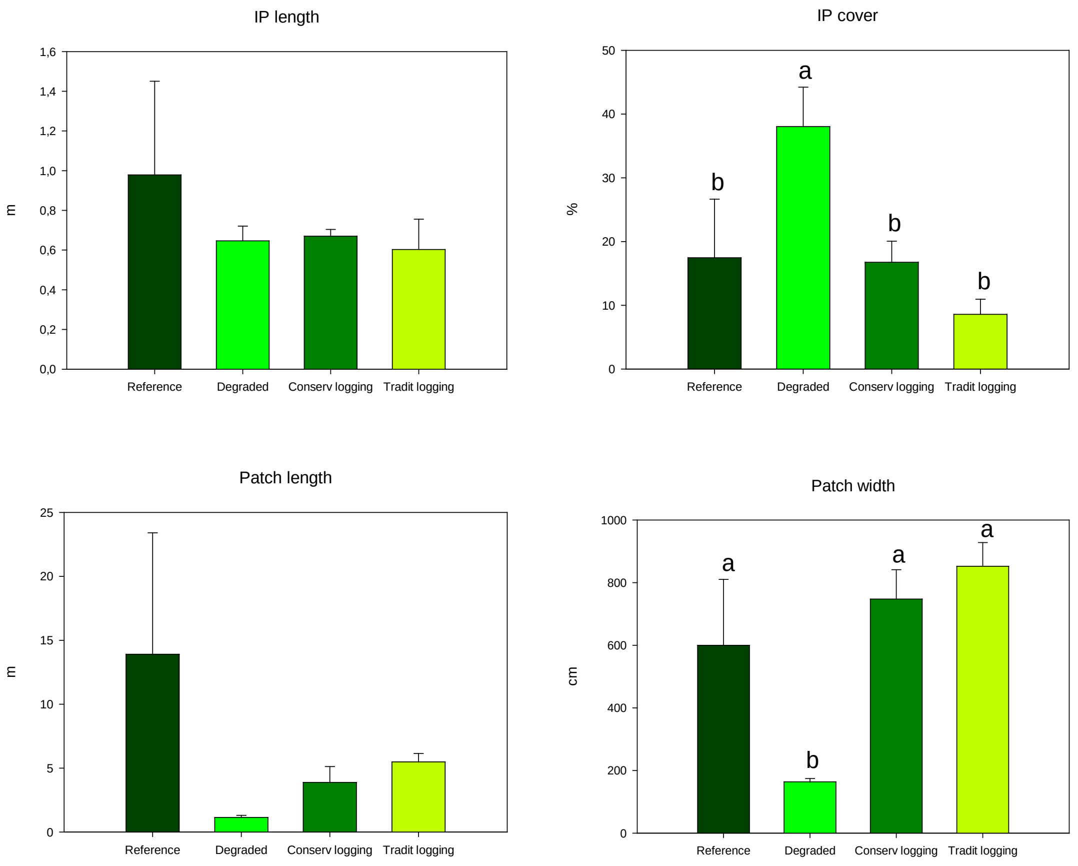

Figure 12. Values of Interpatch length (up, left), cover (up, right), patch length (bottom, left) and width (bottom, right) in the Reference, Degraded and the two Restored states in Várzea field site. Mean and standard errors are shown. Different letters indicate significant differences.

The highest total biomass of the ecosystem was of course observed in the Reference unburned forest but the biomass of the understory component, both woody and grasses, was significantly higher in the Degraded than in the other three states of the ecosystem (Fig. 13). The two restoration approaches required the participation of heavy machinery in the site impacting the recovery and the build up of vegetation biomass. These restored areas, specially the Conservation Logging, showed significant higher litter accumulation than in the Degraded, mainly due to the disposal of plant remains during wood extraction after the fire. Litter in the Restored areas represented 5 and 7 times the total standing plant biomass while in the Degraded this ratio was only 1.3. The two restoration approaches were highly efficient in protecting the soil surface with the remains of the extracted plants. The percentage of bare soil was around 5% in the two Restored areas as compared to 31% of exposed soil surface in the Degraded. Reference plots showed only 1% of unprotected soil surface.

Figure 13. Biomass of the understory (left) and litter accumulation (right) in the Reference, Degraded and the two Restored states in Várzea field site. Mean and standard errors are shown. Different letters indicate significant differences.

The three indexes of functionality of the ecosystem derived from the LFA assessment were similar in the two alternative Restored sites but were significantly improved from the Degraded situation (Fig. 14). The Stability, Infiltration and Nutrient Cycling indexes increased in a 9%, 30% and 45%, respectively, in the best of the Restored options as compared to the Degraded state. However, all indexes are still far from the values of the Reference forest.

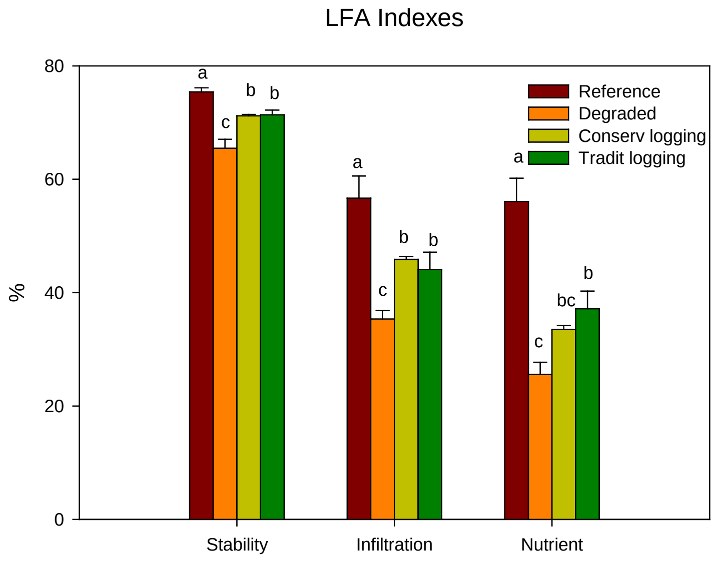

Figure 14. Values of the Stability, Infiltration and Nutrient Cycling indexes derived from LFA in the Reference, Degraded and the two Restored states in Várzea field site. Mean and standard errors are shown. Different letters indicate significant differences.

The calculation of the five ecosystem services considered showed significant differences only in C sequestration and in the combination of all five services (Fig. 15). The Conservation Restoration released very similar results than the Degraded plots while the Traditional Restoration showed a trend to increase all services from these two situations. Absolute values of Water and Soil Conservation, Nutrient Cycling and Biodiversity in the Traditional Restored sites were quite similar to the Reference forest.

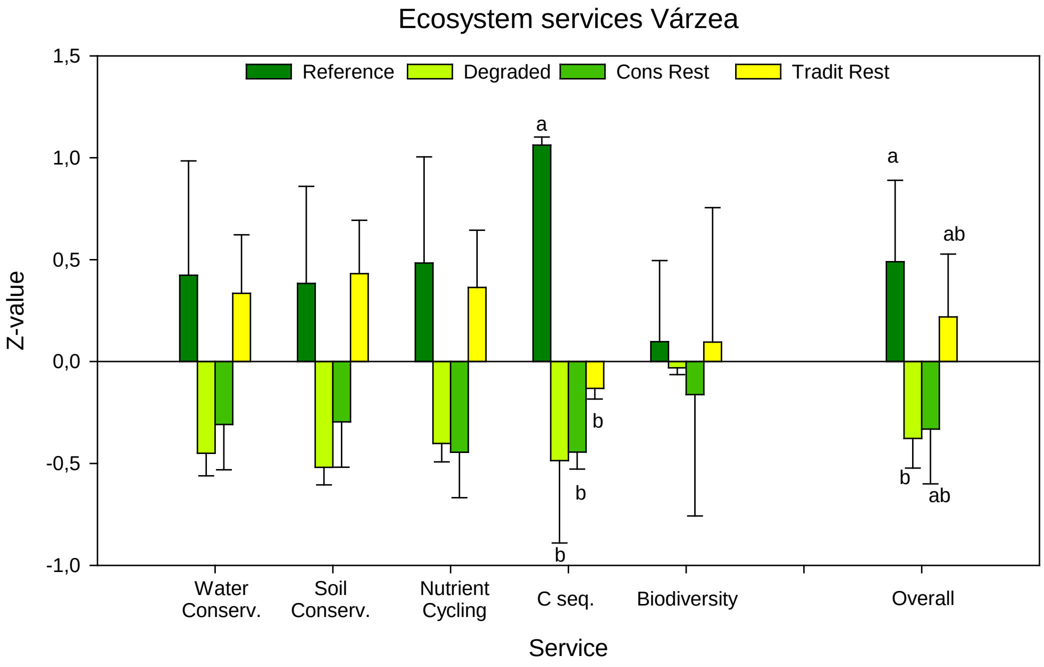

Figure 15. Standardized values of the list of ecosystem services in Várzea, as derived from combinations of the different variables acquired.Mean and standard errors are shown. Different letters indicate significant differences.

Nine out of fifteen ecosystem properties considered in this assessment changed in one or the two Restored areas in relation to the Degraded four-times-burned sites (Fig. 16). Only total plant cover and understory biomass worsened in the Conservation Restoration respect the Degraded. Properties related to the organization of the landscape, such as patch width and length and interpatch cover, released the greater changes. Significant negative changes in Interpatch cover might be interpreted as an improvement of ecosystem functioning. Also the three LFA derived indexes were significantly improved with both Restoration approaches. These findings suggest that actions carried out after the fire improved the ability of the ecosystem to retain resources in situ and, hence, the functionality of the system.

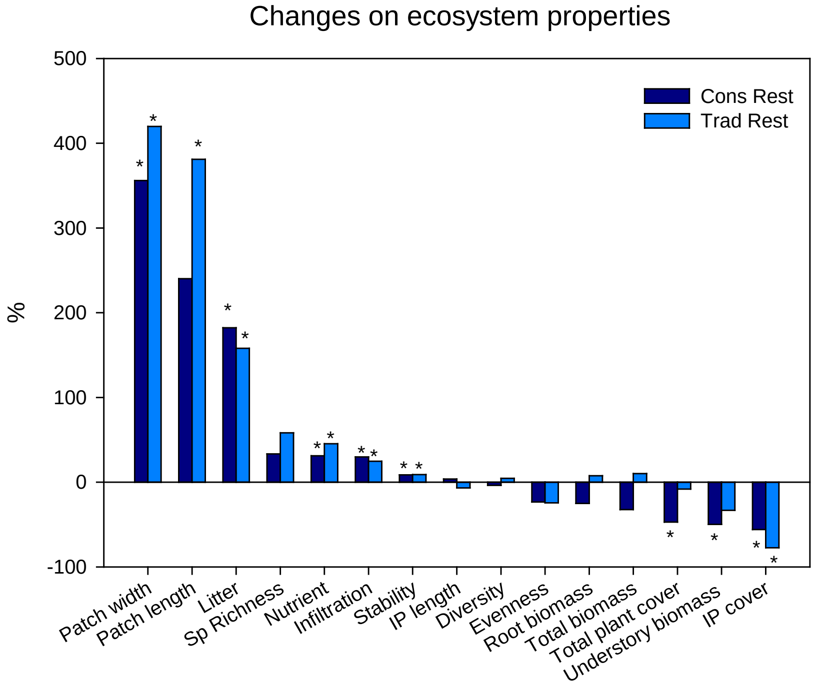

Figure 16. Losses or gains (negative and positive values, respectively) of assessed ecosystem properties in the Restored areas of the Várzea field site in relation to the Degraded areas. Asterisks denote significant differences between the correspondent Restored site and the Degraded one.

Results highlights - Várzea

- The composition of the plant community of areas subjected to Traditional and, especially, Conservation logging is closer to the Reference than the Degraded areas
- The disposal of plant remains on the soil surface after wood removal increased the cover and size of patches and, hence, the conservation of resources
- However, plant remains on soil might hamper the recruitment of some species, especially seeders, with direct consequences of diversity indexes and biomass build up
- Ecosystem functioning assessed as LFA's stability, infiltration and nutrient cycling indexes were improved by the restoration approaches but are still far from the natural forest
- In general, restoration actions improved ecosystem properties and services at the very short term after their implementation although the dynamics of the plant communities were slowdown, probably due to the impact of the heavy machinery on the earliest regenerated plants

5.1.2 Ayora

Plant cover in Ayora ecosystems ranged between 78.8 and 89.6% with significant differences between the degraded and the other two systems (Fig. 17 left). We found fifty-two vascular plant species in the sites, eight of them (subshrubs except the shrub *Rhamnus lycioides*) were exclusively present in the reference forests. The average number of plant species in the restored and the reference sites were significantly higher than in the degraded plots (23.7, 22.7 and 17.0, respectively; Fig. 17 right). Eleven species corresponding to both shrubs and subshrubs were only found in the restored plots, two of them (*Pistacia lentiscus* and *Rhamnus alaternus*) were introduced by planting. The reference communities correspond to pine forests of *Pinus pinaster* and *P. halepensis* (47.9 and 11.4% of cover, respectively) with two dominant species in the understory (*Rosmarinus officinalis* - 26.4% - and *Ulex parviflorus* - 13.3%) and scarce abundance of grasses (*Brachypodium phoenicoides* and *B. retusum* with 8.3 and 6.3%, respectively). The degraded shrublands are dominated by the shrubs *R. officinalis* (44.3%) and *Erica multiflora* (15.7%) and larger herbaceous layer (*B. retusum* and *Helictotrichon filifolium*, 21.3 and 10.8%, respectively). The species dominance inverted in the restored plots, with *B. retusum* being the most abundant species (40.2%) and *R. officinalis* dominating the shrub layer (26.2%). Other species with cover percentages above 10% were the shrub *U. parviflorus* (12.7%) and the grasses *B. phoenicoides* (12.5%) and *Stipa offneri* (10.7%).

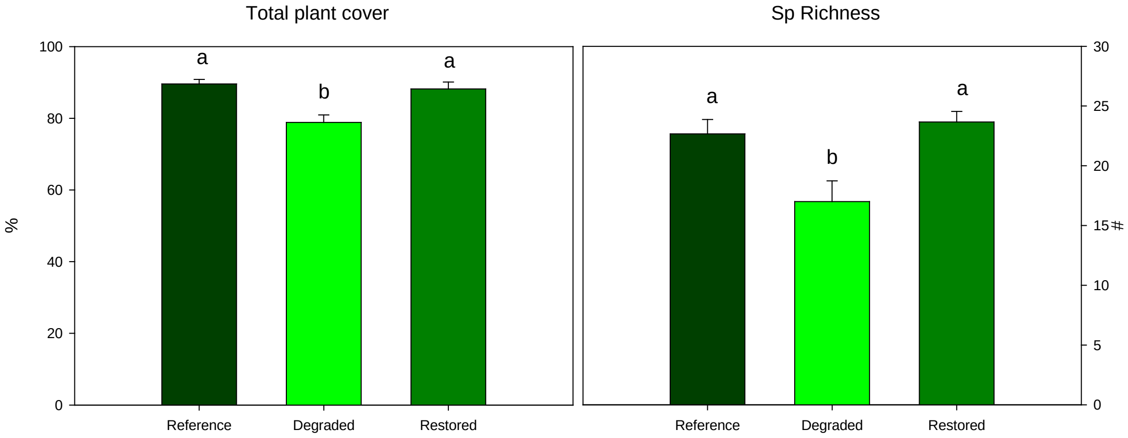

Figure 17. Total plant cover (left) and species richness (right) in the Reference, Degraded and Restored states in Ayora field site. Mean and standard errors are shown. Different letters denote significant differences.

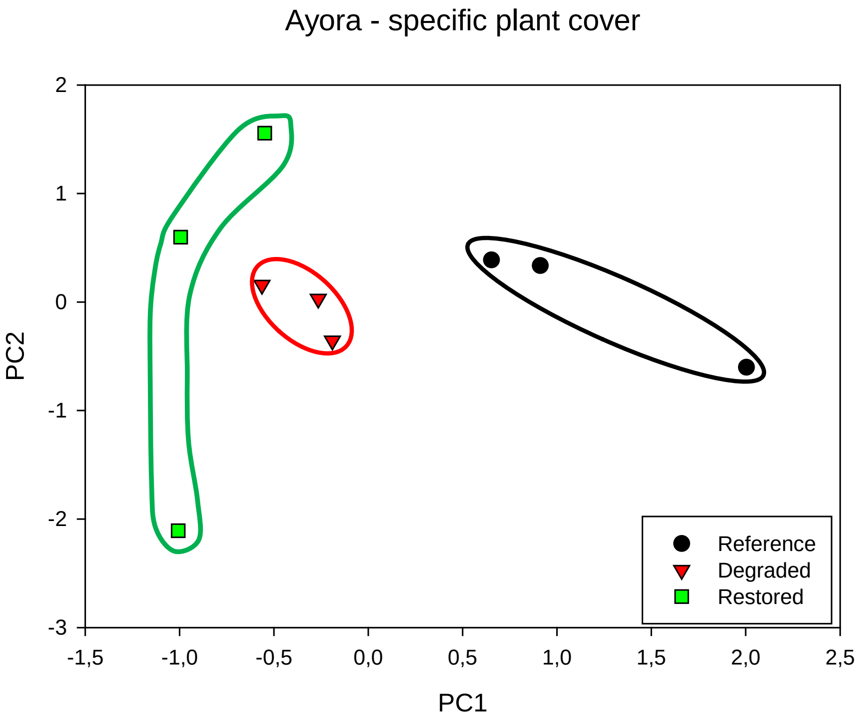

Figure 18. Distribution of Reference, Degraded and Restored plots in Ayora field sites according to the two first axis of PCA conducted on plant cover.

The two first axes of the principal component analysis accounted for 43.1% of the total variance and plots were clearly separated, especially, along the first component (Fig. 18). The reference plots showed the highest positive scores of PC1, the restored plots released the most negative ones while the degraded showed intermediate values but closer to the restored than to the reference sites. We also observed high variability in species composition and contribution within the three spatially replicates restored plots as showed by wide range of values along the second axis. Degraded areas were plotted in a tight group according with these two axes.

Diversity indexes, especially Shannon-Wiener’s, were also affected by restoration (Fig. 19). The H’ value was significantly higher in the restored plots (3.0) than in the degraded (2.2) with the reference areas showing intermediate values. Evenness values were quite similar but in this case differences were not statistically significant.

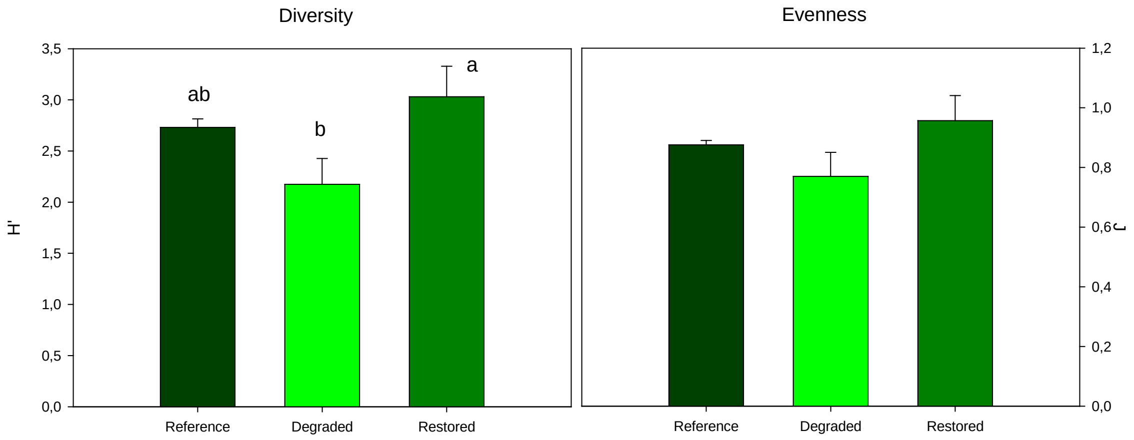

Figure 19. Shannon-Wiener Index of diversity (left) and evenness (right) in the Reference, Degraded and Restored states in Ayora field site. Mean and standard errors are shown. Different lettersdenote significant differences.

Biomass accumulation in the different layers of the ecosystem was also dependent of the ecosystem state (Fig. 20). Obviously, the biomass in the overstory was extremely higher in the reference than in the other two states where pine recovery was almost null. Regarding the understory (both shrubs and grasses), the degraded plots showed higher biomass than the restored areas. In fact, understory biomass in untreated degraded shrublands were two-fold that in the cleared plots. As a consequence, and because the contribution of the overstory in these two types of shrublands is negligible, the total aboveground biomass accumulated in the degraded was significantly higher than in the restored plots.

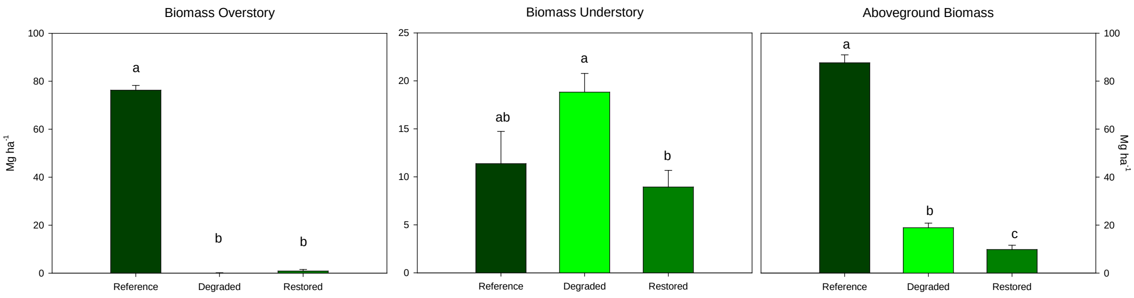

Figure 20. Plant biomass of the tree canopy (left), understory (centre) and total aboveground biomass (right) in the Reference, Degraded and Restored states in Ayora field site. Mean and standard errors are shown.Different lettersdenote significant differences. Note the different scales in Y-axes.

The litter layer in both the reference and the degraded areas were very similar although the origin of the plant material was quite different (mostly pine needles in the reference and fine and coarse shrub debris in the degraded). The restored plots showed significant reductions, above 40%, of litter accumulation (Fig. 21 left). Root biomass in the most superficial 20 cm of soil did not show significant changes due to the state of the ecosystem and ranged between a minimum of 15.8 Mg ha^{-1} in the reference to a maximum of 20.6 Mg ha^{-1} in the degraded plots (Fig. 21 right).

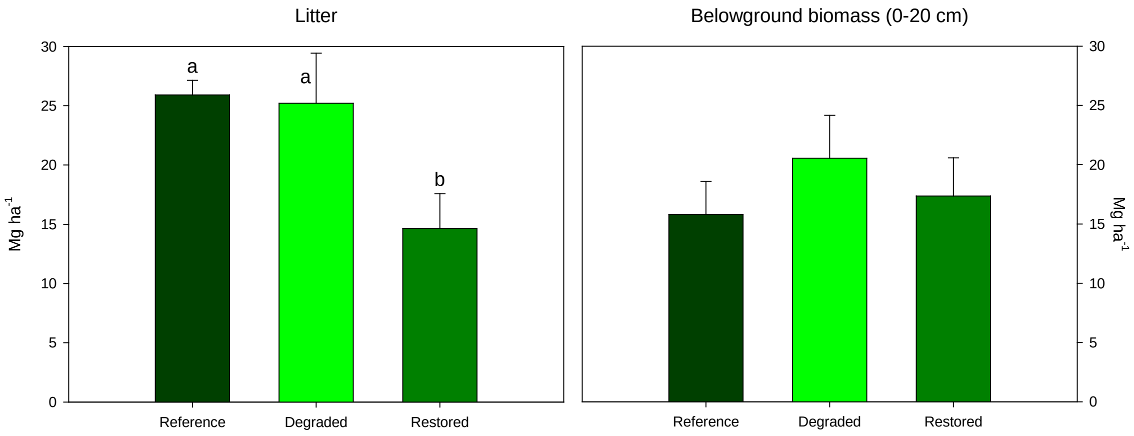

Figure 21. Litter accumulation (left) and root biomass in the uppermost 20 cm of soil (right) in the Reference, Degraded and Restored states in Ayora field site. Mean and standard errors are shown.Different letters denote significant differences.

In relation to the distribution and morphology of the sink and source areas within the landscape, the considered interpatches in Ayora included bare soil but also the matrix of perennial grasses and litter, very abundant in the three ecosystem states as it has already been mentioned. The length and percentage of interpatches in the forest systems were slightly higher than in the two shrubland systems. Interpatches averaged 1.0, 0.9 and 0.7 m in the reference, the degraded and the restored plots, while their cover was 52% in the forest and 33-34% in the two shrubland types (Fig. 22). The size of patches was increased by restoration with an average width of 4.6 m while both in the reference and in the degraded plots patches were 2.9 m wide.

The three indexes derived from the LFA assessment showed a significant reduction in the restored system while no differences were observed between the reference and the degraded states (Fig. 23). The more pronounced reduction was observed in the nutrient cycling that fell from 53% to 36% in the restored plots. Also the infiltration was sharply reduced, with values around 55% in the reference and degraded sites and 40% in the restored. The reduction in the stability index was slightly lower but however differences between states were also significant.

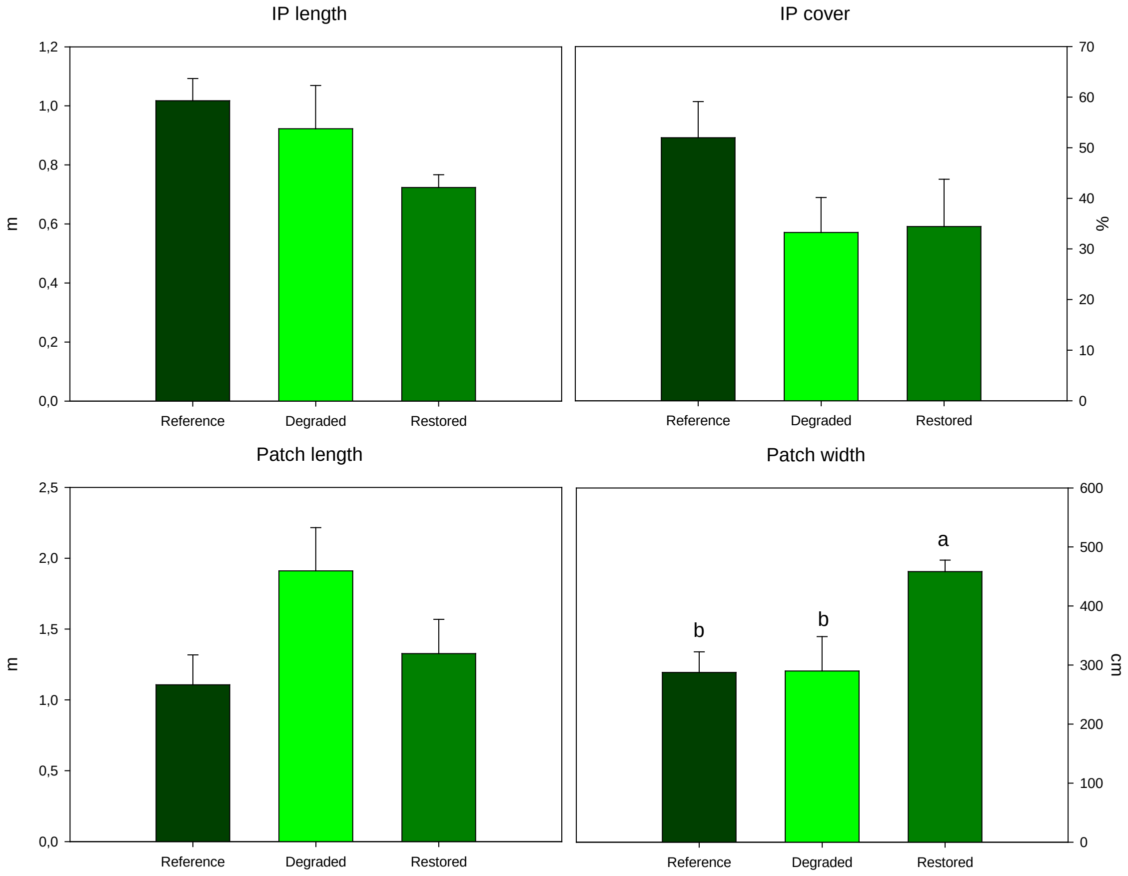

Figure 22. Values of Interpatch length (top left), cover (top right), patch length (bottom left) and width (bottom right) in the Reference, Degraded and Restored states in Ayora field site. Mean and standard errors are shown.Different letters denote significant differences.

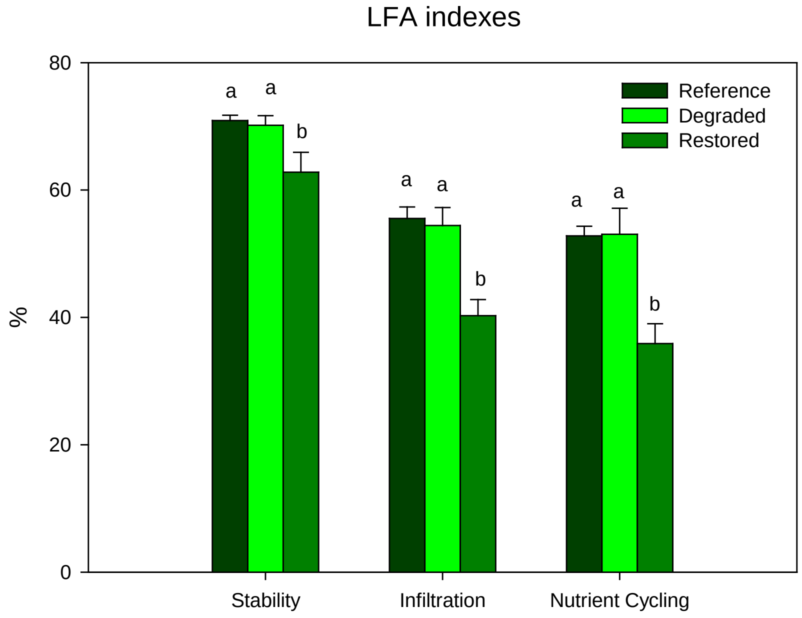

Figure 23. Values of the Stability, Infiltration and Nutrient Cycling indexes derived from LFA in the Reference,Degraded and Restored states in Ayora field site. Mean and standard errors are shown.Different letters denote significant differences.

Half of the ecosystem services were significantly affected by the state of the ecosystem: C sequestration, biodiversity and fire risk reduction (Fig. 24). The other three services (soil and water conservation and nutrient cycling), and the combination of all of them as well, showed the same trend to decrease from the reference to the degraded state while the restored showed intermediate values. The fire and the extremely limited post-fire recovery of pines, combined with the selective clearing of flammable shrubs, resulted in a significant reduction of C sequestration in the restored plots. On the contrary, restoration improved biodiversity of the degraded shrublands even beyond values of the reference state of the ecosystem. The most significant effect of the restoration actions was in reducing fire risk in relation to both the reference and the degraded plots. This was the main objective pursued with the restoration conducted in the degraded areas ten years before this assessment.

The summary of changes of ecosystem properties due to restoration (Fig. 25) showed postive effects on variables related to community composition and structure (diversity indexes and patch-interpatch distribution and morphology) while negative chenges were mainly associated to biomass accumulation (litter, understory and nutrient cycling and infiltration). However, large amounts of biomass in the understory in the degraded areas increases the C sequestred in the system but also increases flammability and, hence, fire risk.

Results highlights - Ayora

- The degraded state represents an ‘old’ shrubland where gorse disappeared by natural senescence and was dominated by rosemary
- The bigger size of patches in the restored areas can be related to the collapse of old shrubs in the degraded plots resulting in openings in the continuous shrubland
- Biodiversity was the most improved service by restoration actions
- But also the restoration approach considered in Ayora, with the reduction of levels of understory biomass, improves the ecosystem service ‘fire risk reduction’ even ten years after the application of treatments

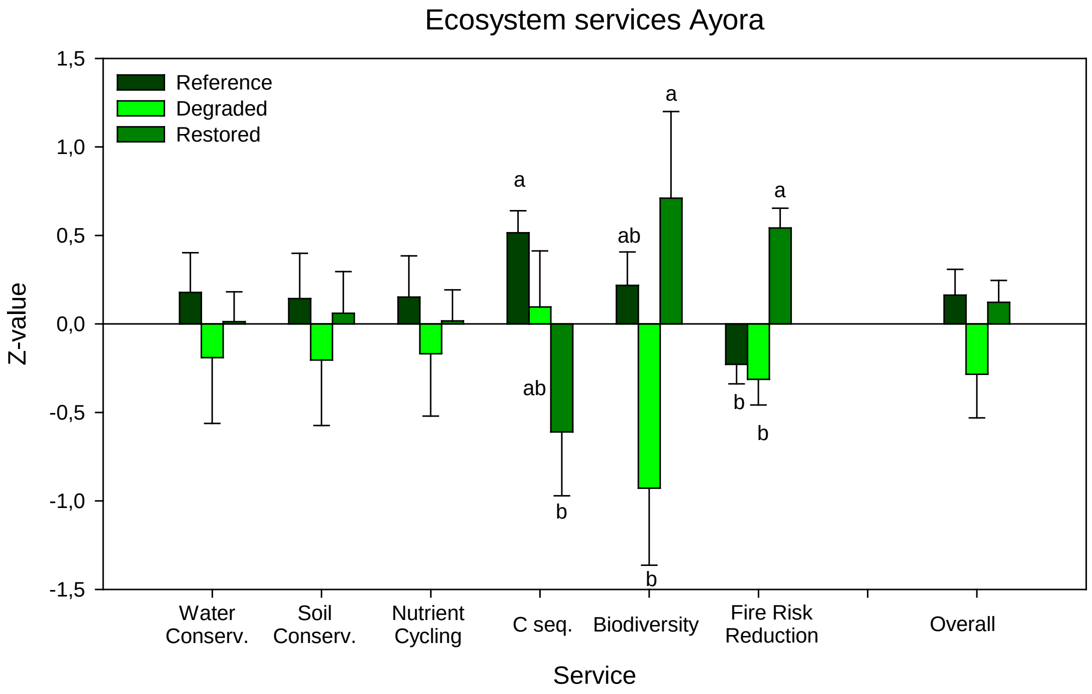

Figure 24. Standardized values (mean and standard errors) of the list of ecosystem services in Ayora, as derived from combinations of the different variables acquired. Mean and standard errors are shown.Different letters denote significant differences.

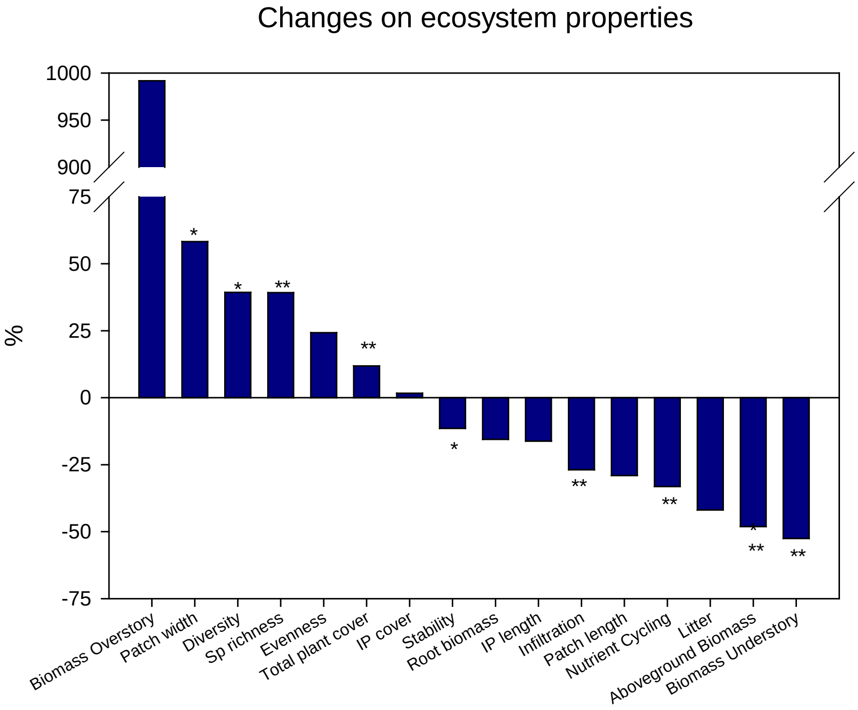

Figure 25. Losses or gains (negative and positive values, respectively) of assessed ecosystem properties in theRestored areas of the Ayora field site in relation to the Degraded. Asterisks denote significant differences between the two ecosystem states(*: 0.10<p<0.05; **: 0.05<p<0.01).

5.2 Grazing Driven Landscapes

5.2.1 Castelsaraceno

5.2.1.1 Overgrazed and Fenced systems

Plant cover in all three situations in Castelsaraceno was above 85 % but significant differences were observed between the Overgrazed and the Fenced areas (Fig. 26 left; 98.9 and 86.1%, respectively). Plant cover in the Overgrazed areas was due to 29 species while in the Fenced sites we found an average of 39.3 species (Fig. 26 right).Diversity indexes (and evenness) did not show significant differences between the three states of the ecosystem (Fig. 27). However, we observed a trend to increase diversity in the restored areas in relation to the degraded ones. Shannon-Wiener’s and evenness increased in a 34.0 and 22.0%, respectively, ten years after fencing the overgrazed areas.

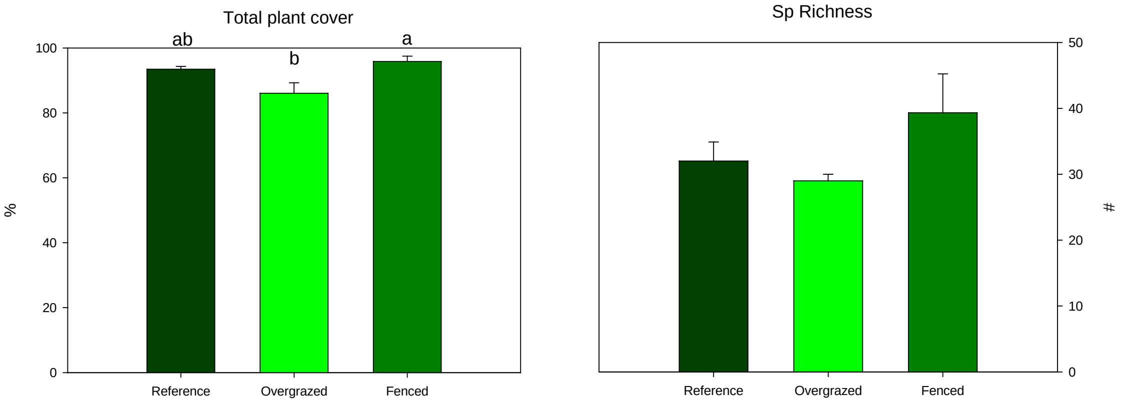

Figure 26. Total plant cover (left) and species richness (right) in the Reference,Overgrazed and Fenced states in Castelsaraceno field site. Mean and standard errors are shown. Different letters denote significant differences.

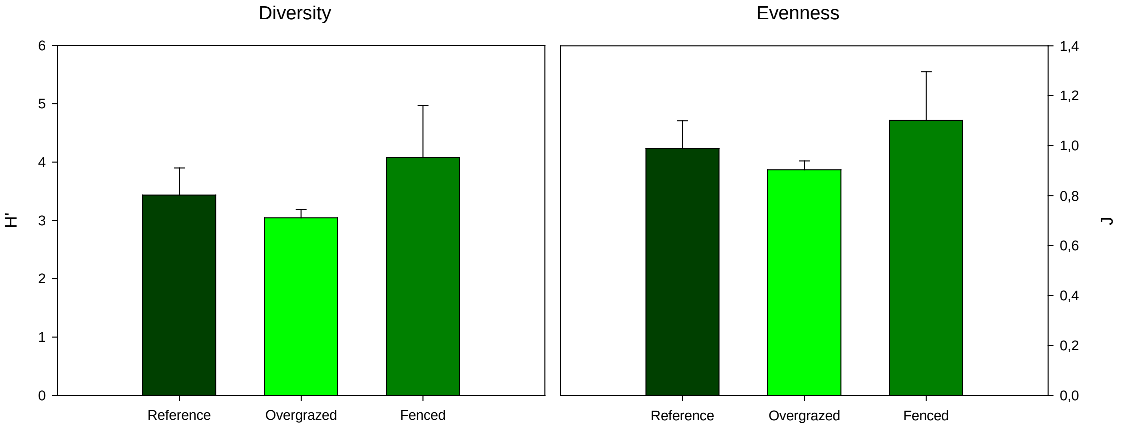

Figure 27. Shannon-Wiener Index of diversity (left) and evenness (right) in the Reference, Overgrazed and Fenced states in Castelsaraceno field site. Mean and standard errors are shown.

A total of 171 species of vascular plants were recorded in the 45 plots established in Casatelsaraceno. Plant composition was different according to the state of the ecosystem in the three experimental sites. We have analyzed plant composition separately in each of the three experimental for clarity in the changes due to the state of the ecosystem. In Favino, only 12 out of 55 recorded species were found in the three states of the ecosystem. In the Reference plots, the species with higher abundance were *Medicago minima* (17.9%), *Poa pratensis* (15.4%), *Trifolium repens* (13.4%) and *Brachypodium rupestre* (10.4%). Two species were the most abundant both in the overgrazed and fenced plots: *Trifolium incarnatum* (22.9 and 12.9%, respectively) and *B. rupestre* (18.4 and 12.9%, respectively). The graphical representation of the two first axis of the Principal Component Analysis on plant cover data clearly grouped the plots by state (Fig. 28). In Favino, with 55 species in the analysis, the first and second components explained 23.8 and 18.9% of the total variance. The extraction of species along the first two components of the PCA is shown in Annex I. The Reference plots showed highest values of the first component while the second component separated the Overgrazed plots (higher values in PC2) and the Fenced plots (lower values).

Sixty-two species were present and included in the analysis in Monte Alpi, 12 of them were present in all three ecosystem states. *Bromus erecti* and *Brachypodium rupestre* showed the highest cover in the Reference community (20.9% both), followed by *Satureja montana* and *Stipa austroitalica* with 14.4 and 10.9%, respectively. *Cynosurus cristatus* and *B. erecti* were the most abundant in the overgrazed (13.4% both) while *Stipa austroitalica* (39.9%), *B. erecti* (29.4%) and *S. minor* (19.4%) abounded in the Fenced plots. Species which characterized the References were *Lonicera caprifolium*, *Medicago lupulina* and *P. hirsutum* (Annex II). The Overgrazed and Fenced communities separated along the second axis with negative values of the Fenced plots and positive values of the Overgrazed ones.

In Piano del Campi, 80 species were found in the 9 plots and only nine species were common to the three states. *Scorzonera villosa* (38.3%), *Bothriochloa ischaemum* (35.8%) and, in a lesser extent, *Triticum ovatum* (17.4%) and *Dactylis glomerata* (11.9%) were highly represented in the Reference community. The Overgrazed plots also presented high cover of *S. villosa* and *B. ischaemum* and *T. ovatum* (36.3, 25.4 and 17.4%, respectively) but also showed high cover of *S. austroitalica* (32.3%). Eleven species showed cover values above 10% in the Fenced areas. *Xeranthemum cilindraceum*, *Daucus carota* and *B. rupestre* showed the highest cover percentages in these areas (27.9, 27.4 and 21.9, respectively). The two first axis of the PCA explained 51.8% of the total variance (34.1 and 17.7%, respectively). In this site, Reference and Overgrazed plots were quite similar in composition as observed in Fig. 28. Fenced plots were clearly separated from the rest along the first axis (Annex III) but showed high heterogeneity along the second axis.

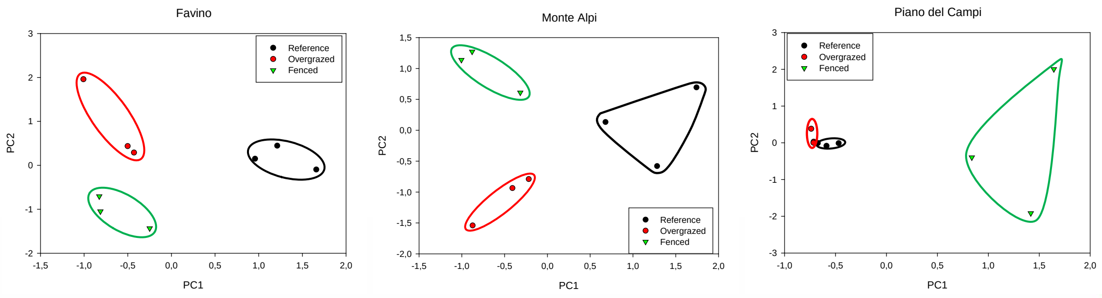

Figure 28. Distribution of Reference, Overgrazed and Fenced plots in the three sites in Castelsaraceno according to the two first axis of PCA conducted on plant cover.

We observed some small changes in the different biomass fractions of the community but these changes were not significant (Fig. 29). Average aboveground biomass was 36.6% higher in the Fenced than in the Overgrazed areas and similar to the values of the Reference ecosystem. Belowground biomass in the uppermost 15 cm of the soil showed the same trend than aboveground biomass but, surprisingly, litter showed the opposite trend with a reduction of about 50% in the Fenced plots. However, data heterogeneity was large enough to prevent significant differences.

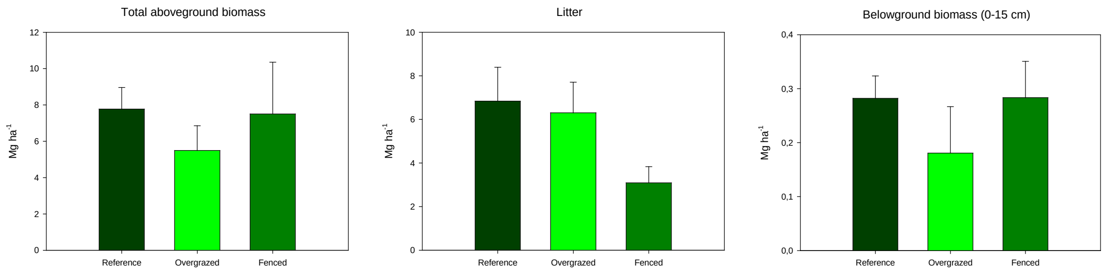

Figure 29. Total aboveground biomass (left), litter accumulation (centre) and belowground biomass on the uppermost 15 cm of soil (right) in the Reference, Overgrazed and Fenced states in Castelsaraceno field site. Mean and standard errors.

Fencing significantly increased the length but not the total cover of interpatches in relation to both the Overgrazed and the Reference situations (Fig. 30). However, in all the systems of Castelsaraceno interpatches do not represent bare soil areas but a matrix of herbs and grasses. Size of patches, woody plants or a mix of woodies and herbs, tended to increase in the Fenced areas with increases aorund 60% both in length and width in relation to the Overgrazed areas.

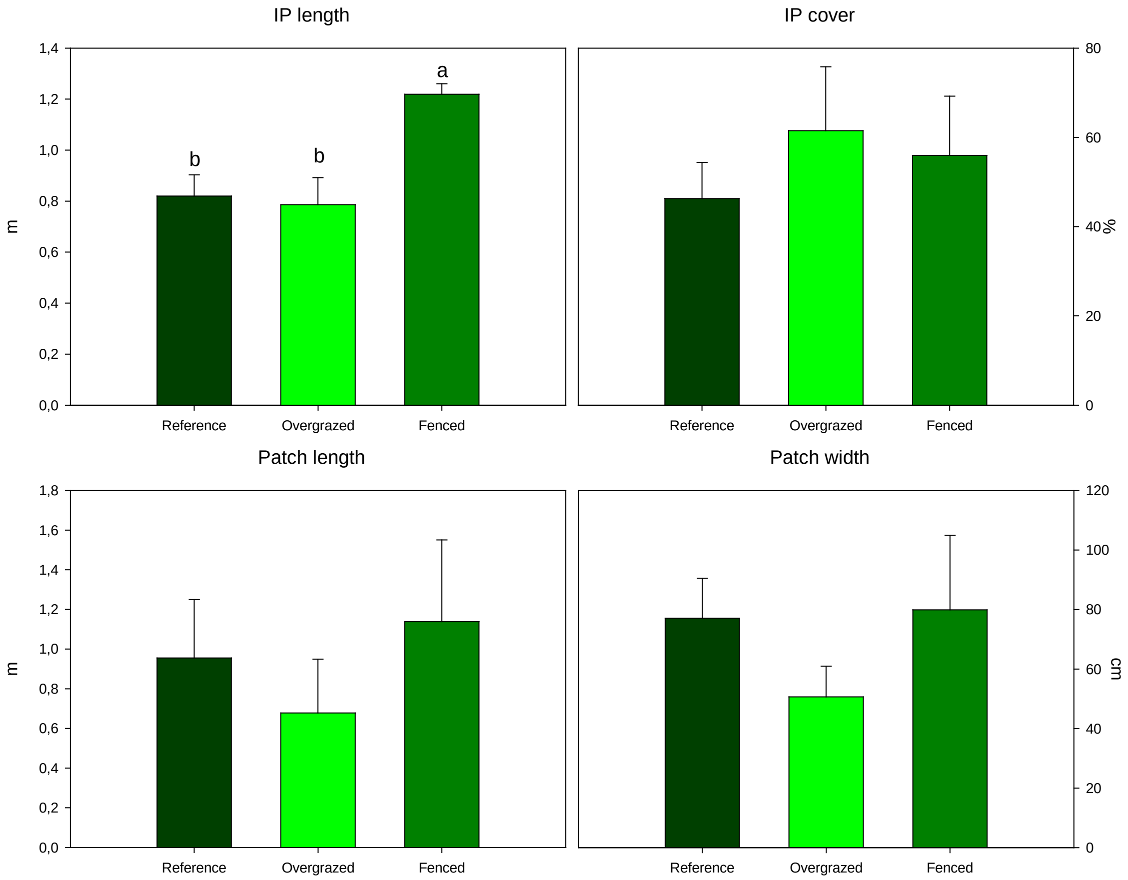

Figure 30. Values of Interpatch length (top left), cover (top right), patch length (bottom left) and width (bottom right) in the Reference, Overgrazed and Fenced states in Castelsaraceno field site. Mean and standard errors are shown.Different letters show significant differences (p<0.050).

The three indexes derived from LFA were quite similar in the three studied situations (Fig. 31). Stability is only slightly reduced from the Reference in the Overgrazed and Fenced areas while the nutrient cycling index is relatively increased in a 19.8% in the Fenced as compared to the Overgrazed plots.

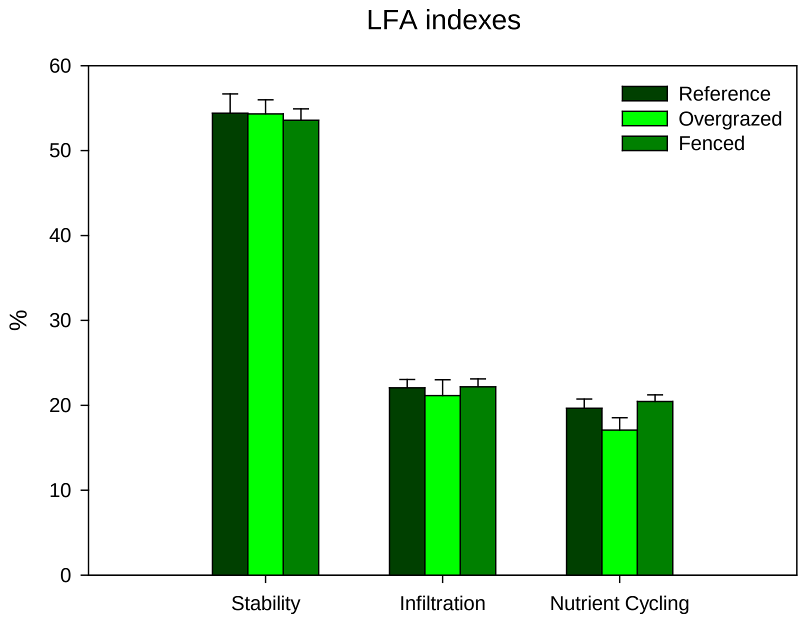

Figure 31. Values of the Stability, Infiltration and Nutrient Cycling indexes derived from LFA in the Reference, Overgrazed and Fenced states in Castelsaraceno field site. Mean and standard errors are shown.

Ecosystem services calculated from these properties are shown in Fig. 32. Restoration by fencing implied an increase (not significant) of nutrient cycling, C sequestration and, especially, biodiversity from the Overgrazed state of the ecosystem. This former service was also well above in the Restored than in the Reference sites. Water and soil conservation did not show important changes due to restoration. The combination of all calculated services showed that the Restored system through fencing overgrazed areas resulted in an increase of ecosystem services but still below the services provided by the Reference ecosystem.

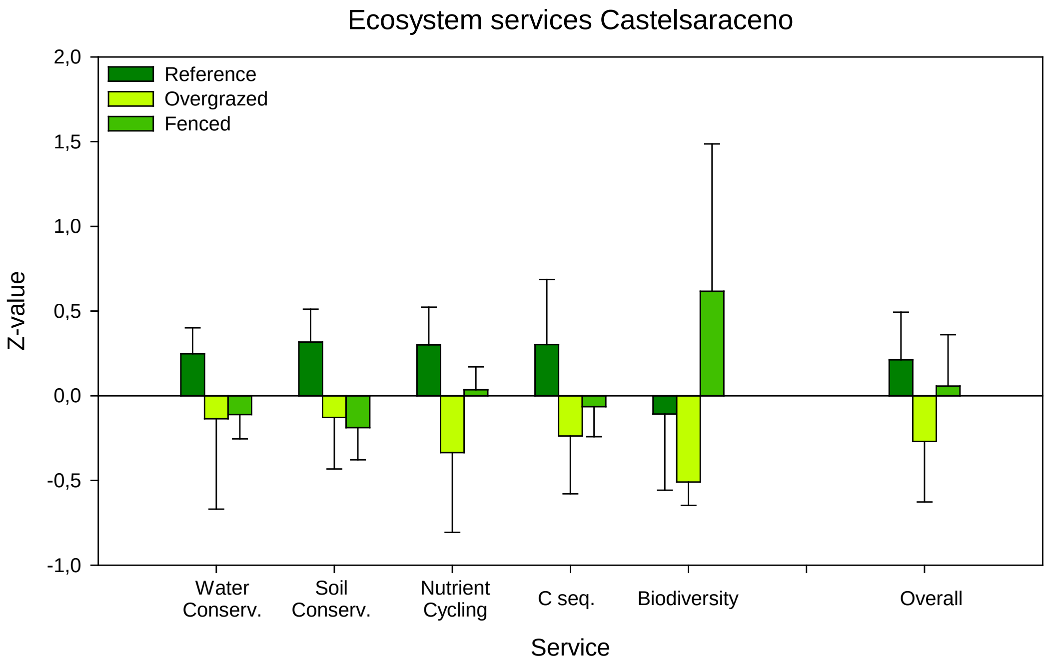

Figure 32. Standardized values of the list of ecosystem services in Castelsaraceno, as derived from combinations of the different variables acquired. Mean and standard errors are shown.

All the ecosystem properties evaluated in this study were higher in the Fenced than in the Overgrazed lands of Castelsaraceno except litter accumulation (Fig. 33). However, these improvements are not yet translated to significantly better ecosystem services after the type of restoration assessed.

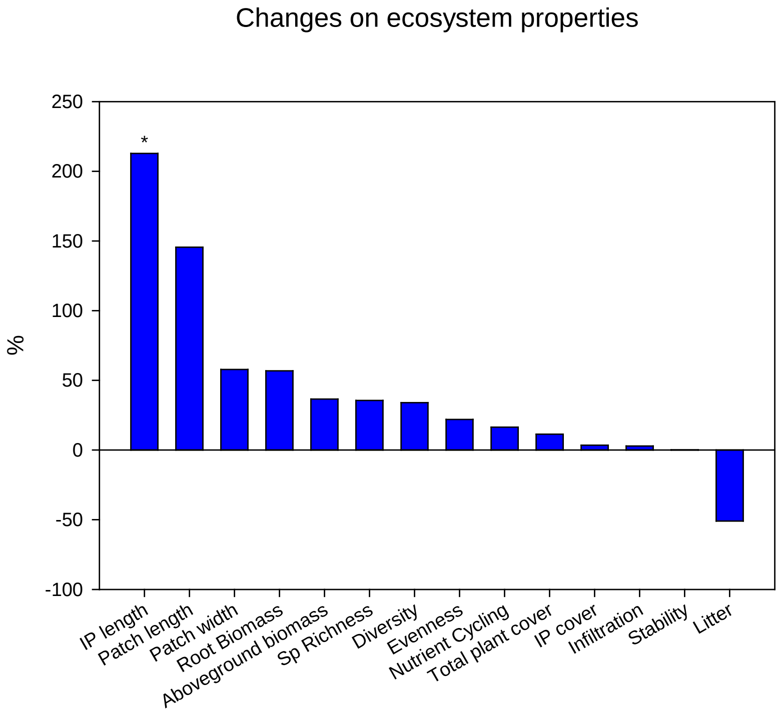

Figure 33. Losses or gains (negative and positive values, respectively) of assessed ecosystem properties in the Fenced areas of the Castelsaraceno field site in relation to the Overgrazed. Asterisks denote significant differences between ecosystem states.

5.2.1.2 Undergrazed and Cleared systems

The opposite to the above mentioned situation is represented by areas in which the grazing pressure is very low, with symptoms of shrub encroachment and where the restoration approach consisted in clearing woody vegetation. In comparison to the reference grassland, both the degraded and the restored plots did not show significant changes either in total plant cover or number of vascular plant species (Fig. 34). Plant cover in all three situations was very high (above 92%) and the total number of plant species found was 142, slightly higher in the Undergrazed and, in a lesser extent, in the Cleared states than in the Reference. Diversity and evenness indexes showed a trend to increase (26% higher values in relation to the degraded state) in the Cleared plots in relation to the other two situations (Fig. 35).

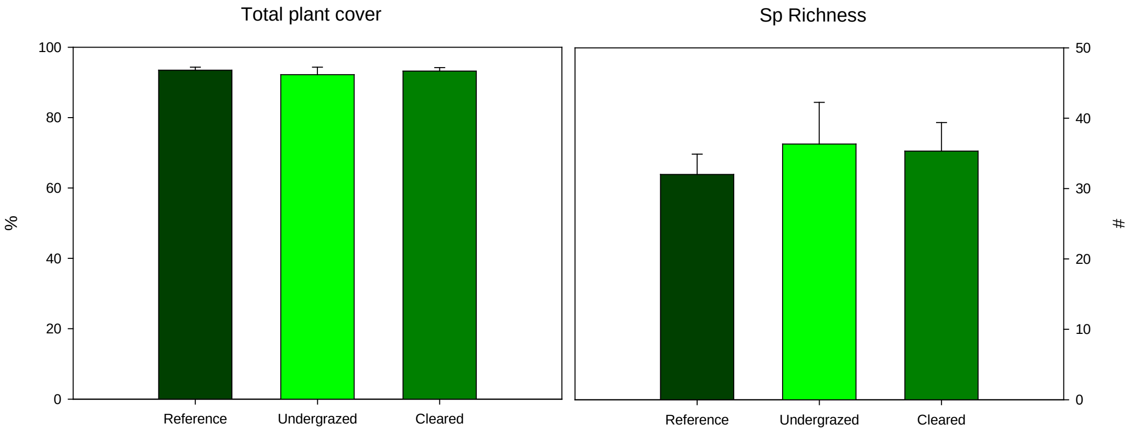

Figure 34. Total plant cover (left) and species richness (right) in the Reference, Undergrazed and Cleared states in Castelsaraceno field site. Mean and standard errors are shown.

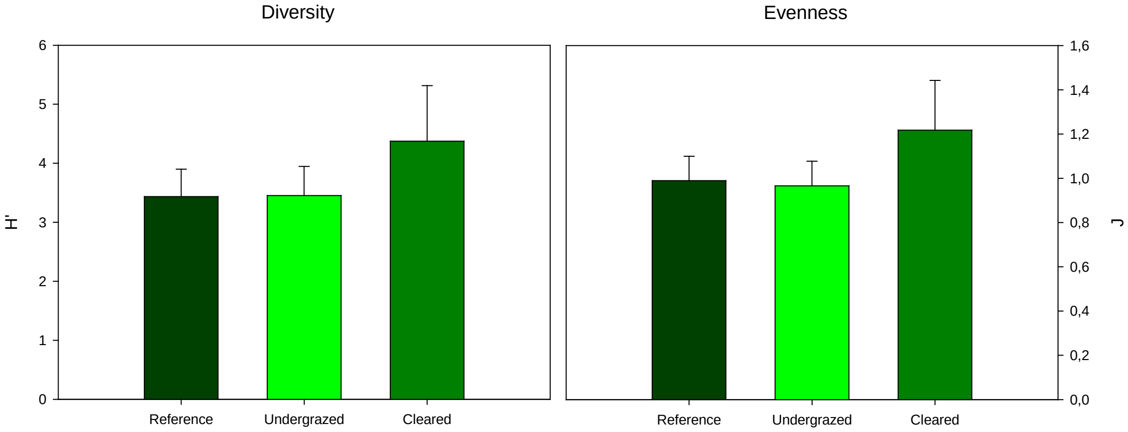

Figure 35. Shannon-Wiener Index of diversity (left) and evenness (right) in the Reference, Overgrazed and Fenced states in Castelsaraceno field site.Mean and standard errors are shown.

Twenty-four species were shared by the three states of the ecosystem. Also 24 species were only found in the Reference plots, 20 in the Undergrazed plots, and 29 were exclusive of the Cleared ones. In Favino, the three most abundant species in the Reference plots (*Medicago minima*, *Poa pratensis* and *Trifolium repens* with 17.9, 15.4 and 13.4%, respectively) were not found in the other states. *Brachypodium rupestre*, one of the species present in all communities, was the most abundant one in the Undergrazed (50.7%), but the second and third species with highest cover (*Spartium junceum* and *Festuca circummediterranea*, with 45.3 and 26.4%, respectively) were specific of the Undergrazed plots. Another woody species, such as *Crataegus monogyna* (15.4%), presented relative high cover in this situation. In the Cleared plots, the species with highest cover was *Agrostis stolonifera* (24.9%) which was absent in the Reference and Undergrazed plots. These contrasted composition of species resulted in clearly separated groups after PCA analysis in the three spatially replicated sites (Fig. 36). The two first components of the analysis in Favino explained 48.9% of the total variance and included 73 species. The three replicates of both the Reference and Undergrazed plots were very close in the graphical representation of these two axes revealing high similarity of plant composition while the Cleared plots showed a wider range of values along these two axes. The extraction of species in axis 1 and 2 of the PCA are shown in Annex IV. Something similar was observed in Monte Alpi (90 species in the analysis and 45.1% of explained variance by the two first axes), with plots plotted close for the Undergrazed and Reference states but more separated, especially along the second axis, in the case of Cleared plots (Annex V). The proximity of the reference and undergrazed groups of plots can be related to the grazing pressure that might not be much contrasted. In Piano del Campi, with 69 species included in the PCA (46.1% of explained variance by the two first axes), the three groups of plots were separated along the first axis and the References also showed lower values of the second axis than the Undergrazed and Cleared groups of plots (Annex VI).

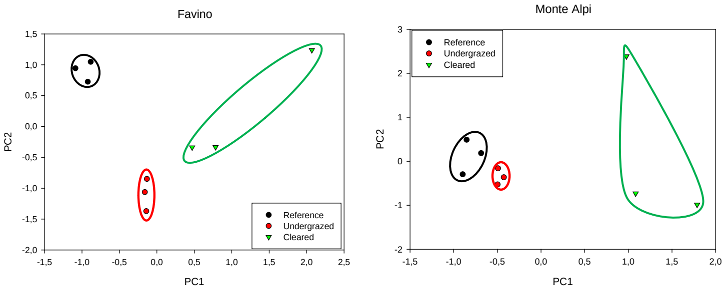

Figure 36. Distribution of Reference, Undergrazed and Cleared plots in the three sites in Castelsaraceno according to the two first axis of PCA conducted on plant cover.

We observed an opposite effect of clearing in above and belowground accumulation of biomass (Fig. 37). Both aerial plant biomass and litter were sharply reduced (but not significantly) to similar values than the reference areas by the restoration treatment implemented in Undergrazed plots. These reductions were around 50%. In contrast, belowground biomass in the uppermost 15 cm of soil was increased in 56.6% in the restored sites in comparison to the Undergrazed. These findings might be related to the relative changes in plant composition and species life traits (life cycle, leaf life span and production, rooting patterns) associated to the clearing treatment.

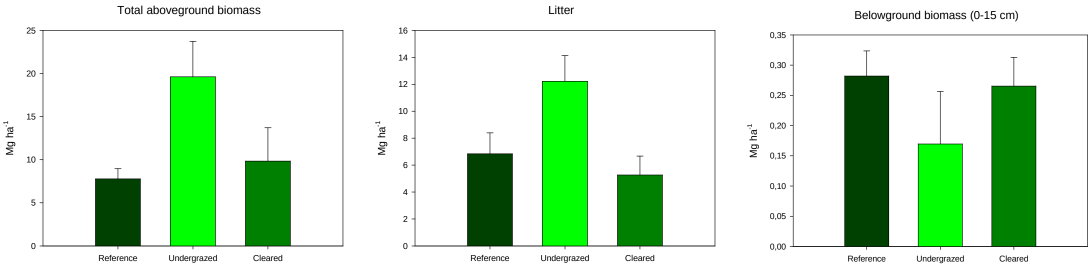

Figure 37. Total aboveground biomass (left), litter accumulation (centre) and belowground biomass on the uppermost 15 cm of soil (right) in the Reference, Undergrazed and Cleared states in Castelsaraceno field site. Mean and standard errors.

The arrangement of vegetation in the space was only slightly changed ten years after restoration. The length and cover of interpatches were very similar in all three situations (Fig. 38). The percentage of land associated to interpatches (a matrix of grasses and forbs) increased in ca. 56% in the Cleared sites as compared to the Undergrazed ones. Conversely, the size of patches (mainly due to woody plants) was sharply reduced in 24 and 29% (length and width, respectively). The Cleared areas were much more similar to the Reference than the degraded Undergrazed ones.

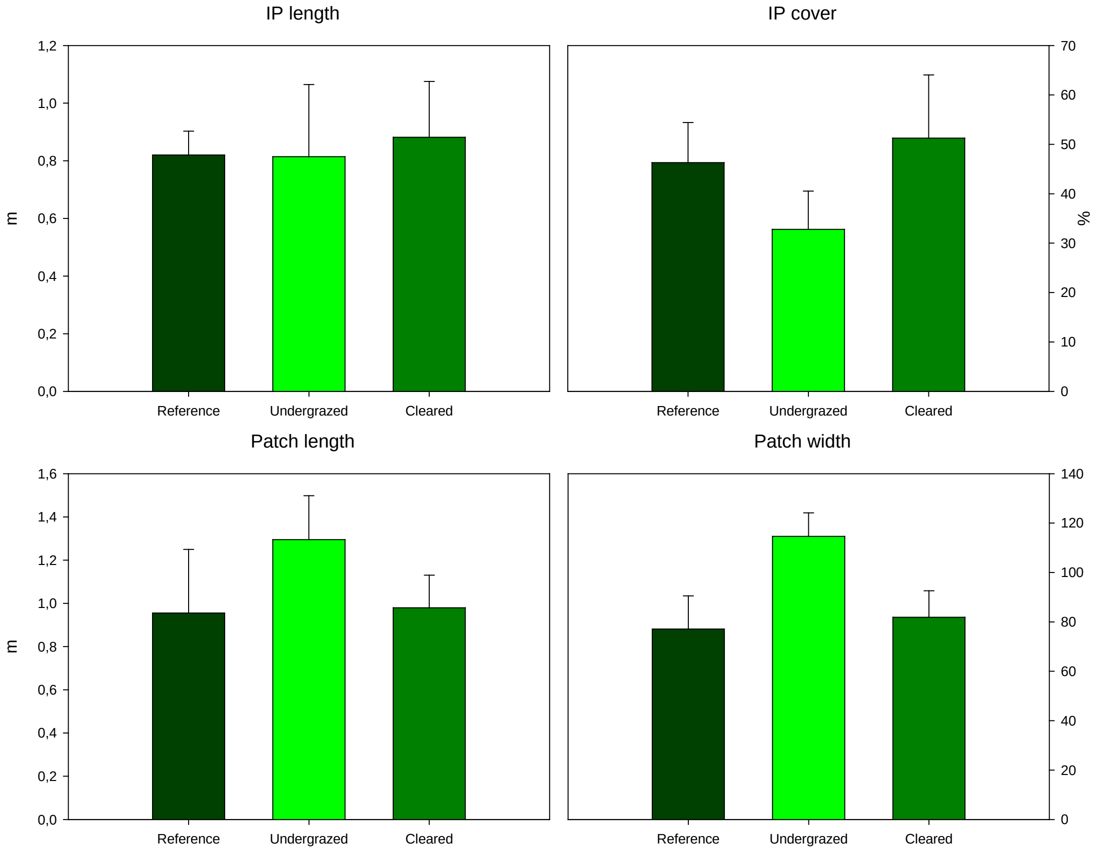

Figure 38. Values of Interpatch length (top left), cover (top right), patch length (bottom left) and width (bottom right) in the Reference, Undergrazed and Cleared states in Castelsaraceno field site. Mean and standard errors are shown.

The stability, infiltration and nutrient cycling indexes derived from the LFA assessment did not show important differences between the three states of the ecosystem (Fig. 39). However, the Cleared plots showed a slight improvement of these indexes (lower than a relative 10% in all cases) in relation to the Undergrazed plots.

All ecosystem services except C sequestration were improved in the Cleared plots in relation to the Undergrazed ones (Fig. 40). The latter showed a clear reduction in soil and water conservation and nutrient cycling as compared both to the Reference and the Cleared sites. On the contrary, the reduction of the grazing pressure increased C sequestration notably in respect to the two alternative situations. The highest value of the combination of all the services considered in this study was observed in the restoration areas. However, there are other provisioning services associated to grazing that might reverse the final balance.

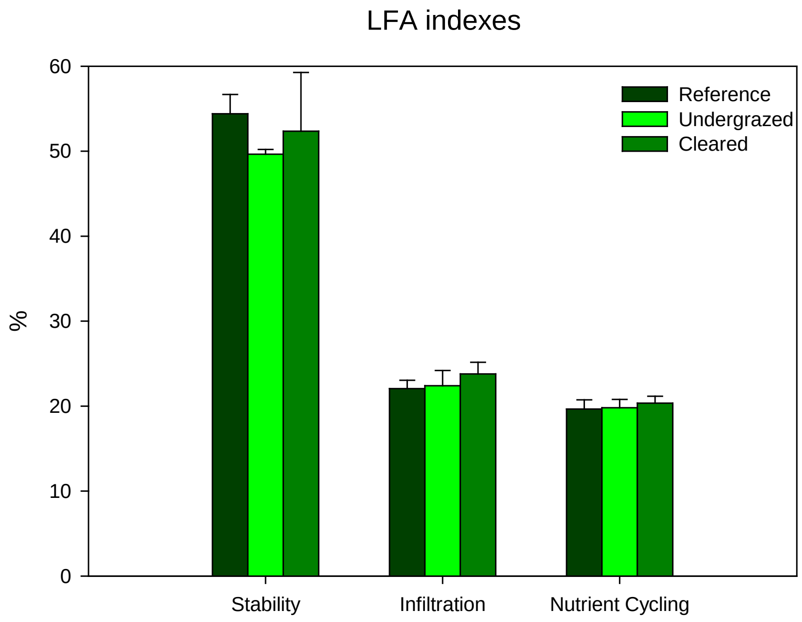

Figure 39. Values of the Stability, Infiltration and Nutrient Cycling indexes derived from LFA in the Reference, Overgrazed and Fenced states in Castelsaraceno field site. Mean and standard errors are shown.

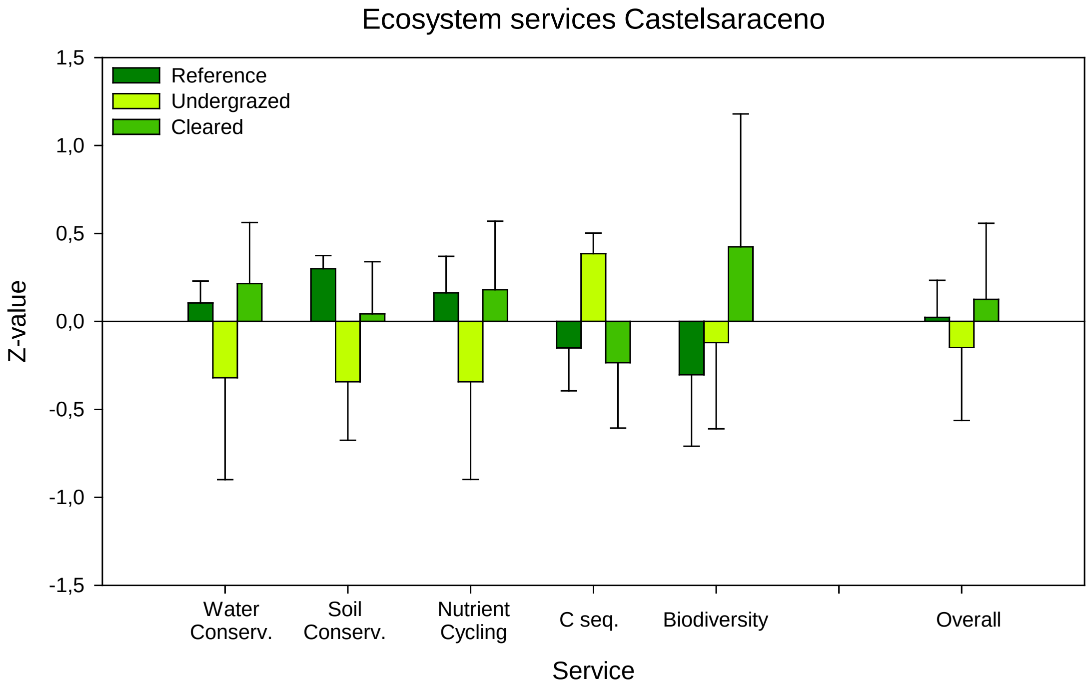

Figure 40. Standardized values of the list of ecosystem services in Castelsaraceno, as derived from combinations of the different variables acquired. Mean and standard errors are shown.

In this case, aproximately half of the ecosystem properties we have evaluated showed improvements and the other half were reducedafter restoration while other such as total plant cover (close to 100% in both cases) and species richness showed very little changes (Fig. 41). Only the size of the patches and litter accumulation were significantly reduced in the Cleared areas in relation to the Undergrazed.

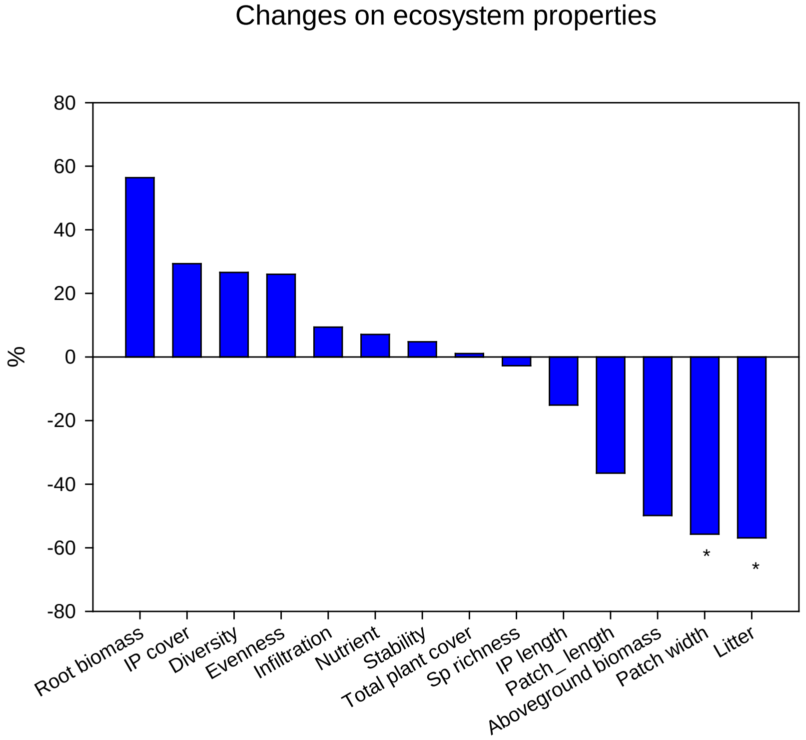

Figure 41. Losses or gains (negative and positive values, respectively) of assessed ecosystem properties in the Cleared areas of the Castelsaraceno field site in relation to the Undergrazed. Asterisks denote significant differences between ecosystem states.

Results highlights - Castelsaraceno

- Different restoration approaches were considered depending on the sense of the grazing pressure: Fencing in case of overgrazing, and clearing woody vegetation in case of undergrazing.
- The degradation due to overgrazing seems more pronounced than that due to undergrazing. The losses of services provided in relation to the reference productive grasslands in the overgrazed are higher than in the undergrazed.
- Ten years after the application of restoration, the ecosystem services evaluated in this study have been slightly improved.
- Biodiversity is the most improved service associated to the two restoration approaches.
- In the areas affected by overgrazing, restoration did not achieve the overall balance of services provided by the references while in the undergrazed areas the restoration through clearing showed the highest balance of services.
- Provisioning services associated to grazing should be specifically considered in Castelsaraceno and integrate them into the final analysis.

5.2.2 Messara

As previously mentioned, plant cover, diversity indexes and biomass estimation (above and belowground and litter) are not yet available for the two restored sites in Messara. The assessment is therefore based just of Landscape Function Analysis variables and derived indexes.

5.2.1.1 Melidochori

The contribution of the interpatches in the Restored plot is higher than in the other three states of the ecosystem (Fig. 42). The high heterogeneity of, especially, the Reference and the Semi-Degraded plots prevented significant differences of interpatch length but not of the cover. However, the total interpatch cover in the restored plot was equally distributed into litter and bare soil interpatches while both in the Reference and the Degraded plots bare soil contributed to more than 60% of the respective interpatches.

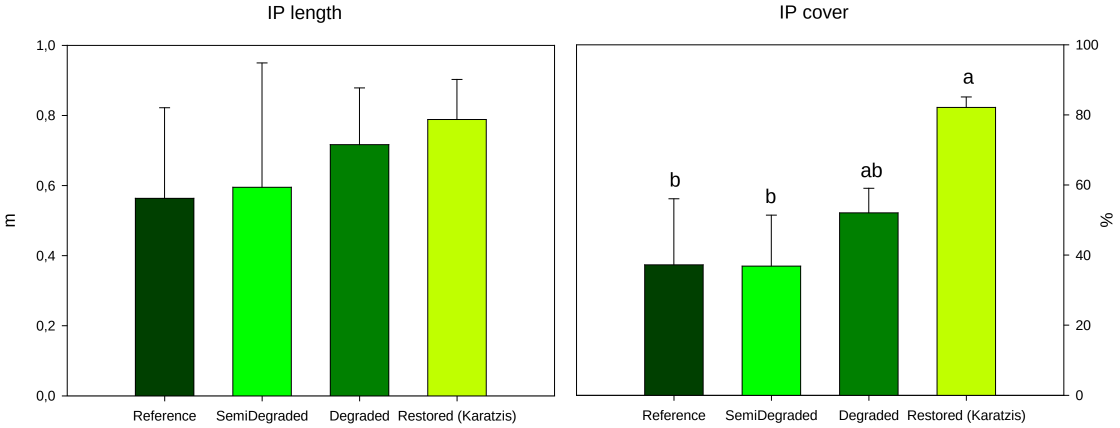

Figure 42. Values of Interpatch length (left) and cover (right) in the Reference, Semi-Degraded, Degraded and the Melidochori Restored plot in Messara field site. Different letters indicate significant differences.

Similarly, patches were smaller, especially their average width, in the Restored plot than in the other three states (Fig. 43). These data are deceptive because, in reality, there are larger patches of vegetation but not on the ground but in the canopy of the carob trees.

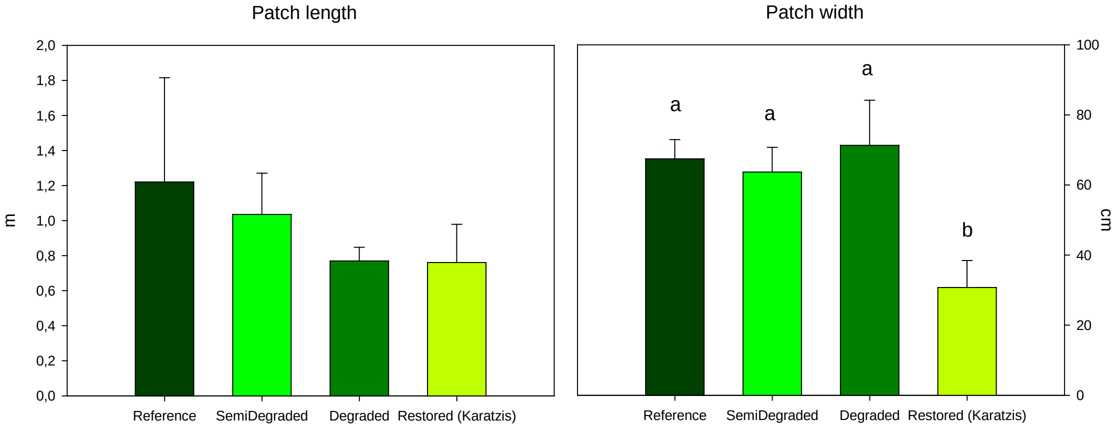

Figure 43. Values of patch length (left) andwidth (right) in the Reference, Semi-Degraded, Degraded and the Melidochori Restored plot in Messara field site. Different letters indicate significant differences.

The combination of the soil surface assessment and spatial contribution of patches and interpatches resulted in LFA indexes not different among ecosystem states (Fig. 44). Restoration improved the infiltration index in relation to the Degraded areas from 24.9% to 30.9% and was also even higher than the observed index in the Reference (29.3%). On the other hand, the highest reduction was perceived in the nutrient cycling as compared to the Reference but it was not so sharp in relation to the Degraded (from 22.2 to 19.3%).

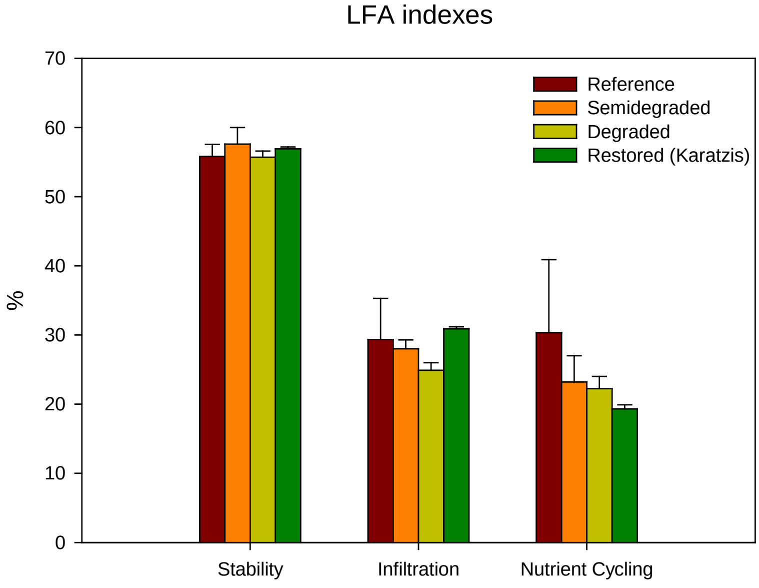

Figure 44. Values of the Stability, Infiltration and Nutrient Cycling indexes derived from LFA in the Reference, Semi-Degraded, Degraded and the Melidochori Restored plot in Messara field site. Mean and standard errors are shown.

5.2.1.2 Odigitria

The percentage of the land corresponding to interpatches in the Odigitria restored plot was slightly lower than in the Degraded areas and rather similar to the Reference and the Semi-Degraded ones (Fig. 45). However, the restored plot showed the longest interpatches (1.1 m vs 0.7 m in the Degraded) although differences were not significant. Interpatches in Odigitria were constituted by a mixture of grasses, stones and bare soil.

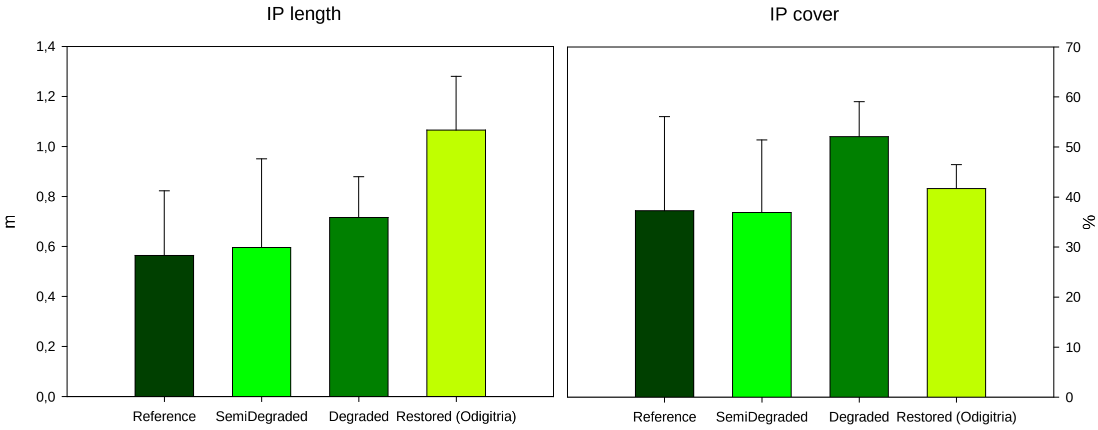

Figure 45. Values of Interpatch length (left) and cover (right) in the Reference, Semi-Degraded, Degraded and the Odigitria Restored plot in Messara field site.

Patches in the restored plot averaged 1.61 m long and 0.95 m wide (Fig. 46) and were mainly constituted by shrubs and subshrubs (24.5 and 22.5% of the total surface area, respectively). The Degraded plots showed smaller patches (0.77 m long and 0.71 m wide) with lower proportion of shrubs (18.0%) and slightly higher cover of subshrubs (27.9%, mainly the unpalatable species *Urginea maritima*). The Reference plots showed the highest diversity of patch types where shrubs were the most abundant (33.8%) followed by subshrubs (15.2%) and tussock grasses (10.0%).

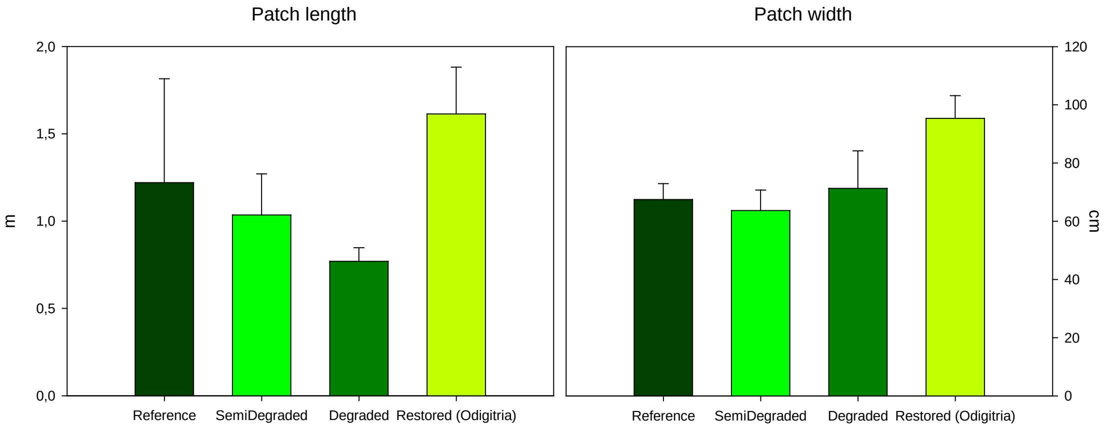

Figure 46. Values of patch length (left) andwidth (right) in the Reference, Semi-Degraded, Degraded and the Odigitria Restored plot in Messara field site. Different letters indicate significant differences.

LFA indexes in Odigitria restoration showed some differences to the restoration in Melidochori. Stability and infiltration did not change in relation to the Degraded plots while nutrient cycling was slightly improved (from 22.2 to 26.6%; Fig. 47). The Reference released the highest values for both infiltration and nutrient cycling indexes while the stability was highest in the Semi-Degraded.

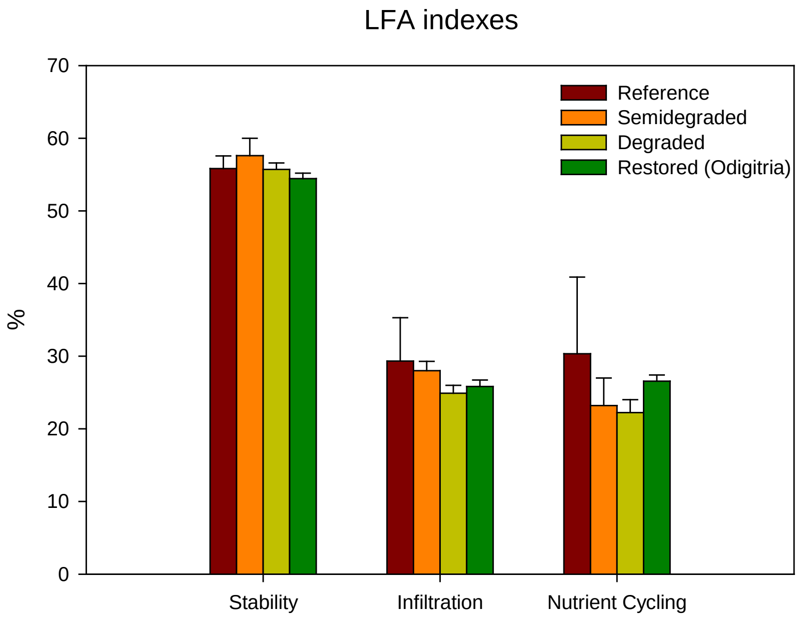

Figure 47. Values of the Stability, Infiltration and Nutrient Cycling indexes derived from LFA in the Reference, Semi-Degraded, Degraded and the Odigitria Restored plot in Messara field site. Mean and standard errors are shown.

In general, most of the ecosystem properties evaluated through the LFA assessment were improved with restoration, especially with the Odigitria approach (Fig. 48). The perceived significant increase of interpatch cover and length cannot be seen as positive changes.

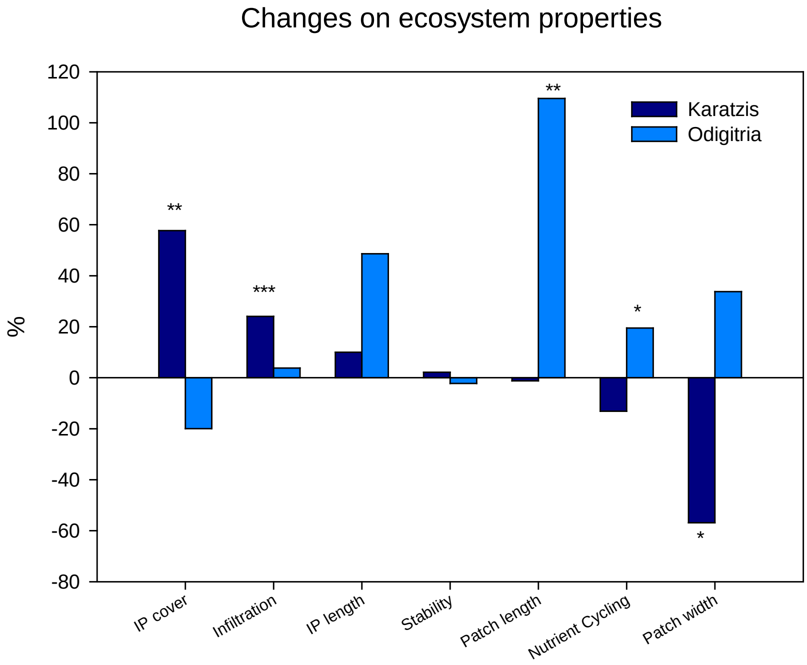

Figure 48. Losses or gains (negative and positive values, respectively) of assessed ecosystem properties in the Restoredplots of the Messara field site in relation to the Degraded areas. Asterisks denote significant differences between ecosystem states (*: 0.05<p<0.10; **: 0.01<p<0.05; ***: p<0.01).

Results highlights - Messara

- The lack of areas with similar biophysical properties and land use histories that underwent any kind of restoration action in the past impeded to fully apply the ecosystem service protocol
- The two restoration plots found included the transformation of overgrazed areas to carob tree orchards
- Contrary to expected, interpatch cover and size were enhanced in the restored areas but the cover of bare soil was reduced as compared to the overgrazed degraded areas
- The Melidochori approach significantly improved the infiltration index from the degraded lands while the Odigitria restoration enhanced the nutrient cycling
- Plant cover, diversity and biomass data are needed to fully calculate regulating ecosystem services.

5.2.3 Randi

The passive restoration in Randi significantly increased plant cover percentage from the Degraded areas to the same values than the References. Plant cover in the Degraded plots is 46.0% and the Reference and Restored are 79.6 and 83.8%, respectively (Fig. 49). The Restored plots showed the highest average number of vascular plants (12.7) in contrast to the Degraded and the undisturbed reference (9.3 and 10.3, respectively). The two diversity indexes evaluated (Shannon-Wiener’s and evenness) showed similar trends in the three states of the ecosystem (Fig. 50): the Degraded areas presented the lowest values of both indexes and the Restored reached the same values than the Reference. Shannon’s doubled from 1.0 to 2.0 while the evenness moved from 0.43 to 0.77.

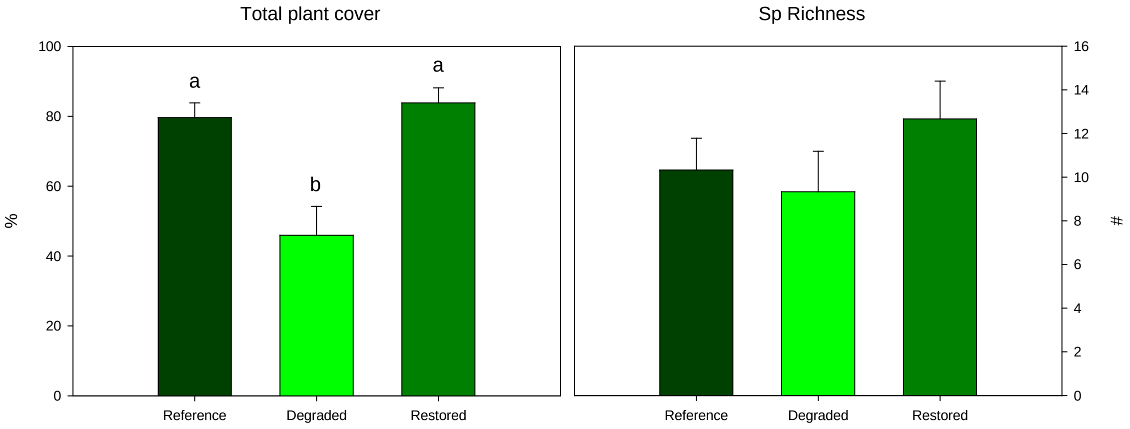

Figure 49. Total plant cover (left) and species richness (right) in the Reference, Degraded and Restored states in Randi field site. Mean and standard errors are shown. Different letters denote significant differences.

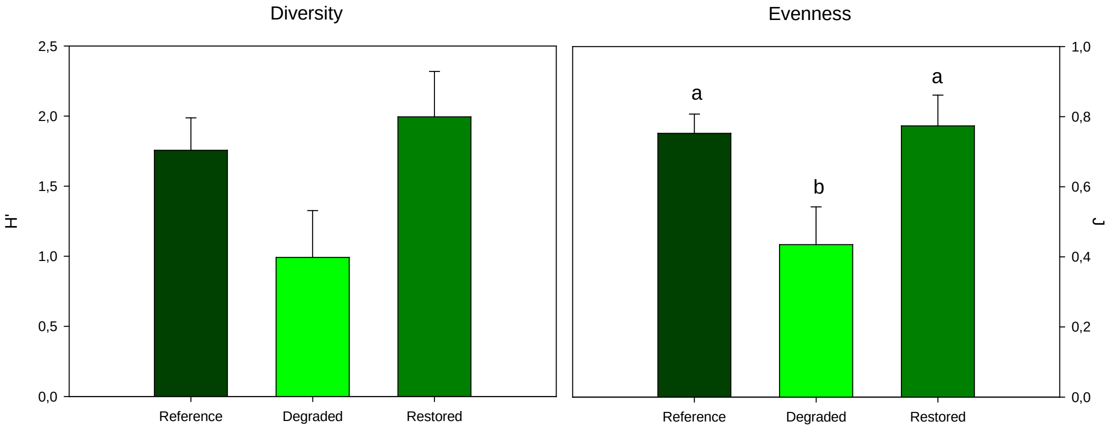

Figure 50. Shannon-Wiener Index of diversity (left) and evenness (right) in the Reference, Degraded and Restored states in Randi field site. Mean and standard errors are shown.Different letters denote significant differences.

Twenty-three species were found in Randi but only six species were present in the three ecosystem states. The References were characterized by high cover of shrubs like *Cistus creticus* (36.7%), *Calycotome spinosa* (15.5%), *Lithodora hispidula* (15.3%) and *Pistacia lentiscus* (12.0%). Other key species that were only present in the undisturbed areas were, in addition to *P. lentiscus*, *Pinus halepensis* and *Rosmarinus officinalis*. The Degraded state showed low plant cover and most of it was due to a species of the family Asteraceae (undetermined) with 12.5%. *Sarcopoterium spinosum* is the most abundant species in the Restored areas (25.1%) with an extended cover of an unidentified grass (24.7%) and, in a lesser extent, *C. spinosa* (11.1%). The graphical representation of the two first axis of the Principal Component Analysis on plant cover data clearly grouped the plots by state (Fig. 51). PC1 and PC2 explained 30.9 and 19.2% of the variance and, hence, jointly explained more than half of it. The extraction of species along the first two components of the PCA is shown in Annex I. Reference plots showed the lowest values of the PC1 and separated these plots from the other two groups. Degraded and Restored plots separated along the second axis, with higher and more heterogeneous values of the restored plots.

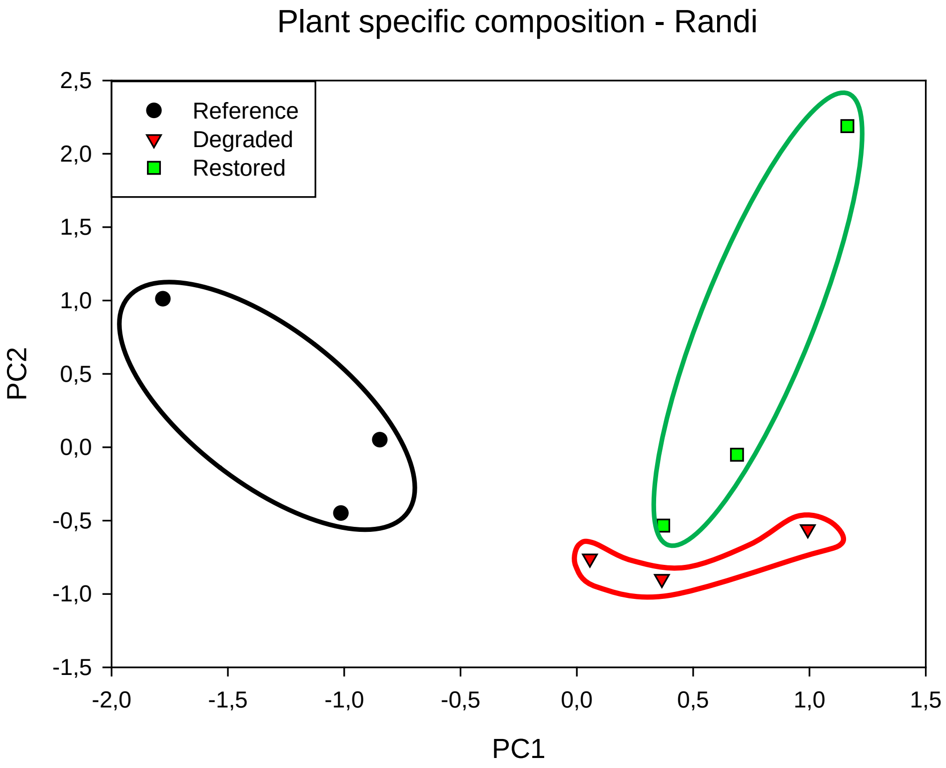

Figure 51. Distribution of Reference, Degraded and Restored plots in Randi field site according to the two first axis of PCA conducted on plant cover.

Biomass accumulation in the different components of the ecosystem of the restored areas is more similar to the references than to the degraded lands (Figs. 52 and 53). Degraded areas showed relatively high amounts of herbaceous biomass and low litter and biomass of woody species. The Restored plots significantly increased the woody biomass (from 3.4 to 9.5 Mg/ha) and the litter accumulation on the ground (20 times larger in the restored than in the degraded) while the biomass of grasses was slightly reduced by the restoration. None of the three biomass components (woodies, grasses and litter) were significantly different in the Restored than in the Reference plots. Total biomass in the Restored areas was two times higher that of the Degraded ones (Fig. 53).

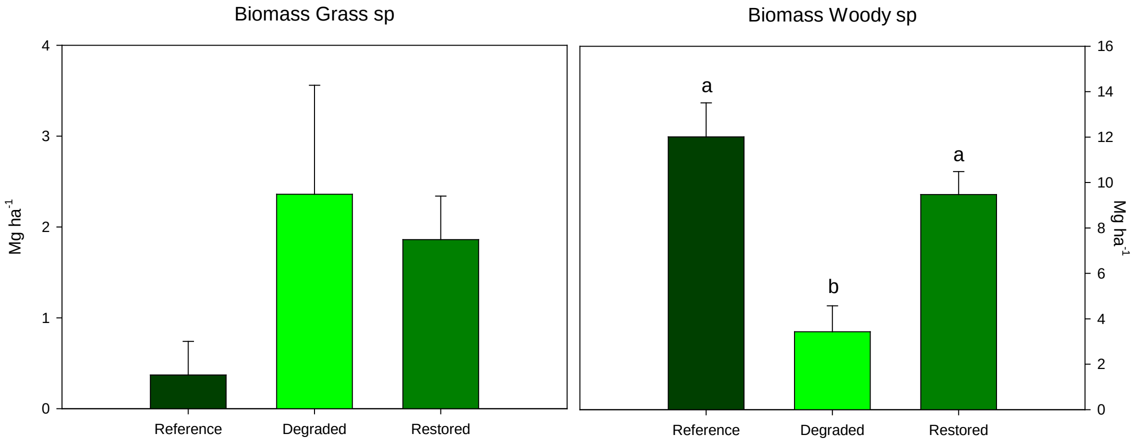

Figure 52. Biomass of herbaceous (left) and woody vegetation in the Reference, Degraded and Restored states in Randi field site. Mean and standard errors are shown. Different letters denote significant differences. Note different scales in the Y-axes.

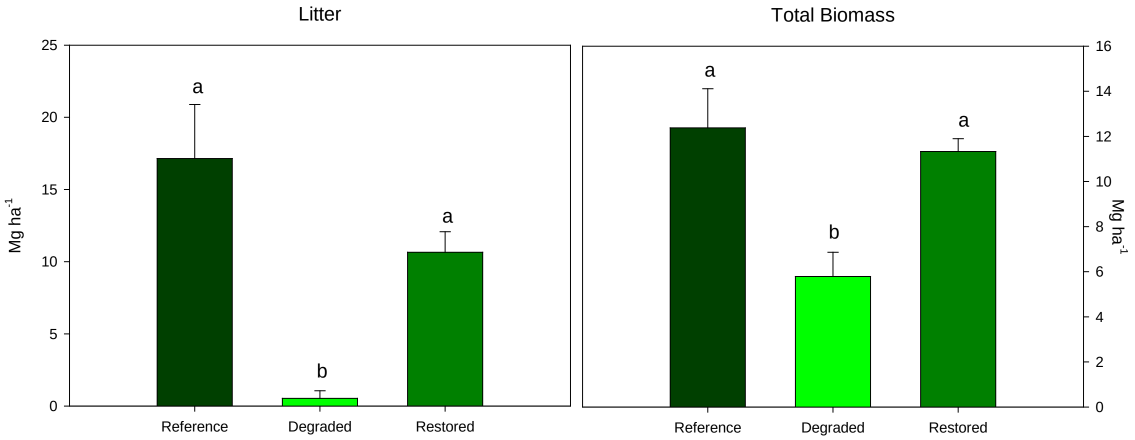

Figure 53. Litter accumulation (left) and total biomass (right) in the Reference, Degraded and Restored states in Randi field site. Mean and standard errors are shown. Different letters denote significant differences.Note different scales in the Y-axes.

Both the spatial distribution and size of patches and interpatches were significantly affected by restoration (Fig. 54). The percentage of land covered by interpatches was reduced from 87.1% in the degraded areas to 61.0% in the restored, much closer to the 51.4% observed in the references. Also the length of the interpatches was lower in the restored than in the Degraded (1.5 and 3.9 m, respectively). Conversely, the size of the patches in the restored plots were among the size of the reference and the degraded areas. Average patch size in the degraded plots were 0.53 m long and 0.81 m wide while in the restored plots they averaged 1.01 m long and 1.98 m wide.

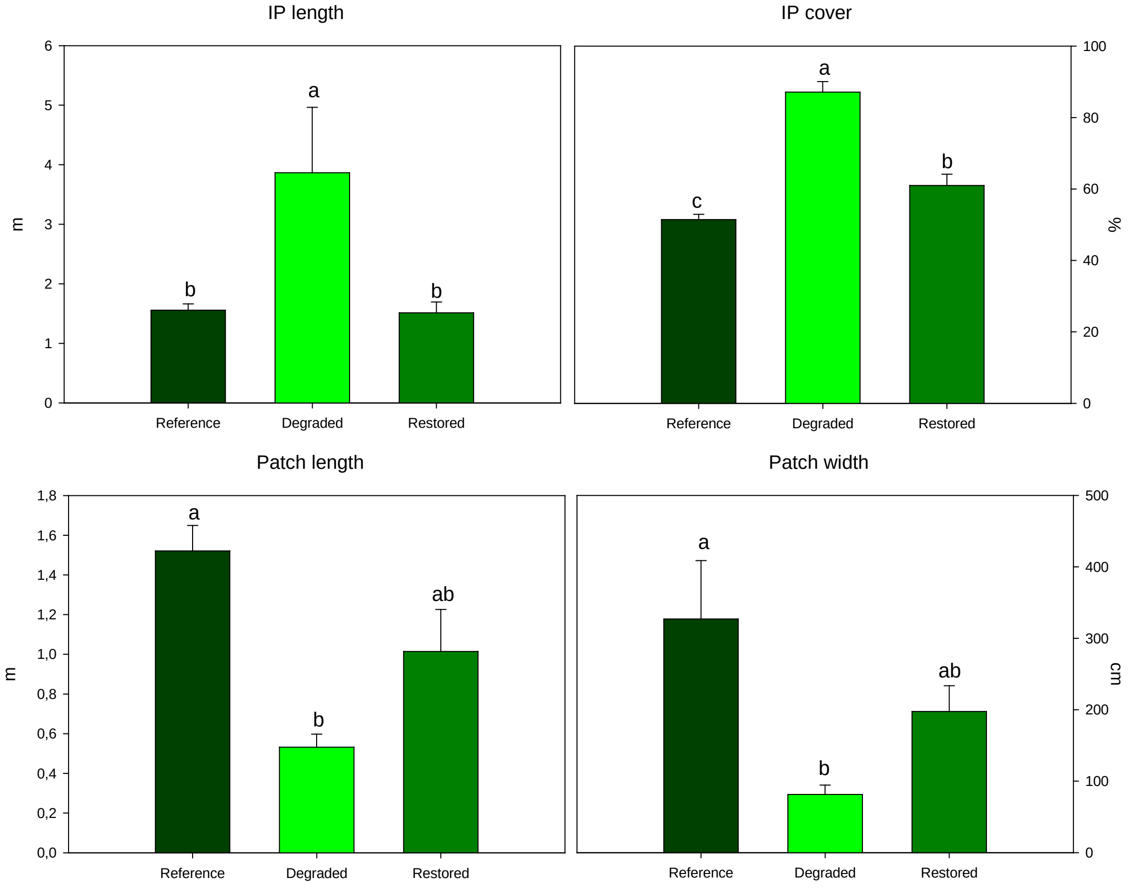

Figure 54. Values of Interpatch length (top left), cover (top right), patch length (bottom left) and width (bottom right) in the Reference, Degraded and Restored states in Randi field site. Mean and standard errors are shown.Different letters show significant differences.

The largest changes in LFA derived indexes were observed in nutrient cycling and infiltration (Fig. 55). These two indexes were significantly improved by restoration, from 9.9 to 29.8% the former and from 20.8 to 34.0% the latter. However, reference plots still showed higher values of the three indexes, especially infiltration and nutrient cycling. Changes in stability were minor and not significant.

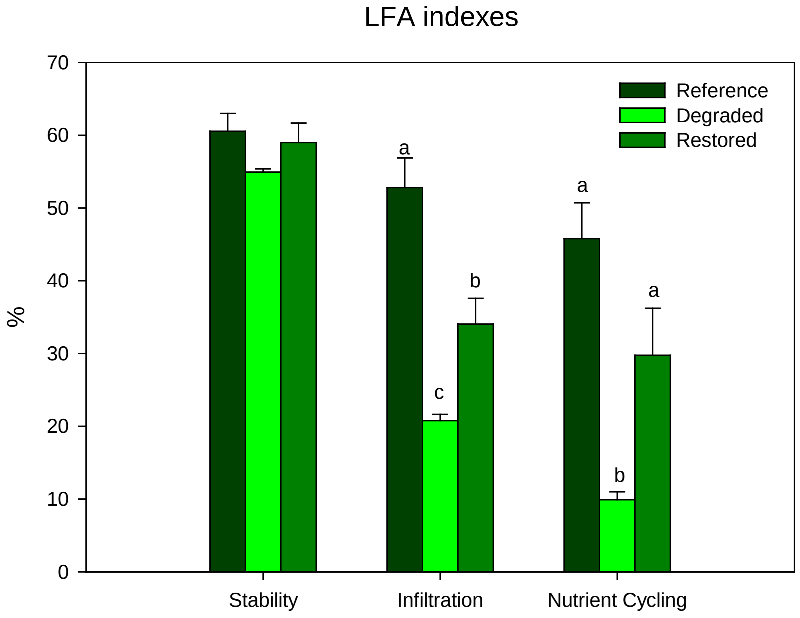

Figure 55. Values of the Stability, Infiltration and Nutrient Cycling indexes derived from LFA in the Reference, Degraded and Restored plots in Randi field site. Mean and standard errors are shown.Different letters show significant differences.

All calculated ecosystem services but biodiversity were improved in the restored plots in relation to the degraded ones as well as the averaged combination of the five services assessed (Fig. 56). Biodiversity was the service that released he highest increase (1.47) but the heterogeneity of the variables included in this service in the different replicates of each ecosystem state prevented significant differences. Nutrient cycling (1.45) and water conservation (1.40) also showed large increase after restoration. All individual services and also their combination showed no difference between the restored and the reference areas suggesting a high effectiveness of the restoration measures (grazing exclusion) in improving ecosystem services.

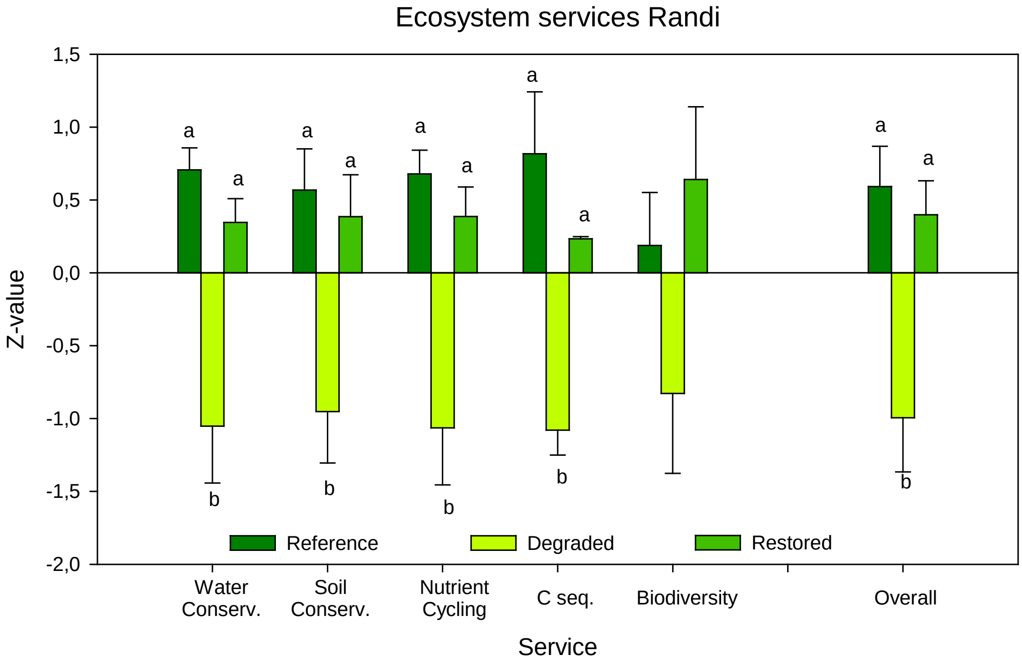

Figure 56. Standardized values of the list of ecosystem services in Randi, as derived from combinations of the different variables acquired. Mean and standard errors are shown.

Figure 57 summarizes the relative changes of the measured and calculated ecosystem properties in the restored in relation to the degraded state of Randi ecosystems. The largest increase in litter accumulation and, in a lesser extent, the build up of woody biomass and patch size led to a significant improvement of nutrient cycling and infiltration indexes.The three properties that showed negative values (herbaceous biomass, and interpatch cover and length) can also be considered as positive symptoms to the recovery of a healthy ecosystem closer to the reference target state.

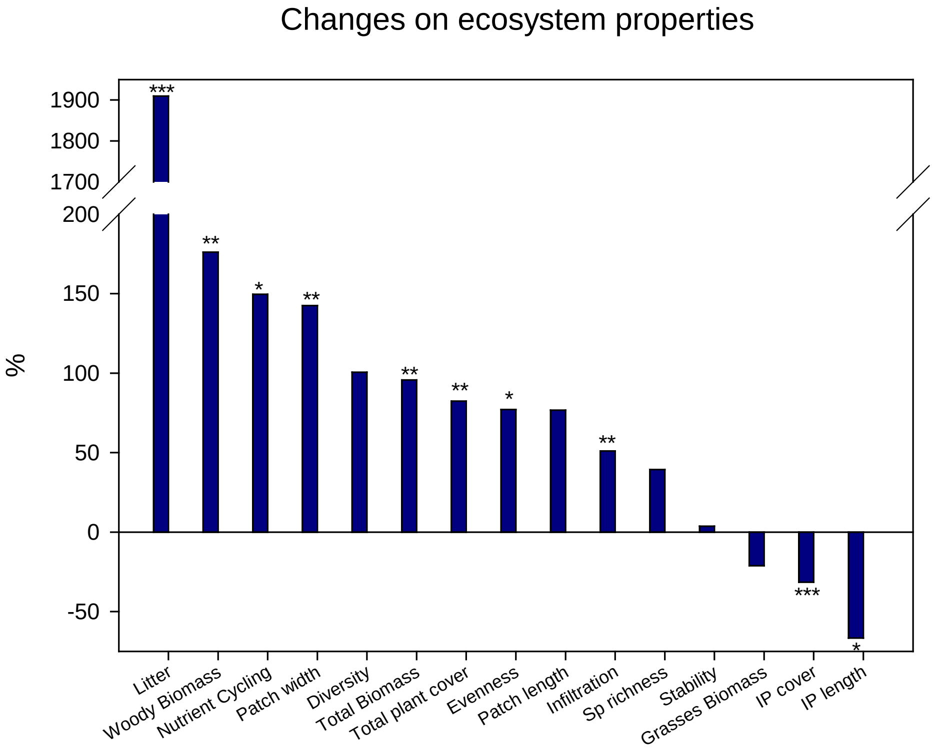

Figure 57. Losses or gains (negative and positive values, respectively) of assessed ecosystem properties in the Restored areas of the Randi field site in relation to the Degraded. Asterisks denote significant differences between ecosystem states (*: 0.10<p<0.05; **: 0.05<p<0.01; ***: p<0.01).

Results highlights - Randi

- Restoration by long-term grazing exclusion increased plant cover, litter accumulation and aboveground biomass to similar levels found in the undisturbed reference areas
- Plant composition and spatial structure of vegetation (cover and size of patches and interpatches) also reflected differences in the three ecosystem states
- Ecosystem functioning, mainly nutrient cycling and infiltration, is sharply improved in the restored areas but are still far to the values observed in the references
- The five ecosystem services calculated did not show differences between the Restored and the Reference areas and were significantly improved form the Degraded lands
- Restoration in Randi can be considered as successful with the approach followed in the project

5.3 Multifactor Driven Landscapes

5.3.1 Albatera

The two restoration approaches did not significantly improved total plant cover in relation to the degraded state (Fig. 58 left). In fact, the old restoration showed the lowest values of plant cover (35.5%), significantly lower than the new restoration (43.6%) and the references (55.6%). Considering the four ecosystem states, we found 32 vascular species in Albatera, being the richest states both the reference and the new restoration plots (17.3 species) and the poorest the old restoration with only 9.7 species present (Fig. 58 right). Four species were present in the four situations (*Fumana thymifolia*, *Fagonia cretica*, and the grasses *Brachypodium retusum* and *Stipa tenacissima*) while other four shrubs were only found in the reference plots (*Whitania frutescens*, *Ephedra sp*, *Pistacia lentiscus* and *Cistus clusii*). Six species were exclusive of the new restored sites, some of them, like *Olea europaea*, *Lygeum spartum* and *Juniperus oxycedrus*, were introduced during the restoration activities.

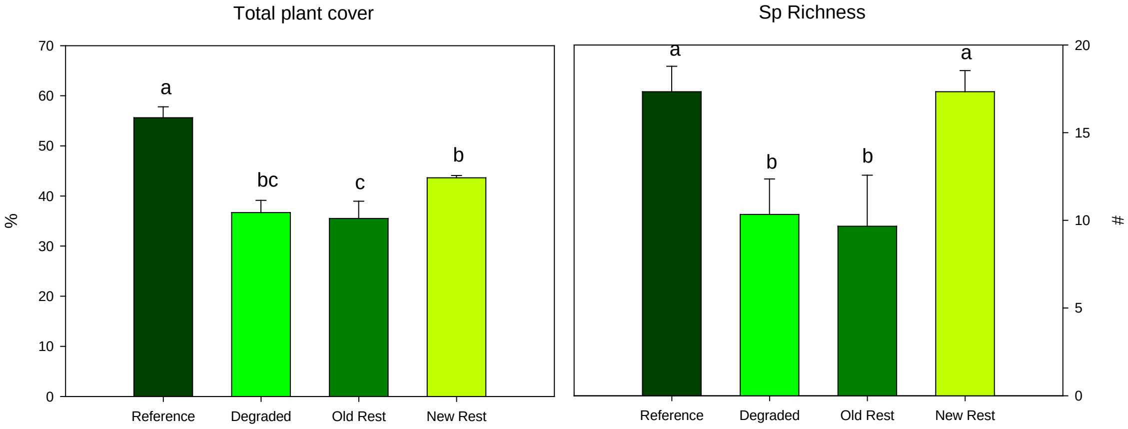

Figure 58. Total plant cover (left) and species richness (right) in the Reference, Degraded and the two Restored states (Old and New Restoration) in Albatera field site. Mean and standard errors are shown. Different letters denote significant differences.

In addition to species richness, diversity and evenness indexes were significantly reduced in the degraded areas (0.92 and 0.40, respectively) as compared to the references (1.89 and 0.66, respectively). We observed a slight trend in the old restoration to increase these indexes while the improvements in the new restoration were statistically significant but did not reach the reference values (1.48 and 0.52, respectively; Fig. 59).

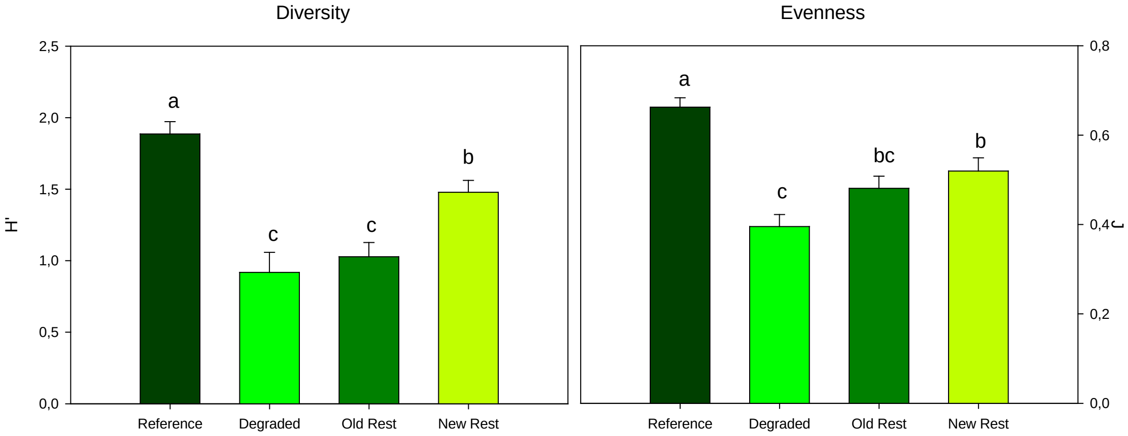

Figure 59. Shannon-Wiener Index of diversity (left) and evenness (right) in the Reference, Degraded and the two Restored states (Old and New Restoration) in Albatera field site.Different letters denote significant differences.

The reference ecosystem is characterized by *Artemisia barrelieri* and *Fagonia cretica*, with 17.9 and 13.1%, respectively, and the species with highest cover in the degraded areas is *Fumana thymifolia* (24.3%), with no other species with cover values beyond 6%.*Globularia alypum* is the most abundant species in the two restored sites (18.8 and 15.5% in the old and new restoration, respectively). The two first axes of the PCA carried out on specific plant cover explained 44.9% of the total variance and grouped all the plots by the defined state of the ecosystem (Fig. 60). Along PC1 references separated from the other three groups of plots while the second axis discriminated the new restoration (highest values), the old restoration (intermediate) and the degraded plots (the lowest) (see Annex I for the species extraction on these two components).

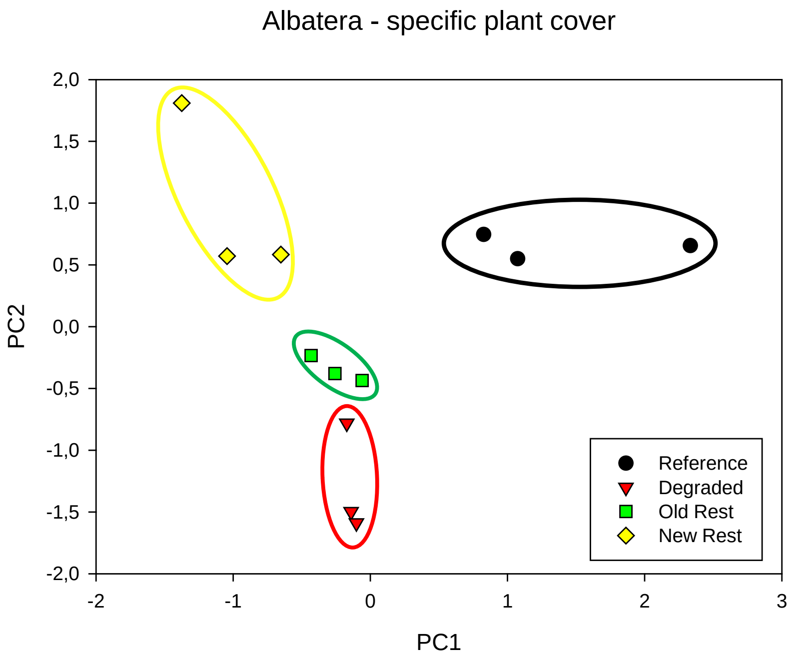

Figure 60. Distribution of Reference, Degraded, Old Restored and New Restored plots in Albatera field site according to the two first axis of PCA conducted on plant cover.

Plant volume (a proxy of plant biomass) in the old restoration was quite similar to that in the reference plots and more than 2.2 times higher than in the degraded areas (Fig. 61). The new restoration showed intermediate values of plant volume (1259 m^3/ha).

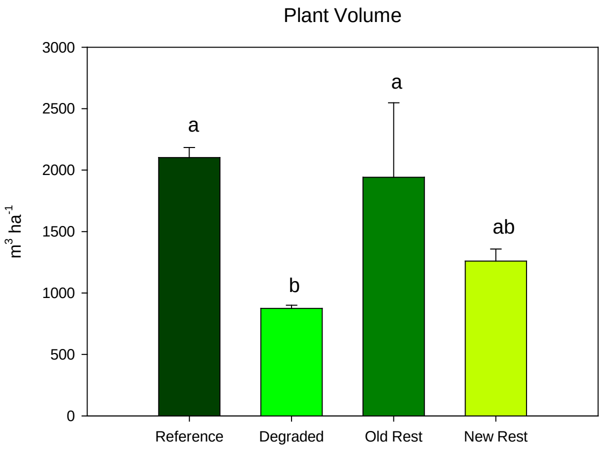

Figure 61. Plant volume (m^3 ha^{-1}) in the Reference,Degraded, Old Restored and New Restored plots in Albatera field site. Mean, standard errors and significance are shown. Different letters denote significant differences.

We noticed significant changes in the spatial arrangement and size of vegetation associated to degradation and the alternative restoration approaches (Fig. 62). Interpatches are significantly shorter in the reference than in the other three states of the ecosystem but we observed a trend to decrease the length (20% shorter than in the degraded) in the new restoration sites. Interpatch cover was also enhanced due to degradation (from 66.9 to 91.8%) but both restorations significantly decreased the percentage of land corresponding to sink areas (82.7 and 84.5% the old and new restoration, respectively). The morphology of patches also changed with degradation and restoration. Patches in the reference averaged 0.50 m long and 0.64 m wide while in the degraded averaged 0.16 m long and 0.19 m wide. Old restoration produced patches significantly longer and wider than in the degraded areas and not different than the references (0.39 m long and 0.82 m wide), and the new restoration did not show significant changes of patch size than in the degraded plots.

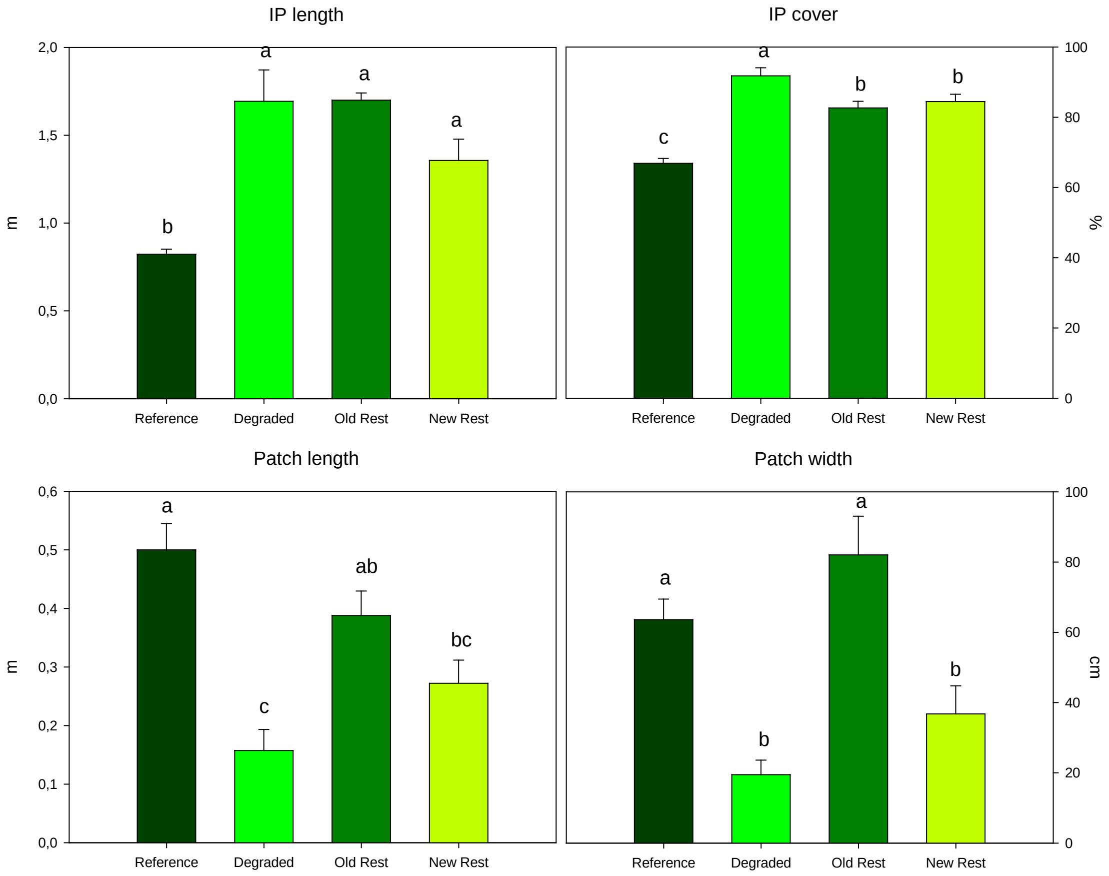

Figure 62. Values of Interpatch length (top left), cover (top right), patch length (bottom left) and width (bottom right) in the Reference,Degraded, Old Restored and New Restored plots in Albatera field site. Mean and standard errors are shown.Different letters show significant differences.

The largest changes in LFA derived indexes were observed in nutrient cycling and infiltration (Fig. 63). However differences were only observed between the reference systems and the other three states of the ecosystem, with no significant change among them. We only noticed a trend to increase nutrient cycling and infiltration in the old restoration while the effects of the new restoration were even lighter. Changes in stability were minor and not significant.

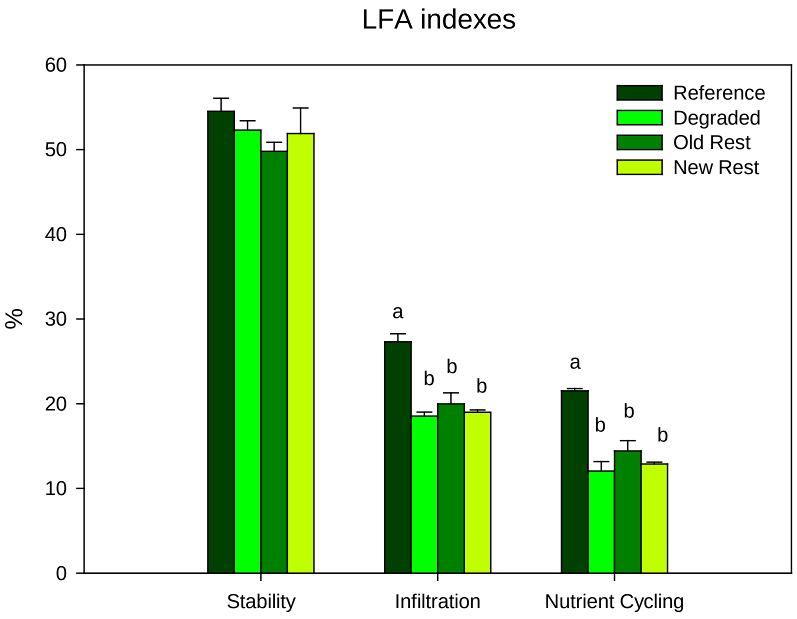

Figure 63. Values of the Stability, Infiltration and Nutrient Cycling indexes derived from LFA in the Reference,Degraded, Old Restored and New Restored plots in Albatera field site. Mean and standard errors are shown.Different letters show significant differences.

Long-term degradation had a severe impact on the services provided by the target ecosystem in Albatera (Fig. 64). The reference state showed the highest values of all services and, consequently, also of the final averaged value. Only one service, biodiversity, was significantly improved by the new restoration approach but these areas also showed signs of improving soil and water conservation and nutrient cycling. The old restoration only tended to improve carbon sequestration. From the two restoration approaches, the new restoration resulted in better balance of ecosystem services than the old restoration. However, the two restored systems are still far from the services provided by the reference systems.

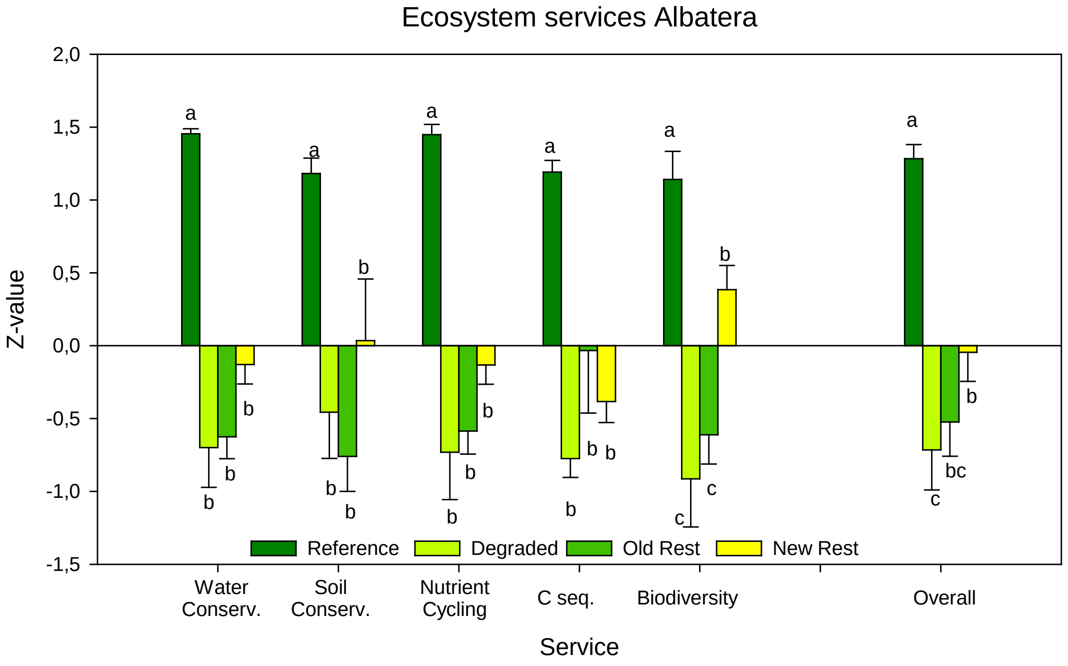

Figure 64. Standardized values of the list of ecosystem services in Albatera, as derived from combinations of the different variables acquired. Mean and standard errors are shown.Different letters show significant differences.

The two restoration approaches studied improved all the ecosystem variables considered in this assessment (negative values of interpatch cover and length are considered better conditions; Fig. 65). The contrasted restoration options in Albatera affected different ecosystem variables, with old restoration affecting patch morphology and interpatch cover while new restoration positively impacted diversity indexes and plant cover and volume.

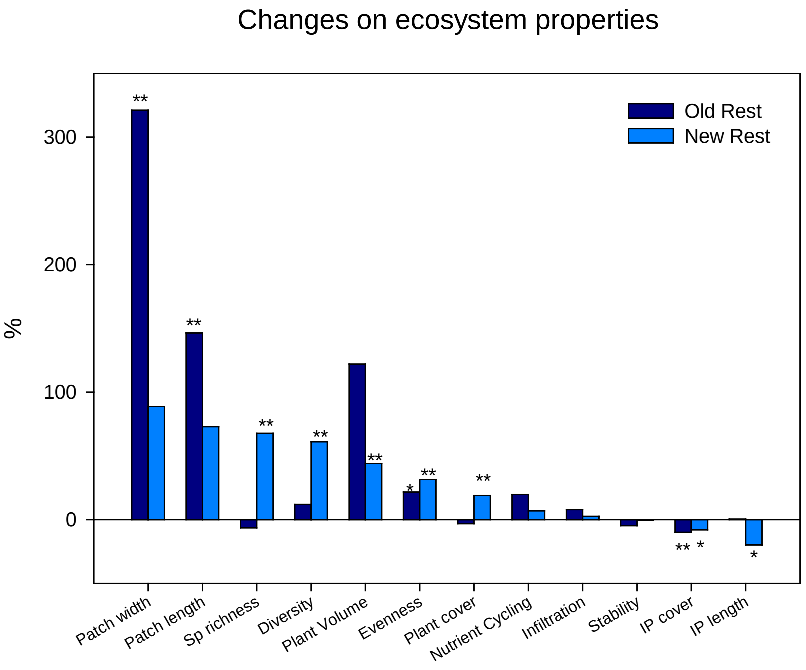

Figure 65. Losses or gains (negative and positive values, respectively) of assessed ecosystem properties in the Restored areas of the Randi field site in relation to the Degraded. Asterisks denote significant differences between ecosystem states (*: 0.10<p<0.05; **: 0.05<p<0.01).

Results highlights - Albatera

- Old restoration especially affected the contributions of sink and source areas to the landscape and their morphology
- New restoration especially affected biodiversity and vegetation structure and biomass
- The extremely harsh conditions in Albatera determine low recovery rates of ecosystem structure and function after restoration
- New restoration improved ecosystem services in higher extent than old restoration in Albatera
- At the medium and long term after restoration, ecosystem services are still far from those provided by natural undisturbed ecosystems

6 GENERAL DISCUSSION

The large heterogeneity of target ecosystems, properties, constraints and conditions of the six field sites resulted in a very wide range of values of the variables evaluated (Table 6). For instance, plant cover percentage ran from less than 40% in the degraded state of Albatera to above 95% in the fenced sites of

Castelsaraceno. Similar disparities of data are found in diversity, productivity, patch size and distribution, and functionality indexes.

We cannot do a generalization of either the impacts or the restoration potential of the two sites of the projects subjected to fire. Although Várzea and Ayora shared the reference ecosystem (pine forest), the restoration approaches and timing of application after fire are not comparable. In Várzea, traditional (salvage) logging resulted in better results than conservation logging in the short term, but extremely far from the reference forest. Natural pine forests show marked differences even with successfully restored forests. For instance, it has been shown that diversity and natural recruitment is significantly lower in restored pine forests than in undisturbed ones, especially under medium to low values of annual rainfall (Ruiz-Benito et al., 2012). These restored habitats also need additional restoration management such as thinning high density stands and increasing diversity through planting. In our case, the assessment of ecosystem properties and services has been carried out two years after the logging treatments were applied. This time frame is very short to detect any significant improvement. Both logging treatments showed reductions in biomass and plant cover in relation to the degraded areas (four times burned) probably related to the use of heavy machinery during the logging procedures. However, the stability, infiltration and nutrient indexes were improved in both restored sites in relation to the degraded.

Traditional logging in Várzea is a common practice in burned forests that, in Mediterranean ecosystems, has been justified to reduce further reforestation costs. However, Leverkus et al. (2012) observed that from an economic point of view a treatment similar to the conservation logging carried out in Várzea may release higher reforestation success than traditional logging with lower costs. Leverkus et al. (2014) reported lower plant species number, diversity and cover at the short-term (two years after treatment establishment) in post-fire salvage logged areas than in unmanaged burned sites or areas where wood debris were left on the ground in the SE of the Iberian Peninsula). Both logging treatments produce a homogenization of the landscape, higher in the traditional logging sites, while degraded areas without any post-fire intervention present higher heterogeneity in microclimatic conditions caused by burned plants that affect heterogeneity of resource distribution (Castro et al., 2011). In addition, salvage logging might decrease the vigor and growth of regenerating pine seedlings due to an increase of water stress, and a reduction of nutrient availability and microclimatic heterogeneity associated to standing dead wood (Moya et al., 2015), as well as increase the susceptibility of alien species to spread within the burned and salvage logged area (Moreira et al., 2013). The naturally regenerated pine seedlings in areas subjected to salvage logging usually show at the medium term lower ecophysiological performance, growth and cone production than those where more conservative logging practices were conducted (Marañón-Jiménez et al., 2013). The extraction of burned wood soon after fire may result in longer-term reductions of C sequestration than if wood had remained to decomposed in situ (Johnson et al., 2005). However, the net effect of salvage logging depends on the serotinity level of the stand (de las Heras et al. 2012). In general, post-fire emergency rehabilitation actions should be applied only to burned pine forests showing high erosion and runoff risk, with slow natural plant recovery rate Vallejo et al. (2012). These observations together with the data we recorded suggest that management activities soon after fire in Várzea may release negative net effects. On the other hand, the creation of piles of at least 50 cm height with the remains of the wood (branches and non-profitable logs), as in the conservation logging carried out in Várzea, enhances the abundance of seed dispersal bird species, especially in winter, and also richness breeding bird species, rodents, and mammals (Rost et al., 2010). Bautista et al. (2004) made some technical recommendations about the management of burned wood after fire. They included the avoidance of salvage logging in vulnerable soils until a protective vegetation cover develops, to keep some individuals as perches for birds nesting and seed dispersal, to conduct logging in patches promoting spatial heterogeneity, and to leave branches, trunks or chipped material on the ground to protect against erosion.

The unexpected better results of the traditional than the conservation logging in many ecosystem properties and services are due to the increase in patch size and cover in the former. This may lead to misunderstanding as patches in the two logging sites are not completely vegetated while interpatches are not exclusively bare soil but brush chip remains. More time is needed to assess whether the traditional and conservation logging treatments affect differently to the recovery of ecosystem properties in Várzea.

At the short term after the fire, passive restoration, e.g. by assisting natural germination or resprouting, is rather preferred than active restoration, e.g. by planting seedlings, due to the high costs and unpredictable results of the latter (Vallejo et al. 2012).

In Ayora, restoration was conducted 23 years after the fire, when a mature shrubland was established and with the main objective of reducing fire hazard and improve vegetation resilience. The assessment was done eleven years after the application of selective clearing and plantation of resprouter seedlings. In this case, restoration at the medium term had positive impacts on most ecosystem properties and services, especially on biodiversity. Both the direct introduction of species that had locally disappeared and the increase of landscape heterogeneity by selective clearing might have promoted the significant improvement of biodiversity indexes in the restored plots. It has been observed that a thick and continuous understory layer reduces plant diversity (Royo and Carson, 2006). All other ecosystem services also improved except C sequestration as the restoration treatments included the removal of seeder fire-prone vegetation and hence the aboveground biomass. However, this fact fulfilled one of the objectives pursued by restoration as it is the reduction of the fire risk. The degraded shrubland presented two times higher amount of standing dead biomass than the restored plots (data not shown). Reduction of fire hazard has been recognized as a regulating ecosystem service (Bagdon et al., 2016). The approach done to Fire Risk Reduction revealed a significant increase of this service in the Restored areas as compared to the Degraded and even to the Reference state of the ecosystem. The fuel model of the degraded community changed to less flammable types in the restored areas, probably from model 4 to 5. This is especially interesting as the reduction of fire hazard, together with increasing the resilience of the plant community, was the main objective of the restoration carried out. We have confirmed that fire risk was still reduced ten years after the application of the vegetation management treatments.

In Ayora, shrublands are quite effective in protecting the soil, show high ecosystem attributes and, when resprouters are abundant, show high resilience (Vallejo et al., 2006). However and due to different reasons mainly related to stakeholders perception, restoring the forest that has been lost might be desirable.

In relation to grazing, Castelsaraceno (overgrazed) and Randi included similar restoration approaches based on grazing exclusion. In the two sites, general improvement of ecosystem properties and services were observed, especially related to enhancing biodiversity. In fact, grazing exclusion 20 years ago in Randi showed the greatest improvement in all the evaluated variables of all the field sites and ecosystem states (Table 6, Fig. 66). Fenced in Castelsaraceno also improved all variables except interpatch length and litter accumulation. Changes in land use, like grazing exclusion as restoration measure, produce a trade-off between provisioning and regulating ecosystem services (Foley et al., 2005). Rong et al. (2014) reported an increase in vegetation cover, height and biomass both of the grass and shrub layers as well as in soil surface properties eight years after grazing was excluded from an arid continental region in China. However, these authors did not find significant differences in any diversity index between the degraded and the restored sites. But the effects of grazing exclusion in improving ecosystem properties are not immediate and straightforward. Li et al. (2012) observed the maximum effect of this practice on plant cover, diversity, biomass, and soil physical and chemical properties in areas with 13 and 26 years of enclosure. Passive restoration actions, such as fencing overgrazed areas or clearing shrub encroached sites, probably do not pursue a well-defined target ecosystem but alternative meta-stable states (Cortina et al., 2006).

On the other hand, and in areas where shrub encroachment is relevant like in Castelsaraceno undergrazed areas, the removal of woody vegetation by clearing may release both positive and negative effects on C sequestration in the soil depending on the precipitation regime of the site. Thus, Alberti et al. (2011) proposed that below 900 mm yr^{-1} of rainfall, soil C increases with clearing woody vegetation while above this threshold (corresponding to Castelsaraceno field site) the net effect of clearing on soil C sequestration is negative. Although we did not evaluate soil C, cleared areas in Castelsaraceno showed higher root biomass but less litter accumulation than undergrazed areas.

Significant changes in the composition of plant communities have been found according to the grazing pressure (overgrazed-reference-fenced and also undergrazed-reference-cleared in Castelsaraceno). The reduction of the relative abundance of unpalatable species in degraded areas or its replacement by other more palatable at the medium term after grazing exclusion has been previously reported in other Mediterranean drylands (Jeddi and Chaieb, 2010) as well as the modification of the relative proportion of different life-forms (Medina-Roldán et al., 2012). These changes in plant community composition are less pronounced in the most arid areas and increase both with precipitation and net primary productivity (Milchunas and Lauenroth, 1993). But not only composition is sensitive to grazing exclusion. Several studies reported an increase in diversity indexes such as those we have evaluated in the project (species richness, Shannon-Wiener's and evenness indexes) after ca 10 years of excluding grazing (Jeddi and Chaieb, 2010; Wang et al., 2016). However, it has also been observed that species composition does not significantly change at moderate levels of grazing at the time that both regulating and provisioning ecosystem services are optimized (Oñatibia et al., 2015). These authors and others (e.g. Medina-Roldán et al. 2012) observed a reduction of C and N stocks in heavily grazed arid rangelands as compared to moderate grazed areas, and recommend a reduction of grazing pressure for increasing C sequestration rather that complete grazing exclusion. Probably, the definition of optimum intermediate stocking rates instead of complete grazing exclusion is a main objective for the management of these areas where grazing represents an important ecosystem service (Papanastasis et al., 2015).

The case of Messara is rather different than the other sites affected by grazing. The restoration did not aim to recover the pre-disturbance or reference state of the ecosystem but a transformation of land use from grazing to carob tree orchards as a silvopastoral or agroforestry system. Carob trees are a landmark of Greek landscapes as it is one of the greatest producers of carob pods (5.600 ha and 22.000 tons of pods in 2013; data from FAO), most of the production is concentrated in Crete. The reference, semi-degraded and degraded states in Messara represent different situations along the degradation trajectory while the restored options built alternative system through replacement following Bradshaw's classical structure-function model (Bradshaw 1984). Under this situation, the assessment based on the spatial arrangement of vegetation, the contribution of patches and interpatches to the landscape, and the evaluation of soil surface properties provides useful insights of ecosystem properties but does not represent a reliable approach of the restoration potential of ecosystem services of these degraded sites. The incorporation of plant cover and plant biomass will surely result in significant improvements of the ecosystem services included in this report. The possibility of getting external funds from the EU Common Agricultural Policy for this agroforestry transformation, as happened in Melidochori, is another aspect to be considered in the final balance of the impacts of this land management alternative. In addition, provisioning services such as fodder or gum products (Papanastasis

1989) can be provided by this transformation from overgrazed areas to carob tree orchards when physical features of the site, especially soil depth, are appropriate.

In the field site with the highest aridity index, Albatera, there was an important improvement of ecosystem services and properties due to the development of new restoration technologies, such as higher number of planted species, species selection based on geomorphological features, compost application or water harvesting structures (Chirino et al., 2009). However, despite both the new and the traditional restoration improved the state of the degraded ecosystem, its properties are still far from the values of the undisturbed reference sites. One of the reasons underlying the better performance of the planted seedlings in the new than in the old restoration approach is that it included the optimization of hydrological properties that have significant effect on restoration success (Urgeghe and Bautista, 2015). It is important to highlight that the new restoration action in Albatera was applied only ten years before the assessment, and even so it already yielded much better results than the traditional approach implemented several decades ago. We expect that the positive effects of this management option will increase over time as ecological processes act at slow rate in these extremely stressed sites (Pugnaire et al., 2006).

Our analytical approach to evaluate the potential to restore areas impacted by fire, grazing or multiple simultaneous stresses has provided useful insights on constraints and opportunities for restoration that may be considered when designing landscape management options. Our assessment is based on biophysical features in the different states of the ecosystem and special weight relies on Landscape Function Analysis. Other services that has not been quantified in this report such as the reduction of fire risk in Várzea or the provisioning services especially in Castelsaraceno could also be considered to better capture the net outcome of restoration actions.

Stakeholders perception about ecosystem services and properties should be incorporated in the decision making (Bullock et al. 2011).

Table 6. Direct comparison of ecosystem properties between the Degraded and the Restored (the best one in case of two alternatives) states of the ecosystem in the six CASCADE field sites.

	FIRE-DRIVEN LANDSCAPES				GRAZING-DRIVEN LANDSCAPES								MULTIFACTOR-DRIVEN LANDS.	
	VARZEA		AYORA		CASTELSARACENO				MESSARA		RANDI		ALBATERA	
	Deg	Trad Log	Deg	Rest	Over	Fenced	Under	Cleared	Deg	Odigitri	Deg	Rest	Deg	New Re
Plant Cover (%)	**86.6**	79.6	78.8	**88.2**	86.1	**95.9**	92.2	**93.2**	62.5	n.d.	46.0	**83.8**	36.7	**43.6**
Sp richness (#)	4	**6**	17	**24**	29	**39**	**36**	35	13	n.d.	9	**13**	10	**17**
Diversity (H)	1.04	**1.09**	2.17	**3.03**	3.04	**4.08**	3.45	**4.37**	1.56	n.d.	0.99	**1.99**	0.92	**1.48**
Evenness (J)	**0.78**	0.59	0.77	**0.96**	0.90	**1.10**	0.97	**1.22**	0.60	n.d.	0.43	**0.77**	0.40	**0.52**
Aboveground Biomass (Mg ha^{-1})	3.5	**3.8**	18.9	9.8	5.5	**7.5**	**19.6**	9.8	30.6	n.d.	5.8	**11.3**	0.9[a]	**1.3**
Litter (Mg ha^{-1})	4.5	**11.6**	25.2	14.6	**6.3**	3.1	**12.2**	5.3	0.1	n.d.	0.5	**10.6**	n.d.	n.d.
Interpatch cover (%)	38.0	**8.6**	33.1	**33.7**	**58.4**	60.5	**39.2**	50.6	52.1	**41.7**	87.1	**59.7**	91.8	**84.5**
Interpatch length (m)	**0.6**	**0.6**	0.9	**0.7**	**0.4**	1.1	0.7	**0.6**	**0.7**	1.1	3.9	**1.3**	1.7	**1.4**
Patch length (m)	1.1	**5.5**	1.84	**1.31**	0.4	**1.0**	**1.0**	0.6	0.8	**1.6**	0.5	**0.9**	0.2	**0.3**
Patch width (m)	1.6	**8.5**	2.9	**4.6**	0.5	**0.8**	**1.1**	0.5	0.7	**1.0**	0.8	**2.0**	0.2	**0.4**
Stability index (%)	65.5	**71.4**	**71.0**	62.8	53.9	**54.0**	50.1	**52.5**	55.7	**54.4**	54.9	**57.0**	**52.3**	51.9
Infiltration index (%)	35.3	**44.1**	**55.1**	40.3	21.5	**22.1**	21.8	**23.8**	24.9	**25.8**	20.8	**31.4**	18.5	**19.0**
Nutrient Cycling index (%)	25.6	**37.1**	**53.7**	35.9	17.5	**20.4**	19.1	**20.4**	22.2	**26.6**	9.9	**24.7**	12.0	**12.9**

[a]: Plant volume (m^{3}). n.d.: not determined. Highlighted in bold red the best values.

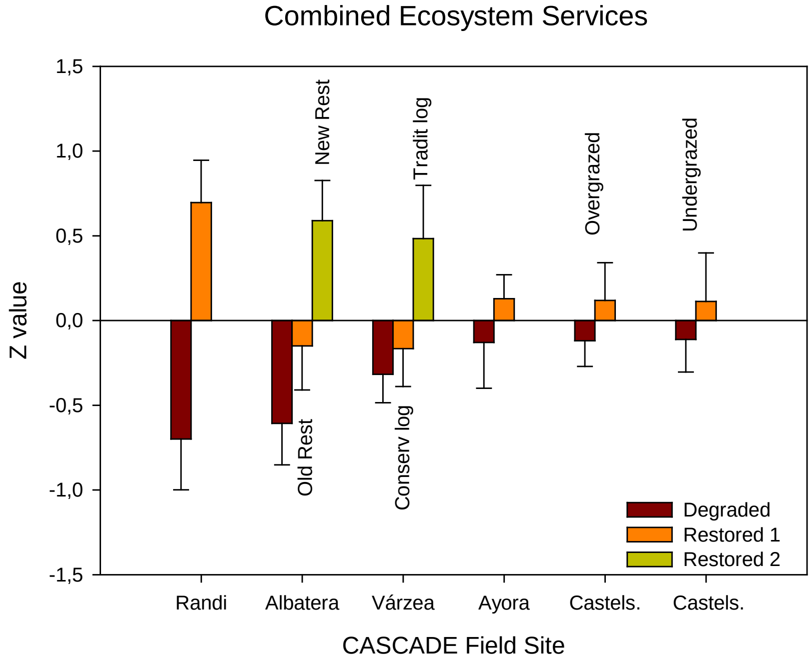

Figure 66. Summary of the changes of standardized ecosystem services due to restoration actions in all CASCADE field sites . Bars represent an average of all five environmental services evaluated.

The study sites represent different degradation drivers, different intensities and duration of pressures, and different climatic, water stress and soil vulnerablity to degradation of representative Mediterranean landscapes. However, the contrast between reference and degraded sites, and between patch and interpatch characteristics constitute an ecologically-sound common indicator of degradation severity. Restoration measures also yielded different outcomes, e.g. different degree of change in ecosystem properties and services. Different restoration treatments and evaluation times after application, and the diverse nature of restoration techniques applied are factors that modulated restoration results. Despite this variation, when the degree of ecosystem change achieved by restoration (relative to degraded states) is analyzed as a function of the relative impact of degradation (relative to the reference state), we observed a global positive relationship between them (Fig. 67), so that the more intense the loss of services the higher the effects of restoration on the recovery of those services. However, one of the sites, Albatera, does not follow this pattern. The stressful conditions in Albatera site (the highest aridity index) determine the slow recovery of ecosystem dynamics and properties even in case of successful restoration practices. Furthermore, despite the multiple degradation factors that drove the ecosystem to its degraded state ceased many decades ago, Albatera did not ever show any sign of self-recovery towards healthier conditions, which indicates that the pressures exerted in the past triggered the shift of the system to a particularly severe degraded alternative state that has proven to be rather stable. Overall, our results suggest that the relationship between restoration potential and degradation level matches a non-linear model, being positive until certain threshold in the loss of services, beyond which the benefits of restoration drop sharply. From the management perspective, the implications of these results are of paramount importance for prioritizing restoration efforts and assessing the cost-benefit of restoration as a function of degradation.

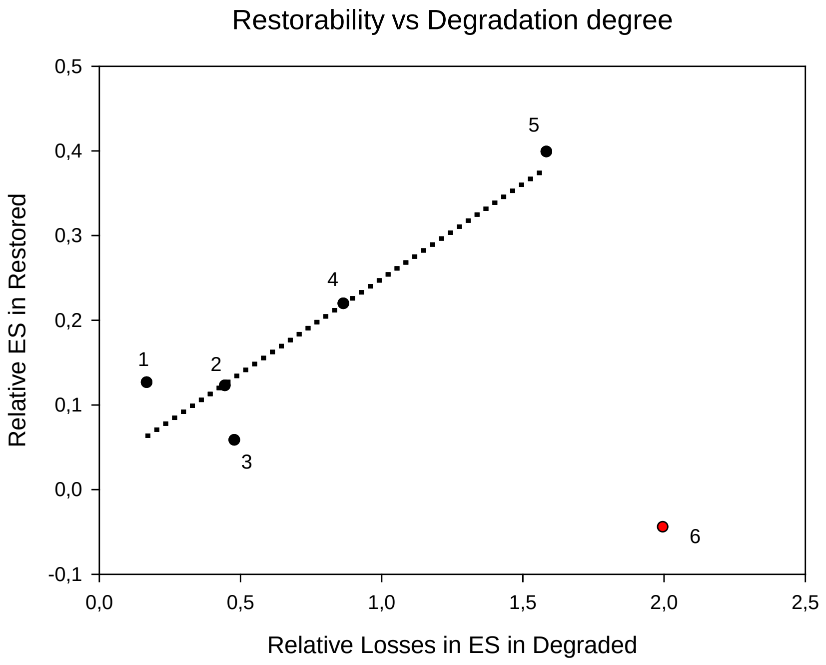

Figure 67. Recovery of ecosystem services by restoration (Z values in Restored plots) in relation to their losses by degradation (Z value Reference – Z value Degraded). The best restoration approach has been selected in the field sites with two alternatives. 1: Castelsaraceno Undergrazed; 2: Ayora; 3: Castelsaraceno Overgrazed; 4: Várzea; 5: Randi; 6: Albatera (red dot).

7 REFERENCES

Alberti, G., Leronni, V., Piazzi, M., Petrella, F., Mairota, P., Peressotti, A., Piussi, P., Valentini, R., Gristina, L., Mantia, T.L., Novara, A., Rühl, J., 2011. Impact of woody encroachment on soil organic carbon and nitrogen in abandoned agricultural lands along a rainfall gradient in Italy. Reg. Environ. Chang. 11, 917–924. doi:10.1007/s10113-011-0229-6

Alkemade, R., Bakkenes, M., Eickhout, B., 2011. Towards a general relationship between climate change and biodiversity: an example for plant species in Europe. Reg. Environ. Chang. 11, 143-150. doi: 10.1007/s10113-010-0161-1

Archer, S.R., Predick, K.I., 2014. An ecosystem services perspective on brush management: Research priorities for competing land-use objectives. J. Ecol. 102, 1394–1407. doi:10.1111/1365-2745.12314

Baeza, M.J., Raventós, J., Escarré, A., Vallejo, V.R., 2003. The effect of shrub clearing on the control of the fire-prone species Ulex parviflorus. For. Ecol. Manage. 186, 47–59. doi:10.1016/S0378-1127(03)00237-8

Baeza, M.J., Vallejo, V.R., 2008. Vegetation recovery after fuel management in Mediterranean shrublands. Appl. Veg. Sci. 11, 151–158. doi:10.3170/2007-7-18339

Bagdon, B.A., Huang, C.-H., Dewhurst, S., 2016. Managing for ecosystem services in northern Arizona ponderosa pine forests using a novel simulation-to-optimization methodology. Ecol. Modell. 324, 11–27. doi:10.1016/j.ecolmodel.2015.12.012

Bautista, S., Gimeno, T., Mayor, A.G., Gallego, D., 2004. El tratamiento de la madera quemada tras los incendios forestales. En: Vallejo, V.R., Alloza, J.A. (eds.) Avances en el estudio de la gestión del monte Mediterráneo, pp. 547-570. Fundación CEAM, Valencia

Bautista, S., Lamb, D., 2013. Ecosystem services. Encyclopedia of Environmental Management. DOI: 10.1081/E-EEM-120046612

Bautista, S., Mayor, A.G., 2010. Ground-based methods and indicators for the assessment of desertification prevention and restoration actions. In: Deliverable D2.1 Working papers on Assessment Methods – PRACTICE Project.

Bradshaw, A.D., 1984. Ecological principles and land reclamation practice. Landscape planning, 11, 35-48.

Bullock, J.M., Aronson, J., Newton, A.C., Pywell, R.F., Rey-Benayas, J.M., 2011. Restoration of ecosystem services and biodiversity: conflicts and opportunities. Trends in Ecology & Evolution, 26, 541-549.

Castro, J., Allen, C.D., Molina-Morales, M., Marañón-Jiménez, S., Sánchez-Miranda, Á., Zamora, R., 2011. Salvage logging versus the use of burnt wood as a nurse object to promote post-fire tree seedling establishment. Restor. Ecol. 19, 537–544. doi:10.1111/j.1526-100X.2009.00619.x

Chirino, E., Vilagrosa, A., Cortina, J., Valdecantos, A., Fuentes, D., Trubat, R., Luis, V.C., Puértolas, J., Bautista, S., Peñuelas, J.L., Vallejo, V.R., 2009. Ecological restoration in degraded drylands: the need to improve the seedling quality and site conditions in the field, in: Forest Management. Nova Science Publ., New York, USA, pp. 85–158.

Cortina, J., Maestre, F.T., Vallejo, R., Baeza, M.J., Valdecantos, A., Pérez-

Devesa, M., 2006. Ecosystem structure, function, and restoration success: Are they related? J. Nat. Conserv. 14, 152–160. doi:10.1016/j.jnc.2006.04.004

Daliakopoulos, I., Tsanis, I., 2013. Historical evolution of dryland ecosystems. CASCADE report series D2.1. 122 pp.

De Groot, R.S., Wilson, M.A., Boumans, R.M.J., 2002. A typology for the classification, description and valuation of ecosystem functions, goods and services. Ecol. Econ. 41, 393–408.

de las Heras, J., Moya, D., Vega, J.A., Daskalakou, E., Vallejo, V.R., Grigoriadis, N., Tsitsoni, T., Baeza, J., Valdecantos, A., Fernández, C., Espelta, J., Fernandes, P., 2012. Post-fire management of serotinous pine forests. In *Post-Fire Management and Restoration of Southern European Forests* (pp. 121-150). Springer Netherlands.

Foley, J. a, Defries, R., Asner, G.P., Barford, C., Bonan, G., Carpenter, S.R., Chapin, F.S., Coe, M.T., Daily, G.C., Gibbs, H.K., Helkowski, J.H., Holloway, T., Howard, E. a, Kucharik, C.J., Monfreda, C., Patz, J. a, Prentice, I.C., Ramankutty, N., Snyder, P.K., 2005. Global consequences of land use. Science 309, 570–4. doi:10.1126/science.1111772

Hilderbrand, R.H., Watts, A.C., Randle, A.M., 2005. The myths of restoration ecology. Ecol. Soc. 10. doi:19

Hooper, D.U., Chapin III, F.S., Ewel, J.J., Hector, A., Inchausti, P., Lavorel, S., Lawton, J.H., Lodge, D.M., Loreau, M., Naeem, S., Schmidt, B., Setala, H., Symstad, A.J., Vandermeer, J., Wardle, D.A., 2005. Effects of biodiversity on ecosystem functioning: a consensus of current knowledge. Ecol. Monogr. 75, 3–35. doi:10.1890/04-0922

Jeddi, K., Chaieb, M., 2010. Changes in soil properties and vegetation following livestock grazing exclusion in degraded arid environments of South Tunisia. Flora Morphol. Distrib. Funct. Ecol. Plants 205, 184–189. doi:10.1016/j.flora.2009.03.002

Johnson, D.W., Murphy, J.F., Susfalk, R.B., Caldwell, T.G., Miller, W.W., Walker, R.F., Powers, R.F., 2005. The effects of wildfire, salvage logging, and post-fire N-fixation on the nutrient budgets of a Sierran forest. For. Ecol. Manage. 220, 155–165. doi:10.1016/j.foreco.2005.08.011

Lammerant, J., Peters, R., Snethlage, M., Delbaere, B., Dickie, I., Whiteley, G., 2013. Implementation of 2020 EU Biodiversity Strategy: Priorities for the restoration of ecosystems and their services in the EU. Report to the European Commission. ARCADIS (in cooperation with ECNC and Eftec). http://ec.europa.eu/environment/nature/biodiversity/comm2006/pdf/2020/RPF.pdf (Accessed 11/04/2016).

Lavelle, P., Dugdale, R., Scholes, R., Asefaw Behere, A., Carpenter, E., Izac, A.M., Karl, D., Lemoalle, J., Luizao, F., Scholes, M., Tréguer, P., Ward, B., 2005. Nutrient cycling. Chapter 12 in: Millennium Ecosystem Assessment, Island Press, Washington, DC.

Leverkus, A.B., Lorite, J., Navarro, F.B., Sánchez-Cañete, E.P., Castro, J., 2014. Post-fire salvage logging alters species composition and reduces cover, richness, and diversity in Mediterranean plant communities. J. Environ. Manage. 133, 323–331. doi:10.1016/j.jenvman.2013.12.014

Leverkus, A.B., Puerta-Piñero, C., Guzmán-Álvarez, J.R., Navarro, J., Castro, J., 2012. Post-fire salvage logging increases restoration costs in a Mediterranean mountain ecosystem. New For. 43, 601–613. doi:10.1007/s11056-012-9327-7

Li, Y., Zhao, X., Chen, Y., Luo, Y., Wang, S., 2012. Effects of grazing exclusion on carbon sequestration and the associated vegetation and soil characteristics at a semi-arid desertified sandy site in inner Mongolia, northern China. Can. J. Soil Sci. 92, 807–819. doi:10.4141/CJSS2012-030

Lindig-Cisneros, R., Desmond, J., Boyer, K.E., Zedler, J.B., 2003. Wetland restoration thresholds: Can a degradation transition be reversed with increased effort? Ecol. Appl. 13, 193–205. doi:10.1890/1051-0761(2003)013[0193:WRTCAD]2.0.CO;2

MA (Millenium Ecosystem Assessment). 2005. MA Conceptual Framework. Chapter 1 in Millennium Ecosystem Assessment. Island Press, Washington, DC

Maestre, F.T., Cortina, J., Vallejo, R., 2006. Are ecosystem composition, structure, and functional status related to restoration success? A test from semiarid mediterranean steppes. Restor. Ecol. 14, 258–266. doi:10.1111/j.1526-100X.2006.00128.x

Marañón-Jiménez, S., Castro, J., Querejeta, J.I., Fernández-Ondoño, E., Allen, C.D., 2013. Post-fire wood management alters water stress, growth, and performance of pine regeneration in a Mediterranean ecosystem. For. Ecol. Manage. 308, 231–239. doi:10.1016/j.foreco.2013.07.009

Medina-Roldán, E., Paz-Ferreiro, J., Bardgett, R.D., 2012. Grazing exclusion affects soil and plant communities, but has no impact on soil carbon storage in an upland grassland. Agric. Ecosyst. Environ. 149, 118–123. doi:10.1016/j.agee.2011.12.012

Milchunas, D.G., Lauenroth, W.K., 1993. Quantitative effects of grazing on vegetation and soils over a global range of environments. Ecol. Monogr. 63, 327–366.

Moreira, F., Ferreira, A., Abrantes, N., Catry, F., Fernandes, P., Roxo, L., Keizer, J.J., Silva, J., 2013. Occurrence of native and exotic invasive trees in burned pine and eucalypt plantations: Implications for post-fire forest conversion. Ecol. Eng. 58, 296–302. doi:10.1016/j.ecoleng.2013.07.014

Moya, D., De las Heras, J., López-Serrano, F., Ferrandis, P., 2015. Post-fire seedling recruitment and morpho-ecophysiological responses to induced drought and salvage logging in Pinus halepensis Mill. stands. Forests 6, 1858–1877. doi:10.3390/f6061858

Oñatibia, G.R., Aguiar, M.R., Semmartin, M., 2015. Are there any trade-offs between forage provision and the ecosystem service of C and N storage in arid rangelands? Ecol. Eng. 77, 26–32. doi:10.1016/j.ecoleng.2015.01.009

Papanastasis, V.P., 1989. Multipurpose woody plants for the Mediterranean arid zone of Greece. Programme de recherche Agrimed.

Papanastasis, V.P., Bautista, S., Chouvardas, D., Mantzanas, K., Papadimitriou, M., Mayor, A.G., Koukioumi, P., Papaioannou, A., Vallejo, R. V., 2015. Comparative assessment of goods and services provided by grazing regulation and reforestation in degraded Mediterranean rangelands. L. Degrad. Dev. doi:10.1002/ldr.2368

Pugnaire, F.I., Luque, M.T., Armas, C., Gutiérrez, L., 2006. Colonization processes in semi-arid Mediterranean old-fields. J. Arid Environ. 65, 591–603. doi:10.1016/j.jaridenv.2005.10.002

Rojo, L., Bautista, S., Orr, B. J., Vallejo, R., Cortina, J., Derak, M., 2012. Prevention and restoration actions to combat desertification. An integrated assessment: The PRACTICE Project. Secheresse, 23, 219-226.

Rong, Y., Yuan, F., Ma, L., 2014. Effectiveness of exclosures for restoring soils and vegetation degraded by overgrazing in the Junggar Basin, China. Grassl. Sci. 60, 118–124. doi:10.1111/grs.12048

Rost, J., Clavero, M., Bas, J.M., Pons, P., 2010. Building wood debris piles benefits avian seed dispersers in burned and logged Mediterranean pine forests. For. Ecol. Manage. 260, 79–86. doi:10.1016/j.foreco.2010.04.003

Royo, A.A., Carson, W.P., 2006. On the formation of dense understory layers in forests worldwide: consequences and implications for forest dynamics, biodiversity, and succession. Can. J. For. Res. 36, 1345–1362. doi:10.1139/x06-025

Ruiz-Benito, P., Gómez-Aparicio, L., Zavala, M. a., 2012. Large-scale assessment of regeneration and diversity in Mediterranean planted pine forests along ecological gradients. Divers. Distrib. 18, 1092–1106. doi:10.1111/j.1472-4642.2012.00901.x

Su, H., Liu, W., Xu, H., Wang, Z., Zhang, H., Hu, H., Li, Y., 2015. Long-term livestock exclusion facilitates native woody plant encroachment in a sandy semiarid rangeland. Ecol. Evol. 5, 2445–2456. doi:10.1002/ece3.1531

Tongway, D.J., Hindley, N.L., 2004. Landscape function analysis manual: procedures for monitoring and assessing landscapes with special reference to minesites and rangelands. Canberra, ACT: CSIRO Sustainable Ecosystems

Urgeghe, A.M., Bautista, S., 2015. Size and connectivity of upslope runoff-source areas modulate the performance of woody plants in Mediterranean drylands. Ecohydrology 8, 1292–1303. doi:10.1002/eco.1582

Valdecantos, A., Baeza, M.J., Vallejo, V.R., 2009. Vegetation management for promoting ecosystem resilience in fire-prone mediterranean shrublands.

Restor. Ecol. 17, 414–421. doi:10.1111/j.1526-100X.2008.00401.x

Valdecantos, A., Vallejo V.R., 2015. Report on structural and functional changes associated to regime shifts in Mediterranean dryland ecosystems. Deliverable 5.1 CASCADE Project (EU GA283068)

Vallejo, R., Aronson, J., Pausas, J.G., Cortina, J., 2006. Restoration of Mediterranean woodlands. Restor. Ecol. new Front. 193–207.

Vallejo, V.R., Arianoutsou, M., Moreira, F., 2012. Fire ecology and post-fire restoration approaches in Southern European forest types. In *Post-Fire Management and Restoration of Southern European Forests* (pp. 93-119). Springer Netherlands.

Vörösmarty, C.J., Leveque, C., Revenga, C., 2005. Fresh water. Chapter 7 in: Millennium Ecosystem Assessment. Island Press, Washington, DC.

Wang, K., Deng, L., Ren, Z., Li, J., Shangguan, Z., 2016. Grazing exclusion significantly improves grassland ecosystem C and N pools in a desert steppe of Northwest China. Catena 137, 441–448. doi:10.1016/j.catena.2015.10.018

Whisenant, S., 1999. Repairing damaged wildlands: a process-oriented, landscape-scale approach. Cambridge University Press, Port Chester, NY, USA.

9 ANNEXES

ANNEX I. Results of PCA on plant species cover in Várzea.

ANNEX II. Results of PCA on plant species cover in Ayora.

ANNEX III. Results of PCA on plant species cover in Favino, Castelsaraceno (Overgrazed and Fenced).

ANNEX IV. Results of PCA on plant species cover in Monte Alpi, Castelsaraceno (Overgrazed and Fenced).

ANNEX V. Results of PCA on plant species cover in Piano del Campi, Castelsaraceno (Overgrazed and Fenced).

ANNEX VI. Results of PCA on plant species cover in Favino, Castelsaraceno (Undergrazed and Cleared).

ANNEX VII. Results of PCA on plant species cover in Monte Alpi, Castelsaraceno (Undergrazed and Cleared).

ANNEX VIII. Results of PCA on plant species cover in Piano del Campi, Castelsaraceno (Undergrazed and Cleared).

ANNEX IX. Results of PCA on plant species cover in Randi.

ANNEX X. Results of PCA on plant species cover in Albatera.

Annex I. Results of PCA on plant species cover in Reference, Degraded and Restored plots in Várzea. Eigenvalues of plant species on the two first components are shown (only species with eigenvalues higher than 0.500 or lower than -0.500).

Species	PC1	PC2
Ulex minor	0,911	-0,093
Pteridium aquilinum	0,881	-0,073
Quercus robur	0,859	-0,068
Pinus pinaster	0,819	-0,012
Erica cinerea	-0,091	0,925
Agrostis delicatula	-0,091	0,925
Halimium lasianthum	-0,164	0,948
Pterospartum tridentatum	-0,590	-0,181
Agrostis curtisii	-0,712	-0,453

Annex II. Results of PCA on plant species cover in Reference, Degraded and Restored plots in Ayora. Eigenvalues of plant species on the two first components are shown (only species with eigenvalues higher than 0.500 or lower than -0.500).

Species	**PC1**	**PC2**
Pinus pinaster	0,896	0,066
Juniperus thurifera	0,814	-0,206
Teucrium capitatum	0,800	-0,211
Satureja montana	0,784	-0,223
Helichrysum serotinum	0,764	-0,068
Rhamnus lycioides	0,751	-0,225
Salvia lavandulifolia	0,751	-0,225
Linum suffruticosum	0,751	-0,225
Centaurea ornata	0,751	-0,225
Lavandula latifolia	0,693	0,152
Phagnalon rupestre	0,681	0,051
Juniperus oxycedrus	0,475	0,346
Rubia peregrina	0,440	0,126
Saxifraga sp	0,435	0,185
Genista scorpius	0,386	0,025
Bupleurum fruticescens	0,373	0,000
Pinus halepensis	0,367	0,202
Plantago sempervirens	0,342	0,126
Arctostaphyllos uva-ursi	0,245	0,146
Ulex parviflorus	0,239	0,532
Dorycnium pentaphyllum	0,112	-0,425
Brachypodium phoenicoides	0,056	0,653
Daphne gnidium	0,020	0,561
Aphylantes monspeliensis	-0,110	0,400
Rosmarinus officinalis	-0,172	-0,402
Dorycnium hirsutum	-0,206	0,584
Leucanthemum vulgare	-0,206	0,584
Erica multiflora	-0,238	0,236
Thymus vulgaris	-0,282	-0,504
Rhamnus alathernus	-0,313	0,468
Thymus piperella	-0,358	0,609
Poligala rupestris	-0,364	0,529
Helianthemum syriacum	-0,373	0,224
Pistacia lentiscus	-0,373	0,224
Helianthemum apenninum	-0,373	0,224
Argilobium zannoni	-0,378	-0,791
Verbascum thapsus	-0,378	-0,791
Lithodora fruticosa	-0,378	-0,791
Fumana thymifolia	-0,394	-0,822

Annex II (cont.)

Species	PC1	PC2
Helianthemum cinereum	-0,399	-0,603
Stipa offneri	-0,402	-0,719
Helictotrichon filifolium	-0,402	0,183
Teucrium polium	-0,416	0,343
Ononis minutissima	-0,441	-0,156
Fumana ericoides	-0,442	0,214
Erinacea anthyllis	-0,445	-0,553
Quercus ilex	-0,485	0,517
Cistus clusii	-0,574	-0,604
Quercus coccifera	-0,613	0,510
Cistus albidus	-0,674	0,333
Carex humilis	-0,723	0,042
Brachypodium retusum	-0,830	-0,127

Annex III. Results of PCA on plant species cover in Reference, Overgrazed and Fenced plots in Favino. Eigenvalues of plant species on the two first components are shown (only species with eigenvalues higher than 0.500 or lower than -0.500).

Species	PC1	PC2
Capsella bursa pastoris	0,950	0,121
Agropyron repens	0,944	0,070
Holcus lanatus	0,814	0,100
Medicago minima	0,813	0,450
Hordeum murinum	0,757	0,170
Trifolium repens	0,678	-0,048
Cirsium vulgare	0,674	-0,138
Parentucellia latifolia	0,622	-0,036
Sixalix atropurpurea	0,622	-0,036
Bromus hordaceum	0,622	-0,036
Ononis spinosa	0,622	-0,036
Bellis perennis	0,615	0,169
Eryngium campestre	0,540	0,621
Poa pratensis	0,531	0,175
Cruciata laevipes	0,360	-0,550
Prunus spinosa	0,205	-0,524
Lathyrus sphaericus	-0,093	-0,538
Saxifraga bulbifera	-0,093	-0,538
Poa annua	-0,167	0,575
Allium sp	-0,209	-0,675
Pyrus pyraster	-0,245	-0,624
Genista tinctoria	-0,285	-0,625
Lathyrus pratensis	-0,305	-0,607
Poa bulbosa	-0,378	0,735
Anthyllis vulneraria	-0,378	0,735
Lotus corniculatus	-0,378	0,735
Sanguisorba minor	-0,378	0,735
Medicago sativa	-0,378	0,735
Scabiosa columbaria	-0,378	0,735
Phleum hirsutum	-0,378	0,735
Galium verum	-0,415	-0,645
Carex caryophyllea	-0,486	-0,670
Ranunculus lanuginosum	-0,514	-0,050
Trifolium incarnatum	-0,644	0,078
Cichorium intybus	-0,708	0,561
Plantago lanceolata	-0,711	0,165
Anthoxantum odoratum	-0,763	-0,196

Annex IV. Results of PCA on plant species cover in Reference, Overgrazed and Fenced plots in Monte Alpi. Eigenvalues of plant species on the two first components are shown (only species with eigenvalues higher than 0.500 or lower than -0.500).

Species	PC1	PC2
Lonicera caprifolium	0,861	0,102
Medicago lupulina	0,854	-0,048
Phleum hirsutum	0,806	-0,007
Satureya montana	0,773	-0,080
Securigera varia	0,772	0,256
Dactylis glomerata	0,761	-0,082
Cytisophyllum sessilifolium	0,728	0,273
Festuca gr rubra	0,698	0,270
Potentilla hirta	0,686	0,234
Orlaya daucoides	0,686	0,234
Poa pratensis	0,653	0,260
Dorycnium pentaphyllum	0,653	0,260
Polygala nicaeensis	0,653	0,260
Melica ciliata	0,653	0,260
Brachypodium rupestre	0,624	0,359
Anthyllis vulneraria	0,592	-0,276
Helianthemum nummularium	0,248	-0,600
Pimpinella tragium	0,153	0,535
Koeleria splendes	-0,057	-0,566
Trifolium campestre	-0,102	-0,736
Pteridium aquilinum	-0,226	-0,584
Calamintha nepeta	-0,328	-0,578
Lotus corniculatus	-0,328	-0,578
Trifolium scabrum	-0,328	-0,578
Trifolium incarnatum	-0,375	-0,817
Trifolium repens	-0,380	-0,718
Cynosurus cristatus	-0,402	-0,836
Avena barbata	-0,449	0,611
Centaurium erythrae	-0,534	0,684
Hieracium pilosselloides	-0,534	0,684
Sanguisorba minor	-0,582	0,652
Euphrasia stricta	-0,623	0,570
Carex macrolepis	-0,718	0,185
Plantago lanceolata	-0,744	-0,392
Anthoxantum odoratum	-0,802	0,201
Thymus pulegioides	-0,867	0,331

Annex V. Results of PCA on plant species cover in Reference, Overgrazed and Fenced plots in Piano del Campi. Eigenvalues of plant species on the two first components are shown (only species with eigenvalues higher than 0.500 or lower than -0.500).

Species	PC1	PC2
Picris hieracioides	0,995	0,004
Daucus carota	0,989	-0,090
Galium verum	0,986	-0,032
Plantago lanceolata	0,971	-0,177
Agrostis stolonifera	0,944	-0,207
Xeranthemum cilindraceum	0,940	-0,239
Avena barbata	0,927	-0,260
Cynosurus cristatus	0,925	0,341
Lotus corniculatus	0,924	0,377
Brachypodium rupestre	0,889	-0,410
Lathyrus hirsutus	0,868	0,025
Odontites rubra	0,847	-0,416
Potentilla hirta	0,732	0,593
Bromus hordaceum	0,717	0,664
Lolium rigidum	0,711	0,692
Trifolium incarnatum	0,699	-0,629
Hieracium pilosselloides	0,695	0,714
Sixalix atropurpurea	0,691	-0,472
Tussilago farfara	0,658	-0,571
Oenanthe pimpinelloides	0,616	0,751
Potentilla reptans	0,616	0,751
Cirsium arvense	0,616	0,751
Mentha pulegioides	0,616	0,751
Juncus inflexsus	0,616	0,751
Leucanthemum vulgare s.l	0,616	0,751
Convolvulus arvensis	0,616	0,751
Trifolium pratense	0,616	0,751
Sanguisorba minor	0,599	0,567
Medicago sativa	0,598	-0,706
Cynosurus echinatus	0,547	-0,479
Malus sylvestris	0,531	-0,719
Linum trigynum	0,531	-0,719
Consolida regalis	0,531	-0,719
Rubus ulmifolius	0,531	-0,719
Bromus erecti	0,531	-0,719
Prunella vulgaris	0,531	-0,719
Blakstonia perfoliata	0,531	-0,719
Centaurium erythrae	0,513	0,100

Annex V (cont.)

Species	PC1	PC2
Eryngium campestre	0,227	0,739
Bellardia trixago	-0,503	0,019
Micromeria graeca	-0,511	0,075
Phleum hirsutum	-0,586	0,026
Stipa austroitalica	-0,598	0,111
Botrhiocloa ischamum	-0,772	0,048
Scorzonera villosa sub columnae	-0,933	0,065
Triticum ovatum	-0,958	0,080

Annex VI. Results of PCA on plant species cover in Reference, Undergrazed and Cleared plots in Favino. Eigenvalues of plant species on the two first components are shown (only species with eigenvalues higher than 0.500 or lower than -0.500).

Species	PC1	PC2
Festuca gr ovina	0,930	0,339
Bromus arvensis	0,921	0,375
Plantago lanceolata	0,901	0,376
Phleum pratense	0,837	0,446
Linum trigynum	0,824	0,416
Anthoxantum odoratum	0,807	0,457
Arrhenatherum elatius	0,776	0,464
Vicia craca	0,776	0,464
Carum carvi	0,776	0,464
Hypericum perforatum	0,776	0,464
Alnus cordata arb	0,776	0,464
Mentha pulegioides	0,776	0,464
Torilis japonica	0,776	0,464
Poa bulbosa	0,776	0,464
Eryngium amethystinum	0,755	0,287
Cruciata laevipes	0,682	0,556
Clinopodium vulgare	0,643	0,101
Trifolium pratense	0,284	-0,678
Galium lucidum	0,168	-0,760
Echinops ritro	0,148	-0,559
Thymus pulegioides	0,022	-0,731
Blakstonia perfoliata	-0,053	-0,514
Lomelosia crenata	-0,053	-0,514
Teucrium chamaedrys	-0,053	-0,514
Carlina vulgaris	-0,053	-0,514
Brachypodium rupestre	-0,061	-0,938
Koeleria splendes	-0,064	-0,590
Xeranthemum cilindraceum	-0,078	-0,629
Pimpinella tragium	-0,082	-0,618
Odontites rubra	-0,083	-0,542
Daucus carota	-0,083	-0,542
Spartium junceum	-0,104	-0,790
Festuca circummediterranea	-0,104	-0,816
Sixalix atropurpurea	-0,138	-0,509
Crateagus monogyna	-0,188	-0,502
Bromus erecti	-0,289	-0,586
Ranunculus lanuginosum	-0,441	0,515
Trifolium incarnatum	-0,504	0,537
Cirsium vulgare	-0,507	0,509

Annex VI (cont.)

Species	PC1	PC2
Rosa canina	-0,529	0,176
Poa pratensis	-0,551	0,575
Potentilla recta	-0,559	0,564
Bellis perennis	-0,559	0,564
Holcus lanatus	-0,570	0,473
Geranium rotundifolium	-0,570	0,473
Hordeum murinum	-0,642	0,626
Agropyron repens	-0,656	0,592
Eryngium campestre	-0,670	0,596
Trifolium repens	-0,673	0,622
Medicago minima	-0,695	0,630
Capsella bursa pastoris	-0,695	0,621

Annex VII. Results of PCA on plant species cover in Reference, Undergrazed and Cleared plots in Monte Alpi. Eigenvalues of plant species on the two first components are shown (only species with eigenvalues higher than 0.500 or lower than -0.500).

Species	PC1	PC2
Cynosurus cristatus	0,971	0,123
Ranunculus lanuginosum	0,965	0,136
Potentilla hirta	0,959	-0,044
Centaurea jacea s.l	0,951	-0,227
Ononis spinosa	0,930	0,253
Trifolium pratense	0,927	-0,307
Oenanthe pimpinelloides	0,912	-0,254
Genista tinctoria	0,908	0,264
Anthriscus nemorosa	0,899	0,105
Prunella vulgaris	0,838	0,330
Cirsium tenoreanum	0,831	-0,485
Plantago lanceolata	0,808	-0,481
Poa trivialis	0,806	-0,463
Lathyrus pratensis	0,795	-0,454
Daucus carota	0,781	-0,338
Picris hieracioides	0,716	-0,555
Calamintha nepeta	0,716	-0,506
Festuca gr ovina	0,671	-0,369
Lolium multiflorum	0,671	-0,369
Alnus cordata arb	0,671	-0,369
Arrhenatherum elatius	0,671	-0,369
Agrostis stolonifera	0,649	0,640
Anthoxantum odoratum	0,588	0,759
Lotus corniculatus	0,574	-0,366
Phleum pratense	0,522	-0,338
Trifolium repens	0,506	-0,329
Holcus lanatus	0,426	0,870
Galium lucidum	0,399	-0,585
Potentilla reptans	0,367	0,895
Trifolum stellatum	0,367	0,895
Lathyrus sylvestris	0,367	0,895
Tussilago farfara	0,367	0,895
Trifolium incarnatum	0,367	0,895
Bromus hordaceum	0,367	0,895
Carex hirta	0,367	0,895
Carlina vulgaris	0,347	0,883
Galium verum	0,282	0,818
Lolium rigidum	0,108	0,881

Annex VII (cont.)

Species	**PC1**	**PC2**
Koeleria splendes	-0,205	0,705
Dactylis glomerata	-0,521	0,055
Satureya montana	-0,526	0,175
Medicago lupulina	-0,530	0,121
Teucrium chamaedrys	-0,550	-0,220
Brachypodium rupestre	-0,565	-0,309
Sanguisorba minor	-0,572	0,101
Pimpinella tragium	-0,595	-0,190
Phleum hirsutum	-0,775	0,042

Annex VIII. Results of PCA on plant species cover in Reference, Undergrazed and Cleared plots in Piano del Campi. Eigenvalues of plant species on the two first components are shown (only species with eigenvalues higher than 0.500 or lower than -0.500).

Species	PC1	PC2
Sanguisorba minor	0,915	0,333
Linum trigynum	0,854	0,491
Stipa austroitalica	0,853	0,369
Micromeria graeca	0,850	0,133
Teucrium chamaedrys	0,837	0,225
Trifolium campestre	0,832	-0,022
Galium lucidum	0,818	0,328
Spartium junceum	0,761	0,358
Ononis pusilla	0,750	0,225
Anthyllis vulneraria	0,685	-0,091
Aristolochia pallida	0,651	-0,029
Lathyrus sylvestris	0,645	0,559
Medicago sativa	0,631	0,126
Carlina vulgaris	0,617	0,437
Triticum ovatum	0,523	-0,704
Crateagus monogyna	0,507	-0,016
Trifolium pratense	0,500	-0,185
Lotus corniculatus	0,500	-0,185
Hypericum perforatum	0,500	-0,185
Polygala monspeliaca	0,500	-0,185
Potentilla hirta	0,500	-0,185
Bellardia trixago	0,396	-0,616
Hyppocrepis comosa	0,327	-0,711
Sixalix atropurpurea	0,233	0,582
Dactylis glomerata	0,228	-0,845
Koeleria splendes	0,129	-0,858
Scorzonera villosa sub columnae	0,000	-0,916
Carex flacca	-0,008	-0,699
Carlina corymbosa	-0,013	0,743
Trigonella gladiata	-0,038	-0,589
Poa molinieri	-0,038	-0,589
Coronilla scorpioides	-0,072	-0,696
Botrhiocloa ischamum	-0,088	-0,839
Rubus ulmifolius	-0,162	0,728
Poa annua	-0,502	0,281
Cirsium vulgare	-0,502	0,281
Bromus erecti	-0,624	0,364
Trifolium stellatum	-0,625	0,390

Annex VIII (cont.)

Species	PC1	PC2
Lolium rigidum	-0,626	0,376
Geranium rotundifolium	-0,635	0,387
Avena barbata	-0,638	0,190
crepis leontodontoides	-0,659	0,388
Bromus hordaceum	-0,701	0,452
Trifolium nigrescens	-0,709	0,428
Medicago minima	-0,796	0,464

Annex IX. Results of PCA on plant species cover in Reference, Degraded and Restored plots in Randi. Eigenvalues of plant species on the two first components are shown (only species with eigenvalues higher than 0.500 or lower than -0.500).

Species	PC1	PC2
Sacropoterium spinosum	0,713	0,503
Rhamnus oleoides	0,630	0,694
Plantago sp	0,579	0,574
Asteraceae	0,533	0,028
Asphodelus aestivus	0,496	-0,072
Taraxacum aphrogenes	0,489	0,032
Urginea maritima	0,454	-0,459
Climber	0,453	0,454
Tulipa cypria	0,436	0,821
Grass unidentified	0,381	-0,019
Olea europaea	0,373	0,891
Arum italicum	0,373	-0,210
Trifolium campestre	0,373	-0,210
Helichrysum italicum	-0,185	0,021
Helichrysum conglobatum	-0,401	-0,028
Genista sphacelata	-0,409	0,731
Lithodora hispidula	-0,585	0,636
Calycotome villosa	-0,602	0,171
Prasium majus	-0,648	0,369
Pinus halepensis	-0,667	0,379
Rosmarinus sp	-0,783	0,318
Pistacia lentiscus	-0,794	0,019
Cistus creticus	-0,867	0,145

Annex X. Results of PCA on plant species cover in Reference, Degraded and Restored plots in Albatera. Eigenvalues of plant species on the two first components are shown (only species with eigenvalues higher than 0.500 or lower than -0.500).

Species	**1**	**2**
Rhamnus lycoides	0,899	0,293
Echium creticum	0,809	0,423
Anthyllis cytisoides	0,775	0,332
Ephedra sp	0,735	0,207
Cistus clusii	0,735	0,207
Pistacia lentiscus	0,735	0,207
Anagallis arvensis	0,721	0,082
Artemisia campestris	0,696	0,247
Brachypodium retusum	0,684	-0,001
Artemisia barrelieri	0,604	0,359
Ephedra fragilis	0,533	0,366
Withania frutescens	0,486	0,215
Fagonia cretica	0,286	0,362
Fumana thymifolia	-0,001	-0,705
Crepis vesicaria	-0,032	-0,500
Asphodelus fistulosus	-0,044	-0,472
Diplotaxis lagascana	-0,054	-0,715
Stipa capensis	-0,114	0,707
Salsola genistoides	-0,163	-0,077
Pinus halepensis	-0,204	-0,167
Juniperus oxycedrus	-0,205	0,184
Stipa tenacissima	-0,259	-0,396
Helichrysum stoechas	-0,329	0,180
Sedum sediforme	-0,337	0,050
Sideritis murgetana	-0,433	0,570
Anthyllis terniflora	-0,474	0,469
Lygeum spartum	-0,540	0,493
Helianthemum almeriense	-0,564	0,584
Asparagus horridus	-0,569	0,593
Fumana ericoides	-0,569	0,411
Olea europaea	-0,597	0,668
Globularia alypum	-0,613	0,271