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Effects of post-land abandonment management strategies on water resources, vegetation dynamics, and soil properties and redistribution in the Central Spanish Pyrenees

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Y a tí Hanin,

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Abstract

Mediterranean mountains have been subject to strong land use and land cover changes during the last century. After centuries of intensive use of resources (livestock, agriculture, construction of settlement and infrastructure, etc.) that led to the deforestation of many slopes, these areas were abandoned due to several environmental and socioeconomic circumstances. Land abandonment subsequently generated important land use and land cover changes that affected the vegetation and the soils endangering the sustainability of regulation and provisioning ecosystem services for lowland areas (particularly water supply). The Spanish Pyrenees are among the most affected regions by land abandonment in the Mediterranean and among the most sensitive to future abandonment episodes. Changes after land abandonment are attributed to both management practices (active restoration/management) and the lack of management (passive restoration/management). The absence of post-land abandonment management implies a natural revegetation process, with the colonization of the abandoned fields by a dense secondary vegetation cover that goes through different succession stages. In other areas, regional administrations (regional or state) intervened through afforestation programs to control soil erosion and accelerate the recovery of the abandoned croplands, or by clearing shrubs to regenerate pastures and reduce forest fires. These land use and land cover transformations affect ecosystem processes, which are also affected by climate change in areas such as the Mediterranean region, that are highly sensitive to its repercussions. Therefore, the scientific community and policy makers should pay more attention to the possible impacts of climate change and land use and land cover change.

In this PhD dissertation, the effects of post-land abandonment management (passive or active) and those of climate change on waters resources, vegetation dynamics, and soils properties and its spatial redistribution are addressed. The methods include the use of an ecohydrological model (the Regional Hydro-Ecological Simulation System, RHESSys) and laboratory soil analysis. RHESSys model was set up, calibrated, and validated using observed streamflow data from the Arnás experimental catchment (Central Spanish Pyrenees) and permitted to study different management and climate scenarios. Fallout ^{137}Cs measurements and other soil properties in 61 soils samples in the Araguás experimental catchment (Central Spanish Pyrenees) permitted to study the effects of land use and land cover changes on soil properties, nutrients, and redistribution rates. Results

showed significant effects on ecohydrological dynamics related to management practices. Shrub clearing increased streamflow by up to 24% and decreased evapotranspiration to almost 8% in 10 years, while afforestation reduced streamflow up to 53% and increased transpiration up to 23% in 50 years compared to natural revegetation. On the other hand, although, changes related to temperature rise were overcome by those linked to land cover, the warming scenario clearly affected the water and vegetation dynamics. The magnitude and timing of the changes were affected by climate variability, vegetation age, and physiological attributes as well as the total managed surface and the intensity of management. The higher the management intensity or the managed surface the higher the changes to water and vegetation dynamics. On the other hand, the analysis of soils properties and nutrients suggested that afforestation could contribute to the recovery of soils, showing the highest soil organic carbon and ^{137}Cs inventories. However, although afforestation showed the lowest erosion rates and the sparsely vegetated areas the highest ones, differences related to land use and land cover were non-significant, suggesting that the generalized erosion process over the Araguás catchment is not only attributed to vegetation change but also to management practices and the historic human activity in the catchment.

This PhD dissertation provides important insights on the effects of land abandonment and climate change on ecosystem processes and evaluates traditional (afforestation) and new (shrub clearing) management practices in comparison to no management. Results from the conducted studies could help elaborate appropriate management plans for the abandoned croplands in Mediterranean mountains that contribute to guaranteeing the sustainability of regulation and provisioning ecosystem services for lowland areas.

Resumen

Las montañas mediterráneas han estado sujetas a fuertes cambios en el uso y la cubierta del suelo durante el último siglo. Después de siglos de aprovechamiento intenso de los recursos (ganadería, agricultura, construcción de asentamientos e infraestructura, etc.) que llevó a la deforestación de muchas laderas, estas fueron abandonadas debido a diversas circunstancias ambientales y socioeconómicas. El abandono de la tierra generó importantes cambios en el uso y la cubierta del suelo que afectaron a la vegetación y a los suelos poniendo en peligro la sostenibilidad de los servicios ecosistémicos de regulación y de aprovisionamiento para las zonas bajas (en particular, el suministro de agua). Los Pirineos españoles se encuentran entre las regiones más afectadas por el abandono de tierras en el Mediterráneo y entre las más sensibles a futuros episodios de abandono. Los cambios tras el abandono de tierras se atribuyen tanto a las prácticas de gestión (restauración / gestión activa) como a la falta de gestión (restauración / gestión pasiva). La ausencia de gestión post-abandono implica un proceso de revegetación natural, con la colonización de los campos abandonados por una densa cubierta vegetal que pasan por diferentes fases de sucesión secundaria. En otras áreas, las administraciones (regional o estatal) han intervenido a través de programas de repoblación forestal para controlar la erosión del suelo y acelerar la recuperación de las tierras de cultivo abandonadas, o bien mediante el desbroce de matorrales para regenerar pastos y disminuir el riesgo de incendios forestales. Estos cambios en los usos y cubiertas del suelo afectan a los procesos de los ecosistemas; además, éstos también se ven afectados por el cambio climático en áreas, como las montañas mediterráneas, muy sensibles a sus repercusiones. Por ello, la comunidad científica y los responsables políticos deben prestar más atención a los posibles impactos del cambio climático y de usos y cubiertas del suelo.

En esta tesis doctoral se abordan los efectos del abandono de la tierra, en función de la gestión posterior (pasiva o activa) y del cambio climático previsto, sobre los recursos hídricos, la dinámica de la vegetación y los cambios en las propiedades del suelo y su redistribución espacial. Entre los métodos empleados hay que destacar el uso de un modelo ecohidrológico (The Regional Hydro-Ecological Simulation System, RHESSys) y el análisis de suelos. El modelo RHESSys se configuró, se calibró y se validó utilizando datos de caudal observados de la cuenca experimental de Arnás (Pirineo Aragonés) y permitió estudiar diferentes escenarios climáticos y de gestión. Las mediciones de

decaimiento del ^{137}Cs y otras propiedades del suelo en 61 muestras de suelos en la cuenca experimental de Araguás (Pirineo Aragonés) permitieron estudiar los efectos del cambio de uso y cubierta del suelo sobre las propiedades del suelo, los nutrientes y las tasas de redistribución.

Los resultados mostraron efectos significativos en la dinámica ecohidrológica relacionada con las prácticas de gestión. Los desbroces de matorral aumentaron el caudal hasta un 24% y disminuyeron la evapotranspiración a casi un 8% en 10 años, mientras que la repoblación forestal redujo el caudal hasta un 53% y aumentó la transpiración hasta un 23% en 50 años en comparación con la revegetación natural. Por otro lado, si bien los cambios relacionados con el aumento de temperatura fueron superados por los vinculados a la cubierta del suelo, el escenario de calentamiento afectó claramente las dinámicas del agua y la vegetación. La magnitud y el momento de los cambios se vieron afectados por la variabilidad climática, la edad de la vegetación y los atributos fisiológicos, así como la superficie total gestionada y la intensidad de la gestión. Cuanto mayor sea la intensidad de la gestión o la superficie gestionada, mayores serán los cambios en las dinámicas del agua y la vegetación. Por otro lado, el análisis de las propiedades y nutrientes de los suelos sugirió que la repoblación forestal podría contribuir a la recuperación de los suelos, presentando los inventarios más altos de carbono orgánico del suelo y ^{137}Cs. Sin embargo, a pesar de que la repoblación forestal mostró las tasas de erosión más bajas y las áreas con escasa vegetación las más altas, las diferencias relacionadas con el uso y la cubierta del suelo no fueron significativas, lo que sugiere que el proceso de erosión generalizado en la cuenca de Araguás no solo se atribuye al cambio de vegetación sino también a las prácticas de gestión y la actividad humana histórica en la cuenca.

Esta tesis doctoral aporta información importante sobre los efectos del abandono de la tierra y el cambio climático en los procesos del ecosistema y evalúa prácticas de gestión tradicionales (repoblaciones forestales) y otras nuevas (desbroces de matorrales) en comparación con la ausencia de gestión. Los resultados de los estudios realizados podrían ayudar a elaborar planes de gestión adecuados para las tierras de cultivo abandonadas en las montañas mediterráneas que contribuyan a garantizar la sostenibilidad de los servicios ecosistémicos de regulación y de suministro para las zonas bajas.

Chapter 1. Introduction

This chapter presents the scientific background of the PhD dissertation. It gives a state of the art on the abandonment of cropland areas in the Central Spanish Pyrenees and its repercussions on water resources, vegetation, and soil. An illustration of the main active and passive management plans in these areas is also presented to introduce the main subject of the presented research studies. Further, a justification of the PhD dissertation, its main objectives and hypothesis are presented. Finally, the chapter summarizes the general structure of the PhD dissertation.

1. Scientific background

Land use and land cover (LULC) provide multiple assets and services to society and, therefore, are of vital importance to humans (Foley et al., 2005; Verkerk et al., 2016). Changes to it are included among the main causes of global warming i.e. the overall natural and human processes modifying biophysical systems, which affects the sustainability of development (Vitousek et al., 1997; Duarte, 2006). Changes in LULC can make significant contributions to climate change (CC) through changes to greenhouse gas emissions (Fleischer et al., 2016), biodiversity loss (Newbold et al., 2015), and the degradation of ecosystem services (Lambin et al., 2001; Millennium Ecosystem Assessment, 2005).

Mountain areas are especially prone to changes in LULC (Körner and Spehn, 2002). Recently, globalization and CC have accelerated their dynamics with significant consequences for the provision of ecosystem services to the adjoining regions (Lambin and Meyfroidt, 2011; Verburg et al., 2013; Magliocca et al., 2015). The scientific literature stresses that changes have been particularly severe in the Mediterranean mountains in Spain. In these mountains, the beginning of the 20^{th} century saw the highest population density and expansion of agricultural areas, followed by depopulation and socio-economic marginalization, gradual loss of agricultural fields and less intense use of land for livestock (MacDonald et al., 2000; Strijker, 2005; Lasanta et al., 2021). The initial and most obvious result of land abandonment was plant recolonization, with the spread of secondary succession shrubs and forests (Vicente-Serrano et al., 2005; Bielsa et al., 2005; Lasanta and Vicente-Serrano, 2007; Arnáez et al., 2008; Vila Subirós et al., 2009). Song et al. (2018) estimated that tree cover on Earth increased by 2.2 million Km^2 (+7.1%) between 1982 and 2016, while bare ground declined by 1.2 million Km^2 (-3.1%) in the same period, with the Mediterranean mountains one of the revegetation hotspots.

On a global scale, the spread of the vegetation is possibly a positive process, as far as it helps to compensate for the loss of large areas of forest, especially in tropical regions (FAO, 2006; Austin et al., 2017). However, it must be remembered that it has significant implications in the production and quality of water, soil erosion, carbon sequestration, fire hazard, the structure and aesthetics of the landscape, biodiversity, and in general ecosystem dynamics and sustainability (García-Llamas et al., 2019; Lasanta et al., 2021). Therefore, the revegetation process is considered to be the change with the highest environmental and socio-economic impact in the Mediterranean mountains (García-Ruiz and Lana-Renault, 2011; San Román et al., 2013). These changes not only affect the mountains, but also society as a whole, given the importance of essential services, that these areas provide to the population, such as water, biodiversity, production of wood and meat, different and attractive landscapes for recreation and leisure, high levels of carbon sequestration, and the prevention of natural hazards, among others (Schmitz et al., 2003; Viviroli et al., 2007; Petanidou et al., 2008). Therefore, Price (2004) reminds us that the mountains supply assets and services to over half the world's population.

1.1. Passive and active management of the Mediterranean mountains

There are two alternatives in managing abandoned agricultural lands: i) passive management which means not intervening in the process of revegetation and re-naturing the landscape, or ii) active management, which requires intervention to control the negative effects of revegetation. The subject was considered sufficiently important, recently, for a discussion to take place between scientists and land managers (Conti and Fagarazzi, 2005; Lasanta et al., 2015; Nogués-Bravo et al., 2016, among others).

Since 1998, the term *rewilding* has been used for passive management. Rewilding sees restoring natural ecosystems and reducing control of the landscape as a highly positive step (Soulé and Noss, 1998; Pereira and Navarro, 2015). Supporters of rewilding state that the large amounts of abandoned land in Europe (MacDonald et al., 2000; Keenleyside and Tucker, 2010; Lasanta et al., 2017) provides an excellent opportunity to promote a more natural landscape for a large-scale recovery of wildlife (Navarro and Pereira, 2012). Passive management also means creating a more natural landscape with higher biodiversity which would support the reintroduction or return of wildlife (Ceausu et al., 2015). It is believed that more wild ungulates would enable a stable and diverse landscape to be maintained, similar to the role extensive livestock farming used to play (San Miguel-Ayanz et al., 2010). Navarro and Pereira (2012) emphasize that in this respect, the Mediterranean landscape is more sustainable the more natural it is, especially because it reduces soil degradation, which is a basic factor for obtaining high levels of biodiversity. In addition, regenerating forests would bring ecosystem services to the population, particularly carbon sequestration and leisure and recreation uses. Finally, they suggest that those wanting to maintain traditional agriculture underestimate the huge effort required for the task.

The concept of rewilding has gradually gained approaches and perspectives for understanding it. At present, there is a wide range of views, from one proposing that uncontrolled ecosystems should be allowed to evolve (Shnitzler, 2014), to others that defend occasional minor actions on the terrain to minimize conflicts between the public and wildlife (Hobbs et al., 2015; Boitani and Linnell, 2015). However, all views agree on promoting a natural landscape and wildlife, although they differ on the approach and means by which the ecological processes should be restored (Deary and Warren, 2017; Fernández et al., 2017). Rewilding has the support of several NGOs, environmental activists and owners of large farms. The Rewilding Europe project aims to restore 1 million hectares of abandoned fields to a natural space (Helmer et al., 2015). During the last few years, the rewilding concept constituted the main topic of multiple scientific articles and became a common issue in journalism and activities linked to nature (Pereira and Navarro, 2015; Corlett, 2016; Svenning et al., 2016).

Figure 1.1. Abandoned agricultural fields undergoing a natural revegetation process in the Arnás catchment (Central Spanish Pyrenees).

On the other hand, some scientists and managers lean toward taking actions on the land in order to control the negative effects of the revegetation process. Such actions aim to keep some features of the traditional mosaic landscape because of its ecological (Moreira et al., 2006), productive (Quetier et al., 2005; Kizos and Koulouri, 2006), cultural and aesthetic (Vanslemsbrouk et al., 2005; Sayadi et al., 2009) value. On the other hand, they try to avoid an over-renatured landscape that is very unlike the traditional ones in Europe (Farina, 2007). Supporters of active management add that the advantages of unbridled renaturing are more apparent than real (Conti and Fagarazzi, 2005). They warn that the landscape in Europe, especially in the Mediterranean, is humanized and shaped by three thousand years of human activity (Grobe and Rackham, 2001; Agnoletti, 2014). Similarly, they argue that the traditional Mediterranean mountain landscape in Europe is cultural and results from cultivating slopes and extensive livestock grazing (Farina, 2007; García-Ruiz and Lasanta, 2018). Taking actions on the land would also be justified by the fact that landscapes should provide assets and services (produce food, water, wood, etc.), as well as maintaining resources and environmental values. Only multi-functional landscapes are sustainable in the medium and long term (Kinzing et al., 2006; Kizos and Koulouri, 2006; Pelorosso et al., 2011).

Figure 1.2. Mosaic landscape in Mediterranean mountains (Estarrún watershed, Central Spanish Pyrenees).

Furthermore, the large extension of abandoned lands in the Mediterranean mountains and the lack of local initiatives, due to the depopulation and social fragmentation recorded throughout the 20th century (García-Ruiz, 1976; Anglada et al., 1980; Collantes, 2005) together provide an excellent opportunity to manage these lands with most present demanded land uses and landscape functions: water and high-quality meat, nature and culture, aesthetic landscapes, etc. (Cogliastro et al., 2003; Benjamin et al., 2008; Keesstra et al., 2018).

Two of the most adopted active management strategies by land managers are afforestation and extensive livestock farming. Large tracts of land were afforested in the 20th century in the Mediterranean countries of Europe, mainly with pine and eucalyptus. In Spain, for example, a large afforestation plan led to 5 million hectares being reforested with conifers from 1940 to 2006. Afforestation has continued in the last few decades with financial support from the European Union (EU), thus increasing wooded areas (S.C.F.C., 2011; Montero and Serrada, 2013). Replanting trees pursues several objectives: economic (to create jobs, reduce depopulation, produce wood and wood pulp), and environmental (reduce soil erosion, control floods, regulate the hydrological cycle, increase the useful life of reservoirs, etc.) (Ortigosa Izquierdo, 1990; Pausas et al., 2004). However, the economic performance is somewhat poor - it creates very few jobs and, if planting is excluded, it does not bring people to settle in rural areas, also wood and pulp production is limited and only in the long-term (García-Ruiz, 1976; Chauvelier, 1990). García-Ruiz (1976) and Ortigosa et al. (1990) state that, in the Pyrenees of Aragón (Spain), tree planting had a very negative impact on local development, as many areas of pasture and shrubs were afforested and limited or prevented livestock grazing, which hastened the departure of the locals, depopulated a large part of the region and reduced the numbers of livestock.

Figure 1.3. Afforestation of abandoned cropland slopes in the Central Spanish Pyrenees.

Environmental results are very varied, depending on the physical conditions of the afforested area, how it was planted and managed post-plantation. Thus, soil erosion is higher and the growth of pines lower if terraces were built by bulldozers, as that removes the top layer of soil which is the most fertile, resulting in shallow pine rooting and an unprotected ground for many years (Ortigosa Izquierdo, 1991). Results were improved if planting was done by digging holes or troughs manually (Ortigosa Izquierdo, 1991). In addition, Nadal-Romero et al. (2016a) compared changes in the soil (fertility, quality and carbon sequestration) among abandoned fields covered by pasture or shrubs, and afforestation. They observed that afforestation did not speed up the recovery of soil properties (fertility and quality) compared to abandoned fields undergoing revegetation processes, although increased organic carbon sequestration (Nadal-Romero et al., 2016b). Finally, it must be highlighted that afforestation frequently creates a highly uniform landscape vulnerable to fire (Pausas et al., 2008; Martínez-Fernández et al., 2013).

Extensive livestock grazing is the most common management strategy in the Mediterranean mountains. This is mainly due to the relative abundance of pastures, and the inappropriate nature of terrain for intensive agriculture (Barry and Ives, 1974). A high proportion of the land is only capable of producing grass or shrubs, which are only profitable if grazed by livestock (García-Ruiz and Lasanta, 1989). Extensive livestock farming adapts well to very depopulated areas with low economic activity. This allows the animals to graze almost freely, with little control by shepherds, which helps reduce labor costs (García-Ruiz and Lasanta, 1989).

However, the low numbers of livestock in wide areas of the Mediterranean mountains and the scant control by shepherds over livestock movements means that some areas are overgrazed, frequently degrading the soil, while others are undergrazed, leading to loss in the quality of pastures and invasion by shrubs (Lasanta, 2010), which has a negative impact on the landscape (homogenization, defragmentation, etc.), biodiversity, and resources (degraded pasture, more fires, loss of water resources). In this context and over the last few decades, extensive livestock farming has been required to provide environmental and landscape functions, as well as the usual economic objective. Land managers deem livestock grazing to be a very useful tool to control the spread of shrubs and preserve cultural landscapes. This is one reason why it is subsidized by public authorities. For example, the EU has given subsidies via the Common Agricultural Policy since 1992 (Mottet et al., 2006; García-Martínez et al., 2008). Extensive livestock farming offers other benefits to profit from public subsidies, as it combines production (meat, wood, milk, etc.), social (settle people in rural areas and maintaining villages and cultural landscapes) and environmental purposes (reduce fire risk and increase biodiversity) (Mouillot et al., 2003; Calvo-Iglesias et al., 2006; Rescia et al., 2008). Gibón (2005) pointed out that extensive livestock grazing can transform a shrub-covered, highly uniform landscape into a more diverse and fragmented one consisting of various land uses, such as forest, pasture, shrubland and some fields for meadows, which is the characteristic landscape of the mountains in Europe (Farina, 2007).

In line with active management, some regional administrations in Spain (Castilla and León, Galicia, La Rioja, etc.), using finance from various agencies, carry out shrub clearing in selected areas, mostly abandoned fields, in order to promote the regeneration of pastures and reduce the risk of fire (Lasanta, 2014). Lasanta et al. (under review) pointed out that between 1986 and 2020, in the Iberian System in La Rioja community, 37,341 ha were cleared, constituting 28.4% of the shrub cover in the cartographies of Lasanta-Martínez and Errea Abad (2001) at the end of the 20th century. The regional government of Aragón has also carried out some shrub clearing recently (Komac et al., 2013). As the results for extensive grazing and fire control are positive (Lasanta et al., 2018 and 2019a), shrub clearing is envisaged to be a very common practice in the near future.

Figure 1.4. Extensive livestock farming in shrub cleared areas in the Iberian system.

The present dissertation took into account some of the requirements of the La Rioja regional government's Plan for Shrub Clearing (PSC), which are explained in detail in chapters 3 and 4. The other regulations in the PSC deal with issues on the environment, such as preventing soil erosion, creating a mosaic landscape, protecting the fauna and encouraging biodiversity (Lasanta et al., 2016).

1.2. The importance of water resources generated in the mountains for the water supply in the Mediterranean region

The availability of water is one of the most pressing problems in the world, particularly in the Mediterranean region. It is expected that water shortages will increase in the next few decades in most Mediterranean countries, due to climate change and the spread of plant cover (Nogués-Bravo et al., 2008; García-Ruiz et al., 2011; Pascual et al., 2015). Population growth (Arnell, 2004) and the high demand for water for irrigation in rural and urban areas (Rodríguez Díaz et al., 2007; Wriedt et al., 2009; Jlassi et al., 2016), are additional factors. Climate models for the Mediterranean region suggest a more irregular rainfall distribution, increased and more severe droughts, and high atmospheric water demand, as a result of rising temperatures (Giorgi and Lionello, 2008; Lionello and Scarascia, 2018). These predicted changes will probably reduce water resources in the Mediterranean basin in the next few decades (García-Ruiz et al., 2011).

This is based on an unassailable fact: water resources in the Mediterranean region are mostly generated in headwaters, i.e. mountain areas, which act as wet islands surrounded by sub-humid and semi-arid terrain (Viviroli et al., 2020 and Immerzeel et al., 2020). On the other hand, lowland areas scarcely produce water resources, while consuming large amounts of water (urban, industrial and agricultural uses), giving rise to marked imbalances between regions. These facts have prompted measures to manage water resources and ensure a supply to flat areas, including the construction of reservoirs (López-Moreno, 2006; Jlassi et al., 2016).

The flow in rivers is mainly determined by the climate and vegetation cover. Changes in the climate and/or the composition, density, structure and extent of vegetation cover give rise to a greater or lesser flow, and an altered river systems i.e in seasonal distribution, frequency and magnitude of floods (Andréassian, 2004). Any change in the vegetation cover in mountains is a threat to the fragile equilibrium between the availability of water, reservoir management, ecological flows and use of water in the lowlands (García-Ruiz and Lana-Renault, 2011). In this respect Gallart and Llorens (2002) examined the land cover changes in Spain in recent decades and the implications for water resources. They found that the vegetation cover had increased, while river flows had decreased. This was attributed to three factors: i) the increase in the population's water consumption, ii) the change in rainfall regime, and iii) higher evaporation in headwaters, most of which were undergoing large revegetation processes. In the Spanish Pyrenees in particular, river flows have become demonstrably lower from the mid-1960s, caused by land abandonment and revegetation, which increases water use by the vegetation and the interception of rainfall (Beguería et al., 2003; García-Ruiz and Lana-Renault, 2011). A study carried out by López-Moreno et al. (2014) in five catchments in the Pyrenees of Aragón concluded that climate change would result in a fall of 13%-23% in annual flows, depending on the studied catchment. If the effects from climate change are added to those from increased vegetation cover, flows would fall by 19%-32%, fluctuating to over 40% in certain seasons. López-Moreno et al. (2011) studied climate and hydrological trends in 88 sub-catchments in the Ebro basin from 1950-2006. The flow was found to be lower in most of the sub-catchments, which was attributed to a combination of climate change and revegetation due to land abandonment, with the consequent rise in evapotranspiration. Morán-Tejeda et al. (2010) came to similar conclusions when studying the hydrological evolution of the Duero headwaters, where they found lower flows, especially in winter and spring.

The widespread revegetation process recorded in the Mediterranean mountains since the mid-20th century therefore plays an important role over water resources. In the future, there may be different scenarios for land uses and cover, ranging from a passively managed landscape, dominated by shrubs and forest, to another actively managed through shrub clearing for livestock grazing or afforestation (Lasanta et al., 2015). Land managers must decide which option to take, either let the vegetation advance or take action on the land by afforestation or shrub clearing, in the knowledge that any steps involving the vegetation cover has implications for the quality and quantity of water resources and erosion processes.

There have been many recent advances in studies on hydrology and soil erosion related to the physical characteristics of the terrain, changes in land use and cover and socio-economic factors. These advances would not have been possible without the development of a set of new techniques and methodologies for field, laboratory and office work, the latter ones supported by IT and computers: Geographical Information Systems (GIS), scenario development, application of models, etc. The opportunities offered by these techniques have enabled challenges to be posed on different spatial and temporal scales, and obtained quantitative information of great value. The use of experimental catchments and hydrological models has been especially interesting for this PhD dissertation. The availability of digitized geographic information and process-based models has helped to tackle the current problem on the availability of hydrological resources on large scales (2^{nd} and 3^{rd} order and regional catchment areas), and improved the knowledge and predictions on runoff, depending on land uses and cover and the projected change in climate (López-Moreno et al., 2014; Morán-Tejeda et al., 2015).

In addition, monitoring small experimental catchment areas (<10 km^2), used as field laboratories, enables hydrological and geomorphological impacts from various uses and land management to be precisely evaluated. This evaluation provides very precise information on a spatial scale and over a long period of time, which contributes to optimum calibration and validation of hydrological and sediment export simulations. The Instituto Pirenaico de Ecología (IPE-CSIC) is monitoring several experimental catchments in the Central Pyrenees that have been gradually set up and monitored since the 1980s (see García-Ruiz et al., 2010; Lana-Renault et al., 2018; Juez et al., 2021): Arnás (a catchment area of abandoned fields undergoing a dynamic revegetation process), San Salvador (covered by beech, pine and oak woods), Araguás Repoblación (afforested in the late 1960’s with pine trees), Araguás (including the afforested Araguás Repoblación, abandoned fields, badlands, and some pastures) and Izas (pasture land, above the tree line). This thesis uses information from Arnás, San Salvador, Araguás Repoblación and Araguás.

Furthermore, hydrological models are useful tools to predict hydrological processes at catchment scale, taking into account various climate and vegetation. These are spatially distributed models used to determine not only variations in flow, but also differences in water sources, as well as temporary storage in the catchment (Merrit et al., 2003). Likewise, these models can enlarge the scale of work in experimental catchment areas to extend to scenarios at valley or regional scale. The thesis uses the Regional Hydro-Ecological Simulation System (RHESSys) model (Tague and Band, 2004) (see Chapter 2 on materials and methods).

Figure 1.5. Experimental catchments of the Pyrenean Institute of Ecology (IPE-CSIC) in the Central Spanish Pyrenees. A. The Arnás catchment: Natural revegetation, B. The San Salvador catchment: Natural forest, C. The Araguás catchment: Afforestation, Natural revegetation, Agriculture and badlands.

1.3. Inter-relationship between vegetation and soil

Vegetation strongly affects soil properties, carbon sequestration and erosion, therefore, any change in plant cover involves improvement or degradation in soil quality (Lasanta et al., 2019b).

In general, revegetation processes improve soil quality by increasing organic material and microbial biomass, reducing bulk density, and generating more stable aggregates which raise infiltration rates and soil humidity (Lesschen et al., 2008; Nadal et al., 2009; Zornoza et al., 2009; Romero-Díaz et al., 2017). This promotes positive feedback by stimulating revegetation, and hindering the introduction of surface erosion processes (laminar erosion and rills) (Cantón et al., 2011; Lasanta et al., 2021). However, the greater presence of roots can lead to water accumulating in the ground and increase mass movement (Cammeraat et al., 2005). If, on the other hand, vegetation advances very slowly or decreases, highly active erosion processes are established, with the consequent loss of soil quality (Pardini and Gispert, 2005; Romero Díaz et al., 2017). Higher density of plant cover usually leads to lower rates of soil erosion (García-Ruiz and López Bermúdez, 2009). It is a logical consequence of the protective effect of vegetation against rain splash and surface runoff. Furthermore, as already stated, soil properties tend to improve with revegetation, promoting infiltration and reducing surface runoff (Martínez-Fernández et al., 1995; Seeger and Beguería, 2003).

The ground is very important for storing soil organic carbon (SOC) in most ecosystems (García-Pausas et al., 2017). Vegetation density and composition condition the rates of carbon sequestration from the atmosphere and the stocks of SOC in the ground, making essential its management (De Baets et al., 2013; Lasanta et al., 2021). Lasanta et al. (2020) studied the SOC content in different land uses and covers in the Iberian mountain system (fist stages of abandonment, shrubs, forest with pasture and forest with shrub undergrowth). It was found that the SOC content was higher in the more complex plant cover structures, particularly in the top 10 cm of soil, and forest with shrub undergrowth had the highest rates of SOC. The authors explain that the larger SOC content in forests is due to the accumulation of mulch on the ground (leaf litter and dead material), and because trees increase the underground biomass. In the Pyrenees, Nadal-Romero et al. (2016a) studied what active management involves (afforestation of abandoned fields with pine trees) compared to passive management (pastures and succession shrubs). They found that shrubs accumulated less SOC than pastures and pine forest. The evolution of SOC content has been also studied by Nadal-Romero et al. (2018) in slopes with mature subalpine pastures (hundreds of years of continuous pasture), recently abandoned in the Pyrenees and subject to secondary succession. They found that the change from pasture to shrubs led to a strong decline in SOC in the soil. In young and mature *P. sylvestris* forest, the SOC partially recovered, but did not reach the subalpine pasture conditions.

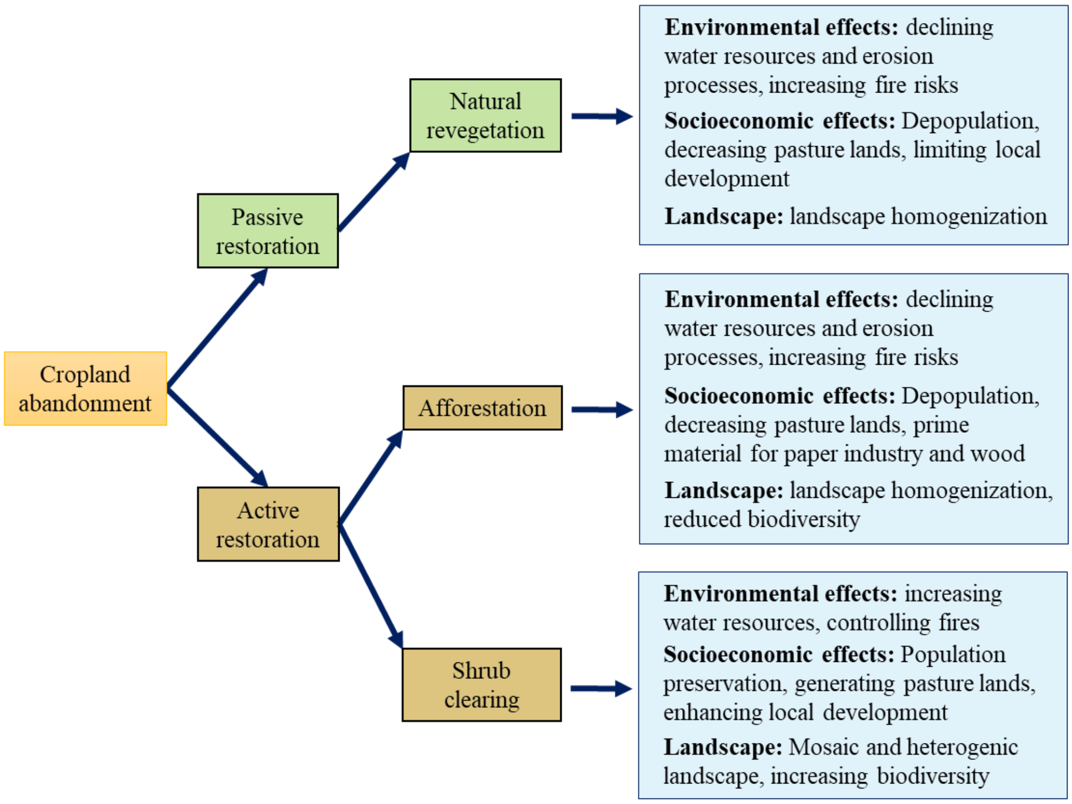

Figure 1.6. Conceptual model on the main strategies of restoration of abandoned cropland areas in the Central Spanish Pyrenees and its environmental, socioeconomic and landscape effects (adapted from García-Ruiz et al., 2020).

2. Justification of the PhD dissertation

The results from this PhD dissertation can have a significant socio-economic impact in areas of the Mediterranean basin. The revegetation of abandoned agricultural land and under-grazing are widespread in the Mediterranean mountains of Europe. This is predicted to continue into the future, with the mountains of North Africa not exempt from the process (Keenleyside and Tucker, 2010; Terres et al., 2015).

In addition, various studies emphasize the reduction in river flows in the Mediterranean basin since the 1970s (López-Moreno et al., 2011; Martínez-Fernández et al., 2013). They also point out that this is not only due to climate issues, but also to revegetation (increased evapotranspiration and water consumption by the vegetation, etc.). In addition, pressure on water resources has grown in the last few decades, linked to a higher demand in urban and rural areas (especially increased irrigation).

Despite the growing imbalance between water production and consumption, there is no management of land uses and covers in the headwaters. At present, the mountains of the

Mediterranean basin are considered marginal areas which are, apparently, a poor economic prospect. They contribute very little to the economy in comparison with the lowlands, where all investment and activities are concentrated. However, it must be borne in mind that mountain catchments can supply several assets and services to society. The main one is the supply of water resources to adjoining areas. They are also the cause of certain extreme events (floods, mass movements, etc.) giving rise to heavy economic and social losses. Therefore, it is fundamental to find out how land cover and management systems impact on flows and sediment export, and to forecast realistic future scenarios from the combination of changes in climate and land use and cover.

The results from this thesis will advance knowledge on the availability of water resources in the lowlands and help experts and public authorities with decision making. The information contained in this thesis will be highly valuable for the supply or expansion of irrigated areas, managing and planning reservoirs, designing conservation measures for land and water, applying the EU Water Framework Directive, or for rural development programs and managing marginal mountain areas.

We also believe that the thesis will advance knowledge of the interrelationships between land cover and the production and quality of water. In this respect, the results can be expected to have a certain scientific and technical impact in the near future by using methodologies on various spatial scales. The results will certainly allow for more realistic strategies in sustainable management of water and land resources. In short, it is expected to make advances on the current knowledge on i) defining realistic scenarios of land use and cover, ii) establishing an ecohydrological model to study changes in hydrological and ecological dynamics under different land covers and warming scenarios, iii) quantifying water production in probable future scenarios and, iv) assessing changes to soil properties and redistribution rates under different land management scenarios of abandoned cropland areas.

3. Hypothesis and objectives

3.1. Initial hypothesis

Although climate change is having a verifiable effect on the functioning of natural systems, it is no less true that the changes seen in land use and cover can be an equally determining factor. Depopulation in the Mediterranean mountains and abandonment of agricultural land during the 20^{th} century has led to natural revegetation by shrubs and forest species. At the same time, many slopes were afforested (particularly by pine trees) for socioeconomic and environmental purposes. Consequently, the region has been revegetated with large hillslopes covered by shrubs and forest, in contrast to the traditional mosaic landscape where crop fields alternate with pasture, shrubs and forest in different vegetation cover structures. The new, more homogeneous and defragmented landscape has a direct impact on the availability of water resources and land, especially in the mid-mountains of the Mediterranean, which are highly sensitive to changes in both the landscape and climate.

The **initial hypothesis** in this PhD dissertation proposes that passive management of the region leads to a scenario with greater shrub and forest cover. Added to this is afforestation by conifer plantations. Increased vegetation means decreases in the water supply and improvements in soil properties, especially in carbon sequestration. On the other hand, shrub clearing to encourage pasture in selected areas, mainly abandoned fields, should lead to higher runoff and flows without significantly altering soil properties. Proving this hypothesis would provide land managers with highly valuable information to implement management strategies to ensure water availability without increasing soil loss or considerably reducing carbon sequestration, taking into account that mountain areas are essential for supplying various assets and services to the adjacent lowlands.

3.2. Objectives of the PhD dissertation

The main objective of the project is to find how much water is produced under different land use and cover scenarios taking into account the current global warming scenario. This is to obtain information for land managers to develop practices in mountain areas in the medium- and long-term that are compatible with both land conservation and water production, in order to guarantee the water supply in the lowlands and extend the useful life of reservoirs.

Within the framework of this proposal, the specific objectives are the study of the following in the Central Spanish Pyrenees:

1. The effects of natural revegetation and shrub clearing in abandoned cropland areas and climate change on the water and vegetation dynamics.

2. The effects of the delay in managing abandoned cropland areas on water resources.
3. The long-term hydrological trends of afforestation and natural revegetation in abandoned cropland areas and climate change.
4. The tradeoffs between carbon sequestration and water yield under natural revegetation and afforestation in abandoned cropland areas.
5. The effects of land abandonment and post-land abandonment management (through natural revegetation and afforestation practices) on soil properties and soil redistribution rates.

4. Structure of the thesis

The thesis is organized into 9 chapters: 4 general chapters and 5 that seek to move forward in each of the 5 specific objectives given above. These 5 chapters constitute the main body of the PhD dissertation. Each one is structured into an abstract, an introduction oriented to the proposal, hypothesis and specific objectives of the subject under study, followed by a section on material and methods that includes a description of the study area and methodology used. It continues with a description of the results, a discussion on these and the main conclusions. Each chapter ends with the bibliographical references. The 4 remaining chapters (the first two and last two) have a freer structure.

Chapter 1 deals with the introduction, which includes this section. The general approach to the subject of the thesis is described, together with some background on the issues to be addressed in the 5 central chapters. Thus, there is reference to passive and active land management, the interrelationship between land uses and land cover and water resources in the Mediterranean mountains, and how changes in plant cover affect the soil. The hypothesis of the thesis and objectives pursued are described briefly. It also includes a short section aiming to justify the scientific interest of the thesis.

Chapter 2 deals with material and methods. The first part of the chapter gives a very general description of the physical aspects of the Central Spanish Pyrenees, which is the general geographical area of the 5 central studies. The specific study sites where the presented studies were carried out: The Arnás catchment, the Estarrún watershed, and Araguás catchment are also presented. This is followed by a short introduction to the methods used, as there is a more detailed explanation in central chapters.

Table 1.1 includes information on chapters 3, 4, 5, 6 and 7. Each chapter contains a study on a specific objective of the thesis; therefore, it has been conceived as a unit structured as a scientific article, as mentioned before. Chapter 3 analyzes the effects of managing abandoned fields in the Arnás catchment, together with those of climate change on vegetation dynamics and the hydrological response. The objective of chapter 4 is to further knowledge on managing areas of abandoned agricultural land to predict changes in plant cover and hydrological dynamics following land abandonment in the Estarrún watershed. Chapter 5 analyzes long-term changes and annual and seasonal trends in streamflow and transpiration in the Arnás catchment under three revegetation scenarios of abandoned land. Chapter 6 provides information on the interactions between carbon sequestration and water yield following natural revegetation or afforestation of abandoned lands. These 4 studies use the ecohydrological RHESSys model to simulate the different vegetation and climate scenarios. Chapter 7 discusses changes in soil properties and redistribution rates in the Araguás catchment.

Chapter 8 makes a brief assessment of the main results obtained from this PhD dissertation. Chapter 9 presents some general conclusions seeking to summarize all those taken from various parts of the PhD dissertation and highlights future work.

Table 1.1. Thesis structure: Specific objectives, main techniques, methods and software used in the different research studies.

Specific objectives	Main techniques and methods	Software
Study the effects of passive and active management of abandoned cropland areas and climate change on the water and vegetation dynamics in the Central Spanish Pyrenees.	- Preprocessing of climate record and preparation of model inputs. - Calibration and validation of RHESSys model in the Arnás catchment. - Definition of areas of shrub clearing in the Arnás catchment. - Run RHESSys in static (no growth) and dynamic (growth) modes. - Combining land cover scenarios with climate scenarios (historic and warming climates).	- Excel - R programing - Cluster computing (Trueno-CSIC) - Geographic Information System (Grass, ArcGis) - RHESSys
Publication: **Khorchani, M.,** Nadal-Romero, E., Tague, C., Lasanta, T., Zabalza, J., Lana-Renault, N., Domínguez-Castro, F., Choate, J., 2020. Effects of active and passive land use management after cropland abandonment on water and vegetation dynamics in the Central Spanish Pyrenees. **Science of the Total Environment** 717, 137160 https://doi.org/10.1016/j.scitotenv.2020.137160.		

Table 1.1. Thesis structure: Specific objectives, main techniques, methods and software used in the different research studies. (continuation)

Specific objectives	Main techniques and methods	Software
Study of the effects of the delay in managing abandoned cropland areas on water resources in the Central Spanish Pyrenees.	- Reconstruction of a long-term climate record and preparation of model inputs. - Upscalling from experimental catchment scale to watershed scale. - Initializing vegetation parameters based on real satellite Leaf area index estimations (Target spin-up using Landsat images). - Modeling vegetation succession using RHESSys and quantifying conversion to shrubs and forest. - Definition of areas of shrub clearing in the Arnás catchment in four spin-up phases. - Running RHESSys in dynamic mode under four different delay scenarios.	- Excel - R programing - Cluster computing (Trueno-CSIC) - Geographic Information System (Grass, ArcGis) - RHESSys
Publication: **Khorchani, M.**, Nadal-Romero, E., Lasanta, T., Tague, C., 2021. Effects of vegetation succession and shrub clearing after land abandonment on the hydrological dynamics in the Central Spanish Pyrenees. **Catena** 204, 105374. https://doi.org/10.1016/j.catena.2021.105374		

Specific objectives	Main techniques and methods	Software
Study of the long term hydrological trends of afforestation and natural revegetation in abandoned cropland areas and climate change in the Central Spanish Pyrenees.	- Reconstruction of a long-term climate record and preparation of model inputs. - Definition of three land cover scenarios including two afforestation scenarios and a natural revegetation scenario. - Running RHESSys in dynamic mode and simulating afforestation plans and temperature increase. - Trend analysis using the MannKendal test. - Estimating the magnitude of change per decade in water fluxes at seasonal and annual scales.	- Excel - R programing - Cluster computing (Trueno-CSIC) - Geographic Information System (Grass, ArcGis) - RHESSys
Publication: **Khorchani, M.,** Nadal-Romero, E., Lasanta, T., Tague, C., 2021. Natural revegetation and afforestation in abandoned cropland areas: Hydrological trends and changes in Mediterranean mountains. **Hydrological Processes** 35, e14191. https://doi.org/10.1002/hyp.14191		

Table 1.1. Thesis structure: Specific objectives, main techniques, methods and software used in the different research studies. (continuation)

Specific objectives	Main techniques and methods	Software
Study of the tradeoffs between carbon sequestration and water yield under natural revegetation and afforestation in abandoned cropland areas in the Central Spanish Pyrenees.	- Reconstruction of a long-term climate record and preparation of model inputs. - Definition of three land cover scenarios including two afforestation scenarios and a natural revegetation scenario. - Running RHESSys in dynamic mode and simulating afforestation plans and temperature increase. - Quantifying annual and seasonal carbon sequestration and water yield under the three land management and two climate scenarios. - Quantifying the Water-use efficiency under the different management and climate scenarios.	- Excel - R programing - Cluster computing (Trueno-CSIC) - Geographic Information System (Grass, ArcGis) - RHESSys
Publication: **Khorchani, M**., Nadal-Romero, E., Lasanta, T., Tague, C., 2022. Carbon sequestration and water yield tradeoffs following restoration of abandoned agricultural lands in Mediterranean mountains. **Environmental Research** 112203. https://doi.org/10.1016/j.envres.2021.112203		

Specific objectives	Main techniques and methods	Software
Study of the effects of natural revegetation and afforestation on soil properties and redistribution rates in the Araguás catchment in the Central Spanish Pyrenees.	- Elaborating land cover, topography and flow maps - Defining soil sampling design. - Soil sampling. - Laboratory analysis. - Analysis of land cover change effects on soil properties. - Quantifying soil redistribution rates.	- Excel - R programing - Geographic Information System (ArcGis)
Publication: **Khorchani, M.**, Gaspar, L., Nadal-Romero, E., Arnáez, J., Lasanta, T., Navas, A. **To be submitted**. Effects of cropland abandonment and afforestation on soil properties and redistribution in a small Mediterranean mountain catchment.		

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Chapter 2. Geographic scope, material and methods

This chapter presents the locations where the different research studies, included in this PhD dissertation, were conducted. A brief general description of the geographic scope, the Central Spanish Pyrenees, and the specific study sites: (i) the Arnás catchment, (ii) the Estarrún watershed, and (iii) the Araguás catchment, are presented. Further, the different datasets used in the research are enumerated, highlighting the preprocessing of model inputs and the preparation and analysis of soil samples. A short description of how RHESSys represents water and carbon fluxes in relation to the four Chapters using the model (Chapters 3, 4, 5 and 6) is included to facilitate the understanding of the modelling approach. Finally, the main used statistics are presented.

1. Geographic scope: The Central Spanish Pyrenees

The Pyrenees are a large mountain chain in the south-west of Europe forming a natural border between Spain and France. The Pyrenees cover an area of more than 450 km in length from west to east and more than 100 km in width in its central sector. In this area, altitudes above 2500 m.a.s.l. are very frequent. The mountain range reaches a maximum altitude in the Peak of Aneto (3404 m.a.s.l.) and has two higher cores around the Monte Perdido and Posets massifs with numerous peaks above 3000 m.a.s.l. The present Pyrenees are the results of water erosion, landslides and glacial movement over millions of years since the mid-Miocene. However, the formation of the mountain range dates between two great tectonic periods, the Variscan orogeny, at the end of the Primary era, and the Alpine orogeny, during much of the Tertiary era (García-Ruiz et al., 2015).

In the central sector of the Spanish Pyrenees, the climate is highly conditioned by the altitude and the proximity to the Mediterranean Sea and the Atlantic Ocean. Temperature distribution is mainly affected by two different gradients. An altitudinal gradient due to the decrease in altitude from north to south and a west east gradient defined by the proximity to the Atlantic Ocean and the Mediterranean Sea (Creus, 1978; García-Ruiz et al., 2015). However, the spatial distribution of precipitations is more complex. The highest elevation and the western part of the Pyrenees are the areas with the highest average annual precipitation (reaching 2500 mm/year) while lower elevation and the eastern part of the range are the areas with the lowest annual average (reaching 700 mm/year) (Batalla et al., 2018).

The high topographic and climatic variability of the Pyrenees resulted in a rich and varied vegetation cover (García-Ruiz et al., 2015), with a mixture between Atlantic, Mediterranean and boreo-alpine species among others (Ninot et al., 2007). However, after centuries of human pressure, this vegetation cover is clearly altered. One of the main Figures of this alteration is the large abandoned cropland areas. Lasanta-Martínez (1988) estimated that 71% of the agricultural areas in the Central Spanish Pyrenees has been abandoned. Some of these abandoned lands were afforested during the last decades while the rest is currently undergoing an advanced process of natural revegetation.

1.1. Selection of study sites

The present PhD dissertation only addresses specific sites in the Central Spanish Pyrenees, the Arnás and the Araguás experimental catchments and the Estarrún watershed (Figure 2.1). These mountain areas are representative of the land use legacy in the Central Spanish Pyrenees given their high cultivation during the 19th century and abandonment during the first decades of the 20th century. After abandonment, Arnás and Estarrún are undergoing a natural revegetation process converting the old agricultural fields to different stages of secondary succession. In some areas where soils are deep and well developed the secondary succession process already reached advanced tree stages, while the steepest slopes where soils are poor and degraded, the shrub cover persists. In Arnás and Estarrún no management plans have been adopted after abandonment making of the two watersheds a good example of passive management in Mediterranean mountains. Furthermore, the availability of a long-term monitoring record was a decisive factor for selecting the Arnás catchment to calibrate and validate RHESSys model.

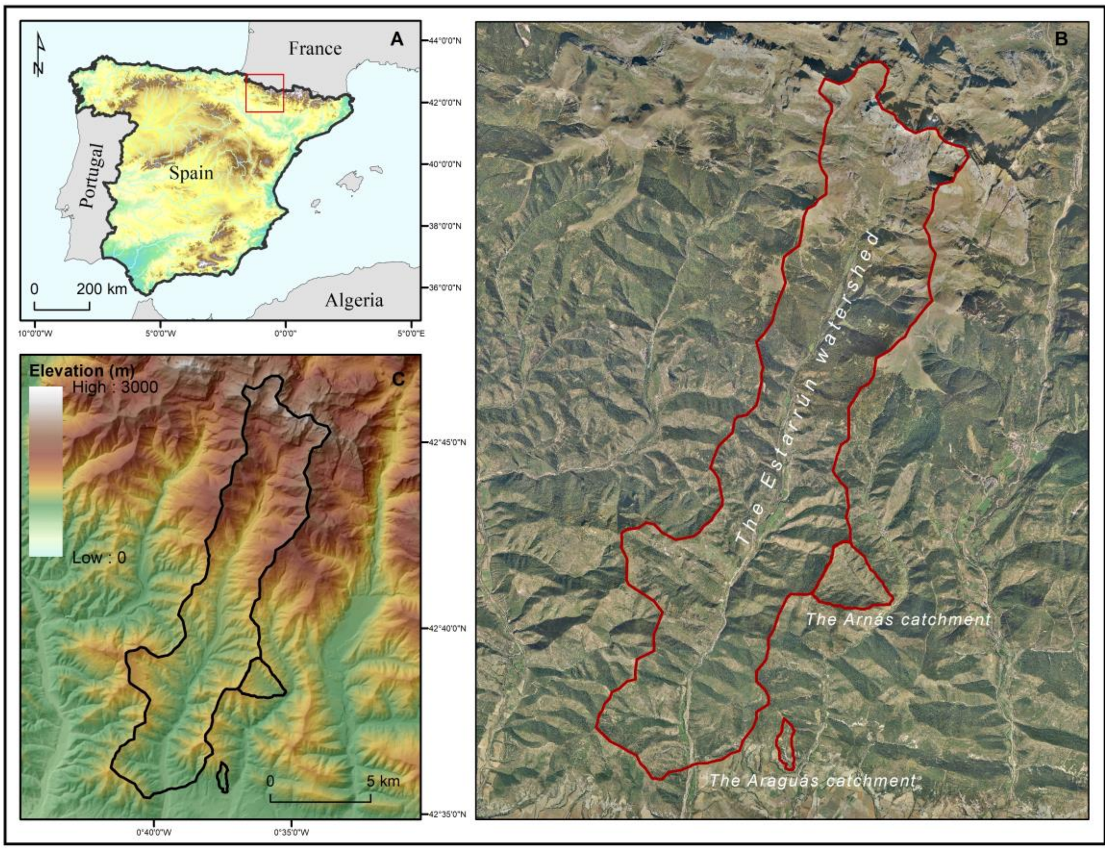

Figure 2.1. The selected study areas in the Central Spanish Pyrenees. A: Localization map. B: Aerial photography of the study area. C: Elevation map of the study area.

In addition, Tague et al. (2013) pointed out the possibility to use similar parameters without model calibration in areas with similar geology. In that sense, the proximity and similarity of the Estarrún watershed to the Arnás catchment (similar geology, climate and vegetation) encouraged an upscaling to a larger spatial scale using the same calibration parameters from Arnás. The calibration of RHESSys in the Arnás catchment and the upscaling in the Estarrún watershed permitted the modeling of realistic passive and active management trajectories coupled with climate change scenarios at small to medium spatial scale. On the other hand, the Araguás catchment was selected for the analysis of cropland abandonment and afforestation effects on soil properties and soil redistribution processes at catchment scale. The selection of Araguás is based on the high geomorphological activity of the catchment and its vulnerability to high erosion events. Araguás was abandoned during the first decades of the 20th century and afforested during the late 1960 making of the catchment a good example of restoration plans in abandoned cropland areas in the Central Spanish Pyrenees.

1.2. The Arnás catchment

Arnás is a small catchment of 0.284 km^2 located in the upper Aragón river in the Central Spanish Pyrenees (Figure 2.2). The west-east orientation of the ravine separates the highly contrasted north-facing and south-facing slopes with altitudes between 900 and 1400 m.a.s.l (Figure 2.2.D). Precipitation, which is uniformly distributed during the year, reaches an annual average of 900 mm. While, average annual temperature, highly affected by Atlantic and sub Mediterranean influence, is around 10 ºC. Sandstone and marl layers of Eocene Flysch define the main lithology of the catchment (Lana-Renault et al., 2007). with shallow-eroded Regosol soils in the south-facing part due to the steep slopes and deep-well developed soils in the gentler north-facing slopes formed mainly by Cambisol and Regosol soils, following the World Reference Base taxonomy (WRB, 2015) (Figure 2.2.E).

Arnás is a good example of the process of cropland abandonment in Mediterranean mountains given the land use legacy in the past decades. During the 19th century, the catchment has been highly cultivated with non-irrigated cereals including areas of steep slopes. During the first half of the 20th century, the catchment has been progressively abandoned. After abandonment, no active plans for the management of the catchment have been adopted, which resulted in a progressive colonization of the abandoned fields by a shrub cover. The process of secondary succession succeeded to reach forest stage in the north-facing slopes while was inhibited by the steep slopes and degraded soils in the south-facing slopes. *Pinus sylvestris* and *Quercus faginea* trees (particularly in the north-facing slopes in the first abandoned fields) are the main trees present in the catchment, while shrubs species mainly consist of *Juniperus communis, Buxus sempervirens*, *Rosa gr*. *canina*, *Echinospartum horridum* and *Genista scorpius*, covering most of the catchment (Figure 2.2.F).

1.3. The Estarrún watershed

The Estarrún watershed (76.8 km^2) is located in the Central Spanish Pyrenees (Figure 2.3). A high north-south altitudinal gradient with altitudes between 700 and 2300 m.a.s.l. crosses the watershed defining the main stream (Figure 2.3.D). This altitudinal gradient is reflected in a high climatic contrast between higher and lower areas. Average annual precipitation reaches 1800 mm in the north while in the south it ranges around 800 mm. The same gradient is also clear in average annual temperature, rounding 9 ºC in the north and 12 ºC in the south. The geology is similar to Arnás catchment with alternating thin layers of sandstones and marls belonging to the Eocene Flysch. Soils are mainly formed by Cambisols and Phaeozem soils, while some Regosols soils are found near stream gauge following the World Reference Base taxonomy (WRB, 2015) (Figure 2.3.E).

This watershed has been highly cultivated until the first decades of the 20^{th} century, when a process of cropland abandonment took place. Similar to many other regions in the Mediterranean mountains, by 1981 most of the cropland area had been abandoned (Lasanta-Martínez, 1988; Lasanta et al., 2019).

Coniferous and deciduous trees (*P*. *sylvestris* and *Q*. *faginea*) predominate the vegetation cover particularly in the north-facing slopes, while shrubs (*E. horridum*, *J. communis*, *G. scorpius, B. sempervirens*) are spread over the south-facing slopes. Around the riparian zone and lowland areas are located the last croplands. This vegetation cover indicates an advanced secondary succession process in the watershed where most of the abandoned cropland areas already reached forest stage (Figure 2.3.F).

1.4. The Araguás catchment

The Araguás catchment is a north-south small catchment, with altitudes between 780 and 1100 m.a.s.l., largely occupied by badlands (Figure 2.4.B, 2.4.D). The climate is sub-

Mediterranean with oceanic and continental influences. Mean annual temperature is 10 ºC and mean annual precipitation is 800 mm. Slopes are steep over most of the catchment particularly in the badland area near the ravine (Figure 2.4.C). Eocene Flysch and Eocene marls form the main lithology of the catchment with relatively deep Cambisol soils in the upper part of the catchment and shallow Regosol soils in the lower part, following the World Reference Base taxonomy (WRB, 2015) (Figure 2.4.E).

The present landscape is a complex mosaic in which alternate: afforested areas of pine trees from the late 1960, abandoned cropland areas colonized by shrub species *(G. scorpius, J. communis, R. gr. canina* and *B. sempervirens*), small untilled pasturelands, and sparsely vegetated badlands characterized by important geomorphological dynamics (Figure 2.4.F).

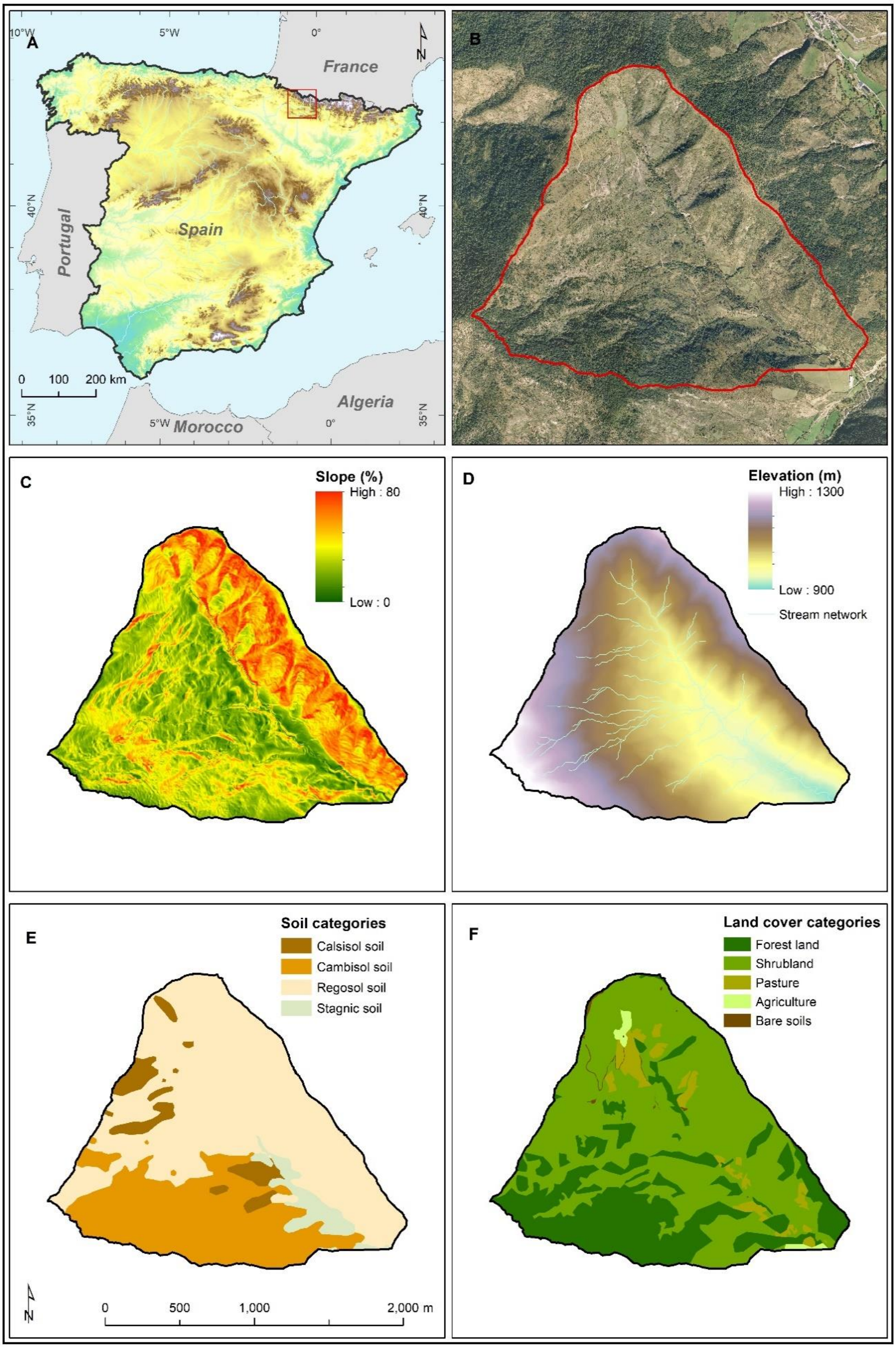

Figure 2.2. The Arnás catchment. A: Localization map, B: Aerial photography, C: Slope map, D: Elevation and stream network map, E: Soil map, F: Land cover map.

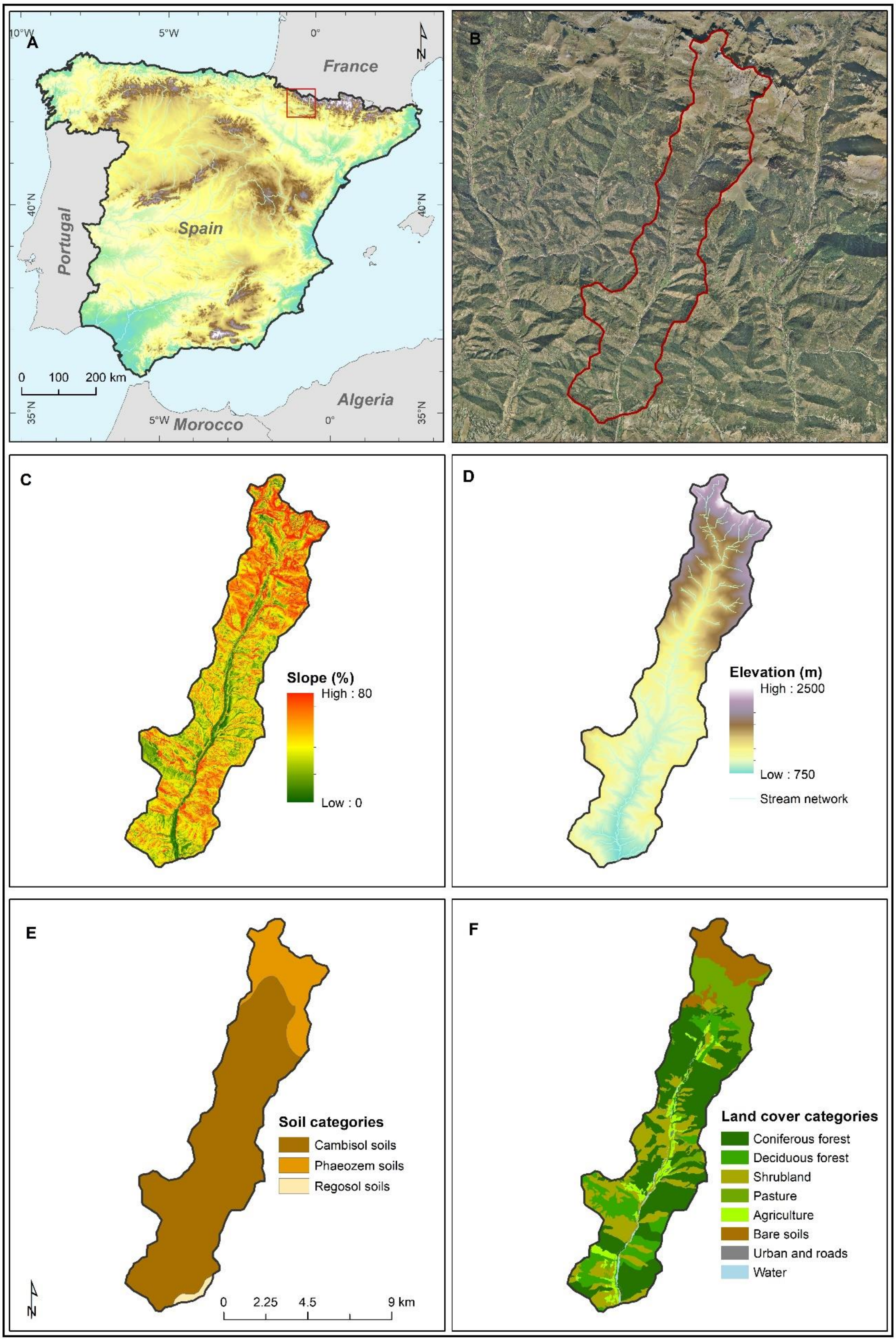

Figure 2.3. The Estarrún watershed. A: Localization map, B: Aerial photography, C: Slope map, D: Elevation and stream network map, E: Soil map, F: Land cover map.

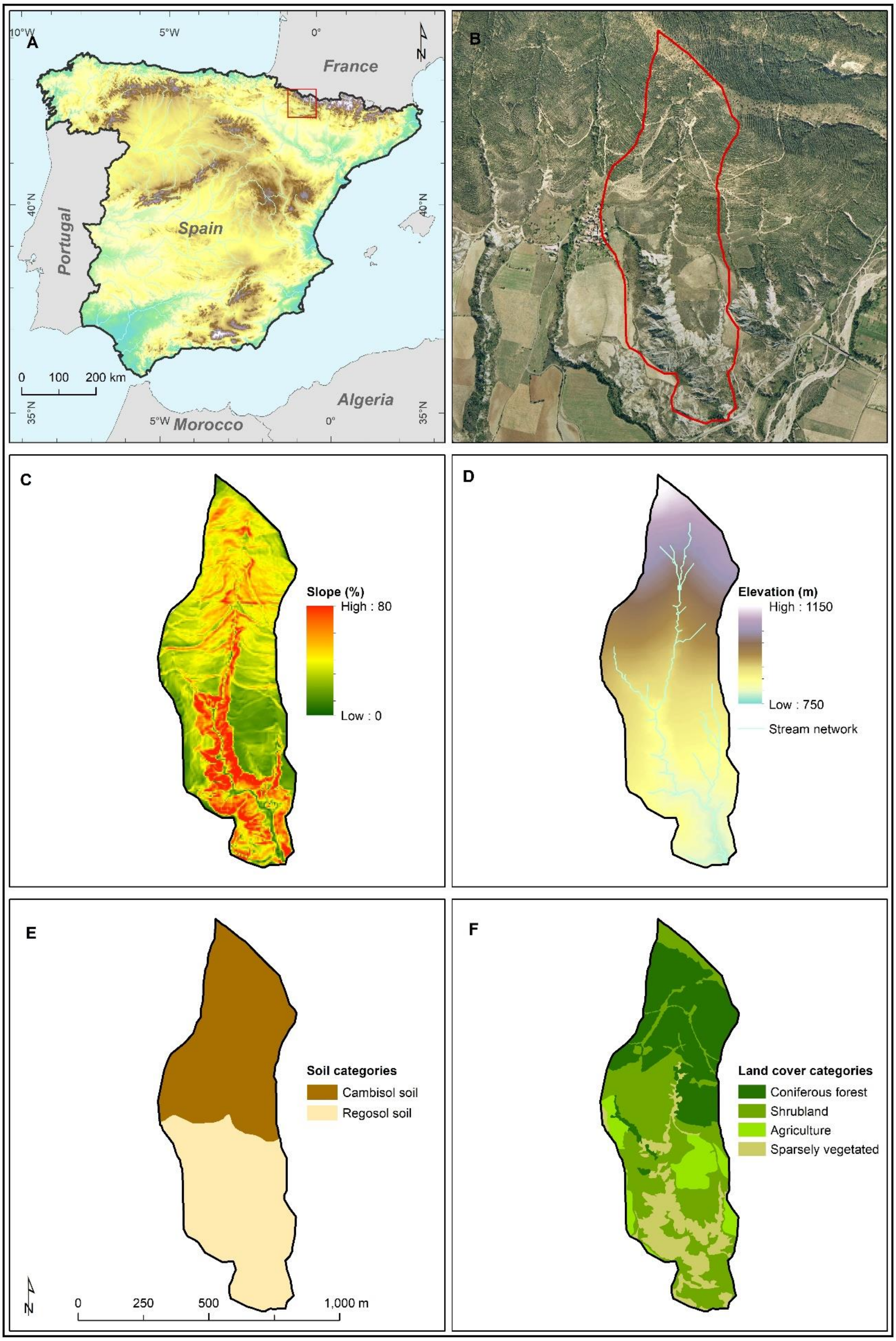

Figure 2.4. The Araguás catchment. A: Localization map, B: Aerial photography, C: Slope map, D: Elevation and stream network map, E: Soil map, F: Land cover map.

2. Materials and methods

2.1. Used information

Several datasets were acquired from different sources for the elaboration of the present PhD dissertation. The data includes geographical information, aerial photography, satellites images, climate and hydrological time series and soil samples among others. The main data sources from which the datasets are acquired include the National Spanish Meteorological Agency (AEMET), the National Center for Geographic Information (CNIG), the United States Geological Survey (USGS) and the local monitoring records from the different experimental catchments of the Pyrenean Institute of Ecology (IPE-CSIC) at Arnás, Araguás, San Salvador and the meteorological station of Esposa. The soil dataset has been collected in 2019. Table 2.1 enumerates the main datasets used and their sources.

Table 2.1. Main materials and data sources of the research studies.

Chapter	Main materials	Data sources
Chapter 3: Effects of active and passive land use management after cropland abandonment on water and vegetation dynamics in the Central Spanish Pyrenees.	◆ Daily maximum and minimum temperature and precipitation	◆ Local meteorological station in the study area (IPE-CSIC) ◆ National Spanish Agency of Meteorology (AEMET)
	◆ Observed streamflow data	◆ Local stream gauge station in the study area (IPE-CSIC)
	◆ Aerial photography (PNOA 2006)	◆ National Center for Geographic Information (CNIG)
	◆ National Territorial Observation Plan (SIOSE 2005)	
	◆ Digital Elevation Model (5 m of resolution)	
	◆ RHESSys default parameters	◆ RHESSys repository at GitHUB

Table 2.1. Main materials and data sources of the research studies (continuation)

Chapter	Main materials	Data sources
Chapter 4: Effects of vegetation succession and shrub clearing after land abandonment on the hydrological dynamics in the Central Spanish Pyrenees.	◆ Daily maximum and minimum temperature and precipitation	◆ Local meteorological station in the study area (IPE-CSIC) ◆ National Spanish Agency of Meteorology (AEMET)
	◆ Aerial photography (PNOA 2006)	◆ National Center for Geographic Information (CNIG)
	◆ National Territorial Observation Plan (SIOSE 2005)	
	◆ Digital Elevation Model (5 m of resolution)	
	◆ Landsat 5 Thematic Mapper surface reflectance 2011 (30 m resolution)	◆ The United States Geological Survey (USGS)
	◆ RHESSys default parameters	◆ RHESSys repository at GitHUB

Chapter	Main materials	Data sources
Chapter 5: Natural revegetation and afforestation in abandoned cropland areas: Hydrological trends and changes in Mediterranean mountains.	◆ Daily maximum and minimum temperature and precipitation	◆ Local meteorological station in the study area (IPE-CSIC) ◆ National Spanish Agency of Meteorology (AEMET)
	◆ Aerial photography (1957)	◆ National Center for Geographic Information (CNIG)
	◆ Digital Elevation Model (5 m of resolution)	
	◆ RHESSys default parameters	◆ RHESSys repository at GitHUB

Chapter	Main materials	Data sources
Chapter 6: Carbon sequestration and water yield tradeoffs following restoration of abandoned agricultural lands in Mediterranean mountains.	◆ Daily maximum and minimum temperature and precipitation	◆ Local meteorological station in the study area (IPE-CSIC) ◆ National Spanish Agency of Meteorology (AEMET)
	◆ Aerial photography (1957)	◆ National Center for Geographic Information (CNIG)
	◆ Digital Elevation Model (5 m of resolution)	
	◆ RHESSys default parameters	◆ RHESSys repository at GitHUB

Table 2.1 Main materials and data sources of the research studies (continuation)

Chapter	Main materials	Data sources
Chapter 7: Effects of cropland abandonment and afforestation on soil properties and redistribution in a small Mediterranean mountain catchment.	◆ 60 soil samples (52 grid samples + 8 reference samples)	◆ Field sampling ◆ Laboratory analysis
	◆ Aerial photography (1957, 1977, 2006, 2012, 2015)	◆ National Center for Geographic Information (CNIG)
	◆ Digital Elevation Model (5 m of resolution)	

2.2. RHESSys Model

The Regional Hydro-Ecologic Simulation System (RHESSys) is a process-based daily time-step ecohydrological model developed to model the effects of land cover and climate change on the ecological and hydrological processes at local to regional spatial scales. RHESSys showed strong suitability to study Global Change effects on the ecological and hydrological dynamics in Mediterranean regions (Morán-Tejeda et al., 2015; Zierl et al., 2007) and has been widely used around the world including the Spanish Pyrenees (Bart et al., 2021; Hanan et al., 2017; López-Moreno et al., 2014; Martin et al., 2017; Saksa et al., 2020; Vicente-Serrano et al., 2015; Zabalza-Martínez et al., 2018; among others).

RHESSys partitions the landscape in a hierarchical representation and associates a range of hydrological, ecosystem and microclimate processes to specific landscape objects (Tague and Band, 2004). Below is a description of the main aspects of RHESSys implementation in this PhD dissertation and how the model simulates water and carbon dynamics.

2.2.1. Hierarchical representation, inputs and outputs

The hierarchical representation of the RHESSys model is designed to enhance the understanding of within watershed processes and facilitate the environmental analysis and the aggregation of water, carbon and nutrient fluxes. Within each spatial units of model hierarchy are associated specific processes modeled by RHESSys. The finest spatial scale is the patch (usually on the order of m^2, 30 m^2 in this case) while the largest scale is the basin (on the order of km^2). Basin is a hydrologically closed area that encompasses a single stream network. Within basin level, are aggregated the different spatial units to determine the net fluxes of water, carbon and nitrogen over the entire study area. Hillslopes, are typically derived upon drainage patterns using Geographic Information Systems (GIS) and define areas draining to the same side of a stream reach. Lateral routing of surface and subsurface water are organized at hillslope level to produce streamflow. Areas of similar climate define zone level, where meteorological variables are derived and extrapolated to characterize landscape spatial variation. The smallest spatial units (patches), denote areas of similar land cover and soil moisture characteristics. Patches include multiple canopy and soil layers were vertical soil moisture and biogeochemical processes are modeled. Basin, hillslopes, zones and patches define landscape spatial heterogeneity. On the other hand, vertical aboveground vegetation layers are defined at canopy strata level where processes such as photosynthesis and transpiration are modeled. Full details on RHESSys hierarchy can be found in Tague and Band (2004).

To run RHESSys model a range of parameters are required, including climate, vegetation, topography and soils data. From these parameters a series of inputs are prepared through time series reconstructions and GIS preprocessing. After preparing model inputs a first step of soil and vegetation initialization is required (spin-up). During spin-up soil carbon and nitrogen pools, as well as vegetation parameters are initialized based on several methodologies. The selection of the spin-up approach was based on the spatial scale and the availability of the data (climate record, satellite data, etc.) in the different study areas. Following spin-up simulations, a calibration and validation step is initiated in which a set of model parameters are modified iteratively within some acceptable criteria in order to ensure a good correspondence between observed and modeled records. Once calibrated and validated, the model is used to simulate the different land cover and climate scenarios. RHESSys outputs are calculated on an hourly, daily, monthly and yearly time scales and some of them can be spatially distributed. Figure 2.5 shows the chronological order of the different steps, as well as, the main inputs and outputs. Full details on the preparation of inputs, the complete list of outputs and user guidelines can be found on RHESSys repository (https://github.com/RHESSys/RHESSys/wiki).

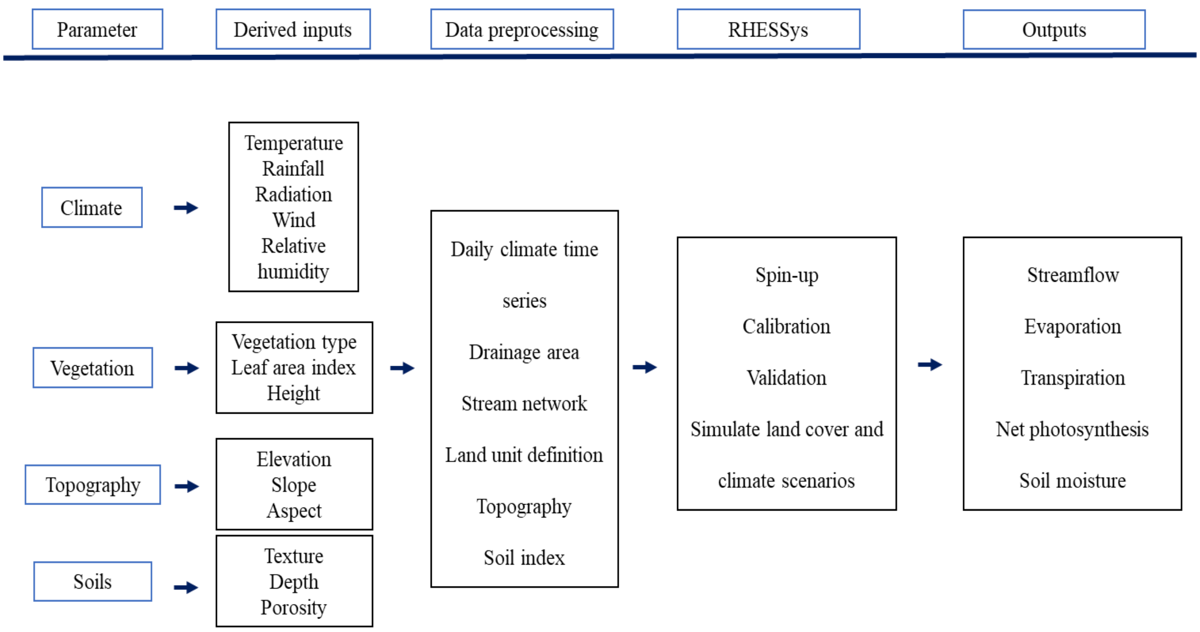

Figure 2.5. Chronological implementation of RHESSys model.

2.2.2. Hydrological cycle

Figure 2.6 shows the main water stores and hydrological processes that RHESSys accounts for in the partitioning of precipitation (water entering the unit). A three-layer model including a surface detention store, an unsaturated and saturated store is used to model vertical soil moisture fluxes. RHESSys also accounts for the water intercepted by the vegetation and litter layers and throughfall from canopy layers and snowmelt. Water is infiltrated from the surface to soil layers (unsaturated and saturated storages) where it can be exchanged through processes of drainage and capillary rise. Evapotranspiration (water exiting the unit) in RHESSys is computed using the Penman-Monteith equation (Monteith, 1965). Evaporation includes the water evaporated from the different surface storages (vegetation, litter, snow and soil), while transpiration accounts for water loss through vascular layers. On the other hand, lateral soil moisture fluxes account for the water reaching the main stream channel from surface, subsurface and deep ground water Table (Figure 2.7). Vertical and lateral soil moisture fluxes are regulated by hydraulic conductivity and porosity of soil layers. Both parameters are permitted to vary with soil depth and are computed as an exponential profile eq2.1 and eq2.2:

$$K_{sat}(z) = K_{sat0}\, e^{\left(-\frac{z}{m}\right)} \qquad \text{eq2.1}$$

$$\Phi(z) = \Phi_0\, e^{\left(-\frac{z}{p}\right)} \qquad \text{eq2.2}$$

Where $K_{sat}(z)$ and $\Phi(z)$ are respectively hydraulic conductivity and porosity at the soil depth z, K_{sat0} and Φ_0 are soil specific parameters defining, respectively, hydraulic conductivity and porosity at soil surface. m and p describe the decay of both variable respectively with depth.

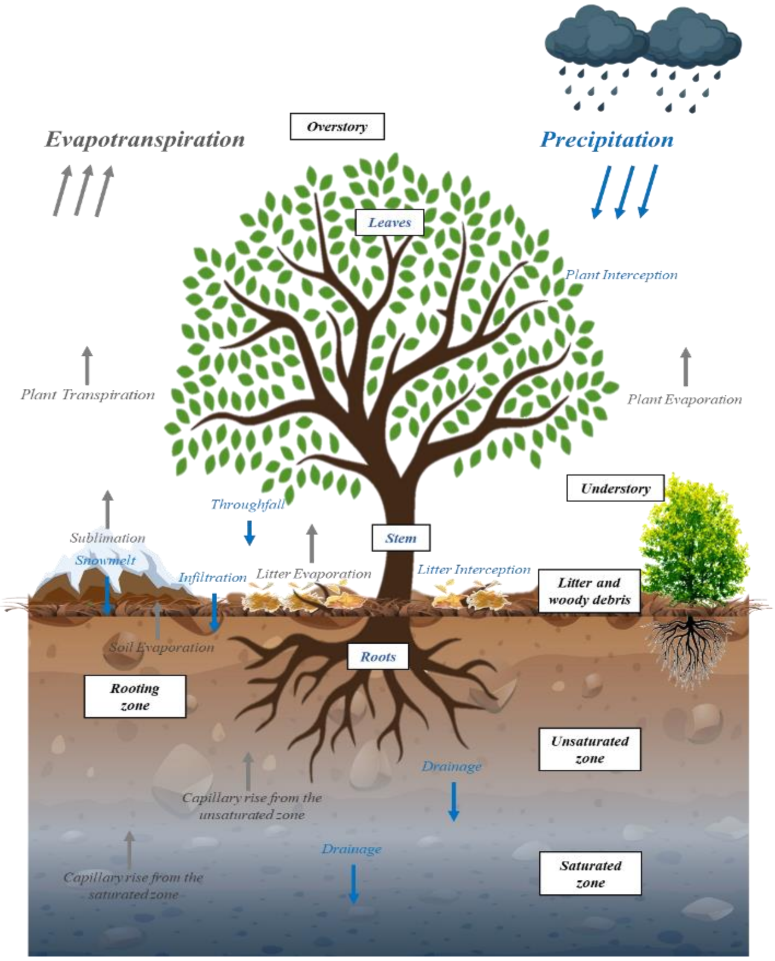

Figure 2.6. Vertical water fluxes in RHESSys model (Adapted from Janet Choate, (2010), unpublished).

Six soil parameters were calibrated on a daily scale against observed streamflow data from the Arnás catchment. *Ksat0*, *m*, *pa* (soil air entry pressure), *po* (soil pore size index), *gw1* (bypass flow) and *gw2* (deep ground water reaching main stream). A Monte Carlo based approach was used to run the model over a range of values for the six soil parameters. The set of parameters producing an optimal accordance between observed and modeled streamflow were selected and used to run the different land cover and climate scenarios. Full details on the calibration and validation processes can be found in Chapter 3. Figure 2.7 illustrates the main vertical and lateral water fluxes modeled by RHESSy and their associated calibration parameters.

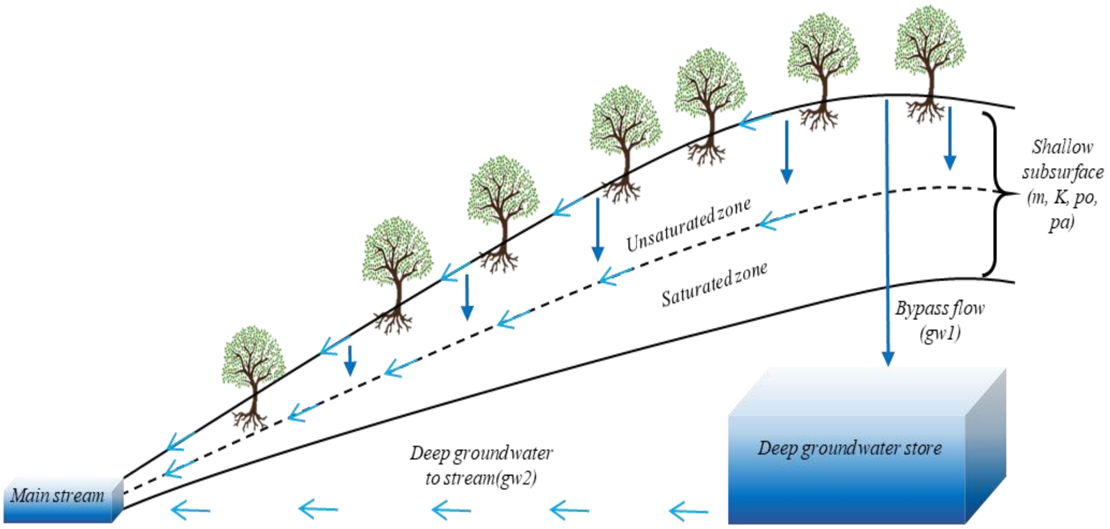

Figure 2.7. Lateral water fluxes and calibration parameters in RHESSys model. (Adapted from Janet Choate, (2010), unpublished).

2.2.3. Carbon cycle

The carbon cycle in RHESSys is directly tied to the nitrogen cycle. However, in this section we only describe the carbon cycle necessary for the understanding of the different research studies. Details on the nitrogen cycle can be found in Tague and Band (2004).

Figure 2.8 shows the main carbon stores in RHESSys model. Carbon enters the system through photosynthesis and exits through respiration. RHESSys uses the Farquhar model (Farquhar and von Caemmerer, 1982) to calculate photosynthesis and Ryan model (Ryan, 1991) to calculate respiration. For live vegetation, carbon stores are partitioned into leaves, stem and roots under tree species, while for grasses, carbon stores are restricted to fine roots and leaves. Carbon is also stored in litter, woody debris and soil layers. The respiration process accounts for carbon release from the decomposition of organic material and plant respiration. The decomposition of organic material occurs in the litter and soil layers.

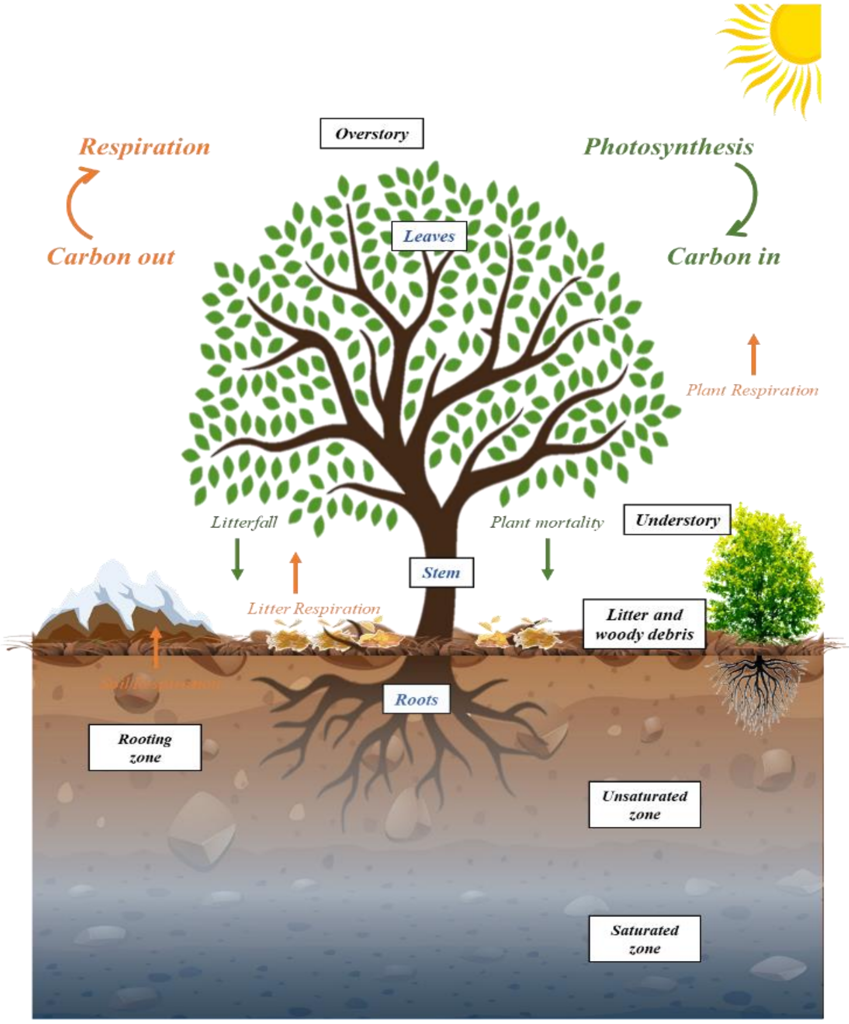

Figure 2.8. Carbon cycle in RHESSys model (Adapted from Janet Choate, (2010), unpublished).

2.3. Soil sampling, preparation and analysis

To study the effects of active and passive restoration after land abandonment on physico-chemical soil properties and soil redistribution rates, an approach based on soil sampling, laboratory analysis and fallout ^{137}Cs measurement was designed in the Araguás catchment. Fallout ^{137}Cs has proved to be a reliable technique for quantifying soil redistribution rates (Alewell et al., 2014; Gaspar et al., 2013; Mabit et al., 2008; Sadiki et al., 2007). The technique has been widely used to investigate mid-term soil loss in the Mediterranean region including the Pyrenees (Alatorre et al., 2012; Gaspar et al., 2019; Lizaga et al., 2018; Navas et al., 2005; Porto et al., 2009; Quijano et al., 2016). Figure 2.9 shows the use of fallout ^{137}Cs in fingerprinting and soil redistribution studies.

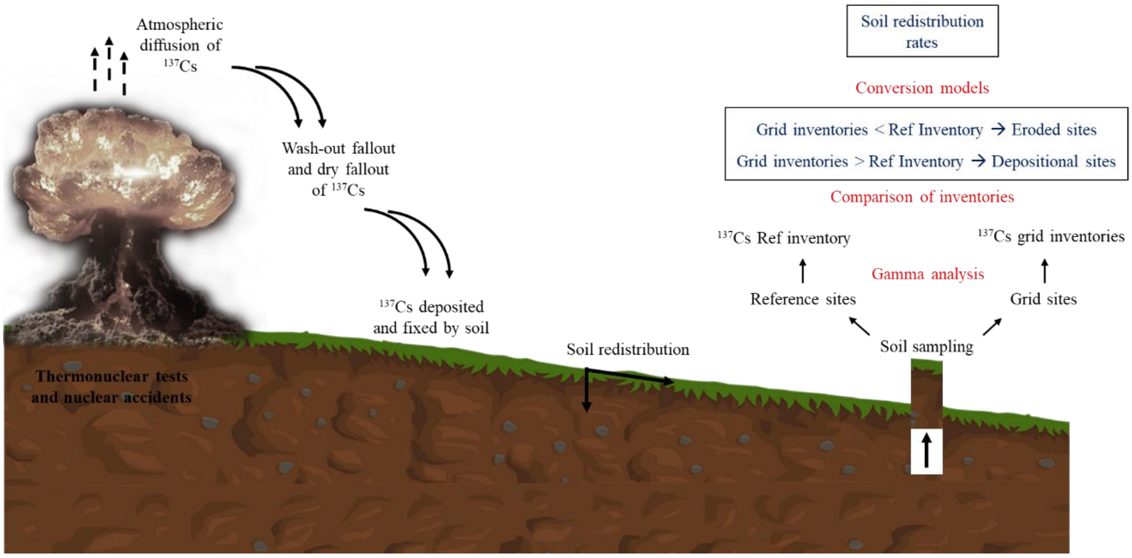

Figure 2.9. Methodology of fallout ^{137}Cs in soil redistribution studies. (Adapted from Gaspar et al., 2011).

A total of 52 soil samples were collected in Araguás during 2019. Soil sampling was distributed across the catchment using a 100 x 100 m grid. From the first 30 to 40 cm depth, two replicates of soil were collected in each sampling point using a steel core (Figure 2.10). Further, 12 soil cores were collected in flat undisturbed locations in order to establish a reference inventory. The reference samples were collected using steel core tubes in the first 40 to 45 cm soil depth (Figure 2.10). The reference soil cores were sectioned in 5 cm intervals to study the vertical distribution of ^{137}Cs. Three references samples were excluded from the analysis based on their ^{137}Cs vertical distribution (disturbed samples).

Soil samples were open and air-dried in the laboratory, homogenized and sieved to ≤ 2 mm. The two replicates of grid samples were mixed before analysis while, the sectioned intervals of the reference samples were analyzed separately. The following physico-chemical soil properties were determined in the laboratory at the Pyrenean Institute of Ecology (IPE-CSIC) and at the Experimental Station of Aula Dei (EEAD-CSIC): Electrical conductivity (EC), pH, soil texture, total carbon (TC), soil organic carbon (SOC), total nitrogen (TN), organic matter (OM), calcium carbonates ($CaCO_3$) and ^{137}Cs mass activity. C:N ratio and stocks of SOC, TN and ^{137}Cs were calculated in each grid sampling point and reference intervals. Soil redistribution rates (Mg ha^{-1} yr^{-1}) were derived from ^{137}Cs inventories applying the conversion model reported by Soto and Navas

(2004, 2008) for uncultivated and cultivated soils. Full details on soil analysis can be found in Chapter 7.

Figure 2.10. Soil sampling, preparation and analysis. A, B: Soil sampling using steel cores. C: Organizing soil samples before laboratory preparation. D: Extracting soil samples from cores. E: Sectioning references samples. F: Air drying. G: Laboratory analysis at the Pyrenean Institute of Ecology (IPE-CSIC).

2.4. Research methodology

The research methodology of the present PhD dissertation consisted of identifying the main research question in relation to the management of abandoned cropland areas in the Central Spanish Pyrenees. Each one of the research questions is associated with specific objectives and research hypothesis to be evaluated. Once study objectives and research hypothesis are identified, the necessary datasets for the analysis are selected and acquired. The data for the present PhD dissertation was acquired in original format from the different sources which implicated a necessary initial process of preprocessing and adaptation of the original data to the specific requirements of each one of the studies. For the soil dataset, full detail on the sampling design, soil preparation and data analysis are presented in Chapter 7. After preparation of soil dataset and preprocessing of climate and hydrological record, data inputs are generated for the different analysis. Analyzed the data, results are discussed and conclusions are made on the management of abandoned cropland areas (Figure 2.11).

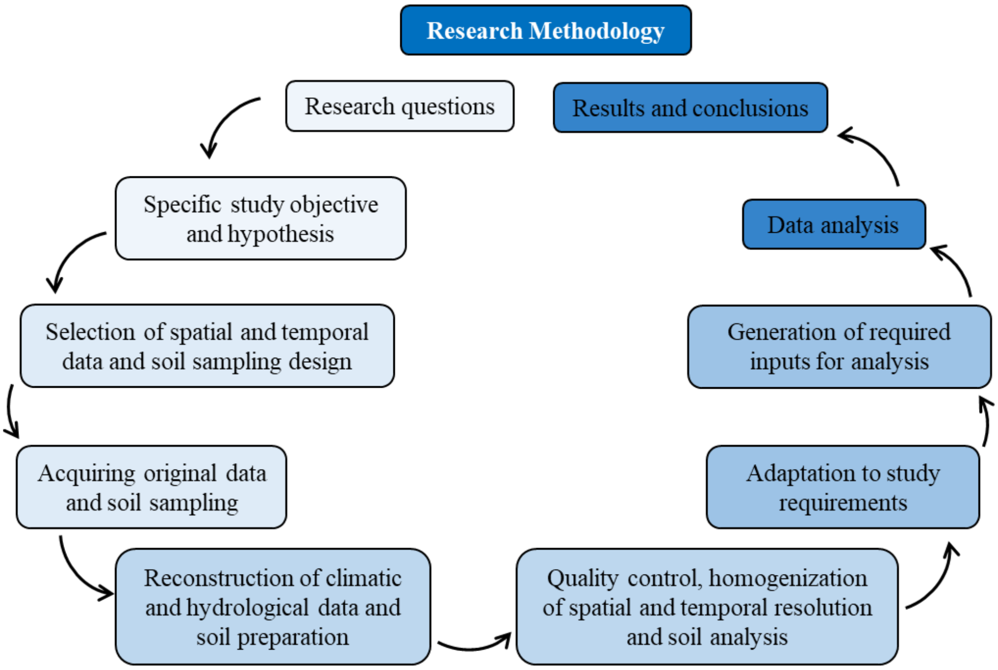

Figure 2.11. Conceptual model of the used research methodology.

2.4.1. Preprocessing of climatic and hydrological times series

Climate time series have been subject to a rigorous preprocessing following approaches of Vicente-Serrano et al. (2010) and El Kenawy et al. (2012). The preprocessing includes identifying and removing data irregularities, homogenization, and gap-filling. For gap filling and reconstruction, we used data from several meteorological stations from AEMET and the IPE-CSIC. The used meteorological stations were selected in a radium of 10 km and at a similar elevation from the subject catchment.

For the hydrological data (observed streamflow), we compiled data from four gauging outlets at Arnás, Araguás, Araguás afforestation and San Salvador catchments. The data was acquired at different temporal resolutions going from 5 min to 30 min. Similar to climatic time series, a process of quality control and homogenization of hydrologic data was conducted. However, we did not fill streamflow data gaps and decided to use the preprocessed record converted to daily scale in the calibration and validation of RHESSys model.

2.4.2. Statistical analysis

This section describes briefly the different statistical analysis used in the research contributions. A 95% confidence interval to test the significance of the p-value of the different tests was used every time it was necessary.

For the calibration and validation of RHESSys model in the Arnás catchment, we used the Nash-Sutcliffe Efficiency (NSE) and the Percent Bias (PBIAS) tests. NSE (Nash and Sutcliffe, 1970) is a normalized statistic widely used to assess the goodness of hydrological models (McCuen et al., 2006), it determines the ratio of the residual variance of the modeled data compared to the variance of the observed data. NSE is computed as follows:

$$\mathrm{NSE} = 1- \left[\frac{\sum_{i=1}^{n}\left(Y_i^{obs}-Y_i^{sim}\right)^2}{\sum_{i=1}^{n}\left(Y_i^{obs}-Y^{mean}\right)^2}\right]$$

PBIAS quantifies the tendency of the modeled data to under or overestimate the observed data. PBIAS is computed as follows:

$$\mathrm{PBIAS} = \left[\frac{\sum_{i=1}^{n}\left(Y_i^{obs}-Y_i^{sim}\right)*100}{\sum_{i=1}^{n}\left(Y_i^{obs}\right)}\right]$$

We followed Moriasi et al. (2007) for the evaluation of model results. Full details on the calibration and validation of RHESSys model can be found in Chapter 3.

Pearson's R is used for correlation analysis and the Shapiro-Wilk test (Shapiro and Wilk, 1965) was used to verify the normality of the distributions. Parametric tests were used to compare normally distributed samples and non-parametric tests were used if normality is not met.

We used the Analysis of variance ANOVA to compare differences between groups and the Tukey's Honestly Significant Difference test (TukeyHSD) (Tukey, 1949) to identify the significantly different groups. With non-normally distributed groups we used the Mann-Whitney Rank Sum test (Mann and Whitney, 1947) to identify significantly different groups.

In Chapter 5, the non-parametric Mann-Kendall trend test (Kendall, 1975) was used to identify statistically significant trends and the non-parametric Sen´s slope test (Sen, 1968) was used to estimate the magnitude of change.

In Chapter 7, to determine the degree to which the different land uses and land covers in the Araguás catchment affect soil properties and ^{137}Cs mass activity, soil samples were analyzed using Principal Component Analysis (PCA). Variables accounting for most of the variation on PC1 and PC2 and the combination of associated variables were identified.

2.4.3. Dealing with model uncertainty

Uncertainty is an intrinsic property of models (Li et al., 2015; Orth et al., 2015). Hydrological models simplify ecosystem processes to simulate its functioning, test change scenarios and support decision making. This simplification often uses conceptual parametrization to deal with the inability to represent complex ecosystem processes. Model structure, parameters, calibration and inputs are all possible sources of uncertainty (Moges et al., 2020). In this PhD dissertation, we tried to minimize the effect of model uncertainty on our results and conclusions. We tried to deal with model uncertainty in three different steps of the work. i) During model setup in the study areas. The Monte Carlo method was used for the calibration of RHESSys model, which relies on repeated random sampling of model parameters. We selected the nine best set of soil parameters, from the calibration and validation, to run RHESSys under the different management and climate scenarios. On the other hand, to initialize vegetation parameters we used a sensitivity analysis to identify the most sensitive vegetation physiological variables. Once identified, values for these variables are defined based on scientific literature in the Central Spanish Pyrenees. ii) During the definition of scenarios. Several land cover and climate scenarios are defined and tested (36 scenarios) to make sure the results give a good general idea on the different trajectories. iii) During the discussion of results. Results and discussion of the different studies consider model uncertainty and emphasize the necessity for deep analysis of ecosystem processes before adopting land management plans in the Central Spanish Pyrenees.

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Chapter 3. Effects of active and passive land use management after cropland abandonment on water and vegetation dynamics in the Central Spanish Pyrenees

This Chapter studies the effects of passive and active management plans in abandoned cropland areas in the Central Spanish Pyrenees on the hydrological and vegetation dynamics. In this study RHESSys model was calibrated and validated using observed streamflow data from the Arnás catchment (2.84 km^2). Five land cover scenarios coupled with two climate scenarios were simulated in static and dynamic modes. Changes to streamflow, evapotranspiration and soil saturation deficit, as well as plant biomass are studied.

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Abstract

The Mediterranean mountains have been subject to significant land abandonment process during the second half of the 20th century. The subsequent natural revegetation following abandonment in rural areas has been widely documented to have substantial implications on the hydrological cycle and the vegetation. The Spanish Pyrenees are one of the most affected areas by these land transformations which could threaten their importance for water supply and agricultural activities in the downstream lowland areas. Land managers as well as scientists around the world have taken different positions on how to deal with these land use changes. Some are in favor of active management (AM) (i.e. density reduction) while others are supporting passive management (PM) (letting the process of revegetation continue). This study aims to investigate the implication of AM and PM on hydrological and vegetation dynamics under different climate trajectories in a representative abandoned cropland catchment in the Central Spanish Pyrenees. A coupled ecohydrologic model is used to estimate the post management response of streamflow (STR), evapotranspiration (ET), soil saturation deficit (SD) and plant carbon (PC) following shrub clearing. Clearing increased annual STR by 16%, while ET and SD decreased by around -9% and -6% respectively during the first year after management with changes to monthly flows. These changes to water regimes may be even higher in wetter years. Over a 10-years period of vegetation recovery annual STR increased between 7.1% and 24.2%, while annual ET and SD decreased between -2.6% to -8.7% and -2.7% to -6% respectively due to shrub clearing, with the highest changes occurring in the first three years of AM. On the effect of climate change, our results show that a 2 ºC increase in temperature could reduce AM effects on water regimes and accelerate the recovery of PC given averaged rainfall conditions.

1. Introduction

Mountain areas constitute the main source of water resources in Mediterranean environments (Viviroli and Weingartner, 2004; De Jong et al., 2008; García-Ruiz et al., 2011), supplying the lowland areas (Cudennec et al., 2007), where water demand has increased in the last decades due to the expansion of irrigated areas (Jlassi et al., 2016), population increase (FAO, 2018; World Bank, 2015), and the concentration of industrial and service activities (Rico-Amorós et al., 2009). While a greater water demand occurs in the lowlands, a decrease in water supply from higher elevations has been observed in the Mediterranean region, and has been linked to increases in temperature and the atmospheric water demand (El-Kenawy et al., 2012), greater rainfall irregularity (García-Barrón et al., 2011) and revegetation processes (Sluiter and de Jong, 2007; García-Ruiz and Lana-Renault, 2011). These socioeconomic, climatic and water demand changes are estimated to increase the rate of global population living under water stress conditions (Arnell, 1999; Vörösmarty et al., 2000).

The fifth report of the Intergovernmental Panel on Climate Change (IPCC, 2013) have predicted an increase of 1.5 to 2 °C of temperature of the globe by the end of this century. Under the Paris deal for climate change, nearly 200 countries have agreed to hold global temperature rise below 2 ºC above preindustrial levels (Paris Agreement, 2015). Nonetheless, the accomplishment of this agreement still debatable and more efforts are needed to its fulfilment. In a study of climate change projections for the Mediterranean region, Giorgi and Lionello (2008) estimated at least a 2 ºC change in surface air temperature between 1961-1990 and 2070-2100. These climate changes are reported to have significant effects on runoff (Beguería et al., 2003; Christensen et al., 2004; López-Moreno et al., 2014a) particularly in the Mediterranean basin as one of the region's most sensitive to climate change (Giorgi, 2006).

The impacts of climate on runoff in Mediterranean may be intensified by land use changes that have occurred in this region for decades. During the 20th century, Mediterranean mountain areas were affected by the abandonment of cultivated fields (MacDonald et al., 2000) and subsequent plant recolonization, either though natural revegetation or accelerated by afforestation (Lasanta et al., 2005; Tasser et al., 2007). The abandonment of cultivated areas and the subsequent revegetation process are particularly striking in Mediterranean mountain areas, including the Pyrenees (García-Ruiz and Lana-Renault, 2011). One of the most relevant impacts of the combination of climate and land use change, is the significant decrease in river flows (Beguería et al., 2003; López-Moreno et al., 2008) and runoff coefficients (García-Ruiz et al., 2008; López-Moreno et al., 2011). Models of future climate and land use change suggest that land abandonment and revegetation processes will continue (Verburg and Overmars, 2009; Keenleyside and Tucker, 2010), which will further reduce water resources in the Mediterranean basin. In recent model-based studies, López-Moreno et al. (2014a, 2014b) estimated that in the Central Spanish Pyrenees, river flows will decrease by the middle of the 21st century between 19% and 32% (depending on the analysed basin) in a moderate climate change scenario (an average annual temperature increase of 1.8 ºC and 10% of annual precipitation decrease) and unmanaged vegetation advance (shrub colonization of some pastures and shrub substitution by forest in some areas).

Land abandonment and revegetation processes produce other negative impacts: an increase of fire risk, degradation of cultural landscapes, reduction of biodiversity and eco-diversity, reduction of natural resources (pastures, wood, forest by-products), loss of productive lands, and reduction of environmental and leisure services (Conti and Fagarazzi, 2004; Sitzia et al., 2010; Lasanta et al., 2015; Romero-Díaz et al., 2019). These negative impacts of land abandonment have made the management of abandoned lands a matter of great interest for scientists and land managers (Gellrich et al., 2007; Lasanta, 2019). Some support a passive management (PM), letting the natural revegetation process continue thinking that the renaturalisation of the landscape provides important environmental benefits (Navarro and Pereira, 2012; Ceausu et al., 2015), while others defend the strategy of intervening in the territory (active management, AM) to control the negative effects (Conti and Fagarazzi, 2004; Lasanta et al., 2015; Nogués-Bravo et al., 2016). Shrub clearing in abandoned croplands is an action carried out by some Spanish regional administrations, with financial support from the European Union, to promote the generation of pastures and reduce forest fires (Lasanta et al., 2018). The AM strategy in La Rioja community has started in 1986 and since then an important decrease has been recorded in the number of forest fires, as well as, the burnt area in the region (Lasanta et al., 2013, 2018).

The hydrological implications of vegetation change following abandonment have been extensively examined worldwide (Bosch and Hewlett, 1982; García-Ruiz and Lana-Renault, 2011). Haria and Price (2000) have noted a direct effect of plant density increase on rainfall partitioning after cropland abandonment. Post-abandonment plant density increase is generally associated with a decrease of runoff (Bosch and Hewlett, 1982; García-Ruiz et al., 2008; García-Ruiz and Lana-Renault, 2011) and a decrease in infiltration due to higher interception rates (Lucas-Borja et al., 2019; Nunes et al., 2011). More generally studies of the effect of forest management in catchment scale have revealed that density reduction due to deforestation or forest thinning will generally result in an increase in runoff coefficients (Andréassian, 2004; Simonit et al., 2015; Yurtseven et al., 2018), although studies that focus on Mediterranean environments often show more muted responses due to compensating high rates of understory and surface evaporation (Tague et al., 2019). While effects of vegetation density changes on the hydrological cycle have been well studied, studies that specifically focus on the hydrologic effects of PM and AM strategies after cropland abandonment are few. At small experimental plots in the Central Spanish Pyrenees, Nadal-Romero et al. (2013) have shown that the runoff coefficients in grassland areas are higher than in shrubland areas. López-Vicente et al. (2011) evaluated the possible effects of land use change after cropland abandonment including a shrub clearing scenario on sediment yield in the Central Spanish Pyrenees. However, studies of the effects of AM strategies on hydrological dynamics after cropland abandonment on a hillslope to watershed scales in Mediterranean systems are still needed (Dung et al., 2012).

One useful approach is to combine experimental catchment data with hydrologic models (Kepner et al., 2012). The results obtained in small experimental catchments can be used to calibrate and validate hydrological models (Tetzlaff et al., 2017), as well as, to predict hydrological functioning under different environmental scenarios. In Mediterranean environments, the importance of water resources has generated a significant increase in the number of experimental catchments with different land uses/land covers during the last decades (García-Ruiz, 2010; Latron and Lana-Renault, 2018). One of these catchments, is the Arnás catchment, located in the Central Spanish Pyrenees. Arnás is a small catchment cultivated until the middle of the 20th century and progressively abandoned during the 60ies and 70ies. After its abandonment, the Arnás catchment has been subject to an intense process of revegetation. These land use changes make of the catchment, representative of the global land abandonment and revegetation processes that affected the Mediterranean mountains during the past decades. The Arnás catchment, has been monitored for more than two decades (from the 1996 up to the present) by researchers from the Pyrenean Institute of Ecology (IPE-CSIC). This long-term monitoring of the post-revegetation period has made of this catchment a widely-appreciated catchment to study soil erosion and sediment delivery (Navas et al., 2005; Alatorre et al., 2012; Nadal-Romero et al., 2012), hydrological dynamics (Lana-Renault et al., 2007, 2011) and modelling soil distribution (Beguería et al., 2013).

Process-based catchment scale models have been a very useful tool to a better understanding of hydrological and ecological processes in the headwaters of the Spanish Pyrenees (Morán-Tejeda et al., 2015). The Regional Hydro-Ecological Simulation System (RHESSys) (Tague and Band, 2004) is one of the numerous models widely used during the last years to assess climate and land use changes (Godsey et al., 2014; Bart et al., 2016; Peng et al., 2016; Bart and Tague, 2017, among others). RHESSys has shown to be effective for hydrologic (López-Moreno, 2014b; Morán-Tejeda et al., 2015) and ecological studies (Vicente-Serrano et al., 2015) in the Spanish Pyrenees. For this reason, this study will use RHESSys to assess the effect of land use changes after cropland abandonment under different land management strategies (PM and AM) and climate change scenarios (2 ºC temperature increase) on water fluxes and vegetation dynamics.

This paper aims to gain more insight into the potential role of AM in land abandonment areas by exploring the following central research question: How do shrub clearing management and expected climate warming (2 ºC warming) affect water yield in Mediterranean humid mountain areas? This leads to the following research hypotheses: (i) shrub clearing increases streamflow (STR), decreases evapotranspiration (ET) and soil saturation deficit (SD) in mountain areas, while an increase in temperature decreases the water yield and increases evapotranspiration and soil saturation deficit, and (ii) the effects over 10 years-period after shrub clearing is diminished due to re-growth after clearing.

For this, we used the ecohydrological RHESSys model in the agricultural abandoned experimental catchment Arnás to compare the effect of AM and PM strategies on water and vegetation dynamics.

2. Data and Methods

2.1. Study area

The Arnás catchment is a 0.284 km^2 catchment located in the Borau valley in the Central Spanish Pyrenees with altitudes between 900-1400 meters above sea level (m.a.s.l)

(Figure 3.1). The average annual precipitation is about 900 mm, which is uniformly distributed during the year. The average annual temperature is around 10 ºC under a sub Mediterranean climate with Atlantic influence. Although, the most part of rainfall occurs between autumn and spring, summer rainstorms are also frequent (García-Ruiz et al., 2000, 2005). In addition, and due to a 0 ºC isotherm of the cold season located at 1600 m.a.s.l. in the region, snowfalls which are relatively frequent in the December-April period, remain just for few days on the soils before melting (Beguería et al., 2003).

The geology of the catchment is comprised of Eocene flysch, alternating with thin layers of marls and sandstones. The ravine has a W-E orientation that resulted in a high contrast between a steeper south slope and the north-facing one. On the steep south slope (0.5 m m^{-1} of gradient), soils are poorly developed, strongly eroded and carbonate rich Regosols, although Calcaric Cambisols have been identified in some bench terraced fields (García-Ruiz et al., 2005; Navas et al., 2005). On the north-facing slope, soils are more developed and organic matter rich (0.28 m m^{-1} of gradient) slope while stagnic conditions dominate at the valley bottom (Supplementary Figure 3.1). The Arnás catchment was cultivated until the middle of the 20th century then progressively abandoned. The main cultivated crops were cereals in non-terraced fields covering most of the catchment area. After its abandonment during the 60ies and 70ies, a natural revegetation process occurred as result of the PM strategy of the catchment (no forest or land management practices have been carried out in the study area since its abandonment). *Pinus sylvestris* and *Quercus faginea* have partially colonized the highest slopes. *Buxus sempervirens*, *Genista scorpius* and *Rosa gr. canina* have covered most of the south-facing slopes, with *Echynospartum horridum* dominated the north-facing one (Figure 3.1 and Supplementary Figure 3.2). At the lower part of the catchment, a gauging station and a weather station were installed in 1996. Precipitation, water level, turbidity and conductivity were measured every 5 minutes (for more details see Lana-Renault et al., 2007).

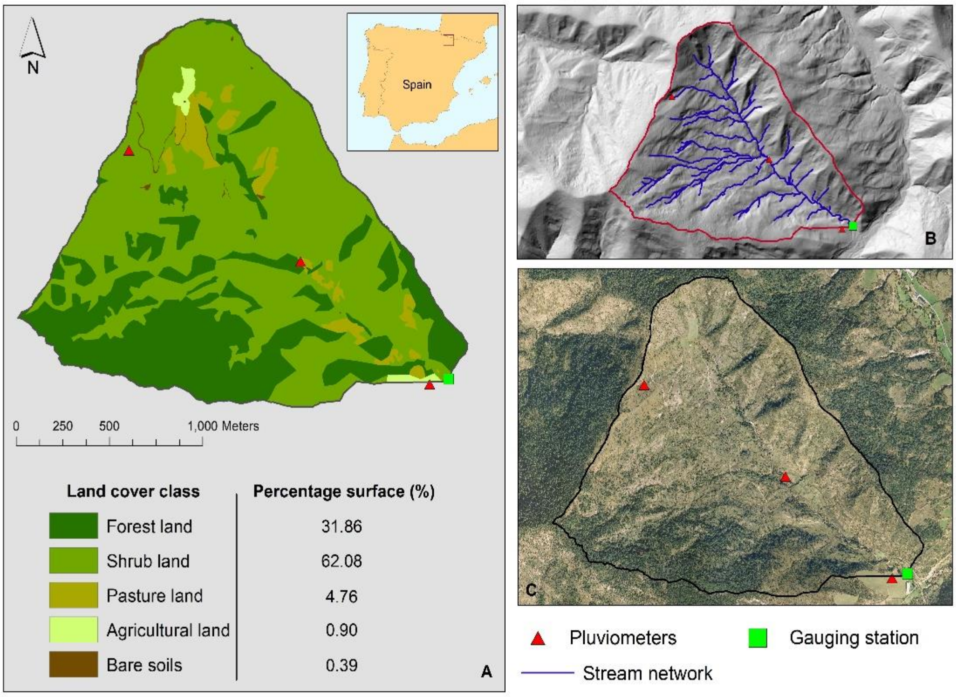

Figure 3.1. The Arnás experimental catchment: A: Simplified land covers and their correspondent percentages of the study area, B: Stream network and instrumentation and C: Aerial photography of the Arnás catchment (PNOA,2006).

2.2. RHESSys Model

The Regional Ecohydrological Simulation System (RHESSys) (Tague and Band, 2004) model was used to conduct all the simulations in this study. RHESSys is a spatially-distributed and dynamic ecohydrological model that has been rigorously tested in different environments including the Spanish Pyrenees (López-Moreno et al., 2014a; Vicente-Serrano et al., 2015; Morán-Tejeda et al., 2015; Saksa et al., 2017; McDonnell et al., 2018; Anderson et al., 2018; Zabalza et al., 2018). RHESSys was developed to study land cover and climate changes effects on the hydrological and ecological processes (Tague and Band, 2004).

RHESSys model establish a hierarchical spatial structure in partitioning the landscape (Tague and Band, 2004). This hierarchical structure permits modeling hydrologic and carbon cycling at a very fine spatial scale (the patch, 30 m in our case), organizing climate drivers within zone level and modeling lateral routing of surface and subsurface flows within hillslope level (Bart et al., 2016).

As well as, vertical hydrologic fluxes (interception, infiltration, canopy transpiration, soil and litter evaporation and groundwater recharge) and lateral redistribution nutrient, moisture and streamflow production, RHESSys includes also mechanistic representation of soils, carbon and nitrogen cycling (Tague and Band, 2004). All this makes of RHESSys model able to represent both plot scale processes (within patch level) and spatial heterogeneity in plot scale environments along hillslopes, accounting for variation in solar radiation, soil water storage and interactions between patches through downslope moisture redistribution. The ability to merge local processes and spatial heterogeneity, enhances the suitability of the model to investigate land use changes effects on hydrological and ecological dynamics. In the same context, Morán-Tejeda et al. (2015) pointed out a high sensitivity of RHESSys compared to SWAT to model land use changes effects in the Pyrenees.

The model continues to evolve with recent updates to nitrogen cycling (Hanan et al., 2016) and vegetation carbon cycling components of the model (Garcia et al., 2016). For this study RHESSys 5.20 .1 was used)

2.2.1. RHESSys Spin-up

Spin-up is a preliminary simulation to initialize the model and allow vegetation and soil carbon and nitrogen stores to reach a state of equilibrium. In this case, soil, litter, and vegetation state variables were initialized at zero value and the model was run dynamically until vegetation and soil conditions reached similar conditions as the Arnás catchment. The five land cover categories were parametrized using plant functional types for forests, shrubs and grasses. More details on RHESSys model could be found in Tague and Band (2004) and Garcia et al. (2016) and its parametrization in White et al. (2000) and the RHESSys website https://github.com/RHESSys/RHESSys/wiki.

2.2.2. Climatic and spatial datasets

Climatic data: Daily maximum and minimum temperatures and rainfall are the three required meteorological variables to run RHESSys model. Daily time series of the three variables were constructed considering all the available historical data from the meteorological station installed in the catchment (1999 to the present) (Figure 3.1). Daily estimates were calculated from the 5-minutal data record. Later, a process of data homogenization, correction and gap filling following Vicente-Serrano et al. (2010) and El-Kenawy et al. (2012) was conducted using data from the 11 available meteorological stations from the National Meteorological Agency (AEMET) in a radius of 10 km.

Land cover map: Using the 2006 aerial photography of the study area from the National Plan of Aerial Photography (PNOA) and the Third National Forest Inventory (2007) and field work, we generated a land cover map for the study area with 5 land cover categories (Figure 3.1). Shrubs are the main vegetation cover in the Arnás catchment with 62% of its surface, however forest (coniferous forest of pines mainly) covers almost 32%. Agricultural land only persisted on around 5% of the Arnás catchment which reflects the strong effect of shrub colonization on the highly cultivated area in the past. The new land cover map has been used to define vegetation type for the spin-up period as well as the calibration and the validation.

Soil map: Four soil types have been considered in the analysis based on the soil map by Seeger et al. (2005) and the soil sampling data by Navas et al. (2005) (Supplementary Figure 3.1). Cambisols (30.2%), Calcisols (6.0%) (relatively developed and deep soils) represents around 36% of the catchment surface and they are mainly located in the north-facing part. Regosols (59.3%) and Stagnic soils (4.5%) (low drained and relatively shallow soils) represents 63% of the catchment surface existing mainly on the south-facing part and the valley bottom. The new generated map was used to define soil types for the different simulations. More details on the required soil parameters could be found in https://github.com/RHESSys/RHESSys/wiki/Parameter-Definition-Files.

2.2.3. RHESSys Calibration and Validation

Despite of the long dataset recorded in the Arnás catchment, there are significant gaps in the data. Given the limited continuity of the daily observed streamflow data, the 01/01/2004-31/12/2005 and the 01/01/2007-30/09/2008 periods were selected for calibration and validation respectively as the most complete range of data on a daily time scale. In this study six soil parameters were calibrated: the decay of saturated hydraulic conductivity with depth **m**, saturated soil hydraulic conductivity at the surface **k**, soil pore size index **po**, soil air entry pressure **pa**, infiltration through macropores **gw1**, lateral water fluxes from the hillslope to the main channel **gw2**. A Monte Carlo approach is used to perform the calibration; it consists on running the model for 1600 simulations with 1600 random set of parameters. Performance was based on the Nash-Sutcliffe Efficiency (NSE, eq1) and the percent bias (PBIAS, eq2), the 9-top set of parameters (Supplementary Table 3.1) have been selected for model simulations under the different scenarios. The average of these simulations has been considered.

$$NSE = 1- \left[\frac{\sum_{i=1}^{n}\left(Y_i^{obs}-Y_i^{sim}\right)^2}{\sum_{i=1}^{n}\left(Y_i^{obs}-Y^{mean}\right)^2}\right] \qquad \text{(eq1)}$$

$$PBIAS = \left[\frac{\sum_{i=1}^{n}\left(Y_i^{obs}-Y_i^{sim}\right)*100}{\sum_{i=1}^{n}\left(Y_i^{obs}\right)}\right] \qquad \text{(eq2)}$$

For both the calibration and the validation period RHESSys model has shown satisfactory performance in reproducing observed daily streamflow data according to metrics of acceptability identified by Moriasi et al. (2007) (Supplementary Table 3.1). Supplementary Figure 3.3 shows, on a daily basis, how the modeled streamflow fits the observed daily runoff in the Arnás catchment. The model has generally reproduced well the evolution of the daily observed streamflows and particularly baseflows and recession periods. On the other hand, the underestimation of some modeled peakflows could be related to an overestimation of the observed flow due to higher sensitivity of gauging stations during high flow events. Nevertheless, Bias under both calibration and validation periods shows that the overall water balance controls appear to be well represented.

2.3. Scenarios

2.3.1. Passive versus active management strategies

This study compares a scenario of active intervention in the abandoned croplands mainly by shrub clearing with a no intervention scenario where there is continuity in forest succession and shrub colonization. For this, we defined two management strategies: PM and AM.

For the PM scenario, the land cover map generated as described above (Figure 3.1) is used to represent natural vegetation colonization. The map represents vegetation condition in the catchment in 2006/2007 approximately forty years after land abandonment. After being abandoned, no AM plans have been carried out in the Arnás catchment. More than 60% of the catchment is covered by shrubs and abandoned pastures however forests cover nearly 35% of the surface (Supplementary Figure 3.2).

The theoretical AM scenario was defined to represent proposed governmental management strategies of shrub clearing, similar to those carried out during the last decades in La Rioja community in the north-western area of Spain. The strategy has been mainly established as a preventive-tool to decrease forest fires and promote the generation of pasture for extensive livestock farmers in the region (see Lasanta et al., 2013)

We used the map of 2006 land cover and a 5 m digital elevation model to limit the target areas for shrub clearing. Areas with more than 30% of slopes, as well as, 5 m from the ravine have been excluded to avoid increasing soil erosion rates (following the established criterion). Only shrub areas and the abandoned pastures have been considered for clearing.

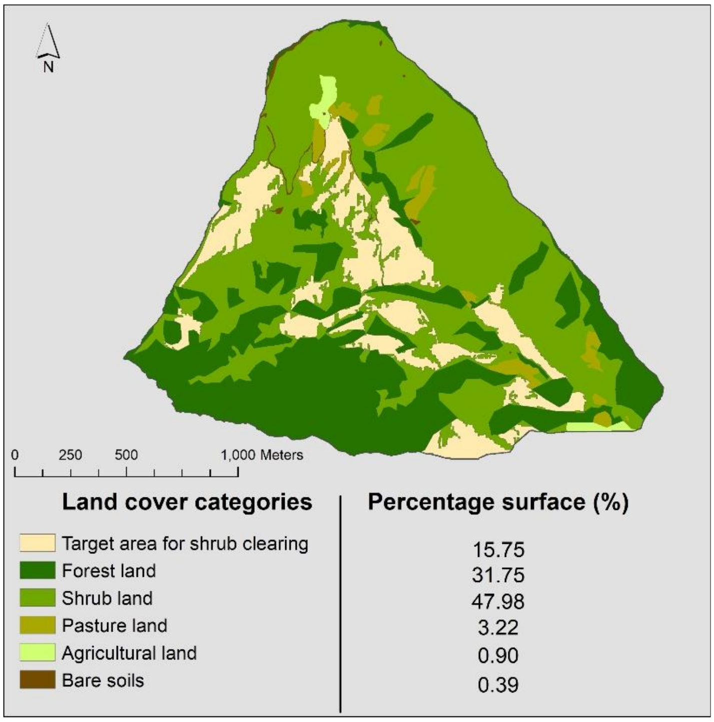

Figure 3.2. Land cover map of the Arnás catchment under the theoretical active management scenario (shrub clearing) and their correspondent percentage of the study area.

Figure 3.2 shows the Arnás catchment under the AM scenario. Target areas for intervention represents 15.75% (43.52 ha) of the study area. Most of these areas are located in north-facing slopes, due to the steeper slopes characterizing the south-facing part of the catchment. Usually the boundaries of the abandoned croplands target for shrub clearing are conserved to protect and feed the fauna (Lasanta et al., 2013). For this and with the aim of comparing different clearing intensities we defined four shrub clearing intensities based on the percentage of the above ground vegetation removed from the target area 95%, 75%, 50% and 25% (41.34 ha, 32.64 ha, 21,76 ha and 10.88 ha respectively). Although, clearing intensities depends also on other criterions when performing shrub clearing (such as accessibility and the vegetation type in the targeted areas), nonetheless, we consider that these ranges could give an idea on the general effect of shrub clearing on the hydrological dynamics of the Arnás catchment. Shrub clearing has been performed theoretically, removing percentage of vegetation carbon stores (leaf, stem and root carbon) in the defined target areas for AM.

2.3.2. Fitting climate variability and effect of climate change

To assess how inter-annual climate variation influences the effect of land management strategies on water and vegetation dynamics in the Arnás catchment we used different start dates for shrub clearings among the existing climate data (17 years of daily maximum and minimum temperature and rainfall).

1. In a first step, we have run RHESSys in a static mode considering only the hydrological component of the model (see Tague et al., 2009) with no inter-annual vegetation changes (only seasonal changes are permitted) and 15 different start dates for shrub clearing (2002-2016) (Figure 3.3). This will allow us detect the net effect of shrub clearing and water balance changes over the first year after the treatment accounting for climate variability in the recent past years (17 years combining dry and wet water years based on the total annual precipitations).
2. To study the net effect of shrub clearing over time and vegetation recovery after the treatment, we have run the model in a dynamic mode (or growth mode: net photosynthesis is allocated to vegetation growth and vegetation structure is able to respond to environmental conditions and in more extreme cases undergoes mortality). Under this mode, only 6 start dates for shrub clearing have been defined (2002-2007) followed by 10 years of vegetation recovery are compared with the baseline PM scenario (Figure 3.3). The recovery of the vegetation over a 10 years-period, will allow us investigate both vegetation and hydrologic dynamics under the different clearing intensities in comparison to the PM scenario.

In addition to climate variability, to account for a possible warming scenario and the effect of climate change on the ecohydrological processes of the Arnás catchment, we repeated all the simulations, under the different land cover scenarios in both static and dynamic modes, with a 2 ºC of temperature rise. However, no significant changes have been found for precipitation over Spain since 1906 (Coll et al., 2017) changes in precipitations have been reported to have strong spatial and seasonal variability over Spain and a strong uncertainty toward extreme events (Vicente-Serrano et al., 2017). In front of this high uncertainty toward precipitation changes we considered that at small catchment spatial scale (as the case of the Arnás catchment) applying changes to the precipitation regime may highly alter our results. For this, we only considered the increase in temperature but we examined how changes vary across inter-annual precipitation variation that is evident in the historic climate record (coefficient of variation 20.1). Table 3.1 shows a summary of all the analyzed scenarios in this study with their correspondent denomination.

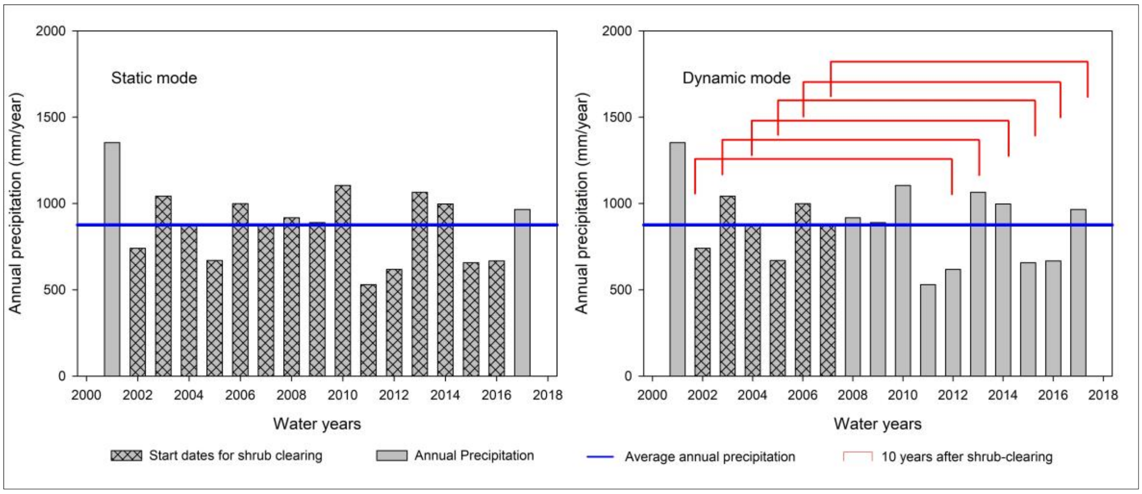

Figure 3.3. Annual total precipitation and selected start dates for shrub clearing under the Static (left plot) and Dynamic (right plot) modes. Water years are defined from October 1 to September 30.

Table 3.1. Simulated scenarios under Passive Management (PM) and Active Management (AM) strategies.

		Model component	Management Scenarios				
			Baseline Scenarios Passive Management (PM)	Clearing intensity under Active Management (AM)			
				25%	50%	75%	95%
Climate Scenarios	No warming climate	Static	BS	CS25	CS50	CS75	CS95
		Dynamic	BG	CG25	CG50	CG75	CG95
	Warming climate (+ 2 ºC)	Static	BSW	CSW25	CSW50	CSW75	CSW95
		Dynamic	BGW	CGW25	CGW50	CGW75	CGW95

B: **Baseline, C: **C**learing, G: **G**rowth (Dynamic), S: **S**tatic (No growth), W: **W**arming*

2.4. Studied variables

To study the effect of shrub clearing on water fluxes in the Arnás catchment three hydrologic variables have been selected for the analysis. Streamflow (STR), evapotranspiration (ET) and soil saturation deficit (SD) (SD as an indicator of soil moisture, higher SD values indicate lower soil moisture).

STR, ET and SD over the first year after the treatment following the different start dates under the static mode were averaged over the different start dates and annual values were obtained and their monthly pattern was analyzed. For the dynamic mode that allowed vegetation growth over a 10 years-period, we used the total plant carbon (PC) to investigate the recovery of the vegetation under the different land cover scenarios. STR, ET and SD in this case are studied on an annual time scale. Figure 3.4 presents a summary of the workflow of this study (compare PM and AM strategies on the hydrologic dynamics of an abandoned cropland area) and the studied variables during the analysis.

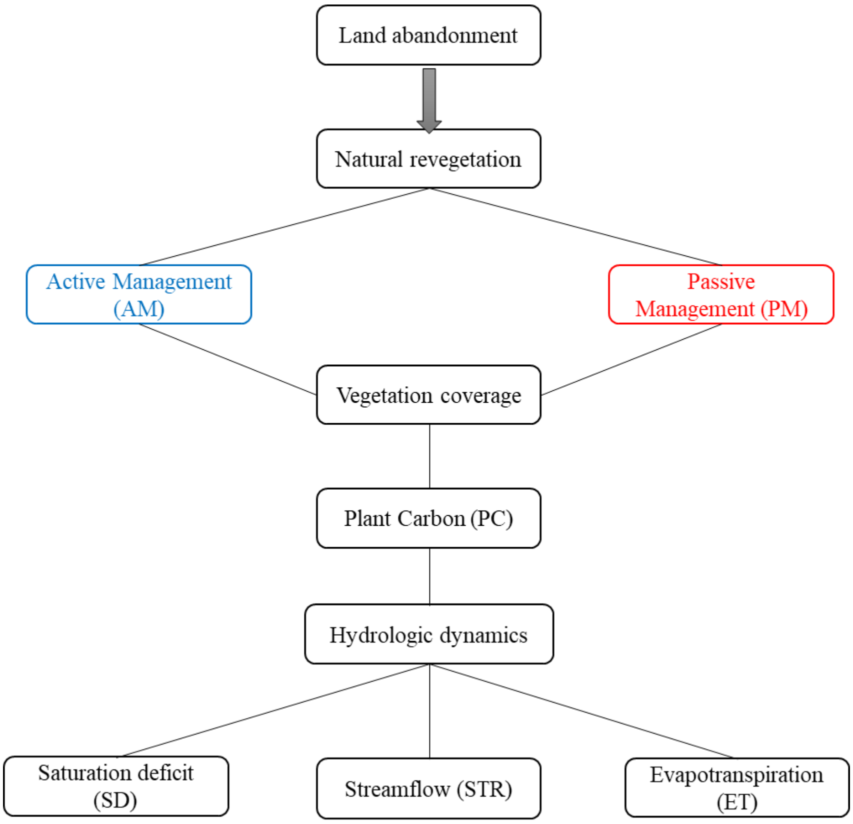

Figure 3.4. Analyzed variables under passive management (PM, natural revegetation) and active management (AM, shrub clearing) scenarios.

2.5. Statistical analysis

The Mann-Whitney Rank Sum and Pearson correlation tests were used to perform the statistical comparisons related to the three hydrological variables. 95% and 90% confidence levels were considered for Pearson correlation test significance. The SigmaPlot 11.0 program was used to perform the tests.

3. Results

3.1. Static mode (no vegetation growth) considering only the first year after shrub clearing

3.1.1. Effects of shrub clearing on the annual STR, ET and SD over the first year after shrub clearing:

For the static mode (no vegetation growth) Figure 3.5 presents boxplots of the total annual STR, ET and SD over the first year after shrub clearing under both land management and climate change scenarios. The values correspond to the first years following the treatment over the 15 selected start dates (2002-2016). CS presents the average of the four clearing intensities under no climate change effect whereas CSW is the average intervention under warming conditions.

Table 3.2. Average annual streamflow (STR), evapotranspiration (ET), and soil saturation deficit (SD) under land management and climate scenarios and their correspondent coefficient of variation in the first year of treatment (period 2002-2016).

	Passive Management (PM)		Active Management (AM)	
	BS	BSW	CS	CSW
STR (average annual streamflow, mm)	145	136	169	157
Coefficient of variation (%)	50.8	50.4	48.1	48.6
ET (average annual evapotranspiration, mm)	739	750	674	689
Coefficient of variation (%)	7.7	7.9	8.4	8.3
SD (average annual soil saturation deficit, mm)	318	327	300	308
Coefficient of variation (%)	14.2	14	14.2	14.2
Average annual precipitation (mm)	867			
Coefficient of variation (%)	20.1			

Boxplots (Figure 3.5) and the coefficient of variation (Table 3.2) of the different variables show high inter-annual variability under both management and climate scenarios. It is notable that the coefficient of variation for STR and SD are substantially higher than that of ET. This variability reflects the impact of the high variability of inter-annual rainfall (20.1%, Table 3.2) (Figure 3.3). The Mann-Whitney Rank Sum Test (Table 3.3) shows a statistically significant change ($p< 0.05$) in ET following shrub clearing while for STR and SD no significant change was found. Table 3.4 shows the percentage change of the three variables relative to the BS scenario (as baseline scenario in this case representing

PM) for the first year after the treatment. Shrub clearing led to a 16.3% of increase in the total annual STR, while ET and SD decreased by 8.8% and 5.8% respectively relative to BS scenario. For the warming scenario, a 2 ºC of temperature rise decreased STR by 6.3% and slightly increased ET and SD by 1.4% and 2.8% respectively compared to the baseline scenario (no shrub clearing under the static mode). Finally, considering the combined effect of shrub clearing and temperature rise relative to the BS scenario, changes are moderate on STR, ET and SD (8.1%; -6.9% and -3.1% respectively). In this case, climate warming reduces the effects of shrub clearing on water fluxes.

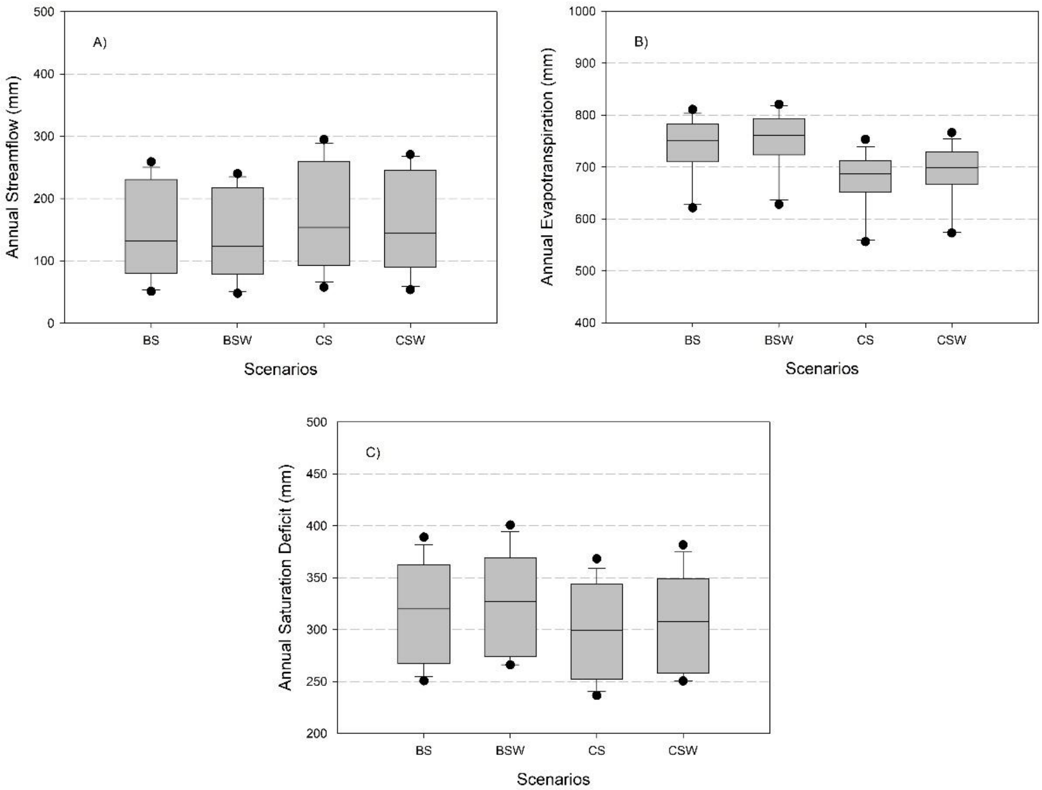

Figure 3.5. Effect of shrub clearing and climate change scenarios on A) annual streamflow, B) annual evapotranspiration and C) annual soil saturation deficit in the first year after treatment. Median value (the line in the middle of boxes), 1st and 3rd quartiles (lower and upper box boundaries respectively), 1st and 9th deciles (lower and upper error lines respectively), outliers (dots). BS: baseline no warming, BSW: baseline warming, CS: average of the clearing intensities under no warming, CSW average of the clearing intensities under warming (Static model).

These changes are sensitive to annual rainfall. The relationship between changes in STR and ET and total annual rainfall show significant positive correlation ($p<0.05$) under both climate scenarios for STR and a significant negative correlation ($p<0.10$) under the warming scenario for ET (Figure 3.6).

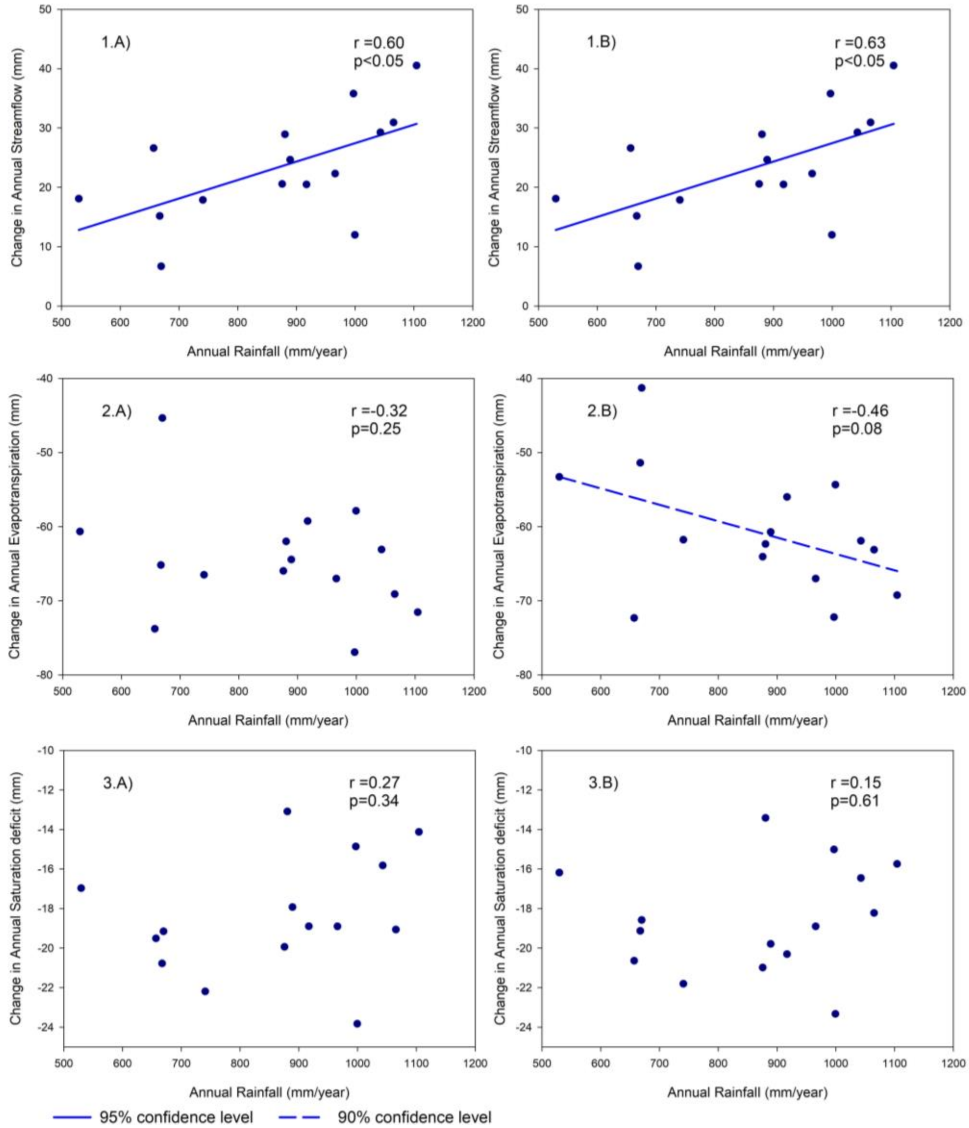

Figure 3.6. Changes in 1) streamflow, 2) evapotranspiration and 3) soil saturation deficit as a function of the total annual rainfall (A: no climate change scenario, B: climate change scenario) (n = 15) (period 2002-2016). Blue solid and dashed lines indicates significant relationship at 95% and 90% confidence levels respectively.

Table 3.3. p-value of the Mann-Whitney Rank Sum Test relative to the baseline scenario BS at 95% confidence level.

	p-value of the Mann-Whitney Rank Sum Test		
	STR	ET	SD
BSW	p = 0.65	p = 0.49	p = 0.41
CS	p = 0.40	**p<0.05**	p = 0.15
CSW	p = 0.57	**p<0.05**	p = 0.32

Table 3.4. Percentage change of annual streamflow (STR), evapotranspiration (ET) and soil saturation deficit (SD) in the first year of treatment, to the baseline scenario (BS) (period 2002-2016).

	Percentage change to Baseline BS (%)		
	Clearing	Warming	Clearing + warming
STR (annual streamflow)	16.3	-6.3	8.1
ET (annual evapotranspiration)	-8.8	1.4	-6.9
SD (annual saturation deficit)	-5.8	2.8	-3.1

3.1.2. Effect of shrub clearing on the monthly STR, ET and SD over the first year after shrub clearing:

The monthly patterns of STR, ET and SD during the first year after intervention were analyzed to investigate the seasonal effects of shrub clearing and climate warming on water fluxes (Figure 3.7). For STR, model results show a clear increase in monthly flows throughout the year under the no warming scenario, with the greatest increases occurring mainly during the wet period (January and May). For ET and SD shrub clearing generally decreased monthly averages with the maximum differences coinciding with the warm season (spring and summer months) (Figure 3.7. BS-CS). In contrast, a 2 ºC increase in air temperature led to a slight decrease of monthly STR during the wet period and an increase in SD (drier conditions). These changes correspond to an increase in early (February-March) season ET. Late season, ET, however is lower under warming suggesting a reduction in ET due to late season water limitations (note however that annual ET generally increases with warming, thus the early season increase in ET dominates and leads to declines in annual STR) (Figure 3.7. BS-BSW). The combined effects of shrub clearing and an increase in temperature (Figure 3.7. BS-CSW) are generally smaller than the effect of either change alone with smaller changes to monthly patterns of the three variables compared to the BS. With combined effects, the greatest changes occur late in the dry season. In this case groundwater stores are higher (lower SD) and ET is lower during the late season relative to BS, with only a very small increase in flow. However, inter-annual climate variability (error bars in the different graphs) is substantial, particularly for STR.

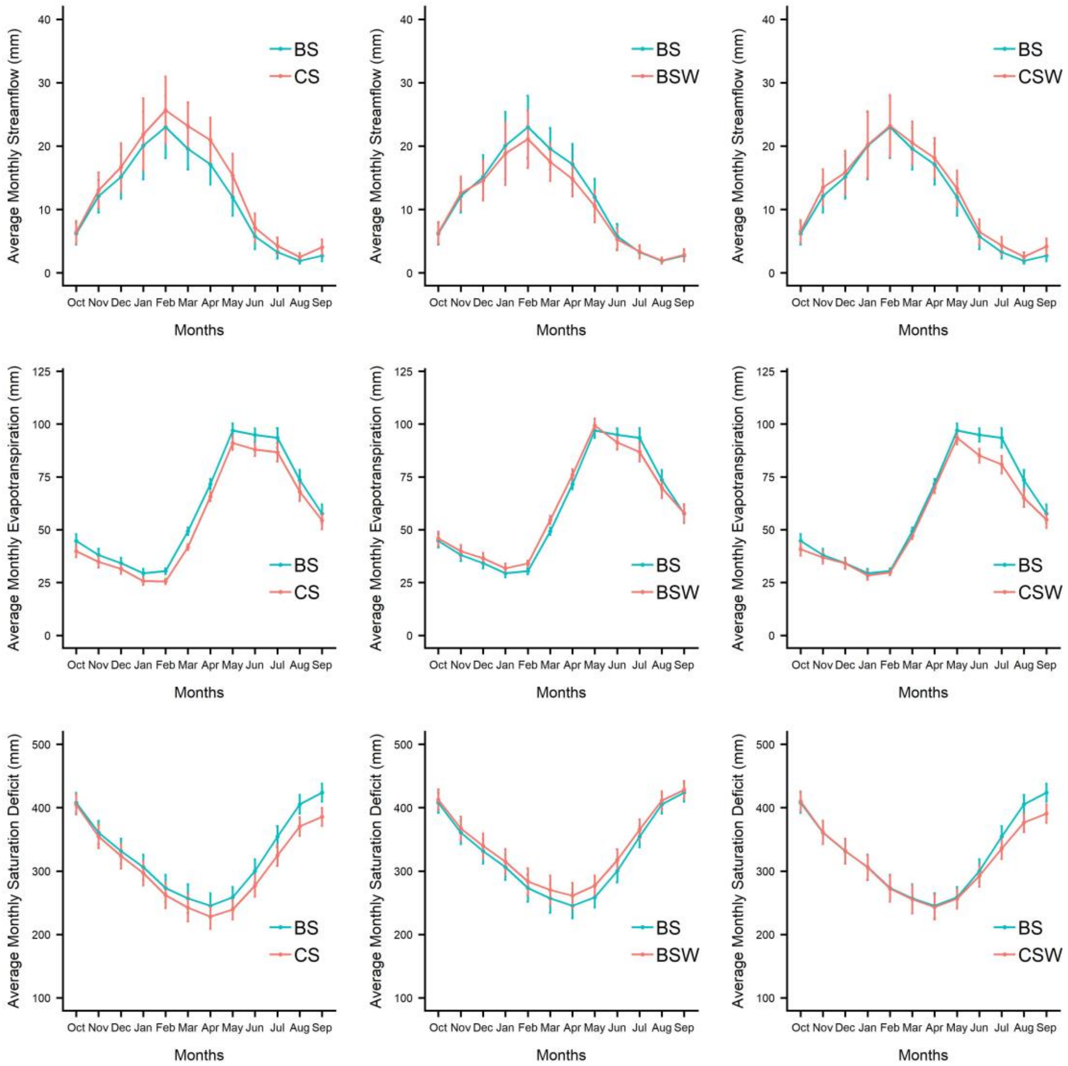

Figure 3.7. Effect of shrub clearing on the monthly patterns of streamflow (top three plots), evapotranspiration (middle three plots) and soil saturation deficit (bottom three plots) averaged over the different clearing intensities over the first year after treatment, vertical bars corresponds to the standard error of the different start dates for shrub clearing (2002-2016). BS: baseline no warming, BSW: baseline warming, CS: average of the clearing intensities under no warming, CSW: average of the clearing intensities under warming (Static model).

3.2. Dynamic mode (with vegetation growth)

3.2.1. Effects of shrub clearing over the first 10 years of vegetation recovery

To study the net effect of shrub clearing over time accounting for vegetation growth and interactions with climate conditions, we ran RHESSys in dynamic mode and evaluate the effects during the 10-years period of recovery following shrub clearing. We repeat using six different start dates to include inter-annual variation in meteorological forcing. We consider baseline and warming scenarios and we compare the year to year differences between AM and PM strategies.

Figure 3.8 presents trajectories of the changes in annual STR, ET and SD under the different clearing intensities during the first 10 years-period following shrub clearing. Boxplots present results of the six different start dates under (A) no climate change effect and (B) with climate change effect. Similar to the no growth simulation, a high variability was observed due to the high inter-annual climate variability (Figure 3.8 and Table 3.5). The direction of change due to clearing over the 10-years recovery trajectories are consistent with changes in the first year. Shrub clearing results in an increase in STR and a decrease in ET and SD under both climate scenarios. Expected differences between the clearing intensities were observed with greater effects associated with higher clearing intensity. Shrub clearing was found to have effects on all three variables throughout the 10 years-period of recovery, however, the timing of the greatest changes differs across the 3 variables. The highest changes in ET occur from the first year following clearing for both climate scenarios. However, for STR and SD although offsetting ET change during the first year their highest changes are delayed to the second year following shrub clearing. The declines in summer ET, thus translate into reduced subsurface storage capacity and consequently greater streamflow in the following year.

Table 3.5. Average annual streamflow (STR), evapotranspiration (ET), and soil saturation deficit (SD) (over 10-year recovery period) under management and climate scenarios and their correspondent coefficient of variation (PM: passive management; AM: active management).

	No warming					Warming				
	PM	AM				PM	AM			
	BG	*CG95*	*CG75*	*CG50*	*CG25*	*BGW*	*CGW95*	*CGW75*	*CGW50*	*CGW25*
Average annual STR (mm)	223	277	266	254	239	219	274	264	253	238
Coefficient of variation (%)	38.5	31	32.2	33.6	35.9	38.7	31.3	32.5	33.8	35.7
Average annual ET (mm)	632	577	589	601	616	636	580	590	602	616
Coefficient of variation (%)	12.2	13.4	13.1	12.7	12.3	12.4	13.6	13.3	13	12.6
Average annual SD (mm)	281	264	266	269	274	287	270	272	275	279
Coefficient of variation (%)	10.1	9.7	9.7	9.9	10.1	9.2	8.8	8.8	8.9	9.2

To quantify the change between AM and PM strategies over multiple years of vegetation recovery, we calculated the average change between the two management scenarios over the recovery period (10 years). With baseline climate and shrub clearing, annual STR increased between 7.1% to 24.2% compared to the baseline scenario (BG) (Table 3.6). Coefficient of variation for STR remains high (>30%) but is slightly lower for PM strategies (Table 3.5). For ET and SD, a lower percentage of change was observed compared to STR. The total annual ET and annual SD decreased by -2.6% to -8.7% and -2.7% to -6% respectively compared to the BG scenario following shrub clearing (depending on the clearing intensity) (Table 3.6).

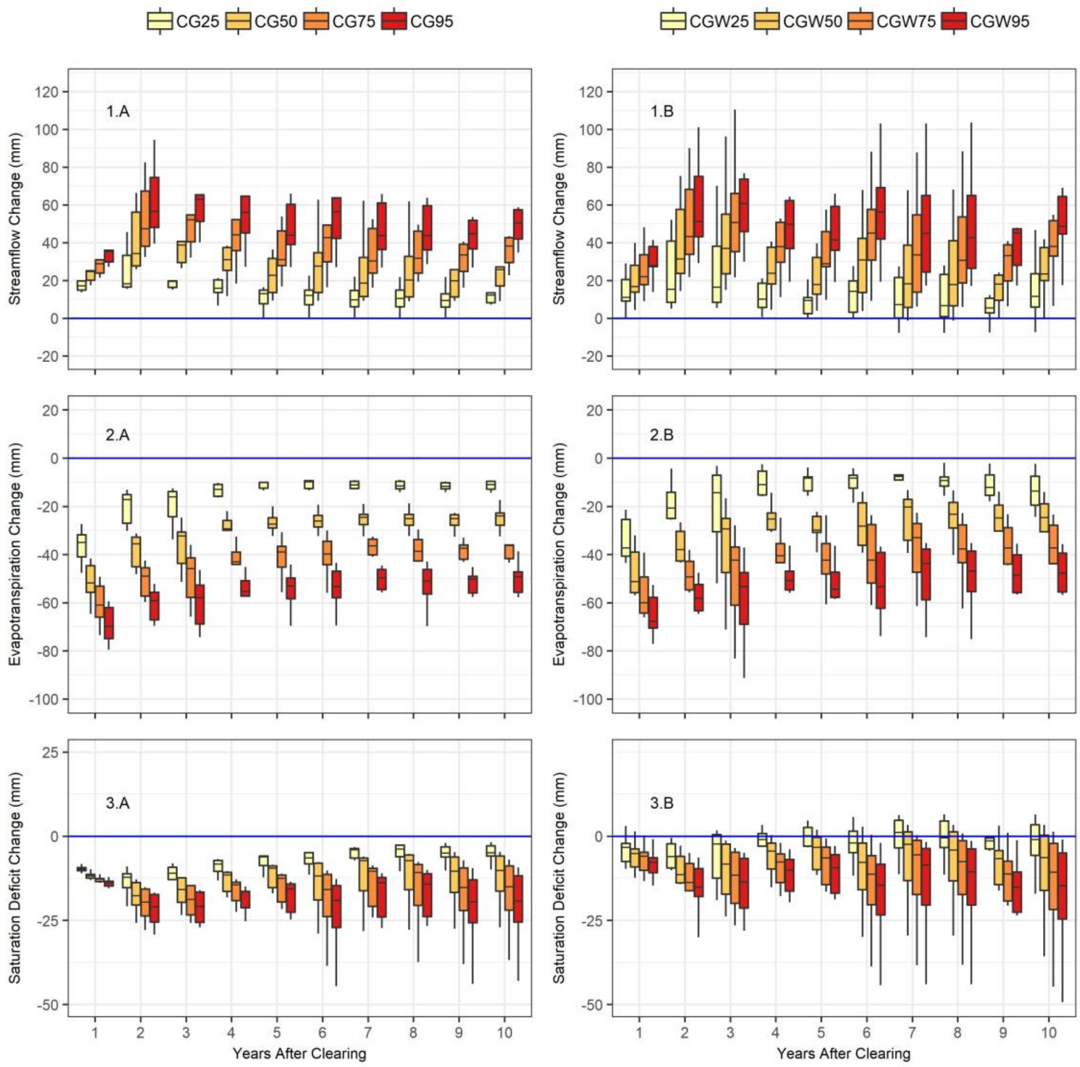

Figure 3.8. Change in 1) streamflow, 2) evapotranspiration, and 3) soil saturation deficit over the 10-year recovery period. A) No Warming, B) Warming. CG25, CGW25:25% clearing intensity, CG50, CGW50:50% clearing intensity, CG75, CGW75:75% clearing intensity, CG95, CGW95:95% clearing intensity (Dynamic mode). Median value (the line in the middle of boxes), 1st and 3rd quartiles (lower and upper box boundaries respectively), whiskers (hanging bars).

Table 3.6. Percentage change of average annual (over 10-year recovery period) streamflow (STR), evapotranspiration (ET) and soil saturation deficit (SD), relative to the baseline scenario BG.

	Percentage change relative to BG (%)								
	Clearing				Warming	Clearing + Warming			
	CG95	*CG75*	*CG50*	*CG25*	*BGW*	*CGW95*	*CGW75*	*CGW50*	*CGW25*
Annual streamflow	24.2	19.1	13.8	7.1	-1.7	23	18.4	13.3	6.7
Annual evapotranspiration	-8.7	-6.8	-4.9	-2.6	0.6	-8.2	-6.5	-4.7	-2.4
Annual saturation deficit	-6	-5.3	-4.3	-2.7	2.2	-3.9	-3.3	-2.3	-0.7

Changes were smaller and in the opposite direction under the warming scenario. STR decreased by -1.7% while ET and SD increased by 0.6% and 2.2% respectively relative to the BG scenario (Table 3.6). Finally, considering the combined effect of warming and management, climate warming reduced the impact of shrub clearing. Results show a smaller increase of STR (6.7% to 23%) and a smaller decrease in ET and SD (-2.4% to -8.2% and -0.7% to -3.9% respectively) following shrub clearing compared (Table 3.6).

3.2.2. Total Plant carbon recovery over the 10 years-period after shrub clearing

The greatest changes to hydrologic dynamics occur during the first three years and then effects decline as vegetation regrows. Figure 3.9 shows the average total PC in cleared scenarios relative to PM scenarios. Boxplots show variation due to the different clearing intensities and climate scenarios for the first 10 years following shrub clearing. Boxplots present results of the six different start dates (2002-2007). The year to year comparison to the BG scenario shows different recovery trajectories depending on the different clearing intensities. As expected, the higher percentage of vegetation removed, the higher is the difference to the BG scenario by the end of the 10-years period. The recovery trajectories show two clear recovery phases: during the first 4 to 5 years a higher recovery rate is estimated and from the fifth year the change in PC begins to stabilize. However, under the 2 ºC warming scenario, although PC shows the same general recovery patterns, the recovery trajectories under the different clearing intensities showed higher variability of the results (Figure 3.9.B).

It's important to note that under the year to year comparison between AM and PM strategies we do not reach PM scenarios levels since they are also changing due to vegetation growth. Supplementary Figure 3.4 shows the accumulated PC during a 10-years period under the baseline scenarios. A clear positive increase of PC rates was found which is mainly due to the continuity of the natural revegetation process with slightly higher growth rate of PC under the warming scenario.

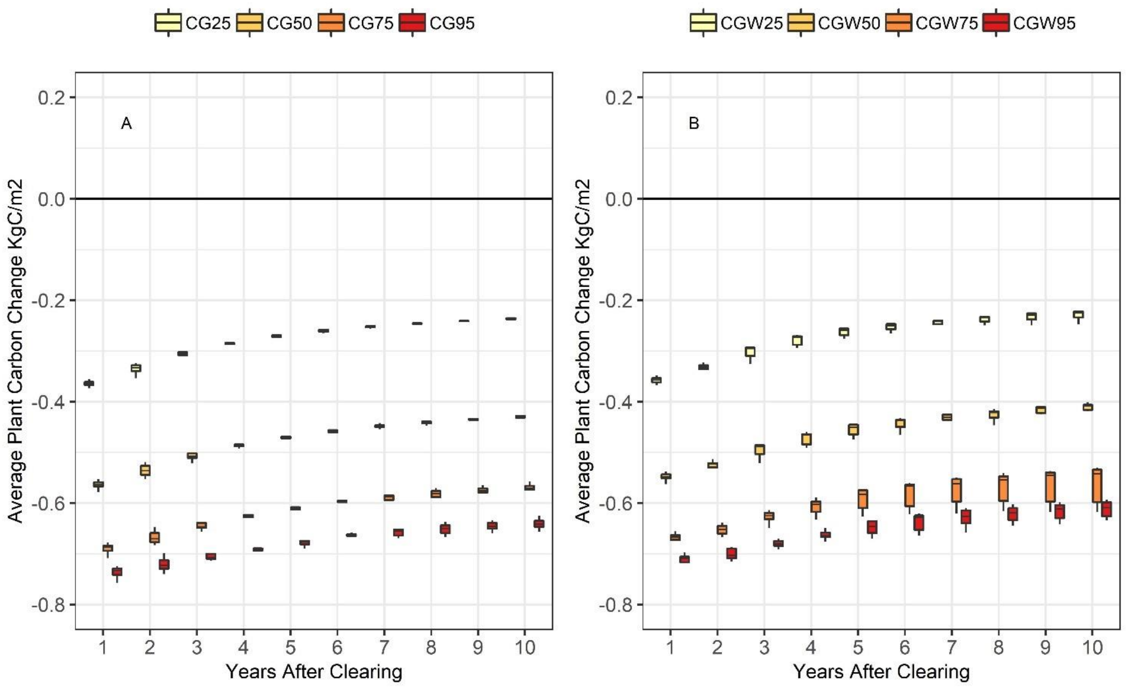

Figure 3.9. Average difference in plant carbon after shrub clearing relative to plant carbon in baseline scenario (BG) (A: no warming, B: warming). CG25:25% clearing intensity, CG50:50% clearing intensity, CG75:75% clearing intensity, CG95:95% clearing intensity (Dynamic mode). Median value (the line in the middle of boxes), 1st and 3rd quartiles (lower and upper box boundaries respectively), whiskers (hanging bars).

Table 3.7 shows the total accumulated PC during the 10-years period of recovery under the different land cover management scenarios. For BG and BGW (representing PM scenarios), the model estimates 0.68 and 0.75 (Kg C/m^2) to be accumulated during a 10-years period which is consistent with the recovery trajectories of PM scenarios (Supplementary Figure 3.4). On the other hand, under AM scenarios, PC recovery ranges between 0.72 and 0.76 Kg C/m^2 under no warming effect. However, the higher recovery rates under the warming scenario ranges between 0.75 and 0.82 Kg C/m^2. Among the different clearing intensities, removing 50% of the shrubs in the target areas gives the highest accumulated PC rates followed by 25% for the both variables and under the two climate scenarios.

Table 3.7. Difference between plant carbon (PC) at the start and end of the 10-year recovery period. Values present the average over the six different start dates (PM: passive management; AM: active management).

	Management Strategy	No Warming		Warming	
PC (Kg C/m^2)	PM	BG	0.68	BWG	0.75
	AM	CG95	0.72	CGW95	0.79
		CG75	0.74	CGW75	0.79
		CG50	0.76	CGW50	0.82
		CG25	0.75	CGW25	0.81

4. Discussion

Over the last decades, several publications have evaluated the effects of land abandonment and revegetation processes on the hydrological response and water resources (López-Moreno et al., 2011; Martínez-Fernández et al., 2013; Lana-Renault et al., 2018). But so far, little is known about how shrub clearing in combination with an increase in temperature may affect water resources and water availability in mountain areas. In this study, we use RHESSys model to study the post response of the hydrological and vegetation dynamics to AM strategy (shrub clearing) in an abandoned catchment of the Central Spanish Pyrenees.

4.1. Short term effects

In first step, we run the RHESSys model in static mode (no vegetation growth). These simulations were performed to study the net effects of shrub clearing on water regimes in the Arnás catchment during the first year after intervention and the relationship with the total annual rainfall. Results indicate an increase in STR and a decrease in ET and SD during the first year after shrub clearing. Precipitation variation in this catchment is high and tends to obscure the impacts of land management and climate change on streamflow. Evapotranspiration change are less sensitive to inter-annual climate variation and show statistically significant changes ($p<0.05$) due to shrub-clearing. Comparing the relative impacts of both land management and warming on water fluxes in the Arnás catchment, model estimates suggested that vegetation change could have a greater impact on streamflow magnitude (increase about 16.3%), evapotranspiration (decrease about 8.8%) and soil saturation deficit (decrease about 5.8%) than the direct hydrologic impacts of temperatures rise (-6.3% for STR, 1.4% for ET and 2.8% for SD). Nonetheless, a warming climate may result in a lower effect of shrub clearing on STR, ET and SD. In fact, model estimates of the effect of shrub clearing on STR, ET and SD have decreased considerably during the first year (8.1%, -6.9% and -3.1% respectively).

These results highlight the importance of considering both land management and climate scenarios to develop reliable water resources management plans. This result was also noted in other studies analyzing different land use changes (i.e. Tong et al., 2012; López-Moreno et al., 2014b). The RHESSys static simulations also show greater increases in annual flows due to shrub clearing in wet years. This makes of annual precipitation a determinative factor in defining areas where shrub clearing could be implemented to increase water yield. In this context, Hibbert (1983) highlighted the potential to increase annual flows in areas where annual rainfall exceeds 450 mm by replacing deep rooted shrubs with shallow rooted grasses.

At monthly time scales, the three variables under AM scenarios followed a similar inter-seasonal pattern as the baseline scenario during the first year after shrub clearing. An increase of STR mainly between January and mid-May (wet period) was observed after intervention, with a corresponding reduction in ET with shrub clearing. A key point of these findings is that although STR increase with AM, this increase occurs in winter months where water supply is less of an issue in this region. The highest water demand for irrigation occurs during the warm season (Casterad and Herrero, 1998) reflecting a clear drying trend in spring and summer period (Coll et al., 2017). Therefore, unless reservoir management can capture the increase in STR, the benefits for water resources management could be lower particularly in medium to large watersheds.

4.2. Longer term effects (10 years of recovery)

In general, most of the forest management studies adopted an increase in water yield following biomass removal due to management or disturbance (Hawthorne et al., 2013; Serengil et al., 2007; Yurtseven et al., 2018). Nonetheless, in all of these studies the magnitudes of water yield impacts are shown to vary with climate, vegetation and type of management.

Under the dynamic simulations of RHESSys, we compared the 10 years following shrub clearing in 6 different start dates for shrub clearing (2002-2007) under four different clearing intensities (25%, 50%, 75% and 95%). Under no effect of climate change, the

AM scenario resulted in an increase of between 7.1% to 24.2% of annual STR compared to the PM scenario. Other studies of the hydrologic effects of clearing show similar levels of increase: Lane and Mackay (2001) revealed an increase between 10% to 31% of annual streamflow during the first 3 to 4 years following a 22% of patch-cut removal and a 12% of forest thinning respectively. Dung et al. (2012) pointed out an increment of 240.7 mm in annual runoff after removing 43.2% of the basal area of a *Chamaecyparis obtusa* forest. Fernández et al. (2006) found even higher rate of increase (47% of annual streamflow) after reducing 70% of the basal area in a *Eucalyptus globulus Labill.* watershed in the NW Spain.

Biomass removal as a land management strategy would reduce canopy interception making more net precipitation reach the ground surface (Lane and Mackay, 2001; Mazza et al., 2011) and reduce the evaporative loss (Sun et al., 2015). These changes would consequently affect a major component of the hydrological cycle which is ET. In consistence with this, a decrease between -2.6% and -8.7% from the baseline scenario was found in annual ET over the first 10 years following management as effect of clearing. Several studies concluded that biomass reduction due to management or disturbance would decrease annual ET. Dore et al. (2012) for example, found a decrease in annual ET by 15% after thinning that reduced 40% of the existing LAI in a pine forest in Arizona. Similarly, Koch et al. (2010) pointed out a decrease in annual ET of 19% a decade after an intensive fire in a ponderosa pine forest in the southwestern United States. Meanwhile, other studies defend the idea that biomass reduction wouldn't necessarily reduce ET since the higher quantity of light reaching forest floor due to biomass reduction may increase ET levels from the understory vegetation and soil (Boczoń et al., 2016) and at some level offset the "gain" by the reduced ET from the overstory vegetation.

Moreover, results of comparing SD after shrub clearing to the BG scenario show a shift in annual SD between -2.7% and -6% after clearing under no climate change effect. This decrease in SD (increase in soil moisture) would enhance surface and subsurface flows and offset part of the increase in annual STR since surface runoff is qualified as a threshold process controlled by catchment wetness conditions (James and Roulet, 2007; Latron and Gallart, 2008; Zehe et al., 2010). In consistence with our results, Tague et al. (2019) pointed out that the existing empirical research revealed declines in water stress within the remaining vegetation following management or disturbance indicating that this water is accessible by the surrounding vegetation.

In addition, results of comparing the clearing intensities under the AM scenario revealed that these effects could be highly conditioned by the cleared area or the clearing intensities. In the same context, Yurtseven et al. (2018) concluded that a low density reduction would evidently results in lower runoff response of watersheds after intervention. While, Bosch and Hewlett (1982) associated the increase rate of the hydrological response with harvesting area and thinning intensity.

Otherwise, under the temperature increase scenario, a combined effect of clearing and warming would result in lower percentages of change to the BG scenario. These lower changes range between 6.7% to 23% for STR, -2.4% to -8.2% for ET and -0.7% to -3.9% for SD. The decrease in the effects of the AM scenario on the hydrologic dynamics may indicate that temperature increase due to climate change could reduce the efficiency of land use management strategies. This makes of climate change a fundamental feature to consider by policy makers when elaborating land management strategies. Besides, our results show that changes to PM scenario due to AM decreased with vegetation recovery among the first 10 years following treatment after achieving their highest changes during the first three years. Although there is a wide agreement on the reduction of post management vegetation response with regrowth and recovery to pre-management levels (Abdelnour et al., 2011; Jones, 2000; Jones and Post, 2004; Sahin and Hall, 1996; Tague et al., 2019), the duration of these effects still debatable in scientific literature depending on the site characteristics and the applied management (Webb and Kathuria, 2012). But in all cases, on short term scale, there is a global agreement that changes to water yield following vegetation removal occur over the first 5 years after management (Brown et al., 2005; Peel, 2009; Tague et al., 2019), which is consistent with our finding. A key point of our results is the occurrence of the strongest changes to the three variables. While ET showed the highest changes during the first year, the highest changes to STR and SD are delayed to the second year although showing some change during the first year. This delayed effect in STR and SD could be related to the date of the clearings (30 September) and to the occurrence of ET changes during the first year following clearing. The decrease in ET as a major component of the hydrological cycle occurs mainly during summer months (the end of the first year after shrub clearing) (result of the static simulations). Consequently, by the end of the first year the catchment presents lower SD resulting from the lower ET. This increase in soil moisture by the end of the first year is enhanced by the second year precipitation generating in return a higher increase in STR during the second year after vegetation removal.

4.3. Recovery of the vegetation

Finally, the recovery of the vegetation after shrub clearing and the clearing intensities are substantial questions for land managers under the relatively high cost of these practices to deal with land abandonment. In fact, we think that the analysis of the subsequent evolution of plant carbon could provide estimates on the ‘optimal’ number of years between clearing events and the adequate clearing intensities that guaranties an efficient economic as well as eco-hydrologic “benefits” of these practices.

Despite the relatively small area considered for shrub clearing compared to the total catchment area (15.75% (43.52 ha)), changes in the average PC showed an evident decrease relative to the PM scenarios proportional to clearing intensities during the first year after management. Comparing differences between the two management strategies, AM show higher accumulated PC during the 10-year recovery period under both climate scenarios following shrub clearing. These differences are likely related to the higher soil moisture following clearing in the study area that facilitates vegetation production.

Under the warming scenario however, an increase of 2 ºC in air temperature would accelerate the recovery process of biomass following shrub clearing. Consistent to our finding, several climate change studies highlighted that climate change has critical implication on plant growth including accelerating LAI and NPP rates (Lin et al., 2016; Nemani et al., 2003). Nonetheless, this behavior may be temporary enhanced by the increased soil moisture for the growing vegetation and at some point we could transit to water availability limitations for the increased rate of growth since water requirement also increase with temperature.

4.4. Land management considerations

Mountains constitute the main source of water for many of the world’s rivers, being considered for many author as “water towers” (Viviroli et al., 2003, 2007) particularly in Mediterranean areas. These areas are subject to changes that will affect the sustainability, quantity, quality and management of water resources (García-Ruiz et al., 2011). In the Pyrenees, it has been demonstrated that the water discharge has been following a downward trend from the mid-1960s, which is explained by land abandonment and plant growth more than by climate change (García-Ruiz and Lana-Renault, 2011). Revegetation process will continue, with a spread of forest in most of the Mediterranean mountains and water scarcity could result in conflicts among different territories, administrations and water uses. For these reasons, land management, particularly in mountain areas, is as important as climatic variability in assessing future water resources.

This study indicated that under these ecohydrologic responses, it is expected that streamflow increases under well managed shrub clearing scenarios (due mainly to changes in evapotranspiration and infiltration rates), however, increases typically occur during the wet period and are greater for wet years. The greatest changes to water regimes occur during the first 3 to 5 years following management. Other studies have already demonstrated positive effects of shrub clearing on fire control (Lasanta et al., 2018), livestock management (Lasanta et al., 2019), landscape and biodiversity (Lasanta et al., 2013) and water quality (Nadal-Romero et al., 2019).

This study has been carried out in a small experimental catchment, representative of the processes that have occurred in Mediterranean mountain areas during the last century. The detailed information obtained in the experimental catchment has allowed to calibrate and validate the ecohydrological RHESSys model. The results obtained in this research and previous studies have validated the use of the RHESSys model in Mediterranean mountain areas (López-Moreno et al., 2014b; Morán-Tejeda et al., 2015; Zabalza et al., 2018). Future studies will analyze the effects of shrub clearing management at larger scale and longer study periods to provide optimum bases for land management related to water resources.

The findings obtained in this study are relevant in view of the post response to land management processes and the time varying response of vegetation processes to interventions in rural and mountain areas and future climatic scenarios. However, further research questions could be also raised in relation to the effects of management practices to soil, plant and animal biodiversity (among others environmental variables).

5. Conclusions

Shrub clearing is a land management strategy that has shown interesting results since its implementation during the last decades to deal with land abandonment. Controlling forest fire risks and enhancing extensive livestock by generating pasture are the initial objectives of these practices. Nevertheless, although studies on the effects of this practice on water resources is scarce most of the studies addressing density reduction (due to fires and thinning in particular) effects on ecosystem services revealed a high impact on the hydrological and vegetation dynamics in the upper headwaters.

This study provides a novel information on the effects of AM strategies on water resources in a representative abandoned cropland area in the Central Spanish Pyrenees. Indeed, it is the first simulation of shrub clearing after cropland abandonment effects on water resources in the Mediterranean Mountain areas.

In consistence with our initial hypothesis, our results highlight the effect of shrub clearing in enhancing annual STR while reducing ET and SD although evidently proportional to clearing intensities. Climate variability and change in this context, are key factors in controlling changes on the hydrological and vegetation dynamics. In fact, our results revealed that shrub clearing effects on the hydrological cycle would be even higher under wetter conditions, whereas, climate change could relatively alter this equilibrium and limit AM effects in these areas. On the other hand, from the analysis of the monthly evolution of STR, ET and SD over the first year following intervention, it is worth mentioning that under the AM scenario, although following the same general pattern, clear differences to the PM scenario were identified (during the wet period for STR and the warm season for ET and SD).

Otherwise, comparing changes in STR, ET and SD to the PM scenario over a 10 years-period following shrub clearing has revealed interesting findings that could strengthen the knowledge on ecosystem response to AM plans over time. In this context, our analysis highlighted a maximum effect of shrub clearing over the first three years after intervention particularly on STR even though these differences still function of climate change and variability as well as the occurrence of drought events. Besides, vegetation recovery is a key factor on determining ecosystem response to shrub clearing and the adequate AM practices to apply considering both eco-hydrological and economic factors (adequate occurrence of shrub clearing events and adequate clearing intensity). Our results revealed that, the recovery of PC to the pre-clearing conditions is function of different factors. Depending on the clearing intensity and the total cleared area PC would take 3 to 9 years to recover to the preclearing levels. Climate change in this case (2 ºC temperature rise) may speed up the recovery process of PC although this process could be temporary enhanced by the increased soil moisture following clearing.

References Chapter 3

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Chapter 4. Effects of vegetation succession and shrub clearing after land abandonment on the hydrological dynamics in the Central Spanish Pyrenees

This Chapter uses the calibration parameters from the previous Chapter to upscale the modeling approach to the Estarrún watershed (76.8 km^2). RHESSys was used to model vegetation succession in abandoned cropland areas and its effects on the hydrological dynamics and the efficiency of shrub clearing. Four vegetation succession projections and their correspondent shrub clearing trajectories are defined. Changes to annual streamflow and evapotranspiration are quantified under the different land cover scenarios.

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Abstract

Cropland abandonment is a global problem that has affected countries worldwide since the 19th century. Abandoned lands usually undergo a process of plant succession. This process has been widely shown to have implications for ecosystem services, including water provision. Mediterranean mountains are currently in the middle of these land transformations: for example, in the Pyrenees, 80% of the agricultural lands have been abandoned. During the last decades, the management of these areas have become a target of environmental policy makers to ensure the sustainability of ecosystem services. Targeting available resources for land management, however, requires estimates of the effectiveness of different treatment options. This study address this need by relating the management of the abandoned lands to plant succession and hydrological dynamics. We used the Regional Hydro-Ecological Simulation System (RHESSys) to predict the changes to land cover and hydrological dynamics following land abandonment in the Estarrún watershed in the Central Spanish Pyrenees. Projections from passive management allowing vegetation succession, and active management via shrub clearing were defined and their effects on annual streamflow (STR) and annual evapotranspiration (ET) are compared. Results show a high increase in forest cover due to the conversion of all the currently existing shrubs to forest during the upcoming three decades. This conversion to forest cover, would decrease STR by 19%, increase ET by 34%, and reduce the target area for shrub clearing by 72% within 30 years of plant succession. However, shrub clearing would similarly increase STR and decrease ET by 6%, if 7.5% of the study area is cleared. For both STR and ET changes decline rapidly if the management is delayed and the vegetation succession process continues. This study confirms the need to take into account succession following land abandonment in designing land management strategies to preserve water resources in Mediterranean mountain areas.

1. Introduction

Cropland abandonment is a worldwide environmental problem (Strijker, 2005). Initially, land abandonment affected countries that became industrialized in the 19th century (United Kingdom, Germany or Switzerland), however in Mediterranean countries, the principal abandonment process occurred in the middle of the 20th century (Walther, 1986). Despite of its relevance, it has been really difficult to know the exact extension of abandoned areas and few data are provided in the literature. Ramankutty and Foley (1999) calculated that approximately 1.5 million km^2 have been abandoned worldwide, and Campbell et al. (2008) estimated that between 385 and 472 million hectares of cultivated areas have been abandoned between 1700 and 2000 m.a.s.l. Numbers that are forecasted to continue for the next few decades (Pointerau et al., 2008; Heinimann et al., 2017).

In the Mediterranean basin, land abandonment has been one of the most important land use and land cover changes during the last decades (Weissteiner et al., 2011). It is mainly concentrated in mountain areas (MacDonald et al., 2000; Lasanta et al., 2005; Tasser et al., 2007; Lasanta et al., 2015), and some semiarid regions (Pugnaire et al., 2006; Alonso-Sarría et al., 2016). In Mediterranean mountains, the process has been very intense, affecting for instance more than 80% of cultivated land in the Spanish Pyrenees (Lasanta, 1988), and around 70% in the eastern Alps (Tasser et al., 2007; Lasanta, 2019).

The first effect of land abandonment is the start of a revegetation process (secondary succession), which has important environmental consequences for water resources and soil quality (Poyatos et al., 2003; García-Ruiz and Lana-Renault, 2011; De Baets et al., 2013; Nadal-Romero et al., 2016; Romero-Díaz et al., 2017; Lasanta et al., 2020), but also for biodiversity or forest fire regime (Soulé and Noss, 1998; Cammeraat and Imeson, 1999; Lasanta et al., 2018). It is a highly complex process, because it depends on multiple climate, natural and biophysical factors, as well as, on human and social factors (Peña-Angulo et al., 2019). Old cultivated fields are transformed into extensive forests and shrublands (Chauchard et al., 2007; Gellrich et al., 2007); and in subalpine grasslands, a progressive revegetation process is also observed, due to the decline in the transhumance system, reducing the area occupied by grasslands (Gartzia et al., 2014; Sanjuán et al., 2018; García-Ruiz et al., 2020a).

Land abandonment and consequent revegetation processes alter the water cycle. The hydrological consequences of land abandonment in Mediterranean environments have been extensively investigated (Andréassian, 2004; García-Ruiz et al., 2011). The revegetation process after land abandonment affects evapotranspiration, throughfall, interception, infiltration, soil-water storage and runoff processes, with marked changes in the main components of the hydrological cycle (Llorens and Domingo, 2007; Beguería et al., 2003; López-Moreno et al., 2011). In general, a reduction in annual runoff coefficients, river discharges and peak flows has been observed (Beguería et al., 2003; Martínez-Fernández et al., 2013), as well as, a decrease in soil water content and aquifer recharge (Maestre and Cortina, 2004; Callegari et al., 2003). For instance, Martínez-Fernández et al. (2013) evaluated 74 rivers in Spain and almost all of them experienced reductions in flows, mainly in spring and summer. Lana-Renault et al. (2018) concluded that land abandonment resulted in a general decrease in hydrological connectivity in different post-management abandoned scenarios.

Water resources management and water availability are key aspects of the environment and socio-economic systems of the Mediterranean region. Mountain headwaters store water in both liquid and solid phases, and produce more than half of the annual runoff, being considered as water towers or humid islands (Viviroli et al., 2007; Viviroli and Weingartner, 2004). Therefore, a land-water-management strategy in the near future must consider what is happening in mountain areas. However, there is no consensus and managers and researchers have serious doubts about how to manage abandoned lands, favouring revegetation or restoration processes (Navarro and Pereira, 2012; Lasanta, 2019; García-Ruiz et al., 2020b).

Shrub clearing of abandoned croplands is an action carried out by some Spanish Regional Administrations, to promote the generation of pastures and stimulate extensive livestock, and reduce forest fires (Lasanta et al., 2018, 2019). In abandoned areas, with less than 30% slope, good accessibility and low risk of erosion, shrubs are removed, respecting the steeper areas and the limits of the old fields, which act as ecological channels. The end result is a mosaic landscape, including cleared fields, small patches of forest and shrubs, and large stands of forest interrupted by abandoned cleared fields (Figure 4.1). In the Northwestern Iberian System between 1986 and 2019, 35,092 ha were cleared, which represents 12.6% of the territory (Khorchani et al., 2020a). The effects of shrub clearing are very positive related to the control of fires (Lasanta et al., 2018), pasture generation (Lasanta et al., 2019), and to the structure of the landscape and biodiversity (Lasanta et al., 2016). Concurrently, recent studies have demonstrated that the clearing of some shrublands would yield a slight increase in water flows. At plot scales, Nadal-Romero et al. (2013 and 2018) suggested that clearing shrubs returned positive effects, as it improves soil quality, while produces high-moderate runoff coefficients (higher than the ones observed in the dense shrub environment). Khorchani et al. (2020b) highlighted that shrub clearing enhances annual streamflow and reduces evapotranspiration, being these effects higher under wetter conditions.

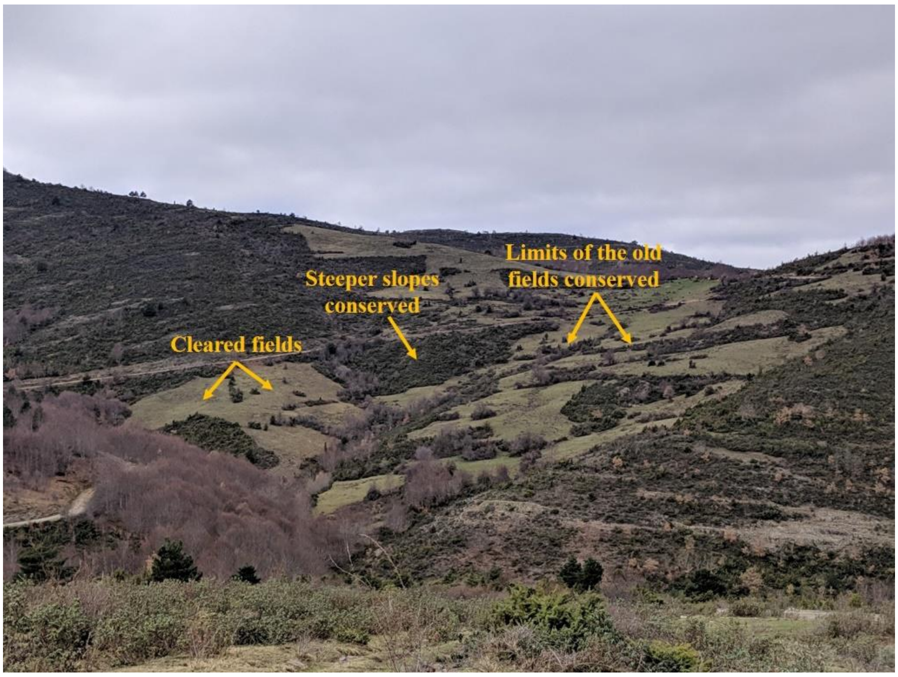

Figure 4.1. Mosaic landscape after shrub clearing.

Most of the Mediterranean mountain areas still maintain a shrub-dominated landscape matrix (Sluiter and de Jong, 2007; Tasser et al., 2007; Tzanopoulos et al., 2007; Stellmes et al., 2013), therefore many areas of the territory could be subject to a shrub clearing intervention. However, in other mountains, such as the Spanish Pyrenees, the process of plant succession is more advanced, with abundant slopes that reached the stage of forest (Lasanta and Vicente-Serrano, 2007). This is explained by the fact that many fields were abandoned several decades ago (mostly before 1950) and by their sub-humid climatic conditions (rainfall above 1000 mm / year), circumstances that have favored a relatively rapid process of plant succession (Poyatos et al., 2003; Lasanta and Vicente-Serrano, 2007). Furthermore, the process of revegetation is likely to continue, shrublands evolving to forests and subalpine pastures being colonized by shrubs. The expected increase in vegetation in these mountain areas during the next decades will have significant hydrological consequences, making water resources availability challenging in

Mediterranean areas (López-Moreno et al., 2014). Thus, there is an urgent need to adopt strategies for landscape management and to evaluate the hydrological implications of shrub clearing in Mediterranean mountain areas and the effects of the delay in managing land abandonment in mountain areas. Nevertheless, shrub clearing is still non-recognized as a management practice to enhance water availability. Its adoption in Mediterranean mountain areas is mainly related to the control of forest fires and pasture generation (Lasanta et al., 2018, 2019). The low adoption of shrub clearing in these areas could explain the lack of empirical studies comparing its hydrological implication respect to other management practices. Process-based models however offer a means to evaluate the potential hydrologic benefits of shrub clearing. In this study we use Regional Hydro-Ecological Simulation System (RHESSys) (Tague and Band, 2004) to study the hydrological implications of management plans in Spanish Pyrenees. RHESSys is a mechanistic model that has been widely used to study the impacts of climate and land use change in Mediterranean type ecosystem (Vicente-Serrano et al., 2015; Zabalza et al., 2018; Boisramé et al., 2019; Saksa et al., 2020). We take advantage of the ecohydrologic processes represented within RHESSys and their sensitivity land use changes (Morán-Tejeda et al., 2015; Khorchani et al., 2020b) to develop land management strategies. We ask the following research question: What are the effects of the delay in managing cropland abandoned areas on vegetation succession and hydrological dynamics? The main objectives are: (i) to analyse revegetation processes (secondary succession) evolution in abandoned areas in the Estarrún watershed (Central Pyrenees); (ii) to identify the target areas for shrub clearing under several revegetation delay scenarios; and (iii) to investigate the effects and the efficiency of shrub clearing under different vegetation succession projections on the hydrological dynamics.

This leads to the following research hypothesis: the delay in managing land abandonment areas will probably limit the efficiency of land management practices and the benefit from its adoption in land abandoned areas in Mediterranean mountain environments. To test this hypothesis, we select a Mediterranean mountain valley (Estarrún watershed, Central Spanish Pyrenees) and simulate revegetation processes and shrub clearing management in abandoned areas under four different vegetation succession stages (current, 10, 20 and 30 years of vegetation succession), and evaluate the hydrological consequences using the eco-hydrological model RHESSys.

2. Material and Methods

2.1. Study area

The study area (76.8 km^2) is the Estarrún watershed in the Central Spanish Pyrenees (Figure 4.2). This watershed has been widely cultivated during the past and from the first decades of the 20th century, a process of abandonment took place. By 1957, 61% of the cropland area had been abandoned and by 1981, 74% of these areas were already abandoned (Lasanta, 1988). This is similar to many other regions in the Mediterranean mountains (Lasanta et al., 2019).

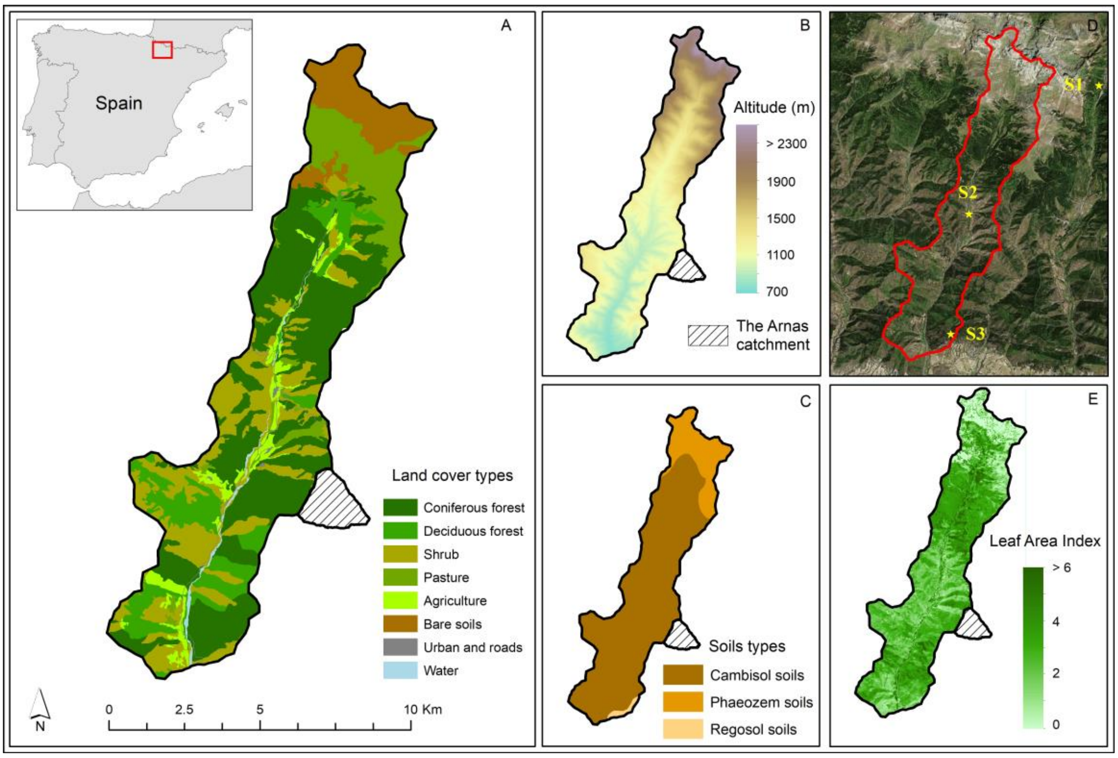

Figure 4.2. The Estarrún watershed. A: Land cover map (reclassified from the SIOSE, 2005 and the aerial photography PNOA, 2006), B: Topographical gradient (Obtained from a 5 m digital elevation model), C: Soil map (reclassified from the Aragón soil map), D: Aerial photography of the study area and E: Leaf area index (Landsat, 2011).

A high north-south altitudinal gradient crosses the watershed and defines the Estarrún main stream with altitudes between 700 and 2300 m.a.s.l (Figure 4.2.B). Forest cover (*Pinus sylvestris and Quercus gr. faginea*) predominates the vegetation coverage particularly in the north-facing slopes (38.4% and 12.3% respectively, Table 4.1), while shrubs (*Buxus sempervirens*, *Juniperus communis*, *Genista scorpius*, and *Echinospartum horridum*) spread over the south-facing slopes (21.2%, Table 4.1). The last cultivated agricultural areas (mainly meadows) represent 4.7% of the study area (Table 4.1) and are concentrated in the riparian zone and the lowland areas of the watershed.

The lithology of the Estarrún watershed is mainly the Eocene Flysch, where the majority of the agricultural fields were concentrated (Lasanta et al., 2017). Cambisols and Phaeozem soils are the main soil categories in the study area (82.2% and 16.7% respectively), while some Regosols soils (1.1%) are found in the bottom lowland areas (Figure 4.2C, Table 4.2).

Table 4.1. Land cover types and correspondent percentage surface of the study area.

Land cover type	Area (Km^2)	Percentage
Coniferous forest	29.5	38.5
Deciduous forest	9.5	12.3
Shrub	16.3	21.2
Pasture	9.5	12.4
Agriculture	3.6	4.7
Bare soil	7.3	9.5
Urban and roads	0.5	0.6
Water	0.6	0.8
Total	**76.8**	**100**

Table 4.2. Soil types and correspondent percentage surface of the study area.

Soil types	Area (Km^2)	Percentage %
Cambisol soils	63.1	82.2
Phaeozem soils	12.8	16.7
Regosol soils	0.9	1.1

2.2. Climatic data

Fifty years' series of daily precipitation, maximum and minimum air temperature were obtained from the National Spanish Meteorological Agency (AEMET). Three meteorological stations relatively well distributed over the study area, were selected due to their large temporal data availability (1968 - 2018) (Figure 4.2D, Table 4.3). Data series from the three meteorological stations were homogenized, corrected and gap filled following Vicente-Serrano et al. (2010) and El-Kenawy et al. (2012). Several meteorological stations from the AEMET and the Pyrenean Institute of Ecology (IPE-CSIC) in a radius of 10 km were involved in the process of data filling and homogenization.

The average annual rainfall ranges from 1846 mm in station 1 to 824 mm in station 3 reflecting a high topographic gradient between the upper and the bottom part of the watershed (Table 4.3). Precipitations are mainly concentrated during autumn and spring while summer months are relatively dry. The high year to year variation in rainfall, (up to 25.6% in station 2, Table 4.3), can be mainly attributed to the strong variations during the wet season (Figure 4.3).

Maximum air temperature oscillates between 13.5 ºC and 18 ºC with a lower year to year variation (4% to 8%) than minimum air temperature, which ranges around 5 ºC with around 19% of variation. While average temperature during summer and winter months oscillates between 16 ºC to 20 ºC and 2 ºC to 4 ºC respectively (Table 4.3).

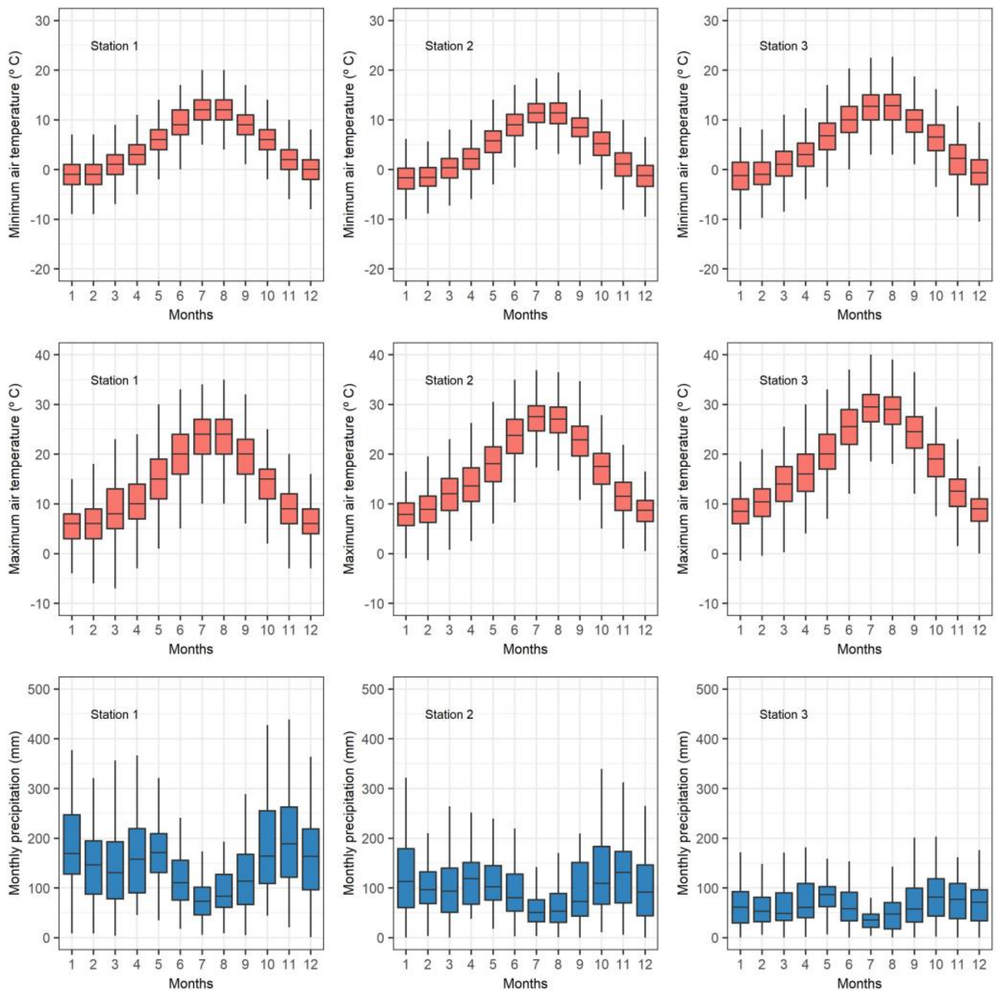

Figure 4.3. Monthly climate data from the three-selected meteorological station (1968 - 2018). Median value (the line in the middle of boxes), 1st and 3rd quartiles (lower and upper box boundaries respectively), whiskers (hanging bars). Note that minimum and maximum plots present different temperature scales.

Table 4.3. Altitude, annual average precipitation (Rain), maximum air temperature (Tmax), minimum air temperature (Tmin), coefficient of variation (c.v) and Average temperature (Tavg) during the summer and winter months in the three meteorological stations.

	Altitude (m)	Rain (mm)	c.v (%)	Tmax (ºC)	c.v (%)	Tmin (ºC)	c.v (%)	Tavg summer (ºC)	Tavg winter (ºC)
Station 1	1295	1846	18	13.5	8.8	4.8	19.3	16.6	2.6
Station 2	999	1246	25.6	16.5	4	4.1	19	18.1	3.4
Station 3	951	824	21.1	18	4.5	5.1	19.4	19.7	4.1

2.3. RHESSys Model

RHESSys (Tague and Band, 2004) has been widely implemented during the last years to conduct hydrological and ecological studies in different environments, including the Pyrenees (Morán-Tejeda et al., 2015; Saksa et al., 2017; McDonnell et al., 2018; Anderson et al., 2018; Khorchani et al., 2020b). RHESSys model was developed to study interactions between ecohydrological processes and land cover and climate changes from local to regional scale (Tague and Band, 2004). RHESSys includes a hierarchical partitioning of the landscape that allows representing both local processes at fine scales and the spatial heterogeneity at coarse scale. Hydrological and carbon cycles are modeled at plot scale (patch level, 50 m in this case) and climate drivers are simulated at zone level, while lateral routing of surface and subsurface flows are modeled at hillslope level (Bart et al., 2016). Details on RHESSys model and using guide can be found in https://github.com/RHESSys/RHESSys/wiki.

2.3.1. Model calibration and validation

To calibrate subsurface storage and drainage parameters and validate the model, observed daily streamflow data from the Arnás catchment was used (see Figure 4.2B). Arnás is an experimental catchment in the Central Spanish Pyrenees widely monitored from the Pyrenean Institute of Ecology (IPE-CSIC) over more than 20 years. The absence of a gauging station in the Estarrún watershed prevented a calibration at the scale of the larger study watershed. Nevertheless, the proximity and particularly the similar lithology between both sites (Eocene Flysch) encouraged the use of the Arnás catchment calibration parameters. The calibration of RHESSys in the Arnás catchment has showed a strong ability of the model to reproduce the observed daily streamflow data. More details on the calibration and validation processes of RHESSys model in the Arnás can be found in Khorchani et al. (2020b).

2.3.2. Target Spin-up and modeling vegetation succession

To make sure that the vegetation cover and its spatial heterogeneity throughout the study area are well modeled, we adopted a target driven spin-up technique following Hanan et al. (2018). The spin-up is a preliminary simulation to reach a state of equilibrium of soil carbon and nitrogen stores and vegetation. In a first step, a 200 years' simulation was made to initialize soil carbon and nitrogen stores. Then, the model was run until a target Leaf Area Index (LAI) value was reached for every specific patch in the watershed. To generate the LAI map for the study area, we acquired Landsat 5 Thematic Mapper surface reflectance at 30 m spatial resolution acquired on June 19, 2011 (Figure 4.2E). More details on the generation of the LAI map could be found in Hanan et al. (2018). This approach allowed a good representation of vegetation spatial heterogeneity within the study watershed.

Given that vegetation germination is not represented in RHESSys model, we modeled vegetation succession in this study by adding small seedlings of understory vegetation to forest, shrubs and pasture patches. A grass layer has been added under forest areas, forest seedlings under the shrub patches, and shrub seedlings under the abandoned pasture patches. No vegetation understory layer has been added to the rest of land cover categories. This way, if all the conditions for vegetation growth are met for the added small seeds (e.g. rooting zone water availability, sunlight etc.), RHESSys will grow an understory layer in selected areas of the watershed. Although vegetation succession is a more complex process involving several climatic, ecological and physical features, we consider that the adopted representation could give a good estimation of when and where type conversion would occur and its subsequent implications on hydrological dynamics.

2.4. Scenarios

After adding understory, four vegetation succession conditions were defined VS0, VS10, VS20 and VS30 (representing the no shrub clearing scenario). VS0 (current situation) represented the control scenario. VS10, VS20 and VS30 are projections of the vegetation after 10, 20 and 30 years of succession respectively. We defined these three projections by spinning up VS0 for 10, 20 and 30 additional years respectively and estimating the areas of conversion (from shrub to forest and from pasture to shrub) under each one of the projections. To determine whether conversion had occurred we used the amount of biomass at the end of each of the 10, 20, and 30 years spin-up. Conversion to forest or shrub was based on the watershed average biomass of forest and shrubs, respectively, under the control situation VS0. All pasture patches, that reached or exceeded the VS0 average watershed shrub biomass, are considered converted to shrub, and all the shrub patches, that reached or exceeded the VS0 average watershed forest biomass, are considered converted to forest patches. The new converted forest areas are added to the coniferous area given the impossibility to determine the species of the new colonizing trees.

To assess the effects of vegetation succession on the efficiency of adopting a shrub clearing management strategy and its effect on water dynamics, we defined four potential shrub clearing scenarios: SC0, SC10, SC20, SC30 which represents the available target area for shrub clearing corresponding to the 4 vegetation succession scenarios respectively (VS0, VS10, VS20, VS30). Target areas for shrub clearing in the watershed were selected by applying the criteria for shrub clearing site selection used by the La Rioja community in Northern Spain. Specifically, only shrub and abandoned pasture lands are considered for shrub clearing while forest and the remaining agricultural lands are excluded. The riparian zone and areas of more than 30% slopes are also masked to avoid soil erosion. More details on defining areas for shrub clearing can be found in Lasanta et al. (2016) and Khorchani et al. (2020b). In RHESSys several options are available to implement shrub clearing and forest thinning. In this study, we selected a harvesting technique that removes above ground biomass. Shrub clearing removes carbon from stem and leaf carbon stores but does not add this removed carbon to the litter carbon pools, while for below ground carbon stores, carbon from roots is removed and added to the litter carbon pools. Corresponding nitrogen pools are also modified to maintain stociometric relationships. Carbon stores are removed equally form overstory and understory in the target areas for shrub clearing. More details and instructions on implementing forest thinning and shrub clearing, could be found in RHESSys website (https://github.com/RHESSys/RHESSys/wiki). This harvesting technique is the most realistic option to simulate a shrub clearing scenario. All shrub clearings are implemented the day before the start of the hydrological year (September, 30), and 100% clearing intensity in the target areas is used (all the above ground biomass is removed in the target areas for shrub clearing).

In order to assess the effect of year to year climate variation (Table 4.3) on land management impacts, we repeated each scenario for multiple start dates. We used 10 different start dates by selecting every 5th year of 50-year climate record. For each of the 10 start-dates, we simulated five years' vegetation growth under the two management scenarios (clearing or no-clearing). In total, we performed 80 simulations (Figure 4.4), comprising shrub clearing or no clearing applied to each of 4 levels of succession (including application to the current land cover state), repeated for each of 10 possible start dates. Note that differ succession scenarios correspond to different timing of shrub clearing (e.g. SC30 reflects waiting 30 years before shrub clearing is initiated). Varying the start dates varies the climate by sampling from the historic record. Simulated annual streamflow (STR) and annual evapotranspiration (ET) are compared across different clearing and no clearing scenarios. The Tukey's Honestly Significant Difference test (TukeyHSD) (Tukey, 1949) was used to perform the comparison between the different scenarios.

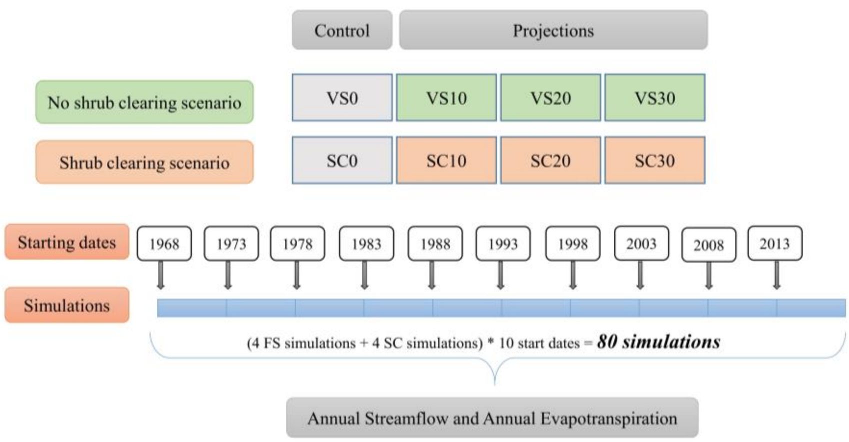

Figure 4.4. Study scenarios.

3. Results

3.1. Effect of passive management on vegetation succession

The spatial patterns of the estimated above ground biomass for the 3 vegetation succession projections (VS10, VS20 and VS30) can be compared with the initial VS0 biomass to evaluate the likely extent of conversion from pasture to shrubs and shrubs to forest in the Estarrún watershed (Figure 4.5). For VS10 (ten years of vegetation succession), no pasture to shrub conversion occurred, but 61% of the existing shrub areas under VS0 has been converted to forest under VS10 (Table 4.4). Following 20 years of vegetation succession (VS20), almost all initial shrubs (98%) would be colonized by forest, and shrub vegetation would expand by 19% due to the colonization of abandoned pasture land, particularly in the highest elevations of the watershed. Under VS30, type conversion would affect 35% of the abandoned pasture land and 99% of VS0 shrub areas (Table 4.4).

Table 4.4. Type conversion as result of management delay. Percentages are computed as the change in total land area relative to the current VS0 condition. VS0, VS10, VS20 and VS30: represent vegetation succession projections.

	Type conversion (%)		
Projections	VS10	VS20	VS30
Shrub to forest	60.7	98.9	99.8
Grass to shrub	0.0	19.6	35.7

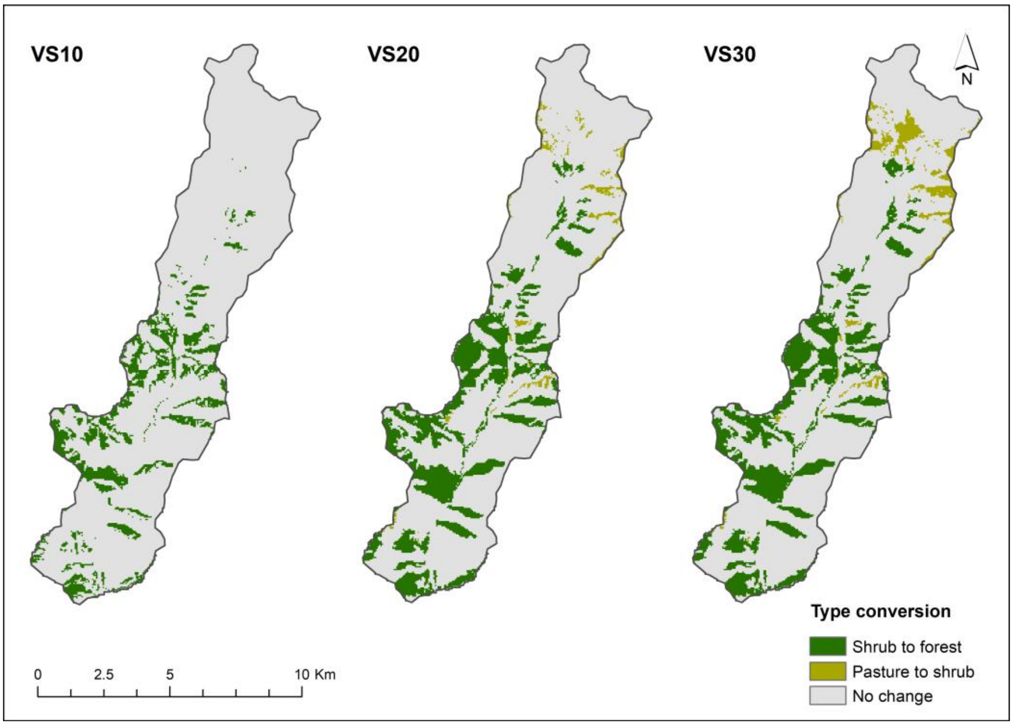

Figure 4.5. Spatial distribution of type conversion as result of management delay. VS10, VS20 and VS30: represent vegetation succession projections in 10, 20 and 30 years.

Figure 4.6 shows the resulting land cover patterns for the different vegetation succession projections (Figure 4.6). Under the VS10 projection, forests would cover more than 63%

of the study area (coniferous forest 51% and deciduous forest 12%, Table 4.5). This increase in forest area will continue, though in a lower rate, such that after 30 years of unmanaged succession, forest would cover 72% of the Estarrún watershed surface (Table 4.5). These new forest areas correspond to the shrub converted areas mainly in the middle and the lowland areas of the watershed. While conversion from pasture to shrub does occur, this conversion rate is less than shrub to forest type conversion. Thus, our results, estimate a long-term decline in the shrub area to 4.4% of the total study area under the VS30, and abandoned pastures would decrease to only 8% of watershed surface (Table 4.5).

Table 4.5. Land cover percentages as results of the different vegetation succession projections. VS0, VS10, VS20 and VS30: represent vegetation succession projections.

	Percentage (%)			
Succession time	VS0	VS10	VS20	VS30
Coniferous forest	38.5	51.3	59.5	59.7
Deciduous forest	12.3	12.3	12.3	12.3
Shrub	21.2	8.4	2.6	4.4
Pasture	12.4	12.4	10.0	8.0
Agriculture	4.7	4.7	4.7	4.7
Bare soil	9.5	9.5	9.5	9.5
Urban and roads	0.6	0.6	0.6	0.6
Water	0.8	0.8	0.8	0.8

An increase in forest cover in the Estarrún watershed would consequently decrease the target areas for shrub clearing. Table 4.6 shows the total target areas for shrub clearing under the different vegetation succession scenarios relative to the current, control condition VS0. Steepest declines occur in the first decade. The total target areas for shrub clearing under VS10, would decline to less than 61% of the area available in VS0 (Table 4.6). For VS20, if no shrub clearing management is adopted before, the decrease in the total target area for shrub clearing would decline by 72%, relative to conditions in VS0, and remain the same for VS30 (Table 4.6). Almost all the available areas for shrub clearing in control situation located in the middle and the bottom of the study area would be excluded under the different future projections for vegetation succession. Only small areas (2.9% of the study area, Table 4.6), in the highest part of the watershed (Figure 4.6) would remain included in a shrub clearing strategy after a 30 years-period of vegetation succession. Our results show no changes in the total target area for shrub clearing between VS20 and VS30 projections (Table 4.6).

Table 4.6. Reduction in the target areas for shrub clearing.

	Shrub clearing projections			
	SC0	SC10	SC20	SC30
Target area (%)	7.5	3.7	2.9	2.9
Reduction from SC0 (%)		-61.1	-72	-72

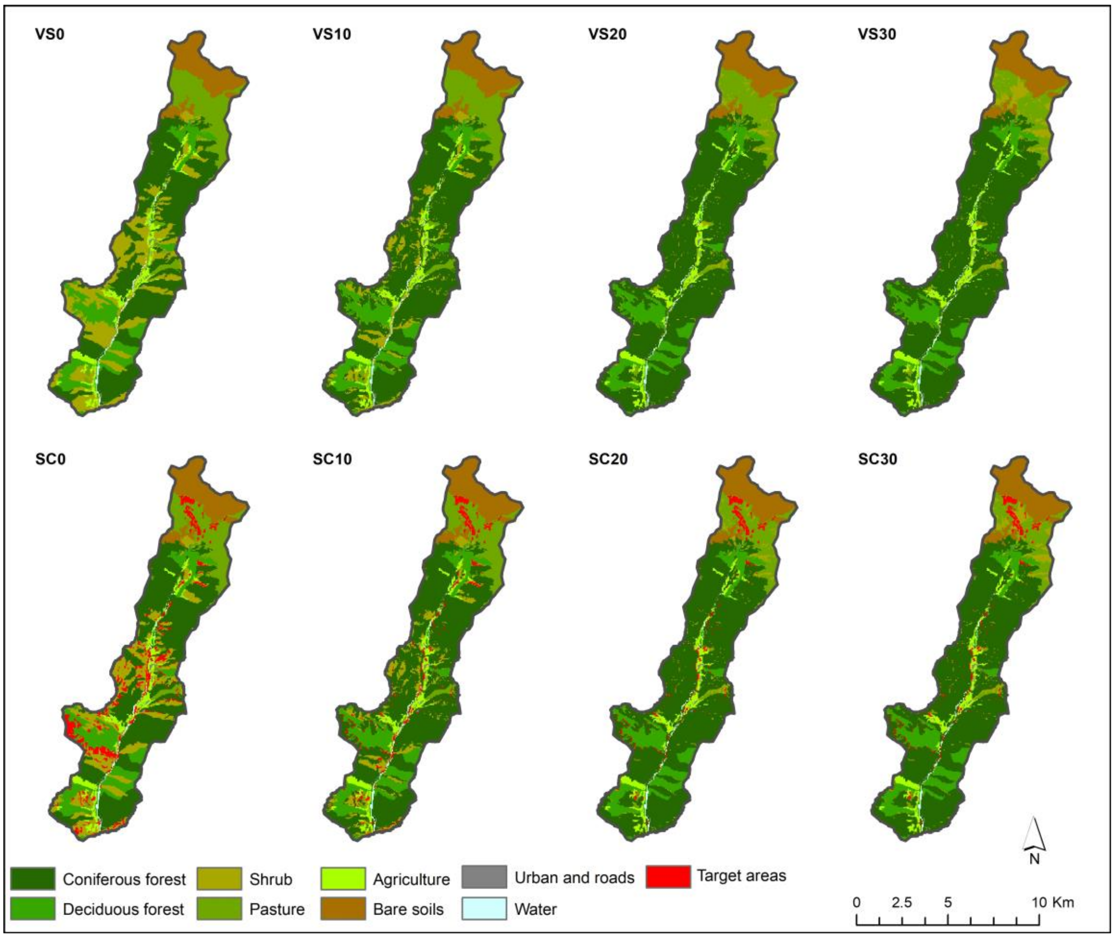

Figure 4.6. Target areas for shrub clearing under the different succession projections. VS0, VS10, VS20 and VS30: represent vegetation succession projections. SC0, SC10, SC20 and SC30: represent shrub clearing projections.

3.2. Effect of delaying land management of abandoned croplands on STR and ET

The results of these plant colonization of abandoned lands would affect the hydrological dynamics of the watershed. Figure 4.7 shows the differences between annual STR and ET under the no clearing and the clearing scenarios as a function of the age of succession. In general, results show a high annual variability across simulation start dates (box-plot widths), particularly for STR, which is mainly related to the high variability in annual precipitation. For the control scenario VS0, results estimate an average annual STR of

1112 mm/year and an ET of about 607 mm/year (Table 4.7). Although the differences are not statistically significant between VS0 and VS10, STR would decrease to 883 mm/year within 10 years of vegetation succession (VS10), which represents 12% of decrease in STR. However, for ET, a statistically significant ($p < 0.05$) increase in ET by 26% (766 mm/year under VS10) after 10 years of vegetation succession was found (Table 4.7 and Table 4.8). Beyond 20 years of vegetation succession, the differences to VS0 become statistically significant ($p < 0.05$) for both variables (Supplementary Tables 4.1 and 4.2). STR would decrease by more than 19% (around 815 mm/year) while ET increase by around 34% (813 mm/year) under the no clearing scenario (Tables 4.7 and 4.8 and Figure 4.7). Whereas, differences between the two projections VS20 and VS30 where not significant under both variables (Supplementary Tables 4.1 and 4.2).

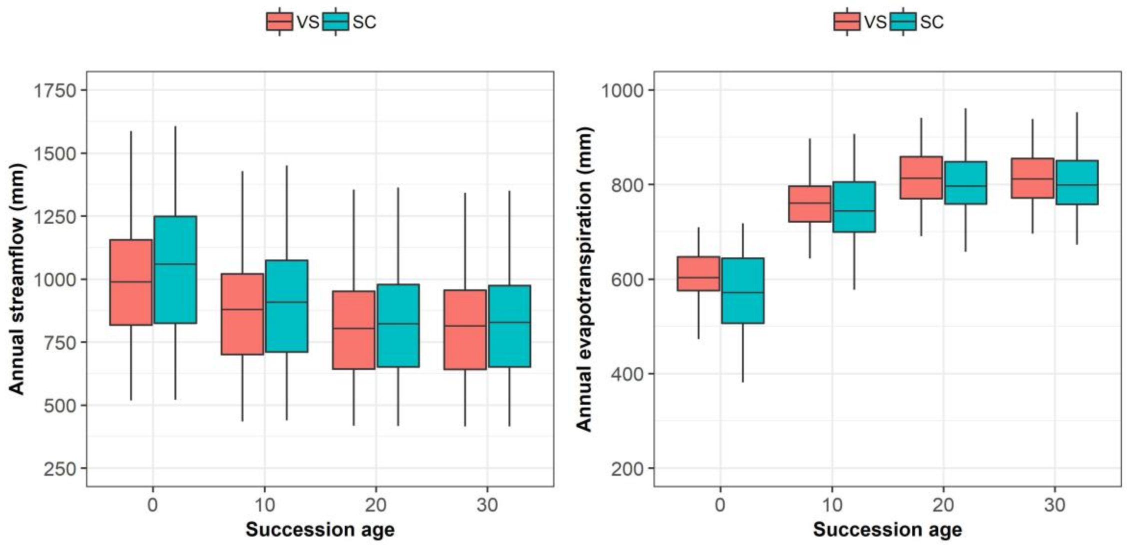

Figure 4.7. Boxplots of the resultant effect of vegetation succession (VS) and shrub clearing (SC) on annual streamflow (STR) and evapotranspiration (ET) under the different succession ages. Median value (the line in the middle of boxes), 1st and 3rd quartiles (lower and upper box boundaries respectively), whiskers (hanging bars).

The differences between the clearing and the no clearing scenarios are not statistically significant (Supplementary Tables 4.1 and 4.2). This could be mainly due to the total limited target area for shrub clearing compared to the surface of the study area. Even though the differences are not statistically significant, important changes are found between the clearing and the no clearing scenario particularly under the control scenario. Results estimate an STR of 1073 mm/year and an ET of 570 mm/year under the control scenario SC0 (Table 4.7). This represents a 6% increase in STR and a 6% decrease in ET from the no clearing scenario VS0 (Table 4.9). However, beyond 10 years of vegetation succession, a statistically significant ($p < 0.05$) decrease in the effects of shrub clearing is shown between the three clearing projections (SC10, SC20, SC30) and the control scenario SC0 (Supplementary Tables 4.1 and 4.2). Shrub clearing would only result in a 3% increase in STR and a 2% decrease in ET (Table 4.9) under SC10. This corresponds to a 15% lower STR (910 mm/year) and a 30% higher ET (745 mm/year) than a shrub clearing scenario SC0 (Tables 4.7 and 4.8). Beyond 20 years of vegetation succession, shrub clearing effect would drop to only 2% of increase in STR and a 1% decrease in ET (Table 4.9). This represents 22% lower STR (831 mm/year) and around 40% increase in ET (803 mm/year) from the SC0 situation (Tables 4.7 and 4.8).

Table 4.7. Average annual streamflow (STR) and annual evapotranspiration (ET) for the different projections under clearing and no clearing scenarios. VS0, VS10, VS20 and VS30: represent vegetation succession projections. SC0, SC10, SC20 and SC30: represent shrub clearing projections.

	No clearing			Clearing	
	STR (mm)	ET (mm)		STR (mm)	ET (mm)
VS0	1011.8	607.6	SC0	1073.2	570.5
VS10	882.6	765.6	SC10	910.9	745.2
VS20	816.4	812.4	SC20	831.7	803.2
VS30	814.2	814.5	SC30	831.3	803.9

Table 4.8. Percentage decrease in annual streamflow (STR) and increase in annual evapotranspiration (ET) from the VS0 and SC0 situations respectively for the different projections under the clearing and no clearing scenarios. VS0, VS10, VS20 and VS30: represent vegetation succession projections. SC0, SC10, SC20 and SC30: represent shrub clearing projections.

	No clearing			Clearing	
	STR	ET		STR	ET
VS10	-12.8	26.0	SC10	-15.1	30.6
VS20	-19.3	33.7	SC20	-22.5	40.8
VS30	-19.5	34.1	SC30	-22.5	40.9

Table 4.9. Average difference between clearing and no clearing under the different vegetation succession conditions in average annual streamflow (STR) and annual evapotranspiration (ET). SC0, SC10, SC20 and SC30: represent shrub clearing projections.

Year	STR (%)	ET (%)
SC0	6.1	-6.1
SC10	3.2	-2.7
SC20	1.9	-1.1
SC30	2.1	-1.3

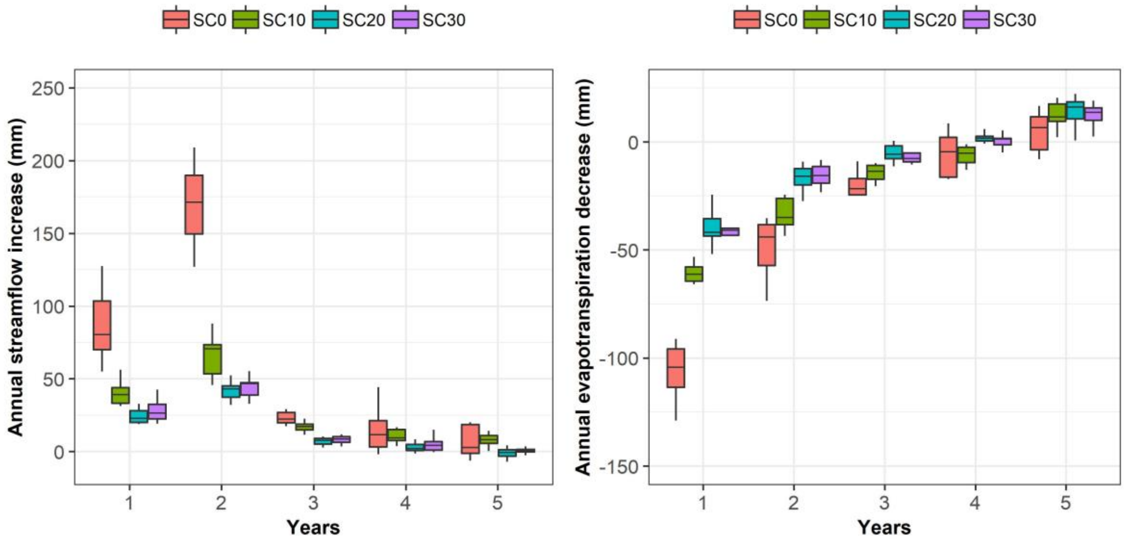

Figure 4.8. Boxplots of the resultant effect of shrub clearing on STR and ET under the different succession projections. Median value (the line in the middle of boxes), 1st and 3rd quartiles (lower and upper box boundaries respectively), whiskers (hanging bars). VS0, VS10, VS20 and VS30: represent vegetation succession projections. SC0, SC10, SC20 and SC30: represent shrub clearing projections.

Figure 4.8 represents the general temporal pattern of changes to STR and ET following shrub clearing, and any subsequent regrowth. The highest increase in STR due to shrub clearing would be recorded in the second year with a smaller increase during the first year. For ET, the highest decrease is observed during the first year after clearing. During the end of the fourth year and beginning of the fifth year following shrub clearing, STR and ET return to the situation before vegetation removal. Streamflow changes are largely related to the accumulation of the highest flows during the early wet season, while ET is concentrated during the late dry season (Figure 4.9). Thus, the highest increases in STR are delayed until the second year of management, when the first post-clearing wet season occurs; while the greatest reduction in ET occur during the first summer.

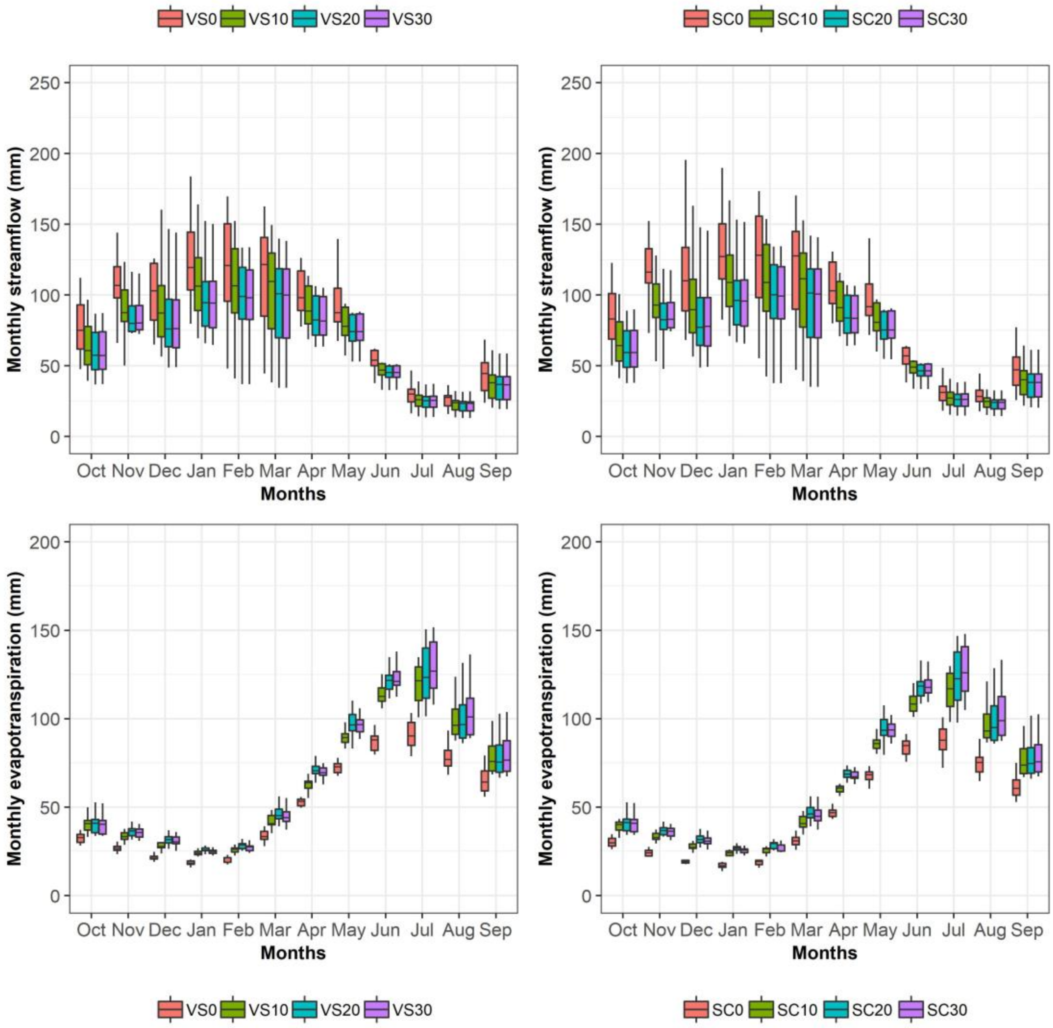

Figure 4.9. Monthly streamflow and evapotranspiration under the different succession projections and the management scenarios. Median value (the line in the middle of boxes), 1st and 3rd quartiles (lower and upper box boundaries respectively), whiskers (hanging bars). VS0, VS10, VS20 and VS30: represent vegetation succession projections. SC0, SC10, SC20 and SC30: represent shrub clearing projections.

4. Discussion

4.1. Changes to land cover due to passive management and effects on shrub clearing efficiency.

Projections for a passive management scenario, allowing secondary vegetation succession show important land cover changes over the Estarrún valley. We present a forecast for an upcoming 30-years of vegetation succession (note that the process of secondary succession started during the late 50ties). Our results point to an important colonization of cropland and pasture abandoned areas by shrubs and trees. This is consistent with the general agreement in scientific literature about the strong increase in vegetation cover over northern Spain during the past decades (FAO, 2010; Vicente-Serrano et al., 2005; Hill et al., 2008; Julien et al., 2011) and an expected pattern to increase during the upcoming decades (Verburg and Overmars, 2009; Lasanta et al., 2017). Model estimates for conversion from shrubs to forest suggest conversion of almost all the shrubs existing under the control scenario VS0 within 30 years. Molinillo et al. (1997) summarized the succession process in the Pyrenees in three general stages: (1) a first invasion by herbaceous plants within the first 0 to 5 years of abandonment, (2) woody shrub cover take place over the herbaceous colonized areas, and (3) a final stage toward a climax ecosystem defines the beginning of the appearance of trees and the retraction of the shrub cover. This is generally consistence with our finding giving the age of abandonment in the study area that dates before 1960. In that sense, Lasanta et al. (2005) estimated 100 years' period of succession for an abandoned area to reach a forest stage. This estimation does not contradict our results given that our projections of succession is only based on the VS0 above ground biomass of forest and shrubs.

The decrease in the shrub cover and the conversion to a tree vegetation cover would evidently limit the surface of the target areas for shrub clearing. Our projections for the shrub clearing scenario, if the succession continues, show a strong decline in the target area for shrub clearing that could reach 72% lower than the current situation. This could be mainly related to a final stage in the succession process characterized by a quick colonization of the abandoned areas by trees in the Estarrún watershed. This acceleration has been already perceived in the first stages of the process of plant succession in the region (Pueyo and Beguería, 2007; Komac et al., 2013). The reduction in the target area for shrub clearing with the succession process is an alarming signal for stakeholders to take measures toward a sustainable hydro-ecological ecosystem.

4.2. Hydrological implications of passive and active management in abandoned lands.

From the hydrological side, our results show a progressive decrease in STR due to the natural revegetation process reaching more than 19% within 30 years of succession. The decrease in STR is related to a strong increase in ET rates in the study area under the no clearing scenario. In fact, we estimate a 34% increase in ET during the upcoming three decades. These findings are in accordance with several studies that have analyzed the effects of vegetation succession process on water resources in Mediterranean environments. For instance, López-Moreno et al. (2014) concluded that an increase in forest cover in the Pyrenees could decrease annual streamflow by 16%. Beguería et al. (2003) analyzed the evolution of the flows of the Pyrenean rivers between 1945 and 1995, pointing out a decrease in the flows that they attribute to the land abandonment and the subsequent plant succession. Morán-Tejeda et al. (2010) observed a negative trend in the flows of the tributaries of the Duero river, especially in winter and spring as a consequence of the increase in vegetation. López-Moreno et al. (2011) studied the climatic and hydrological trends of 88 sub-basins of the Ebro Basin during the period 1950-2006. The results showed the decrease of the flows in most of the sub-basins, which was attributed to the combination of climate change and revegetation due to land abandonment. Some studies attribute, this decrease in STR to the parallel increase of ET as tree covers have relatively higher ET rates than most other land cover types (Calder, 1998). Nevertheless, the impact of forest cover on water yield is still debatable between researcher working at local scales and others modeling larger spatial scales (Ellison et al., 2012). At local scales, several studies related the increase of tree cover to an increase in vegetation interception and ET leading to reduction in runoff and a decline in water availability, particularly in areas of no forest cover for long period of time (Andréassian, 2004; Brown et al., 2005; Jackson et al., 2005; Malmer et al., 2009; Rosenqvist et al., 2010). Whereas at regional to global scales, some authors suggest that the overall role of forest cover is improving water availability by contributing to rise atmospheric moisture vapor and increasing the likelihood of precipitation events and sustaining runoff (Millán et al., 2005; Makarieva et al., 2006; Makarieva and Gorshkov, 2007).

Projections from the shrub clearing scenario in this study reflect the local scale of interaction between forest and water availability. At this local scale, shrub clearing was found to increase STR and decrease ET by 6% on both sides during the current situation of vegetation succession if only 7.5% of the study area is cleared. Similar results were found in Khorchani et al. (2020b) removing shrubs from 15% of a small catchment in the Central Spanish Pyrenees. On this point, local and global opinions agree that these changes to water cycle are temporary until a new forest cover is established (Ellison et al., 2012, a review). However, if vegetation succession continues and the management is delayed, changes to the efficiency of shrub clearing on STR and ET would occur. Results from the shrub clearing projections show a statistically significant decrease in the effects of shrub clearing on STR and ET compared to the control scenario SC0. This is mainly due to the decline in the available area for shrub clearing as result of the colonization of the abandoned fields by tree cover. The available area for shrub clearing is crucial in the magnitude of the changes to "benefits" from the shrub clearing practices. In fact, the small area available for shrub clearing in this study could explain the non-significant differences between the clearing and the no clearing scenarios. This is in part is due to the advanced stage of succession in the study area as most of the abandoned cropland area achieved a tree cover stage. Several studies pointed out the significant effect of shrub clearing as a low cost treatment technique to reduce fire hazard in Mediterranean environments (Madrigal et al., 2017; Lasanta et al., 2018). Thus, the reduction in the available area for shrub clearing could also have drastically increase wildfire risks and the economic cost for its management. Hence, these findings urge environmental policy makers to establish a convenient land management plans for the abandoned rural areas in the Pyrenees before a drastic shift in the efficiency of these management practices.

Our results also emphasize that timing of hydrologic changes related to shrub clearing. At annual scale, there is a global agreement that the magnitude of changes in water yield typically diminish rapidly in years following disturbance and the highest changes to water yield occur during the first five years following vegetation removal (Brown et al., 2005; Peel, 2009; Tague et al., 2019). This is consistent with our results giving that the effects to STR and ET progressively return to initial yield during the fourth or fifth year. It should be also highlighted, at annual scale, that the slight increase in STR and the highest increase in ET observed during the first year; while the highest STR change is reached during the second year after management. These patterns are related to offsets in the seasonal timing of STR generation and ET. This is crucial to the management of the water hydrostructure that is designed and operated to capture the increase in water yield (Khorchani et al., 2020b).

4.3. Management considerations

Our study found that passive management (letting the process of secondary succession continue) reduces STR and increases ET, while shrub clearing in selected areas leads to the opposite effect. However, it can be considered that the results obtained in this work regarding the increase in water yield due to shrub clearing are moderate: 6% if 7.5% of the study area is cleared. This is mainly due to the small area that meets the requirements for the shrub clearing practice established in the La Rioja community. Additionally, this area will substantially decline if the vegetation succession process continues, resulting in a lower efficiency of the shrub clearing practice.

The advanced stage of succession in the study area (since the late 50ties) and the relatively steep slopes are the main limiting factors in selecting areas for shrub clearing in the Estarrún valley. This could be solved if the 30% slope criterion is reconsidered in areas of low erosion risks and shrub clearing is combined with forest management practices such as forest thinning and clear cutting in selected areas.

5. Conclusions

In the Estarrún watershed, natural revegetation processes have occurred since the onset of land abandonment in the early part of the 20th century. Our ecohydrologic model projects that the revegetation processes are likely to continue and intensify in the next decades. We estimate that an uninterrupted revegetation process in the Estarrún watershed would result in the conversion of most of the shrubland areas to forest within the next three decades. This succession to increased forest cover will also have substantial hydrologic impacts strongly decreasing STR (up to 19%) and increasing ET (up to 34%) within three decades. Shrub clearing can slow this successional process but increasing tree cover as succession advances will likely limit the target areas for shrub clearing under current criteria (less than 30% slope and only shrub areas) and reduce the effect of the shrub clearing on STR and ET. Therefore, the delay in managing abandoned cropland areas in the Central Spanish Pyrenees could limit the efficiency of the shrub clearing practice in these areas. In this case, land managers would need to resort to high cost management practices, such as forest thinning and clearcutting, to ensure water availability. These results urge environmental policy makers to decide on the management of the abandoned cropland areas in the Mediterranean mountains considered of crucial economic, environmental and social interest. This study provides important outcomes on this “hot topic”, whereas further work is fundamental to control all the aspects of the management of cropland areas, particularly climate change.

References Chapter 4

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Chapter 5. Natural revegetation and afforestation in abandoned cropland areas: Hydrological trends and changes in Mediterranean mountains

This Chapter studies the long-term hydrological trends of passive and active restoration plans in the abandoned cropland areas in the Central Spanish Pyrenees. RHESSys was used to model the first five decades following abandonment, through natural revegetation or afforestation. Changes to annual and seasonal streamflow and transpiration are studied and related to changes in vegetation physiological attributes.

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Abstract

Water resources availability is one of the main concerns for policy makers around the world in present and future management plans. In the Mediterranean basin, this concern is increased given the extreme variability in climate and the intrinsic aridity conditions. Water resources in the Mediterranean region depend mainly on surface and subsurface supply from mountain areas. Because evapotranspiration comprises a substantial portion of the water budget, recent land cover changes due to cropland abandonment may change transpiration (TRANS) and water supply. Therefore, land management plans must account for these potential hydrologic changes to guarantee water availability in the upcoming decades. Short-term changes to water yield have been shown to follow afforestation or natural revegetation, the main management strategies in abandoned cropland areas. Studies comparing long-term trends of these management practices, however, are scarce due to the lack of long-term hydrological data. In this study we use the Regional Hydro-Ecological Simulation System (RHESSys), to analyze long-term changes and annual and seasonal trends in streamflow (STR) and transpiration following management of abandoned cropland areas. Annual mean values show significant differences between the three management scenarios for both streamflow and transpiration, while differences between climate scenarios are not significant. The Mann Kendall trend analysis shows significant changes to water yield compared to the situation before management. Depending on the total afforested area, afforestation could significantly decrease annual streamflow between 2.3%·decade^{-1} and 5.9%·decade^{-1} and increase annual transpiration between 1.1%·decade^{-1} and 3.5%·decade^{-1}. These trends are attributed to changes during the first 30 years after management, while during the fourth and fifth decade, changes to water yield tend to stabilize or decrease. These results are substantial to optimize land management plans, ensuring sustainable hydrological and ecological ecosystem services.

1. Introduction

The availability of water resources and their quality constitutes, possibly, the most important environmental problem in many countries around the world, including those of the Mediterranean basin (García-Ruiz et al., 2011). In fact, the Mediterranean basin is considered a "hot spot" on a global scale due to: (i) the scarcity of water resources (Giorgi, 2006), (ii) climate change (increased temperatures, less snow accumulation and greater atmospheric evaporative demand) (Gónzalez-Hidalgo et al., 2016; López-Moreno et al., 2008; Polade et al., 2017; Vicente-Serrano et al., 2014), and (iii) the significant expansion of the vegetation cover during the last century (Chauchard et al., 2007; García-Ruiz and Lana-Renault, 2011; Sitzia et al., 2010; Vicente-Serrano et al., 2020; Vicente-Serrano and Heredia-Laclaustra, 2004). Water resources are a limitation for economic development in the Mediterranean basin and are an essential factor to satisfy the increasing needs of society, as a consequence of the increase in population, urbanization and living standards (García-Ruiz et al., 2011), the development of irrigated agriculture (Iglesias et al., 2011; Jlassi et al., 2016) and industrial and tourist activities (Morote et al., 2016; Rico-Amoros et al., 2009).

Water resources in Mediterranean regions depend mainly on mountain areas, which act as “moisture islands” in the middle of drier areas (de Jong et al., 2009; Immerzeel et al., 2020; Viviroli et al., 2011; Viviroli and Weingartner, 2004). However, during the last century, Mediterranean mountains, especially those in Europe, have experienced important changes in land use and cover due to depopulation and the collapse of primary activities (García-Ruiz and Lasanta-Martínez, 1990; Price, 2004). From a hydrological perspective, the abandonment of agricultural lands and pastures is one the most important features of these changes (García-Ruiz et al., 2020; Lasanta, 1988; Macdonald et al., 2000), which triggered important revegetation processes on the slopes (García-Ruiz and Lana-Renault, 2011; Poyatos et al., 2003; Sanjuán et al., 2018).

On the other hand, active afforestation carried out by land administrations also constituted a major feature of land cover changes during the last century. In Spain, for example, in 1939 an important afforestation plan (Plan de Repoblación Forestal de España) proposed the afforestation of 6 million hectares in 100 years (Ortigosa Izquierdo, 1991). Afforestation pursued two objectives: (i) economic: increasing forest production to meet the growing industrial demand, and (ii) environmental: reducing soil erosion and regulating the hydrology of slopes (Ortigosa Izquierdo, 1991; Vallejo et al., 2006). This plan had already reached 5 million hectares by 2006 (Lasanta et al., 2015).

The combination of natural revegetation and active afforestation have led to a substantial increase in forest cover in Spain during the last decades. In fact, the forest area increased by 130,000 km^2 in 45 years (MAPA, 2020). At present, more than 55% of the national territory is covered by trees and shrubs, concentrated particularly in mountain areas (Vicente-Serrano et al., 2020). Several studies relate the progressive decrease in river flows, particularly in mountain areas, to this increase in vegetation and its impact on interception, infiltration, runoff and river-slopes connectivity processes (Beguería et al., 2003; Germer et al., 2010; Muzylo, et al., 2012). At the same time, climate change has had significant impacts on water resources in Mediterranean mountains (García-Ruiz et al., 2011) through changing precipitation patterns and atmospheric energy demand (Trenberth, 2011). These environmental changes can combine to lead to significant decrease in river flows and runoff coefficients (López-Moreno et al., 2008; Martínez-Fernández et al., 2013; Morán-Tejeda et al., 2014; Polade et al., 2017), and have already forced the adaptation of reservoir management, limiting water discharge and using multi-year impoundment cycles (López-Moreno et al., 2008; Zabalza-Martínez et al., 2018). These changes are expected to intensify in the coming decades. López-Moreno et al. (2014) for instance, projected that annual flows from Pyrenean rivers will decrease by 2050 between 13% and 23%, depending on location and forest regeneration patterns. In spite of the impacts to water resources, there is science-based support for natural revegetation process that highlight the important benefits of forest regeneration for biodiversity and ecosystem services (Perino et al., 2019). This support underlines promoting natural succession or passive revegetation (rewilding) (Navarro and Pereira, 2012) and active revegetation in abandoned cropland areas (Benayas and Bullock, 2012).

Future water management plans in these regions must take into account not only water demand, but also how climate and revegetation may change available water. Interactions between land use and climate change and their combined impacts on water resources are of valuable interest to hydrological decision makers, particularly in rapidly changing Mediterranean environments (Beniston, 2003; Nogués-Bravo et al., 2008; Viviroli et al., 2011). Deepening knowledge on these interactions can also enhance land management planning to ensure the sustainability of water resources. In this sense, a very important unresolved question is: what is the long term (next few decades) impact of passive and active revegetation of abandoned lands on vegetation water use and surface water supply? To answer this question this study uses a well-established ecohydrologic model (Regional Hydro-Ecological Simulation System (RHESSys) (Tague and Band, 2004), to estimate long term annual and seasonal trends and changes to streamflow (STR) and transpiration (TRANS) following management of abandoned cropland areas under both historic and a warming climate scenario. To parametrize RHESSys, we used data from the monitoring of a small catchment in the Central Spanish Pyrenees (Arnás) representative of the process of cropland abandonment.

2. Materials and methods

2.1. Study area

Arnás is a small catchment (0.284 km^2) of the upper Aragón river located in the Central Spanish Pyrenees (Figure 5.1). The lithology is Eocene Flysch with sandstones and marl layers alternating and slopping northward (Lana-Renault et al., 2007). The ravine, with a west-east orientation, separates a strongly contrasted north-facing and south-facing slopes. Poor eroded soils characterize the steep south-facing slopes. While, soils in the gentler north-facing slopes are deep and well developed (Navas et al., 2005; Seeger et al., 2005).

The Arnás catchment is a good example of the process of cropland abandonment in Mediterranean mountains. After being highly cultivated with non-irrigated cereals in the 19th century including the steep slopes, Arnás has been progressively abandoned during the first half of the 20th century. Following abandonment, a process of secondary succession of shrubs followed by trees (approximately 100 years following abandonment) took place and a natural vegetation cover colonized most of the catchment. Since abandonment, no active management plans have been adopted in the catchment. Consequently, the present vegetation cover is the result of the natural revegetation process. Present vegetation cover consists mainly of *Pinus sylvestris* and *Quercus faginea* trees (particularly in the north-facing slopes in the first abandoned fields) and *Genista scorpius, Buxus sempervirens, Rosa gr. canina, Juniperus communis* and *Echinospartum horridum* covering most of the catchment. Since 1996, the Pyrenean Institute of Ecology (IPE-CSIC) is monitoring Arnás through the installation of a meteorological station, a gauging outlet and three pluviometers in the catchment (Figure 5.1).

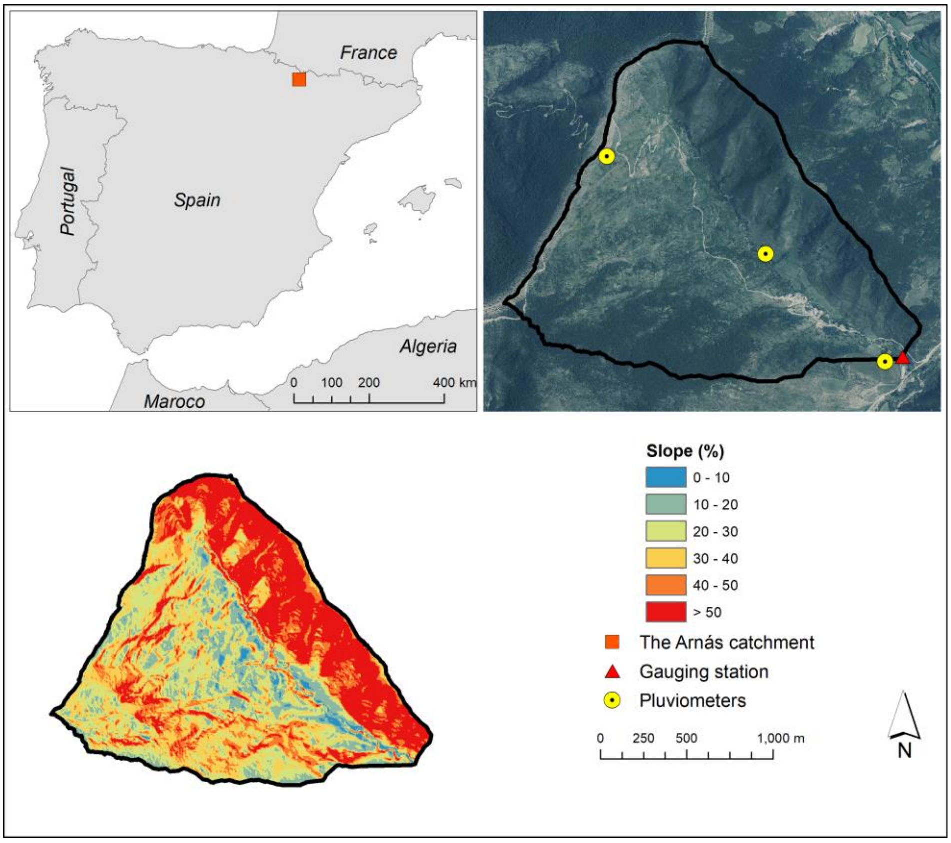

Figure 5.1. The Arnás catchment. Top left: Localization of the Arnás catchment in Spain. Top right: Aerial photography of the catchment (2015). Bottom left: Slope map of the catchment.

2.2. Management of abandoned cropland areas and study scenarios

We develop three land management scenarios that are used to parameterize our ecohydrologic model. All scenarios begin from an initial scenario that follows the historic land abandonment prior to 1960 (Lasanta, 1988). To define this initial state, we used an aerial photography of the catchment from 1957 and developed a land cover map. We then defined the three simulated scenarios (Figure 5.2, Table 5.1). These three scenarios represent three theoretical management trajectories of Arnás catchment following abandonment.

Table 5.1. Vegetation cover under the different management scenarios.

	Scenario 1	Scenario 2	Scenario 3
Target areas for afforestation	-	30.1	61.1
Forest land	3.2	3.2	3.2
Shrub land	90.7	61	30.5
Grass land	6.1	5.7	5.2

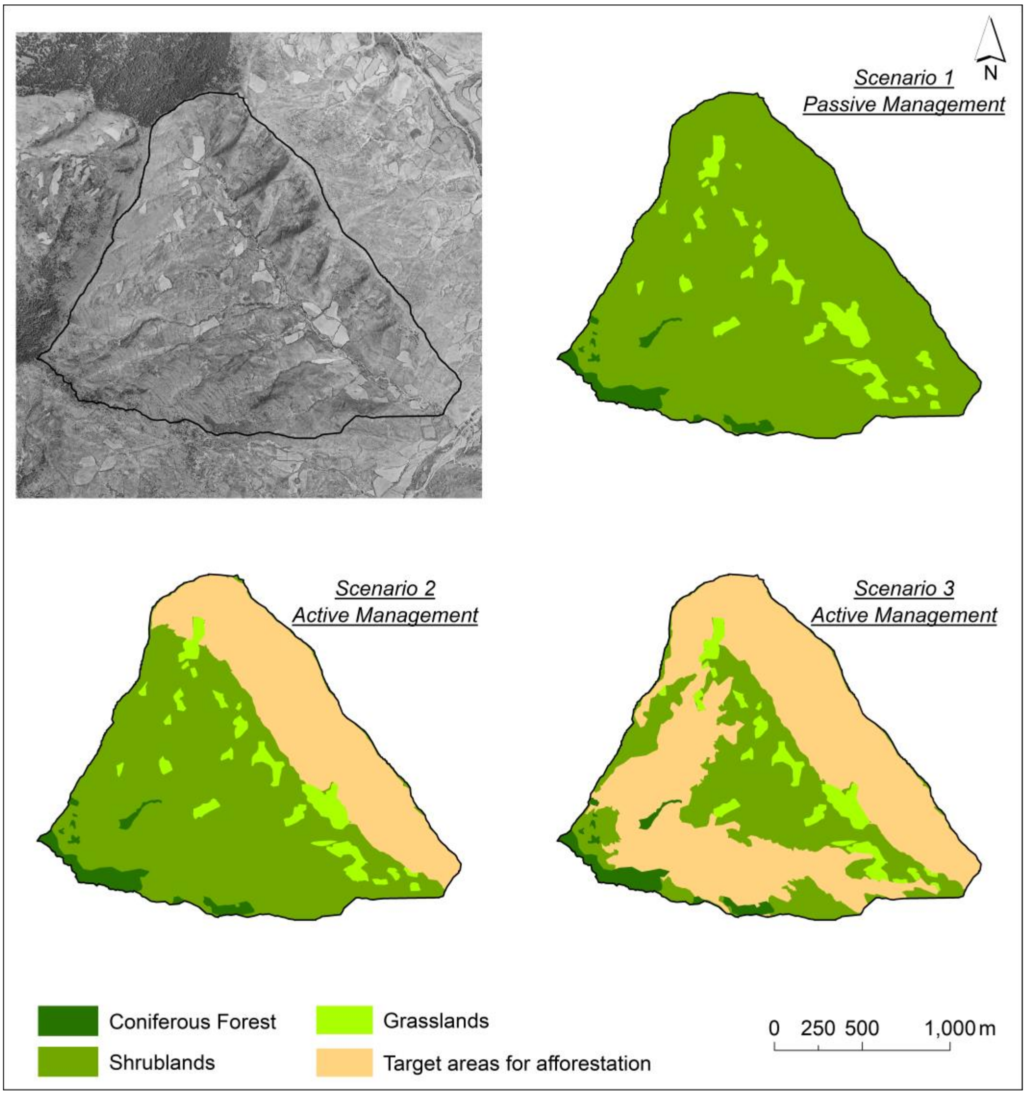

Figure 5.2. Theoretical studied land cover scenarios for the Arnás catchment. (i) Scenario 1. Passive management through shrub expansion in most of the study area; (ii) Scenario 2. Active management scenario through afforestation of areas of high erosion risk in the south-facing slopes; (iii) Scenario 3. Active management scenario through afforestation of areas of high erosion risk in both north-facing and south-facing slopes.

<u>Scenario 1 (PM and PM.W)</u>: representing early successional passive management allowing a natural revegetation by shrubs of the catchment following abandonment. Scenario 1 is a representation of the beginning of the colonization process by shrub species (*G. Scorpius*, *B. sempervirens*, *E. horridum*…) before the entry of a secondary forest. We note that the actual Arnás was abandoned in stages so a transition from shrub to forest occurred at different time in different parts of the catchment. To simplify here we use this to represent a shrub scenario to compare with afforestation catchments where trees are actively introduced.

Scenario 2 (AFF1 and AFF1.W): representing active management through afforestation of areas of high erosion risk (> 30% of slope) in the south-facing slopes while the rest of the catchment is left to undergo natural revegetation by shrubs. The target areas for management represent 30% of the study area in this case (Table 5.1). The gentler slopes and the deeper soils developed on the north-facing part of catchment would enhance a slow natural regeneration of a tree cover without management around 100 years after abandonment (Gibon et al., 2010; Pueyo and Beguería, 2007).

Scenario 3 (AFF2 and AFF2.W): Also representing active management, but in this case, all areas of high erosion risk in the study area are afforested. While the rest of the catchment is left to natural revegetation by shrubs. The target areas for management in this scenario represent 60% of the study area (Table 5.1).

To study the effect of climate change on the trends of STR and TRANS, we coupled these three scenarios with a warming scenario. The European Environmental Agency (EEA, 2020) and the Pyrenean Climate Change Observatory (OPCC, 2018) predicted an increase between 1-2 °C by 2030 and 2.5-5 °C by the end of this century in the Pyrenees with respect to the 1961-1990 and 1971-2000 reference periods respectively. Nevertheless, projections for the precipitation are found to be non-significant during the 21st century given that the number of models predicting an increase in precipitations is similar to those predicting a decrease (OPCC, 2018). To translate these climate projections in our climate warming scenario, we considered an increase of 3 ºC (as a midpoint between the lowest and the highest projections) in maximum and minimum temperature. Whereas, due to the uncertainty in projections, no precipitation change was applied to the climate warming scenario to avoid altering model results.

2.3. RHESSys Model

The Regional Hydro-Ecological Simulation System (RHESSys) model (Tague and Band, 2004) was used to simulate the different management scenarios. RHESSys is a dynamic process-based model that calculates carbon, nitrogen and water fluxes over variable terrain from local to regional spatial scale. It was developed to study the effects of land cover and climate changes on the hydrological and ecological processes (Tague and Band, 2004). RHESSys uses a hierarchical representation of the landscape that subdivides ecohydrological processes into different spatial units. The hierarchical representation of the landscape, enhances the ability of the model to represent both plot scale processes (at patch level) and hillslopes spatial heterogeneity, accounting for interactions between the smallest spatial units through downslope redistribution. Merging local process and landscape spatial heterogeneity enhances the sensitivity of the model to changes in ecohydrological processes due to climate and land cover changes.

RHESSys partitions precipitation to snow and rain based on air temperature. Total evapotranspiration fluxes, including evaporation of intercepted water by the vegetation and transpiration from canopy layers, are modeled using the Penman-Monteith equation (Monteith, 1965). The intercepted water by the vegetation is modeled as a function of the size and the type of the vegetation. Surface and subsurface hydrological processes include vertical and lateral moisture fluxes. Vertical fluxes include the infiltration of water to soil and its drainage through rooting and unsaturated soil layers. Lateral fluxes may drain saturated water to reach the main stream via surface and subsurface water flow or groundwater flow. Vertical and lateral water fluxes are controlled by the vertical hydraulic conductivity profile. In RHESSys, the saturated hydraulic conductivity is computed as: K_{sat} (z) = $K_{sat0} e^{(-\frac{z}{m})}$

Where z is soil depth, K_{sat0} is hydraulic conductivity at the surface and m represents the decay of conductivity with soil depth.

RHESSys model vegetation growth and mortality through a carbon cycling model that includes estimates of photosynthesis, respiration and carbon allocations to leaves, stems and roots. Details on the modeled processes in RHESSys model can be found in (Tague and Band, 2004).

The calibration and validation of RHESSys have been performed using data records from meteorological and gauging stations installed in the Arnás catchment since 1996 by the Pyrenean Institute of Ecology (IPE-CSIC). For details on the calibration and validation process see Khorchani et al. (2020). Information and guidelines on RHESSys model can be found in https://github.com/RHESSys/RHESSys/wiki.

We parameterized the vegetation in this study as standard classes of conifers, shrubs and grasses using the 1957 land cover map. Parameters of these standard vegetation classes can be found in https://github.com/RHESSys/RHESSys /wiki/Parameter-Definition-Files. Before simulating management scenarios, a first 100-year spin-up simulation followed by vegetation stores removal, was conducted to initialize soils carbon and nitrogen stores. All the simulations are run with RHESSys model in dynamic mode which simulates vegetation growth throughout time.

2.4. Climate record and analysed variables

The climate record from the Arnás monitoring station barely covers 20 years due to high data gaps and instrument problems. To generate a long term data record, for a better analysis of the hydrological trends, several meteorological stations from the Spanish Meteorological Agency (AEMET) and the Pyrenean Institute of Ecology (IPE-CSIC) in a radium of 10 km from the Arnás catchment and at a similar altitude, were used. We followed Vicente-Serrano et al. (2010) and El Kenawy et al. (2012) for the homogenization and gap filling processes. 50 years of daily precipitation, maximum and minimum air temperature were used for the simulations.

To address the effects of management and climate scenarios on the trends of STR and TRANS and reduce the effect of annual and seasonal precipitation variability on the interpretation of results, the trend analysis used normalized STR and TRANS by total precipitation. STR and TRANS are calculated as follows:

$$\text{Normalized STR}_{\text{(Annual, Seasonal)}} = \frac{STR}{Precipitation}_{\text{(Annual, Seasonal)}}$$

$$\text{Normalized TRANS}_{\text{(Annual, Seasonal)}} = \frac{TRANS}{Precipitation}_{\text{(Annual, Seasonal)}}$$

Changes to vegetation biomass, Leaf Area Index (LAI) and rooting depth were also analyzed to study potential effects on waters fluxes trends.

2.5. Statistical analysis

In this study we analyze 50 year trends in STR and TRANS for annual, wet and dry seasons - across 3 management scenarios with both historic and warmer climates for a total of 6 scenarios (3 * 2). For each scenario, we compute annual averages and trends.

The Tukey's Honestly Significant Difference test (TukeyHSD) (Tukey, 1949) was implemented to compare differences between mean annual STR and TRANS under the different management and climate scenarios.

The non-parametric Mann Kendall trend test (Kendall, 1975; Mann, 1945) was used to test the significance of the trends in STR and TRANS. Being a non-parametric test enhances the robustness of the Mann Kendall test, since it is not sensitive to outliers and no prior distribution of the data is assumed (Lanzante, 1996). The significance of the trends in this study was assessed at a 95% of confidence level. The non-parametric Sen's slope (Sen 1968) estimator was used to calculate the magnitude of change relative to each management and climate scenario.

Trends and magnitude of change were computed at annual and seasonal scales. Lana-Renault et al. (2007), using water-table data, classified seasonal hydrologic regimes in the Arnás catchment as follows: a wet period during winter and spring seasons and a dry period during summer months while a strong variability characterizes the transition between wet and dry conditions. STR data in the Arnás catchment similarly shows a strong seasonality with a wet winter and summer dry period (Supplementary Figure 5.1). Based on these definitions and to simplify the results, we defined two seasonal scales: a "wet period" from December to May (winter and spring) and a "dry period" from June to November (summer and autumn). These two periods were used to study the seasonal changes of water regimes due to management and climate scenarios.

We also computed a 30-year moving trend to study the stabilization of the trends under the different management and climate scenarios. Based on the age of management, three phases were defined: 0 to 30, 10 to 40 and 20 to 50 years of management.

3. Results

3.1. Annual Streamflow and Transpiration under management and climate scenarios

At annual scale, the results show high inter-annual variability particularly for STR with a coefficient of variation > 37% (Figure 5.3, Table 5.2). Significant differences between the three management scenarios are found for annual STR and annual TRANS. However, no significant differences between the climate scenarios (warming vs. no warming) are found for either STR or TRANS (Supplementary Table 5.1).

For STR, annual averages over the climate record (50 years), decrease drastically with afforestation scenarios (AFF1 and AFF2) compared to the control scenario (PM). Results show a decline in annual STR by 27% and 50% under AFF1 and AFF2 respectively compared to PM (Figure 5.3, Table 5.2). Whereas, under the warming scenario, the decrease in annual STR, compared to PM, is slightly higher under AFF1.W (28%) and

AFF2.W (53%) (Figure 5.3, Table 5.2). Differences between historic and warming climate scenarios show a slight decrease in annual STR due to temperature increase nevertheless these differences are non-significant (Table 5.2, Supplementary Table 5.1).

Table 5.2. Average annual streamflow (STR) and transpiration (TRANS) over the study period (50 years) under the different management and climate scenarios and the change to the control scenario (PM). Scenario 1 (PM and PM.W: Passive management), Scenario 2 (AFF1 and AFF1.W: Active management), Scenario 3 (AFF2 and AFF2.W: Active management). PM.W, AFF1.W and AFF2.W represent warming climate scenarios.

	Scenario 1		Scenario 2		Scenario 3	
	PM	PM.W	AFF1	AFF1.W	AFF2	AFF2.W
Annual Streamflow (mm)	584	570	428	418	293	272
Coefficient of variation (%)	37.8	37.1	48.9	47.9	66.6	66.8
Change to PM (%)	-	*-2*	*-27*	*-28*	*-50*	*-53*
Annual Transpiration (mm)	260	263	364	366	490	501
Coefficient of variation (%)	16.8	18.2	18.9	20	21.7	22.7
Change to PM (%)	-	*1*	*40*	*41*	*89*	*93*

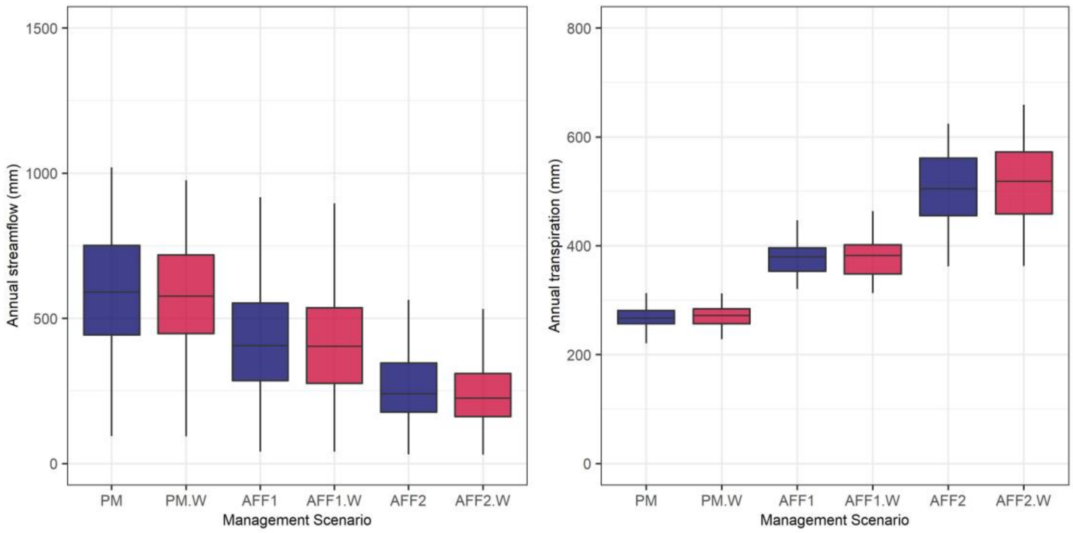

Figure 5.3. Boxplots of annual streamflow (STR) and annual transpiration (TRANS) under the different management and climate scenarios. Scenario 1 (PM and PM.W: Passive management), Scenario 2 (AFF1 and AFF1.W: Active management), Scenario 3 (AFF2 and AFF2.W: Active management). Blue and red colors correspond to the no warming and warming scenarios respectively. Median value (the line in the middle of boxes), 1st and 3rd quartiles (lower and upper box boundaries respectively), whiskers (hanging bars).

For annual TRANS, due to the lower inter-annual variability (less than 23%, Table 5.2), differences between the three management scenarios are clearer than STR (Figure 5.3). Significant differences are found between the three management scenarios, while differences between climate scenarios are non-significant (Table 5.2, Supplementary Table 5.1). Afforestation strongly increased annual TRANS with 40% and 89% higher values under both AFF1 and AFF2 respectively compared to PM. These differences are higher under the warming scenario (41% and 93%) (Table 5.2, Supplementary Table 5.1).

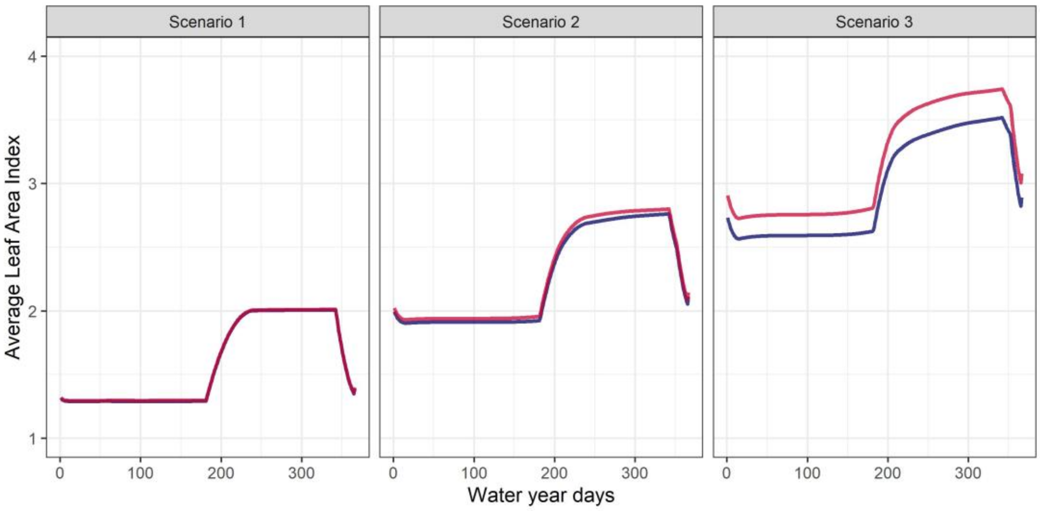

Figure 5.4. Average water year day Leaf Area Index under the different management and climate scenarios. Scenario 1 (PM and PM.W: Passive management), Scenario 2 (AFF1 and AFF1.W: Active management), Scenario 3 (AFF2 and AFF2.W: Active management). Blue and red colors correspond to the no warming and warming scenarios respectively.

Table 5.3 shows three structural vegetation variables that directly affect the hydrological fluxes (vegetation biomass, LAI, rooting depth). All of these variables are computed by RHESSys, evolving over time and respond to available energy and moisture. Comparing these three variables under the different scenarios helps to explain the difference in hydrological fluxes across scenarios. Compared to PM, afforestation scenarios presented a higher vegetation biomass (up to 127%), a more developed LAI (up to 99%) and a deeper rooting depth (up to 27%, though differences are not significant in this case) (Table 5.3, Supplementary Table 5.2). These differences between management scenarios contribute to greater TRANS (and correspondingly lower STR). However, vegetation structural variables do not show significant differences between historic and warmer climate scenarios (Supplementary Table 5.2). At seasonal scale, AFF3 and AFF3.W presented higher differences in LAI between historic and warming climate scenario compared to PM and AFF1 scenarios. However, the duration of the growing season was not changed by the increase of temperature, reflecting the dominance of water availability as the main control on growing season length (Figure 5.4).

Table 5.3. Average Vegetation biomass, Leaf Area Index and Rooting depth under the different management and climate scenarios over the study period (50 years) and percentage change to the control scenario (PM). Scenario 1 (PM and PM.W: Passive management), Scenario 2 (AFF1 and AFF1.W: Active management), Scenario 3 (AFF2 and AFF2.W: Active management). PM.W, AFF1.W and AFF2.W represent warming climate scenarios.

	Scenario 1		Scenario 2		Scenario 3	
	PM	PM.W	AFF1	AFF1.W	AFF2	AFF2.W
Vegetation biomass (kg C · m^{-2})	2.8	2.4	4.4	4.2	6.0	6.3
Change to PM (%)	-	*-13*	*59*	*51*	*118*	*127*
Leaf Area Index	1.6	1.6	2.3	2.3	3.0	3.2
Change to PM (%)	-	*0*	*42*	*45*	*87*	*99*
Rooting depth (mm)	851	754	985	919	1069	1083
Change to PM (%)	-	*-11*	*16*	*8*	*26*	*27*

3.2. Annual and seasonal trends of normalized Streamflow and Transpiration under management and climate scenarios

3.2.1. Effect of climate variability and trends

To make sure the effect of trends in the climate record on our results are minimum, we calculated trends and changes in precipitation, maximum, minimum and average temperature to study if the climate record already presented significant trends. Results show non-significant trends in precipitation in both annual and seasonal scales (Supplementary Figure 5.2). This confirms that recent trends in precipitation would not have significant impacts on STR. However, we note that future trends in precipitation may be more substantial and potentially accelerate declines. Maximum temperature showed significant decrease by 0.26 ºC·$decade^{-1}$ during the dry period, while during the wet period and at annual scale the trends are non-significant. Minimum temperature showed significant increase at annual and seasonal scale (0.2 ºC. $decade^{-1}$ at annual scale and 0.19 ºC·$decade^{-1}$ during the dry period) (Supplementary Figure 5.2). Nevertheless, trends are not significant for average temperature, which reduces the likelihood of substantial effects on the trends in STR and TRANS at annual and seasonal scales.

3.2.2. Annual trends and magnitude of change

At annual scale, the trend analysis of the different management and climate scenarios show significant changes to normalized STR and TRANS. For PM scenario, STR significantly decreases by 2.4%·decade^{-1} and 2.3%·decade^{-1} under historic and warming climates respectively. However, though trends are non-significant, TRANS decrease by around 1% under both climate scenarios (Figure 5.5, scenario 1).

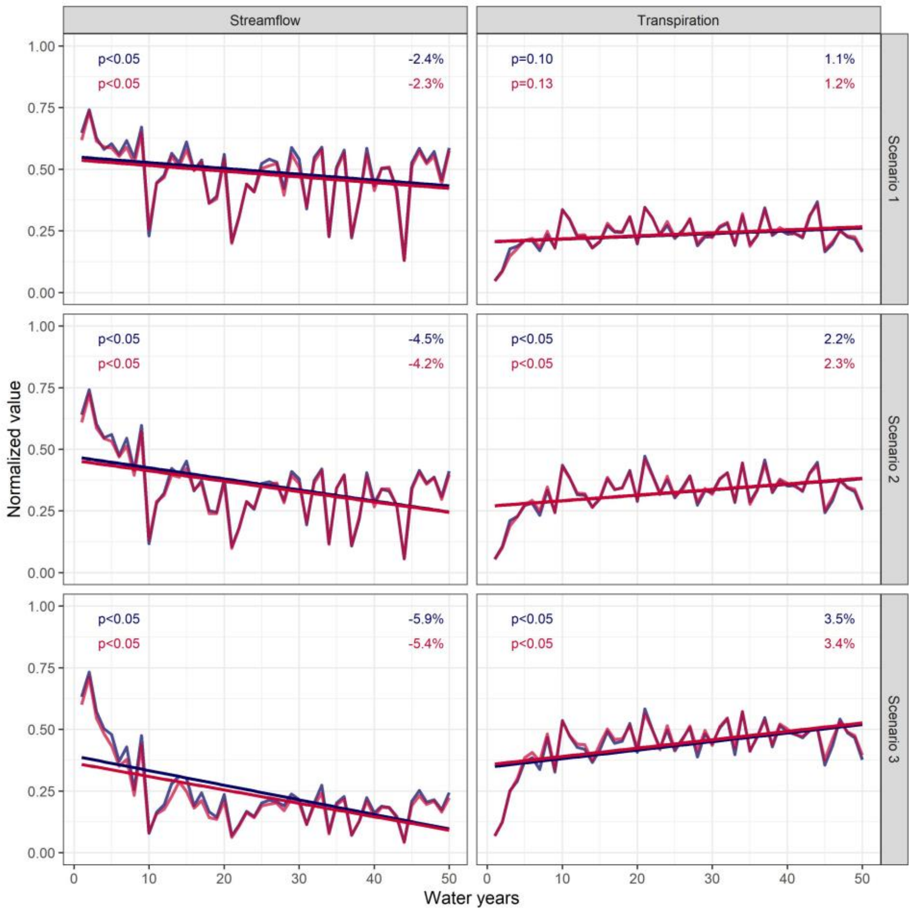

Figure 5.5. Trends and magnitude of change in annual normalized streamflow and transpiration under the different management and climate scenarios. Scenario 1 (PM and PM.W: Passive management), Scenario 2 (AFF1 and AFF1.W: Active management), Scenario 3 (AFF2 and AFF2.W: Active management). Blue and red colors correspond to the no warming and warming scenarios respectively. 95% level is used for the significance of the p-value of the Mann Kendall test. Note that percentages indicate the magnitude of change per decade.

Under AFF1 scenario, significant trends in STR and TRANS are found for both climate scenarios. For STR, the decrease is of 4.5%·decade^{-1}, under no warming effect, and of

4.2%·decade^{-1} under warming. TRANS shows an increase of 2.2%·decade^{-1} and 2.3%·decade^{-1} without and with warming effect respectively (Figure 5.5, scenario 2).

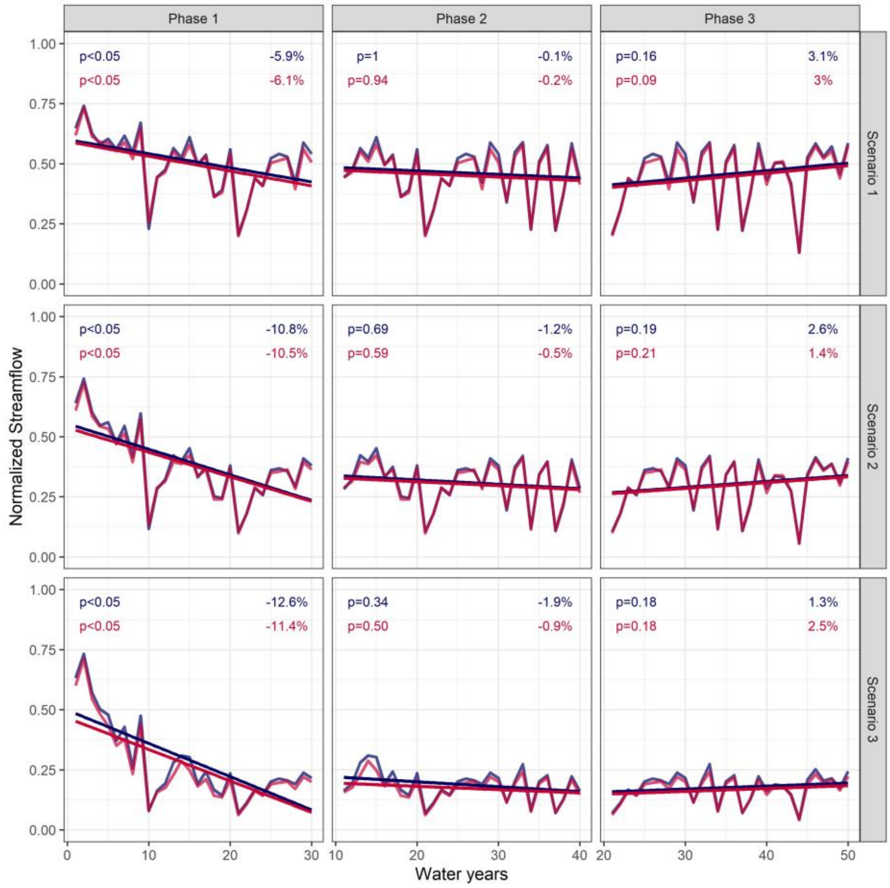

Figure 5.6. 30 years moving trend and magnitude of change in annual normalized streamflow under the different management and climate scenarios. Scenario 1 (PM and PM.W: Passive management), Scenario 2 (AFF1 and AFF1.W: Active management), Scenario 3 (AFF2 and AFF2.W: Active management). Blue and red colors correspond to the no warming and warming scenarios respectively. 95% level is used for the significance of the p-value of the Mann Kendall test. Note that percentages indicate the magnitude of change per decade.

The highest changes to STR and TRANS are found under AFF2 scenario. Results shows a strong significant decrease in STR by 5.9%·decade^{-1} under no climate change effect and by 5.4%·decade^{-1} under a 3 ºC increase in air temperature. While TRANS presented a significant increase by 3.5%·decade^{-1} under no warming effect and 3.4%·decade^{-1} under warming (Figure 5.5, scenario 3).

The significant annual trends are attributed mainly to changes occurring during the first phase of the study period (first 30 years of management) (see Figures 5.6 and 5.7). In fact, analyzing a 30-year moving trend of the normalized STR and TRANS in three management phases revealed only significant trends during the first phase while during the second and the third phases all the trends are non-significant for both variables under the different scenarios (see Figures 5.6 and 5.7). During phase 1, STR presented significant negative trends all management and climate scenarios. The magnitude of change in the significant trends oscillates between -5.9%·decade^{-1} to -12.6%·decade^{-1} (Figure 5.6).

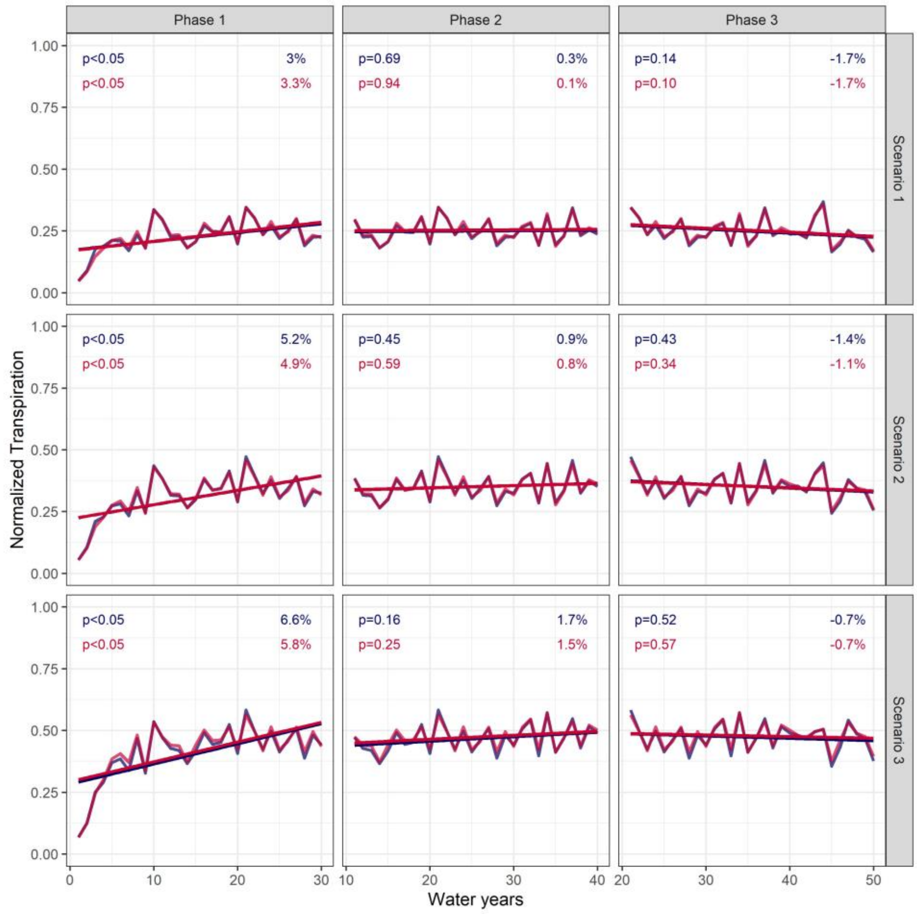

Figure 5.7. 30 years moving trend and magnitude of change in annual normalized transpiration under the different management and climate scenarios. Scenario 1 (PM and PM.W: Passive management), Scenario 2 (AFF1 and AFF1.W: Active management), Scenario 3 (AFF2 and AFF2.W: Active management). Blue and red colors correspond to the no warming and warming scenarios respectively. 95% level is used for the significance of the p-value of the Mann Kendall test. Note that percentages indicate the magnitude of change per decade.

Similarly to STR, significant trends in TRANS are only found during phase 1. Under the different management and climate scenarios, TRANS shows significant positive trends.

The magnitude of change in TRANS ranges between 3%·decade^{-1} to 6.6%·decade^{-1} (Figure 5.7). During phase 3, although the trends are non-significant, a clear increase in water yield is observed, marked by higher STR and lower TRANS values (Figures 5.6 and 5.7).

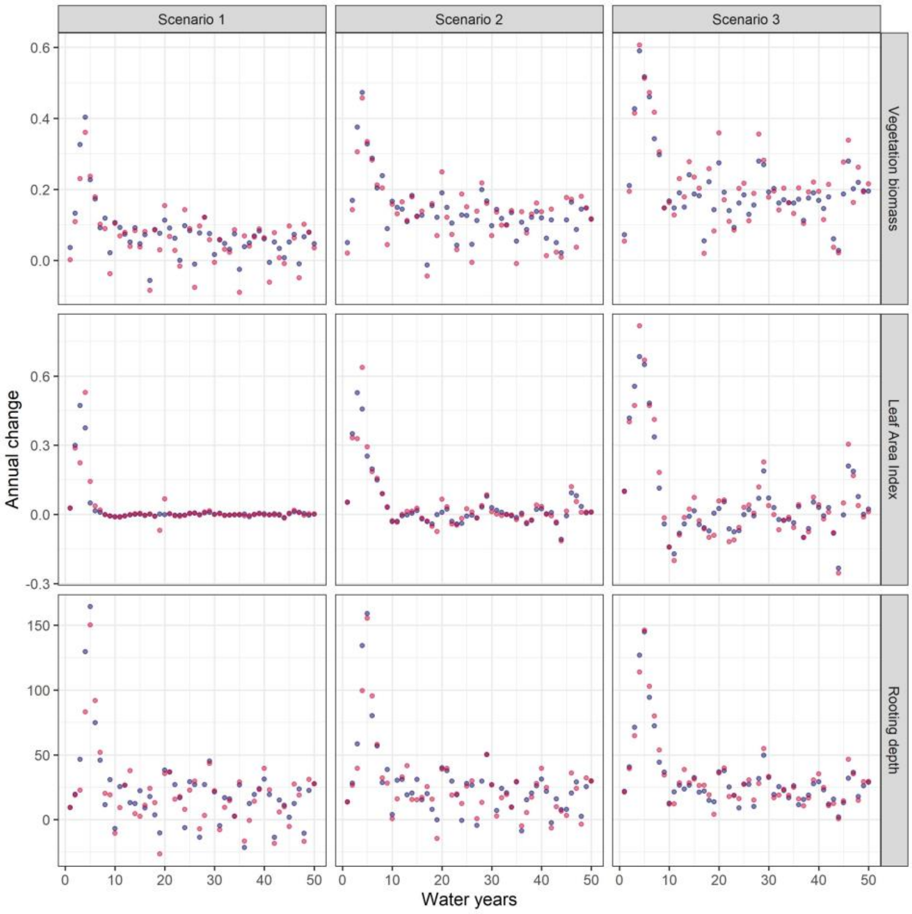

Figure 5.8. Annual change in vegetation biomass (kg C · m^{-2} · year^{-1}), Leaf Area Index and Rooting depth (mm · year^{-1}) under the different management and climate scenarios. Scenario 1 (PM and PM.W: Passive management), Scenario 2 (AFF1 and AFF1.W: Active management), Scenario 3 (AFF2 and AFF2.W: Active management). Blue and red colors correspond to the no warming and warming scenarios respectively.

These trends are highly related to changes in vegetation structural variables. Figure 5.8 shows the year to year change in vegetation biomass, LAI and rooting depth. Results show a clear increase during first years of management corresponding with a growing vegetation cover. Annual changes to vegetation variables reach a maximum and start decreasing until relatively stabilizing around 10 years after management (Figure 5.8). These changes are reflected in vegetation water use marked by significant changes to water fluxes during first years of management.

3.2.3. Seasonal trends (dry period and wet period) and magnitude of change

At seasonal scale, results show high inter-annual variability in both STR and TRANS under the different management and climate scenarios (Supplementary Figures 5.3 and 5.4). For PM scenario, STR and TRANS presents non-significant trends during both seasonal periods and climate scenarios (Tables 5.4 and 5.5, scenario 1).

Table 5.4. Trends and magnitude of change in annual normalized streamflow under the different management and climate scenarios during dry and wet periods. 95% level is used for the significance of the p-value of the Mann Kendall test. Scenario 1 (PM and PM.W: Passive management), Scenario 2 (AFF1 and AFF1.W: Active management), Scenario 3 (AFF2 and AFF2.W: Active management). PM.W, AFF1.W and AFF2.W represent warming climate scenarios.

		Dry-Period		Wet-Period	
		p-value	change (%·decade^{-1})	p-value	change (%·decade^{-1})
Scenario 1	PM	$p = 0.08$	-2.3	$p = 0.32$	-1.4
	PM.W	$p = 0.09$	-2.4	$p = 0.32$	-1.3
Scenario 2	AFF1	$p = 0.06$	-2.8	$p<0.05$	-4.4
	AFF1.W	$p<0.05$	-2.7	$p<0.05$	-4
Scenario 3	AFF2	$p<0.05$	-1.7	$p<0.05$	-5.7
	AFF2.W	$p<0.05$	-1.8	$p<0.05$	-4.3

For AFF1, under no warming, STR only shows a significant decrease during the wet period by 4.4%·decade^{-1}. While results of the warming scenario AFF1.W show a significant decrease of STR by 2.7%·decade^{-1} and 4%·decade^{-1} during dry and wet periods respectively (Table 5.4). Trends in TRANS, are significant only during the wet period with increases of up to 2.4%·decade^{-1}. While, during the dry period, the trends are non-significant (Table 5.5).

The highest changes to STR and TRANS are found during the wet period under AFF2 scenario. STR significantly decline during the wet period by 5.7%·decade^{-1} and 4.3%·decade^{-1} for historic and warming climates respectively (Table 5.4, scenario 3). However, during the dry period, low significant decreases are found (Table 5.4). TRANS trends on the other hand are only significant during the wet period with increases of 3.5%·decade^{-1} and 3.9%·decade^{-1} for historic and warming climates respectively (Table 5.5, scenario 3). Figure 5.9 presents a summary of the obtained results under the different management and climate scenarios for STR and TRANS.

Table 5.5. Trends and magnitude of change in annual normalized transpiration under the different management and climate scenarios during dry and wet periods. 95% level is used for the significance of the p-value of the Mann Kendall test. Scenario 1 (PM and PM.W: Passive management), Scenario 2 (AFF1 and AFF1.W: Active management), Scenario 3 (AFF2 and AFF2.W: Active management). PM.W, AFF1.W and AFF2.W represent warming climate scenarios.

		Dry-Period		Wet-Period	
		p-value	change (%·decade^{-1})	p-value	change (%·decade^{-1})
Scenario 1	PM	p = 0.37	0.9	p = 0.32	1.2
	PM.W	p = 0.26	1	p = 0.32	1.3
Scenario 2	AFF1	p = 0.11	2.1	p<0.05	2.2
	AFF1.W	p = 0.12	1.7	p<0.05	2.4
Scenario 3	AFF2	p = 0.08	3.1	p<0.05	3.5
	AFF2.W	p = 0.11	2.5	p<0.05	3.9

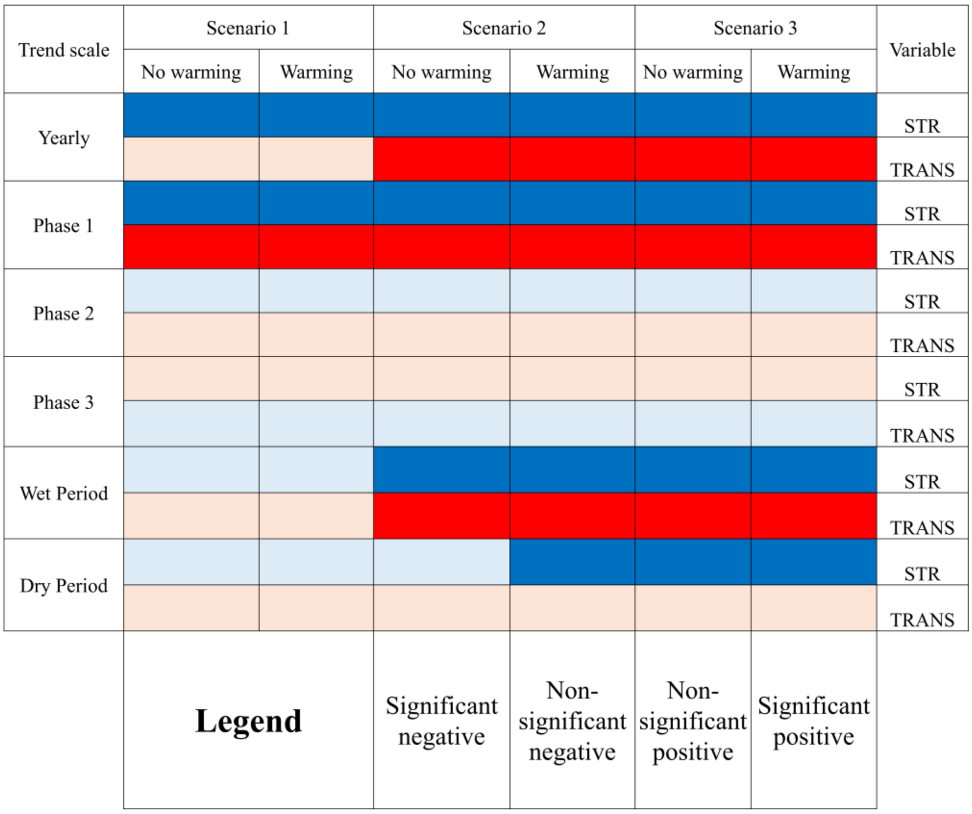

Figure 5.9. Summary of the obtained trends under the different management and climate scenarios for normalized streamflow (STR) and transpiration (TRANS) in annual and seasonal scales. Phase 1: from year 1 to 30 of management, Phase 2: from year 10 to 40 of management, and Phase 3: from 20 to 50 years of management. Wet Period: December to May, Dry Period: June to November.

4. Discussion

This study compares several management and climate scenarios in a small catchment representative of the process of land abandonment in the Central Spanish Pyrenees. Three scenarios of management are compared: (i) a passive management (first 50 years of natural shrub revegetation) and a low and an intense active management scenario through afforestation (first 50 years after afforestation). Simulations are computed with historical climate data and a warming scenario where 3 ºC was added to historic record to test climate change effects.

4.1. Changes to water yield following management

Comparing mean values between the different scenarios revealed significant differences to annual STR and TRANS in Mediterranean mountain areas. Annual STR was lower (27%-53%) and TRANS significantly higher (40% to 93%) under afforestation scenarios when compared to the control scenario PM. These results are consistent with other studies that generally associate afforestation with significant decreases in water yield (Buendia et al., 2016; Jackson et al., 2007; Piao et al., 2007). For instance, in a global analysis of 504 catchment observation, Farley et al. (2005) estimated a decrease by 44% and 31% of annual STR after afforesting grasslands and shrublands respectively. Silveira and Alonso (2009), pointed out that afforestation of 26% of the watershed with pines and eucalyptus species reduces annual STR by between 8.2% and 36.5%, depending on annual precipitation. Guzha et al. (2018), revealed that an increase in forest cover decreases surface STR by 25%.

The trend analysis of annual and seasonal STR and TRANS shows how these declines in STR and corresponding increases in TRANS evolve over time. Here again, results for the Arnás catchment are consistent with previous studies on STR trends during the past decades that generally point to a significant decrease in annual flows in most of the sub-basins in northern Spain (García-Ruiz et al., 2011; Lorenzo-Lacruz et al., 2012; Vicente-Serrano et al., 2019). In these studies, authors attributed the decrease in annual STR to the expansion of a dense vegetation cover due to the management of abandoned areas. Our modeling study for the Arnás catchment confirms that the increase in canopy structural variables (including biomass, LAI, rooting depth) associated with a developing forest corresponds with these hydrologic changes, while climate trends are generally negligible.

It is generally accepted that water consumption of forest is larger than that of shorter vegetation such as shrubs and grasses in hydrological studies (Bosch and Hewlett, 1982; Brown et al., 2005; Farley et al., 2005; Peel, 2009; Sahin and Hall, 1996). Water consumption by vegetation is mainly related to its morphological attributes (Zhou et al., 2016) and in particular its LAI (Wang et al., 2014). TRANS is strongly linked to canopy conductance, which is directly related to LAI and height (Good et al., 2014; Wang et al., 2014). Although, TRANS is the main component of vegetation-evapotranspiration, interception and soil-evaporation contributes substantially in altering water yield in forest, particularly in coniferous plantation (Cannell, 1999). Trees tend to have higher TRANS, interception and evaporation rates than shrubs, which leads to higher losses of water to the atmosphere (Farley et al., 2005). In our modeling study, the greater LAI, greater vegetation biomass and the deep and better developed rooting system of forests, that are more prevalent in AFF1 and AFF2 scenarios, resulted in higher TRANS rates.

In contrast with the significant increases in TRANS associated with afforestation, changes related to climate change are not significant as might be expected given high vapor pressure deficits and a potentially longer growing season with warmer temperatures. Temperature increase in our study didn’t affect the duration of the growing season and changes to vegetation biomass are non-significant under the warming scenario. Warmer temperatures increase vapor pressure deficits and evaporation rates that could increase soil saturation deficit making water less available for the vegetation (Khorchani et al., 2020; Komuscu et al., 1998; Pangle et al., 2014). The lower water availability in the soil inhibits vegetation growth since soil moisture is a potential limiting factor for gross primary production (Green et al., 2019; Seneviratne et al., 2010) making less water transferred to the atmosphere as TRANS (Seneviratne et al., 2010).

The significant trends found at annual scale are mainly related to trends during the first 30 years of management. The moving trend analysis shows that changes to STR and TRANS are only significant during the first 30 years. This is in part consistent with some studies that identified significant effects during the first decades following management (Farley et al., 2005; Peel, 2009), nevertheless, long-term trends to water yield are less certain (Peel, 2009). Significant trends during the first 30 years corresponds to a high increase in vegetation structural variables (biomass, LAI and rooting depth) during first years of management. Van Dijk and Keenan (2007) suggested that most afforestation species (as conifers) show rapid growth rates with annual biomass increase peaking at relatively young ages, and it is reflected in the water use. Notably results suggest that the system stabilizes after the first 30 years (with a long term decline in mean STR and increase in TRANS). This stabilization points to a decrease in water use by the vegetation due the decrease of vegetation growth rates following the first 10 years of management. This is consistent with previous studies highlighting a decrease in evapotranspiration with vegetation age (van Dijk and Keenan, 2007). Scott and Prinsloo (2008) related the stabilization of the effects to water yield to vegetation maturity and pointed out a recovery of pre-afforestation levels depending on the planted trees (30 years in the case of pine plantation). The age of the vegetation therefore, could be of a great interest to better predict changes to water yield after management.

The analysis of trends at seasonal scale show that significant trends to STR and TRANS are mainly concentrated during the wet period (December-May), which makes sense as this is when plants are most active during the year. During the dry period, STR does decline and these declines are significant for afforestation scenarios while trends in dry season TRANS are non-significant. The declines in dry season STR likely reflect carryover of the effects of higher wet season TRANS which reduces groundwater levels that subsequently support summer dry season flows.

4.2. Land management consideration

Land abandonment and afforestation is widespread in parts of the Mediterranean and particularly in Spain (Vicente-Serrano et al., 2020), and there are large abandoned areas in which the process of plant succession has not ended yet (Lasanta et al., 2017; Lasanta et al., 2021). Results from this study suggest that this type of afforestation can accelerate hydrologic change, leading to significant declines in STR that intensify for several decades. If these hydrologic changes occur over a large area, the STR declines would have important repercussions for water resource management, as highlighted García-Ruiz et al. (2011) and Vicente-Serrano et al. (2017), among others. Under these circumstances, water resource management become a delicate matter if it must guarantee the availability of water resources.

In addition to hydrological implications, the increase in the vegetation cover resulting from afforestation and even natural regrowth strategies could strongly affect the sustainability of ecosystem services in these areas. For instance, the expansion of forest cover could strongly increase fire risks and limit the availability of pasture lands (Lasanta et al., 2018). We caution, however, that there are benefits from afforestation including reduced soil erosion (Huang et al., 2017; Porto et al., 2009), increased infiltration rates (Ilstedt et al., 2007) and increased biodiversity (Buscardo et al., 2008). Adoption of afforestation plans in abandoned areas, however, should balance these benefits with potential hydrologic impacts and consider alternative land management strategies. In areas where water supply is a critical issue, natural revegetation combined with shrub clearing could be an effective management strategy to ensure lower fire risks (Lasanta et al., 2018) and higher water availability (Khorchani et al., 2020). On the other hand, afforestation plans could be an effective strategy in areas with sufficient annual precipitations to sustain the high water use of trees. In the Central Spanish Pyrenees, as a transitional region between humid high elevations and drier low elevation areas, tailoring management practices to the local setting will be important. In some areas, combining afforestation with natural revegetation and shrub clearing could enhance the hydrological and ecological processes in these areas and guarantee the availability of water resources for lowland areas. Optimizing hydrological and ecological benefits of active and passive management plans therefore, could be a promising objective for future studies in Mediterranean mountains.

5. Conclusions

In this study, we aim to gain more insight on the changes to water regimes in abandoned cropland areas due to management and climate change. We used an ecohydrologic model to show that afforestation in abandoned areas would significantly decrease annual STR between 2.3%·decade^{-1} and 5.9%·decade^{-1} depending on the total afforested area. These trends were linked with the growth of vegetation over time, leading to increases in TRANS losses, particularly during the wet season. These changes ultimately stabilize as vegetation reaches maturity. Changes in TRANS and STR with a warming climate on the other hand were smaller.

The multi-decadal scale of this study has permitted to underline important findings on the effects of management plans and climate on water regimes in Mediterranean mountains. Our study demonstrated that the adopted management plan, the spatial scale of management and the age of management as well as climate could condition the hydrological response in Mediterranean mountain areas. These results could be of high interest to help quantifying the potential changes to hydrological processes following management of abandoned cropland areas. Such quantification is of crucial importance when elaborating land management plans particularly in water limited environments as it is the Mediterranean region. Nevertheless, studies at medium to large spatial scales are of big interest to validated the obtained results and ensure the efficiency of future land management plans.

References Chapter 5

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Chapter 6. Carbon sequestration and water yield tradeoffs following restoration of abandoned agricultural lands in Mediterranean mountains

This Chapter studies the tradeoffs between carbon and water fluxes under natural revegetation and afforestation programs in Mediterranean mountains. RHESSys was used to study the interactions between carbon sequestration and water yield following the restoration of abandoned cropland areas. Three land cover scenarios coupled with two climate scenarios were modeled and changes to net ecosystem production, water yield and water-use efficiency were quantified.

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Abstract

Abandoned cropland areas have the potential to contribute to climate change mitigation through natural revegetation and afforestation programs. These programs increase above and belowground carbon sequestration by expanding forest cover. However, this potential to mitigate climate change often involves tradeoffs between carbon sequestration and water availability. Particularly in a water limited environments such as the Mediterranean region, any loss of recharge to groundwater or streamflow can have critical societal consequences. In this study, we used an ecohydrologic model, Regional Hydro-Ecological Simulation System (RHESSys), to quantify these tradeoffs for land management plans in abandoned cropland areas in Mediterranean mountains. Changes to Net Ecosystem Production (NEP), water yield and Water-Use Efficiency (WUE) under different land management and climate scenarios were estimated for Arnás, a catchment with similar geology, vegetation and climate to many of the locations targeted for land abandonment restoration in the Spanish Pyrenees. Results showed significant changes to both carbon and water fluxes related to land management, while changes related to a warming scenario were not significant. Afforestation scenarios showed the highest average annual carbon sequestration rates (112 g $C \cdot m^{-2} \cdot yr^{-1}$) but were also associated with the lowest water yield (runoff coefficient of 26%) and water use efficiency (1.4 g $C \cdot mm^{-1}$) compared to natural revegetation (-27 g $C \cdot m^{-2} \cdot yr^{-1}$, 50%, 1.7 g $C \cdot mm^{-1}$ respectively). Under both restoration scenarios, results showed that the catchment ecosystem is a carbon sink during mid-February to July, coinciding with peak monthly transpiration and WUE, while during the rest of the year the catchment ecosystem is a carbon source. These results contribute to understanding carbon and water tradeoffs in Mediterranean mountains and can help adapt restoration plans to address both carbon sequestration and water management objectives.

1. Introduction

The abandonment of agricultural lands is a global issue that has been affecting regions around the world for decades. Land abandonment directly and indirectly alters ecosystem services by changing the quantity and quality of water resources, soil properties, biodiversity, carbon sequestration, disturbance risk (mainly of forest fires), agro-pastoral resources, among others (Conti and Fagarazzi, 2004; Bernués et al., 2014; Lasanta, 2019). Rates of abandonment of agricultural lands around the world vary considerably among studies, but they are generally high (Gibbs and Salmon, 2015). For instance, Ramankutty and Foley (1999) estimated that there is about 1.5 million km^2 of total global abandoned agricultural land, while Campbell et al. (2008) estimates that this total ranges between 3.85 and 4.72 million km^2. The drivers of this abandonment are related to multiple ecological, biophysical, and socio-economic factors, including agricultural practices that have led to soil degradation (Benayas et al., 2007). Globally, Europe has been one of the most affected areas during the past decades (MacDonald et al., 2000; Lasanta et al., 2017) and land abandoment is forecasted to continue in this region (Rounsevell et al., 2006; Castillo et al., 2021). By 2030, 11% of agricultural land in Europe (more than 200,000 km^2) are projected to be under high risk of abandonment. Spain will be the only country to lose more than 10,000 km^2 from its agricultural surface (Castillo et al., 2020).

During the last decades, abandoned lands are attracting interest due to their potential to mitigate climate change through carbon sequestration (Novara et al., 2017; Vilà-Cabrera et al., 2017; Tommaso et al., 2018; Pellis et al., 2019; Bell et al., 2021). Conventional agricultural activities can reduce soil carbon stocks over time and contribute large amounts of greenhouse gases to the atmosphere (Lal, 2013; Carlson et al., 2017). However, following abandonment of cultivation, the recovery of these stocks has been identified as a promising avenue to mitigate climate change (Kurganova et al., 2014; Kämpf et al., 2016; Wertebach et al., 2017). Forest plants, through photosynthesis, assimilate and store atmospheric CO_2 in biomass, deadwood, litter and soils (Farooqi et al., 2021). Forest regeneration, through natural revegetation or large-scale afforestation can sequester significant amounts of carbon, contributing to reducing atmospheric CO_2. However, especially in Mediterranean and semi-arid environments, the implications of land abandonment and post-abandonment vegetation growth for water resources is an important concern (Viviroli et al., 2003; García-Ruiz et al., 2011). Consequently, there is still no scientific consensus regarding the most effective management strategies to achieve multiple objectives related to soil erosion, carbon sequestration and water resources on abandoned lands (Agnoletti, 2014; Schnitzler, 2014).

The potential for carbon sequestration in abandoned agricultural lands is well recognized (Post and Kwon, 2000; Deng et al., 2014). Nevertheless, there is still substantial uncertainty around the magnitude of carbon sequestration and how it evolves over time following land abandonment in Mediterranean environments (Hoogmoed et al., 2012). Passive or active restoration (i.e. natural revegetation or afforestation) of abandoned agricultural areas could require long periods to reach efficient carbon sequestration levels (Bonet, 2004; Garcia-Franco et al., 2014). Grasses and shrubs could require more than 100 years to transit to a natural forest (Pueyo and Beguería, 2007; Gibon et al., 2010). The rate of carbon stock and forest recovery have implications for the selection of active restoration practices particularly in areas of high soil degradation risk.

At the same time, the transition from abandoned agricultural lands to forest, via either natural revegetation or afforestation, can have important impacts on water resources. In non-limited water regions, forests sustain local water dynamics by retaining rainwater through improving infiltration and litter and soil water storage and avoiding immediate losses from runoff events (Wu et al., 2020). However, in water limited environments such as the Mediterranean region, forest regeneration involves tradeoffs (Ovando et al., 2019). Forest regeneration can reduce streamflow (Zhang et al., 2017) and deplete local water availability (Doelman et al., 2020). Changes to water yield are linked to changes in transpiration, evaporation and interception, all of which tend to increase when agricultural abandoned lands are converted to forest. However, the magnitude and timing of these responses can vary considerably between natural revegetation and afforestation (Lacombe et al., 2016; Khorchani et al., 2021a). In Mediterranean regions, water availability is often an issue and is expected to be increasingly problematic given high sensitivity of these regions to climate change (Giorgi, 2006; Giorgi and Lionello, 2008).

Given these concerns, tradeoffs between carbon sequestration and water resources must be adequately assessed in the design and evaluation of restoration strategies. Existing research on carbon sequestration and hydrology mostly use different methodologies and operate at different spatiotemporal scales (Farooqi et al., 2021), making it challenging to quantify tradeoffs between carbon sequestration and water yield. Process-based models, that explicitly account for carbon cycling, hydrology and their interactions, offer a good tool to assess land management impacts and how these may change with climate. In contrast to traditional estimates of forest carbon budget, involving biomass harvesting (Tang et al., 2018) and soil sampling (Nadal-Romero et al., 2016a; Lasanta et al., 2020), process-based models offer less labor intensive, more generalizable evaluation of carbon and water cycles and their interaction. The Regional Hydro-Ecological Simulation System (RHESSys) is a process-based model that simulates carbon and water cycles from local to regional spatial scale (Tague and Band, 2004). RHESSys has been widely used to assess biomass-water balance dynamics around the world (Bart et al., 2016; Martin et al., 2017; Chen et al., 2020a) including studies in the Pyrenees (López-Moreno et al., 2014; Zabalza-Martínez et al., 2018; Khorchani et al., 2021b). In many of these applications model estimates have shown a high sensitivity of carbon and water to land use and land cover changes (Morán-Tejeda et al., 2015; Khorchani et al., 2020).

Restoration practices after cropland abandonment should be decided based on scientific knowledge to understand carbon and water tradeoffs in Mediterranean mountain areas, and ensure an appropriate functioning of ecosystem services. In this study we demonstrate how coupled eco-hydrologic modeling can be used to provide this information. We used RHESSys to evaluate carbon sequestration and water yield changes following passive and active restoration of an abandoned agricultural catchment in the Central Spanish Pyrenees. Annual and monthly changes in carbon stocks and water yield are estimated, along with Water-Use Efficiency (WUE) for natural revegetation and afforestation scenarios. We hypothesized that there would be a significant decrease in water yield with afforestation relative to water yield under natural revegetation. Further, we expected that any decrease in water yield would be associated with significant changes to carbon fluxes and finally that these changes would vary with vegetation physiological attributes and inter-annual and longer term climate variation.

2. Materials and methods

2.1. Study area

Arnás is a small catchment of the Borau valley located in the headwaters of the Aragón river in the Central Spanish Pyrenees (Figure 6.1). The catchment covers 0.28 km^2 with 900 to 1400 m of altitudinal gradient. Eocene Flysch, with thin alternating sandstones and marl layers, forms the lithology of the catchment. The steep slopes in the south-facing part of the catchment results in abundant sheet wash and eroded patches close to the ravine (García-Ruiz et al., 2005). The climate of the region is subject to Mediterranean and

Atlantic influences (López-Moreno et al., 2008) with average annual precipitation of 1100 mm and average temperature of 10 ºC (Seeger et al., 2005). Although precipitation mainly occurs during autumn and spring months, summer short duration rainstorms are relatively frequent (Lana-Renault et al., 2018).

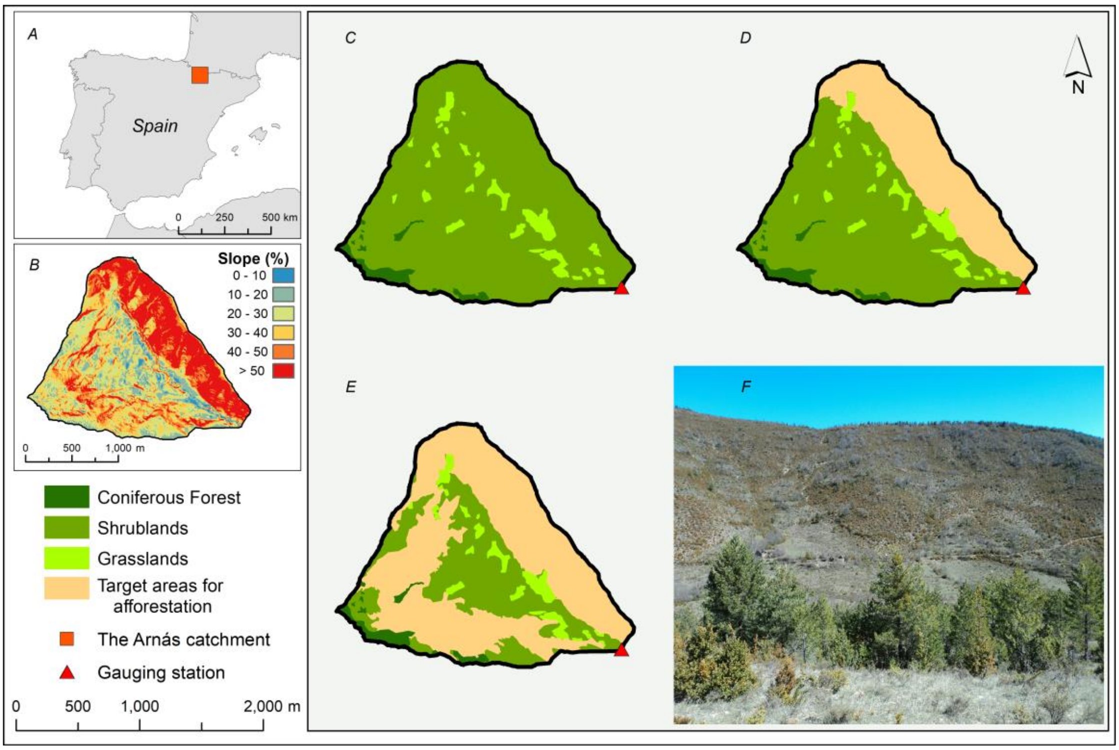

Figure 6.1. The Arnás catchment. A: Localization in Spain. B: Slope map. C: Natural revegetation scenario. D: Low afforestation scenario. E: Intense afforestation scenario. F: Current situation of the Arnás catchment.

The Arnás catchment is an experimental site monitored by the Pyrenean Institute of Ecology (IPE-CSIC) since 1996 to present. The catchment is a good example of the process of land abandonment in Mediterranean mountains. After being intensively cultivated during the 19th century, the catchment has been progressively abandoned during the first decades of the 20th century. Secondary succession processes have taken place after abandonment resulting in the colonization of most of the abandoned croplands by shrubs and trees. *Pinus syslvestris* and *Quercus faginea* are the main tree species that colonized the first abandoned fields mainly in the north-facing part of the catchment (Lasanta et al., 2005; Lasanta and Vicente-Serrano, 2007). In the south-facing slopes, soil degradation due to cultivation inhibited the succession process marked by a slow growing shrub cover (Figure 6.1D, 6.1F). *Genista scorpius*, *Rosa gr. canina*, *Buxus sempervirens*, *Echinospartum horridum* and *Juniperus communis* are the main shrub species covering large areas of the catchment. The geology, climate and vegetation of Arnás catchment is similar to many of the upslope regions of the Spanish Pyrenees and Mediterranean mountains. Thus, it is a good candidate for investigating the likely consequences of land use/land cover change in this Mediterranean region where water resources and carbon sequestration are both critical concerns. Notably the relatively long-term monitoring record at this site (from 1996 to present) supports calibration and validation of an ecohydrological model that can be used to project land use/land cover change impacts on water and carbon under a range of climate conditions. The potential for the application of RHESSys to other similar locations to more generally support the evaluation of land restoration in the context of water and land management is considered in the discussion.

2.2. Scenarios and climate record

Three land cover scenarios are used to parameterize RHESSys in the Arnás catchment (Table 6.1). Land abandonment of most of the catchment occurred prior to 1960 (Lasanta, 1988). We use this as a starting point for land abandonment scenarios. A land cover map derived from 1957 aerial photography was used to define these initial conditions for different land cover/land management scenarios (Figure 6.1A, 6.1B, 6.1C).

Table 6.1. The different land cover and climate scenarios of the study.

Land cover	Natural revegetation		Low-intensity afforestation		High-intensity afforestation	
Scenario	PM	PM.W	AFF1	AFF1.W	AFF2	AFF2.W
Climate	historic	warming	historic	warming	historic	warming
Basis	First successional stages before the entry of a secondary forest.		Areas of more than 30 % of slope in the south-facing part of the catchment are afforested		Areas of more than 30 % of slope in the all the catchment are afforested	
Land cover	Forest: 3.2%, Shrubs: 90.7% and Grass: 6.1%		Afforestation: 30.1%, Forest: 3.2%, Shrubs: 61% and Grass: 5.7%		Afforestation: 61.1%, Forest: 3.2%, Shrubs: 30.5% and Grass: 5.2%	

The natural revegetation scenarios (PM and PM.W) represent no management scenarios, a theoretical representation of the first slow successional stages before the entry of a secondary tree cover. Natural secondary succession needs around 100 years to establish a tree cover after cropland abandonment (Pueyo and Beguería, 2007; Gibon et al., 2010). The natural revegetation scenario represents the beginning of the colonization process by shrub species (*G. scorpius, E. horridum, B. sempervirens*) if no management plan is adopted in the catchment (typically leading to 50 years of shrub cover). We note that the process of land abandonment in the Arnás catchment occurred progressively in space and time. To simplify the representation of the natural revegetation scenario we assume a uniform abandonment of the Arnás catchment as the initial condition for all scenarios. We use this shrub cover scenario (natural revegetation) to compare with catchments where trees are actively introduced.

The eroded and degraded soil strata and the steep slopes in the south-facing part of the catchment inhibit the establishment of a natural tree cover. The low afforestation scenarios (AFF1 and AFF1.W) represent active management scenarios where trees are introduced in areas of high erosion risk (> 30% of slope) in the south-facing slopes while the rest of the catchment undergoes natural revegetation by shrub species (thus 30.1% of the Arnás catchment is afforested). This scenario is similar to management practices where abandoned croplands are afforested to enhance soil restoration and limit erosion processes (Ortigosa et al., 1990; Tomaz et al., 2013). The gentler slopes and the deeper soils in the north-facing part of the catchment would facilitate a slow natural regeneration of a forest cover.

The intense afforestation scenarios (AFF2 and AFF2.W) represent active management scenarios where trees are introduced in areas of high erosion risk in the entire catchment (thus 60.1% of the Arnás catchment is afforested). Areas of gentler slopes are left to natural revegetation by shrubs.

To study the effect of a potential climate change scenario on carbon sequestration and water fluxes, we ran RHESSys simulations for a historic and a warming climate scenario. The climate change scenario includes an increase in air temperature, however, we decided not to change the precipitation given the high uncertainties in future projections (OPCC, 2018). In the Pyrenees, the Pyrenean Climate Change Observatory (OPCC, 2018) predicted an increase between 1 and 2 °C by 2030, with respect to the 1961-1990 reference period, while The European Environmental Agency (EEA, 2020) projected an increase of up to 5 °C by the end of this century, with respect to 1971-2000 reference period. In this study, we considered an increase of 3 ºC as a midpoint between the two projections for the warming scenario. To compute the warming scenario, we uniformly add 3 ºC to the historic climate record. We simulate 6 scenarios in the Arnás catchment (3 land covers scenarios and 2 climate scenarios).

RHESSys requires daily series of precipitation and maximum and minimum temperature. To generate a long-term 50-year climate record for these three variables, we used the available climate data from the existing meteorological station in the Arnás catchment. The climate record from the catchment has been homogenized, quality controlled and gap filled following Vicente-Serrano et al. (2010) and El Kenawy et al. (2012). During the process of data reconstruction and quality control, several meteorological stations from the IPE-CSIC and the Spanish Meteorological Agency (AEMET) at a similar altitude in a radium of 10 km were used. This 50-year climate record was used for the different scenarios.

2.3. RHESSys model

We used the Regional Hydro-Ecological Simulation System (RHESSys) (Tague and Band, 2004) to run the different scenarios. RHESSys is a process-based dynamic model that was developed to study the effects of changes in the vegetation and the climate on the hydrological and ecological processes (Tague and Band, 2004). RHESSys has been continually updated as documented on the RHESSys website (https://github.com/RHESSys/RHESSys/wiki). For this study we used RHESSys5.20 in "dynamic mode", which includes vegetation growth. RHESSys simulates carbon, nitrogen and water fluxes over complex terrain from local to regional spatial scale. Its hierarchical representation of the landscape, subdivides ecohydrological processes into different spatial units (basins, hillslopes, zones, patches and canopy strata). This partitioning of the landscape enhances the ability of the model to represent both local processes such as transpiration and photosynthesis (at patch level) and landscape scale spatial heterogeneity in radiation and climate inputs and downslope water redistribution. Coupling local process and landscape spatial heterogeneity enhances the sensitivity of the model to climate and land cover changes.

As well as modeling surface and subsurface vertical and lateral water fluxes, RHESSys includes a carbon cycling model that estimates photosynthesis, respiration and carbon allocation to leaves, stems and roots. The total assimilated carbon from the atmosphere is estimated using the Farquhar equation (Farquhar and von Caemmerer, 1982). Carbon is transferred back to the atmosphere through respiration from plants, litter and soil (Tague and Band, 2004). Plant respiration represents carbon that is lost to support plant maintenance and growth, while litter and soil respiration accounts for the carbon lost during the process of decomposition of organic material by microbial biomass. Plant respiration is calculated using the model developed by Ryan (1991) while respiration from soil and litter layers is based on an approach developed by Thornton (1998). Allocation of net photosynthesis to leaves, stems and roots varies with plant leaf area as well as availability of water and nutrients (see Garcia et al. (2016) for details).

In this study, we parameterized the vegetation as standard classes of trees, shrubs and grasses. To initialize these standard vegetation classes, we used parameter default files from https://github.com/RHESSys/RHESSys/wiki/Parameter-Definition-Files and the three vegetation maps representing the three land cover scenarios (Figure 6.1C, 6.1D, 6.1E). Soil carbon and nitrogen stores were initialized through a first 100-year simulation followed by vegetation carbon and nitrogen store removal. A soil map developed by Khorchani et al. (2020) was used to initialize soil drainage and storage parameters. Details on RHESSys model and its implementation can be found in https://github.com/RHESSys/RHESSys/wiki. Details on the process of calibration and validation of the model in the Arnás catchment can be found in Khorchani et al. (2020). In this study we use the calibration parameters obtained by Khorchani et al. (2020).

2.4. Studied variables and statistical analysis

In this study, we analyze carbon and water fluxes under different land cover and climate scenarios. For each land cover and climate scenario, we compute a time series of daily Net Ecosystem Production (NEP) and then aggregated to monthly and annual time scales for analysis. NEP is the difference between ecosystem production (carbon inputs) and respiration (carbon outputs) and is a good measure of ecosystem carbon balance as it accounts for above and belowground carbon stores. In this study, negative NEP values indicate carbon source balance while positive NEP values indicate carbon sink balance.

NEP is estimated at annual and monthly scales using Eq1:

$$NEP_{(annual,\ monthly)} = GPP_{(annual,\ monthly)} - RESP_{(annual,\ monthly)} \qquad \text{(Eq1)}$$

Where GPP is the total gross primary productivity and RESP is the total annual respiration. RESP is computed using Eq2:

$$RESP = RESP_{plant} + RESP_{soil} + RESP_{lit} \qquad \text{(Eq2)}$$

Where $RESP_{plant}$ is the total allocation of carbon lost for plant maintenance and growth and $RESP_{soil}$ and $RESP_{lit}$ is the total carbon lost due to the decomposition of organic material in the soil and the litter components.

Changes to carbon fluxes are compared to changes in water fluxes and the interactions between the two regimes and climate and land cover changes are discussed. Annual and monthly carbon and water fluxes are compared under the different land cover and climate scenarios. We computed an ecosystem Water-Use Efficiency (WUE) to link water and carbon cycles as a function of land cover and climate change. WUE is a conceptual measure that quantifies tradeoffs between carbon gains and water losses at ecosystem level (Li et al., 2019). WUE is defined as the ratio of the total amount of assimilated carbon (GPP), through photosynthesis, and the total amount of water consumption through evapotranspiration (ET) (Yang et al., 2016). WUE is computed using Eq3:

$$WUE_{\text{(annual, monthly)}} = GPP_{\text{(annual, monthly)}} / ET_{\text{(annual, monthly)}} \qquad \text{(Eq3)}$$

Pearson's correlation coefficients were used to assess the relationships between the different variables. The Analysis of variance ANOVA was used to compare differences in carbon and water fluxes under the different land covers and climate scenarios. We used Tukey's Honestly Significant Difference test (TukeyHSD) (Tukey, 1949) in order to determine the specific differences. All calculations and statistical analysis were conducted using R 4.0.3 software.

3. Results

3.1. Changes in carbon fluxes

Figure 6.2A shows annual changes in annual and monthly NEP estimates under the different land cover and climate scenarios. Afforestation significantly increases carbon sequestration compared to natural revegetation scenarios (PM and PM.W) (Table 6.2, Supplementary Table 6.1). Afforestation scenarios show positive average annual carbon budgets indicating that the total annual carbon assimilated by the vegetation through the process of photosynthesis is higher than that lost due to vegetation and soil respiration processes. Under afforestation scenarios, the ecosystem shows a carbon sink budget with an average annual NEP reaching up to 112 g $C \cdot m^{-2} \cdot yr^{-1}$ and up to 5.6 kg $C \cdot m^{-2}$ during five decades, with these values being higher under the intense afforestation scenario (Table 6.2). However, under natural revegetation scenarios, the ecosystem shows negative carbon budget with annual average NEP declining to -74 g $C \cdot m^{-2} \cdot yr^{-1}$ and to -3.7 kg $C \cdot m^{-2}$ in five decades (Table 6.2). These negative values indicate that the ecosystem is converted to a carbon source due to high carbon losses to the atmosphere during respiration processes that are not compensated by carbon inputs from photosynthesis.

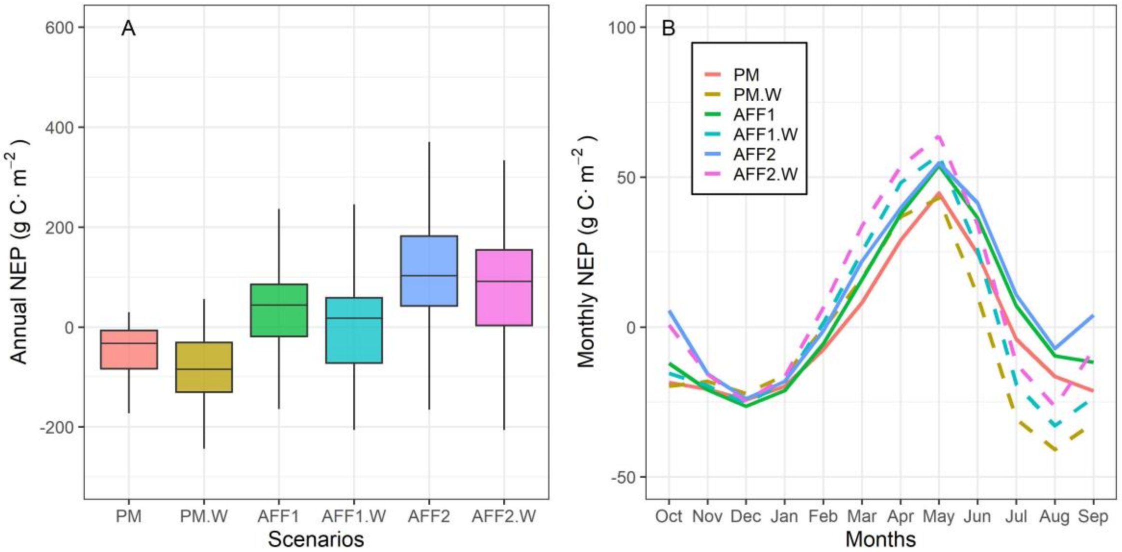

Figure 6.2. Annual and monthly Net Ecosystem Production (NEP) under the different land cover and climate scenarios. A: Annual, B: Monthly. PM and PM.W: No management: natural revegetation. AFF1, AFF1.W: Active management, low afforestation, AFF2 and AFF2.W: Active management, intense afforestation. PM.W, AFF1.W and AFF2.W represent warming climate scenarios. Positive NEP values indicate carbon sink while negative values indicate carbon source. Median value (the line in the middle of the boxes), 1st and 3rd quartiles (lower and upper box boundaries respectively), whiskers (hanging bars).

Table 6.2. Changes to carbon fluxes under the different land cover and climate scenarios. NEP: Net Ecosystem Production. PM and PM.W: No management: natural revegetation. AFF1, AFF1.W: Active management, low afforestation, AFF2 and AFF2.W: Active management, intense afforestation. PM.W, AFF1.W and AFF2.W represent warming climate scenarios. Positive NEP values indicate carbon sink while negative values indicate carbon source.

Scenarios	PM	PM.W	AFF1	AFF1.W	AFF2	AFF2.W
Average annual NEP (g $C \cdot m^{-2}$)	-27	-74	43	3	112	89
Total carbon change (g $C \cdot m^{-2}$)	-1336	-3700	2150	161	5594	4426

Differences related to climate change, however, are less important than those linked to afforestation (Figure 6.2A, Table 6.2). The increase of temperature decreases annual NEP for all land cover scenarios, nevertheless, differences are non-significant between historic and warming climate scenarios (Table 6.2, Supplementary Table 6.1). The decrease in annual NEP is higher under the natural revegetation scenario compared to afforestation scenarios (Figure 6.2A, Table 6.2).

The seasonal timing of the carbon cycle is similar across all scenarios. The ecosystem is a carbon sink between mid-February to July, while the rest of the year the ecosystem shows a negative carbon balance and is a carbon source (Figure 6.2B). The highest carbon uptake during the sink phase is reached in May with average monthly NEP of around 50 g $C \cdot m^{-2} \cdot month^{-1}$. However, during the source phase the ecosystem releases up to 40 g $C \cdot m^{-2} \cdot month^{-1}$. Warming scenarios clearly affected peak carbon fluxes and the duration and the transition between carbon emission and uptake. As well as vegetation type, changes to NEP through time and across scenarios can be attributed to modelled vegetation physiological and structural variables, such as leaf area, height, root depth and biomass that change with age and respond to climate. Figure 6.3 shows the temporal pattern of carbon sequestration under the different land cover and climate scenarios. During the first three years of management, our modeling results show a negative carbon balance, which rapidly changes with vegetation growth to a positive balance and reaches the highest sequestration rates during the first 10 years of management. After reaching a peak sequestration rate, carbon uptake starts decreasing until relatively stable value is reached. Compared to natural revegetation, afforestation reduces the frequency of carbon emission years and strongly increases the stable sequestration rate (Figure 6.3).

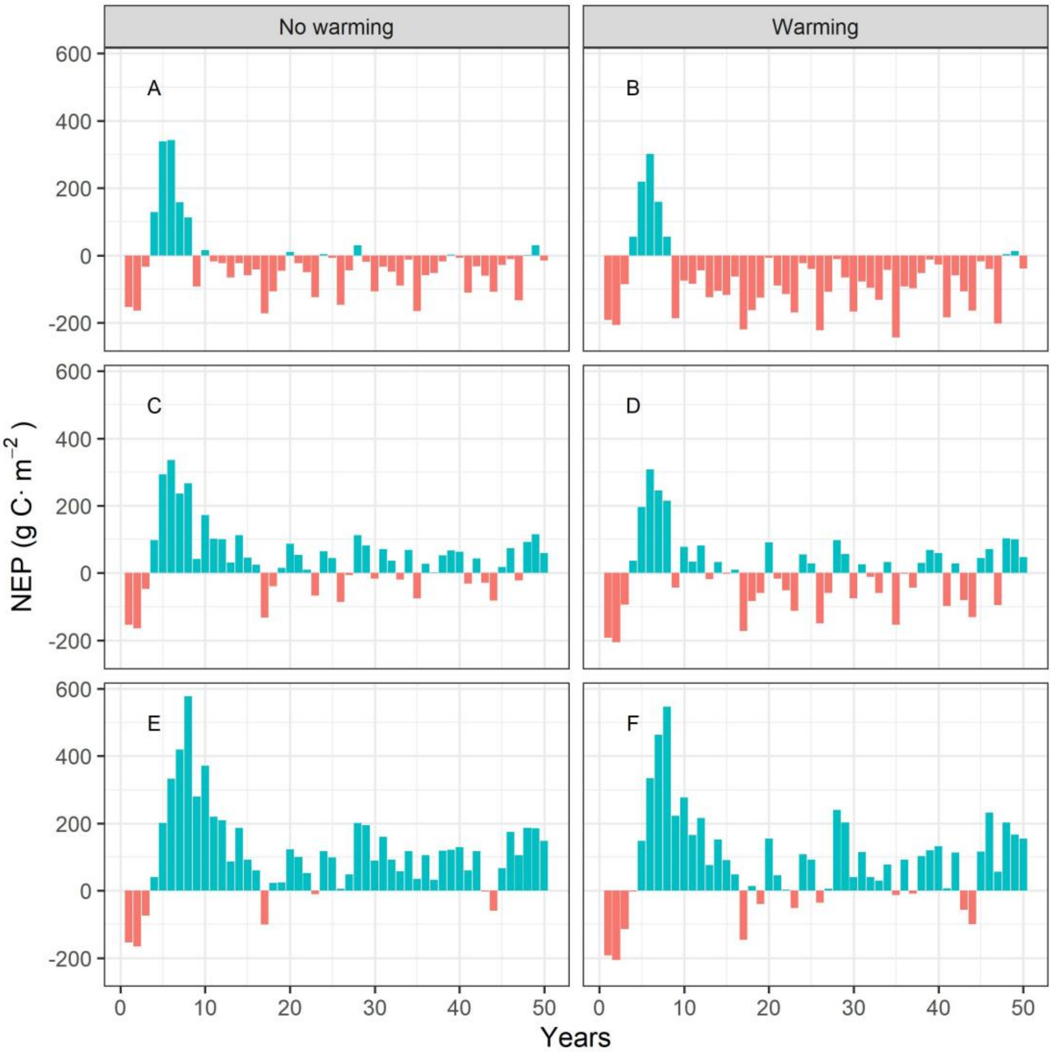

Figure 6.3. Temporal pattern of annual Net Ecosystem Production (NEP) under the different land cover and climate scenarios. A: PM and B: PM.W (No management, natural revegetation), C: AFF1 and D: AFF1.W (Active management, low afforestation), E: AFF2 and F: AFF2.W (Active management, intense afforestation). B, D and F represent warming climate scenarios. Positive NEP values indicate carbon sink while negative values indicate carbon source.

In addition to vegetation physiological and structural properties, changes to carbon sequestration are related to climate variability. Figure 6.4 shows the correlation between annual NEP and annual rainfall. Results show significant, though low, positive correlation between the two variables under the different land cover and climate scenarios (Figure 6.4). Correlation coefficients range between 0.35 to 0.46 and are higher under the warming climate (0.38, 0.43 and 0.46 for PM.W, AFF1.W and AFF2.W respectively) compared to historic climate (Figure 6.4B, 6.4D and 6.4F).

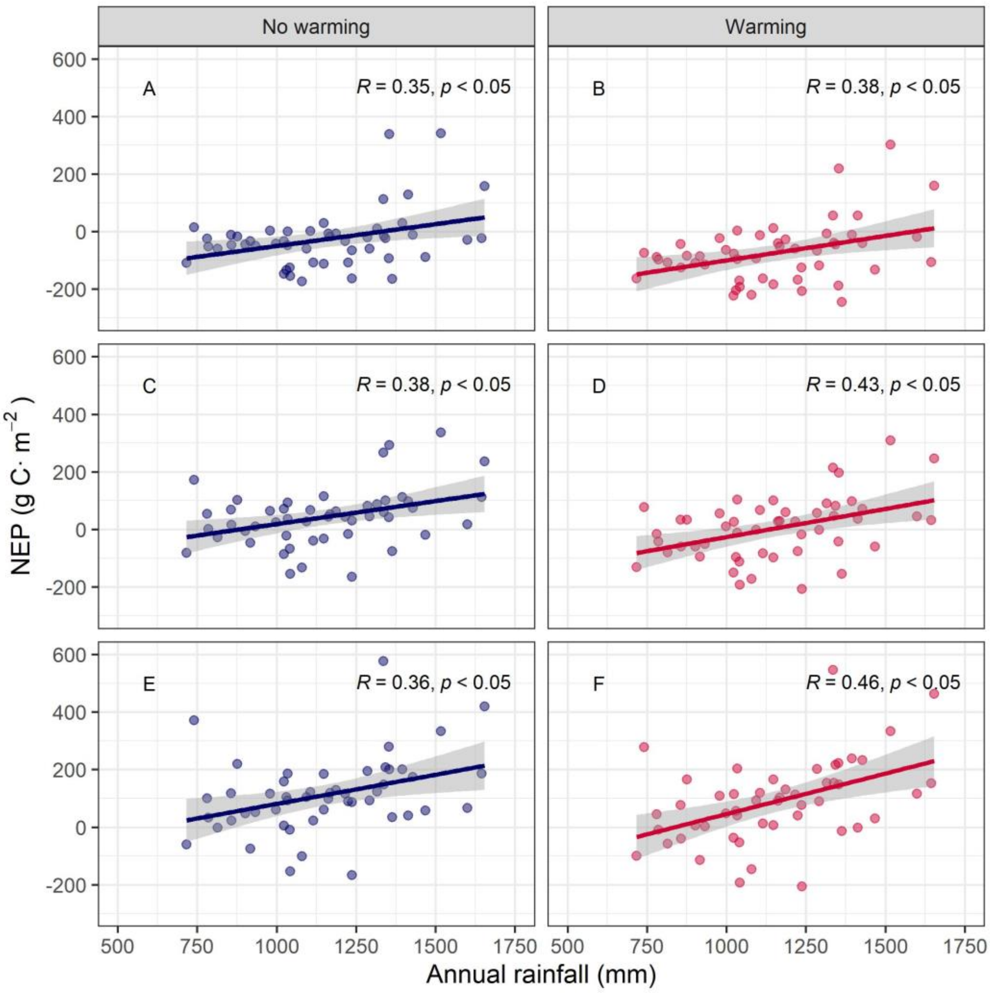

Figure 6.4. Effect of annual rainfall on annual Net Ecosystem Production (NEP) under the different land cover and climate scenarios. A: PM and B: PM.W (No management, natural revegetation), C: AFF1 and D: AFF1.W (Active management, low afforestation), E: AFF2 and F: AFF2.W (Active management, intense afforestation). B, D and F represent warming climate scenarios. The shaded area around the regression line represent a pointwise 95% confidence interval.

3.2. Changes in water fluxes

As well as carbon, water fluxes have also showed important changes related to changes in vegetation cover and climate. Figure 6.5 shows annual changes to streamflow, transpiration and evaporation under the different land cover and climate scenarios. Afforestation significantly decreases annual streamflow and increases annual transpiration and evaporation compared to the natural revegetation scenario, with higher changes under AFF2 compared to AFF1. However, changes related to the increase of temperature are non-significant (Table 6.3, Supplementary Table 6.2). Annual runoff coefficients decrease from 50% to up to 26%, while transpiration increases from 22% to up to 41% and evaporation from 28% to up to 34% of annual precipitation (Figure 6.5, Table 6.3). The total decline in water yield is associated with a significant increase of evapotranspiration. However, changes to transpiration are clearly higher than those of evaporation reflecting a higher percentage of water loss through stomatal conductance that supports the photosynthesis process (Figure 6.5B and 6.5C, Table 6.3, Supplementary Table 6.2).

Table 6.3. Average annual percentage of water fluxes from annual rainfall. PM and PM.W: No management: natural revegetation. AFF1, AFF1.W: Active management, low afforestation, AFF2 and AFF2.W: Active management, intense afforestation. PM.W, AFF1.W and AFF2.W represent warming climate scenarios.

	PM	PM.W	AFF1	AFF1.W	AFF2	AFF2.W
Annual Streamflow (%)	50	49	38	37	28	26
Annual Transpiration (%)	22	23	30	31	40	41
Annual Evaporation (%)	28	29	32	33	33	34

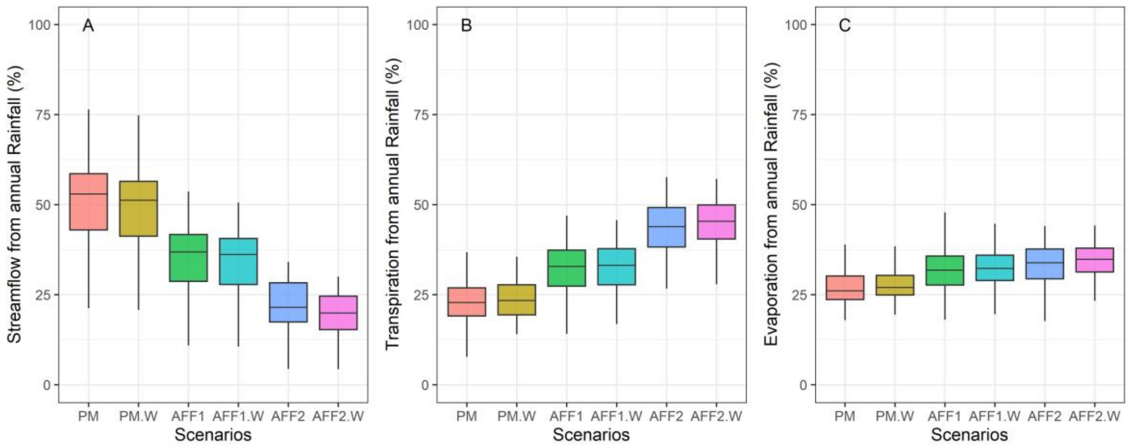

Figure 6.5. Annual percentage of water fluxes from annual rainfall under the different land cover and climate scenarios. A: Streamflow, B: Transpiration, C: Evaporation. PM and PM.W: No management: Natural revegetation. AFF1, AFF1.W: Active management, low afforestation, AFF2 and AFF2.W: Active management, intense afforestation. PM.W, AFF1.W and AFF2.W represent warming climate scenarios. Median value (the line in the middle of boxes), 1st and 3rd quartiles (lower and upper box boundaries respectively), whiskers (hanging bars).

At the monthly timescale, the differences across scenarios are clear for streamflow and transpiration (Figure 6.6A and 6.6B) while for evaporation differences are substantially smaller (Figure 6.6C). The highest changes in streamflow occur during the wet period between December and May months. Changes to transpiration coincide with the start of the growing season and last until the end of summer. In contrast, evaporation showed a relatively stable pattern during the year with smaller differences between land cover and climate scenarios. These water fluxes patterns indicate a higher effect of transpiration patterns on annual streamflow compared to evaporation.

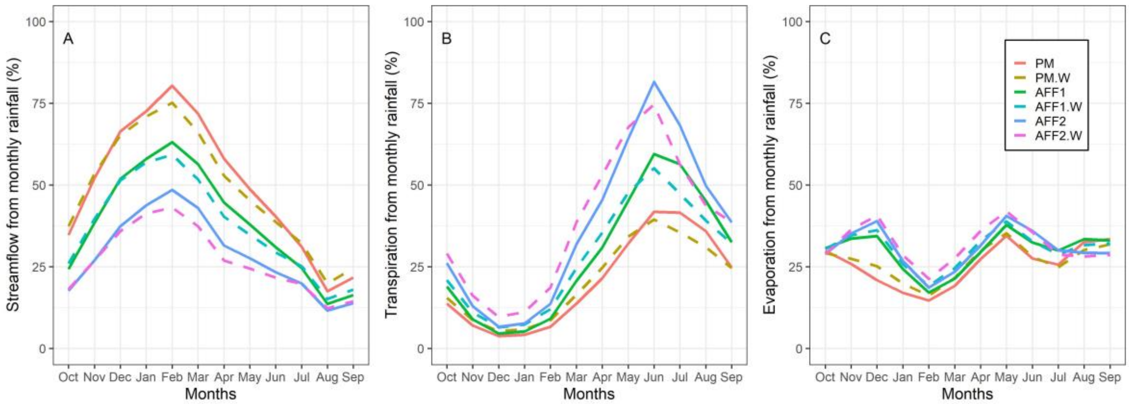

Figure 6.6. Average monthly percentage of water fluxes from monthly rainfall under the different land cover and climate scenarios. A: Streamflow, B: Transpiration, C: Evaporation. PM and PM.W: No management: natural revegetation. AFF1, AFF1.W: Active management, low afforestation, AFF2 and AFF2.W: Active management, intense afforestation. PM.W, AFF1.W and AFF2.W represent warming climate scenarios.

3.3. Changes to Water Use Efficiency

Results show important changes to WUE following management. Afforestation decreases annual WUE compared to natural revegetation scenarios. Average WUE ranges between 1.4 g C·mm^{-1} and 1.7 g C·mm^{-1} with significant differences between PM and AFF2 scenarios and relatively high inter-annual variability (Figure 6.7A, Table 6.4, Supplementary Table 6.3). Differences in WUE related to climate change, though non-significant, are more clear under afforestation scenarios where the warming climate results in an increase in WUE (Figure 6.7A, Supplementary Table 6.3).

At a monthly timescale, results show lower WUE during winter months that increases to reach a maximum during summer months. Land cover and climate scenarios show similar patterns across the year, while the highest differences mainly coincide with the growing season (March - August) (Figure 6.7B), reflecting the higher vegetation activity during that period.

Table 6.4. Annual water-use efficiency under the different land cover and climate scenarios. PM and PM.W: No management: natural revegetation. AFF1, AFF1.W: Active management, low afforestation, AFF2 and AFF2.W: Active management, intense afforestation. PM.W, AFF1.W and AFF2.W represent warming climate scenarios.

Scenarios	PM	PM.W	AFF1	AFF1.W	AFF2	AFF2.W
Water-use efficiency (g C·mm^{-1})	1.7	1.7	1.5	1.5	1.3	1.4
Coefficient of variation (%)	24.9	26.1	25.3	26.4	29.2	29.3

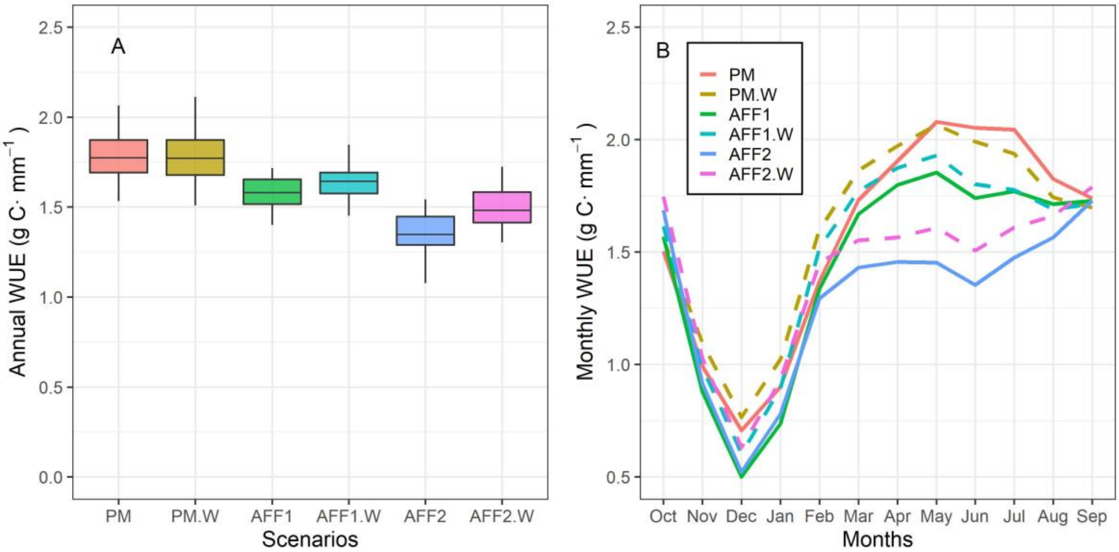

Figure 6.7. Annual and monthly Water-Use Efficiency (WUE) under the different land cover and climate scenarios. A: Annual, B: Monthly. PM and PM.W: No management: natural revegetation. AFF1, AFF1.W: Active management, low afforestation, AFF2 and AFF2.W: Active management, intense afforestation. PM.W, AFF1.W and AFF2.W represent warming climate scenarios.

4. Discussion

4.1. Carbon cycle

Forests are well recognized as carbon sinks in global carbon analysis (Pan et al., 2011; Harris et al., 2021), although there is no scientific consensus regarding the relative roles played by unmanaged and managed forests. Afforestation may accelerate the living-biomass carbon fixation (Law et al., 2018; Nave et al., 2019), and increase its accumulation in soil layers (Nave et al., 2018). However, the net carbon uptake varies considerably between areas, climate, forest type and age of the stands. Fang et al. (2007) reported a net carbon sink in a temperate pine plantation while secondary forests were nearly in a balanced carbon exchange. In a tropical Asian rainforest, Zhang et al. (2010) pointed out an average carbon sink of 168 g C·m^{-2}·yr^{-1}. Valentini et al. (2000) reported a large variation in European forest carbon sequestration ranging between 660 g C·m^{-2}·yr^{-1} to -100 g C·m^{-2}·yr^{-1}.

In this Mediterranean location, afforestation increases annual NEP compared to natural revegetation. Average annual NEP under afforestation scenarios reached up to 112 g C·m^{-2}·yr^{-1} while under the natural revegetation scenario, NEP is negative (-74 g C·m^{-2}·yr^{-1}). Afforestation NEP is within ranges noted above and consistent with other studies in Mediterranean regions. For instance, in an arid Mediterranean *Pinus halepensis* afforestation, Grünzweig et al. (2003) reported an average annual net carbon sequestration budget of 150 g C·m^{-2}·yr^{-1}. These estimates confirm that afforestation in the Spanish Pyrenees is likely to increase carbon sequestration.

On the other hand, while differences between the historic and the warming climate scenarios are non-significant, increases in temperature tended to decrease annual NEP particularly under the natural revegetation scenario. At a monthly time scale, the begining of the growing season marks the transition between the ecosystem as a carbon source between mid-February and July, to a carbon sink during the rest of the year. Notably under warming scenarios summertime carbon source strength increased substantially. While this was mostly balanced by greater carbon uptake during the winter, results may suggest a threshold where carbon losses will dominate - particularly if there is any decline in precipitation. Serrano-Ortiz et al. (2015) presented similar results in alpine and subalpine sites in southeastern Spain with high dependency of carbon fluxes on water availability and temperature. When the photosynthetic activity decreases substantially in autumn and winter months, carbon emission from respiration process exceeds low uptake from photosynthesis.

The multi-year temporal pattern of annual carbon sequestration under the different vegetation scenarios shows a first phase of carbon emission followed by a substantial increase in annual NEP. After reaching the peak carbon sequestration, annual NEP starts decreasing and generally stabilizes 10 years after management. These results follow general patterns of carbon uptake under afforestation. In a global analysis of carbon fluxes under different forest types, Chen et al. (2020b) revealed low to negative carbon balances in afforestation of less than 10 years while during the first 10 to 20 years, carbon sequestration reaches its maximum levels. Similar results were provided by Bond-

Lamberty et al. (2004), concluding a low carbon source NEP in the youngest stands of a boreal forest in Canada, while middle-aged stands are the highest carbon sinks and the oldest stands are relatively neutral. Carbon fluxes patterns and differences between land cover scenarios are highly related to vegetation structural properties. The higher Leaf Area Index (LAI) and the deeper rooting system of forests, relative to grasslands and shrubs, ensures higher photosynthetic activity particularly in seasons and years with greater water availability, all of which contribute to a higher carbon uptake under afforestation compared to natural revegetation.

Changes to carbon cycle are also related to climate variability. In general, annual precipitation determines ecosystem moisture, which affects LAI and GPP (Law et al., 2002) and therefore evapotranspiration and NEP. Our modeling results show significant positive correlations between annual NEP and rainfall under the different land cover and climate scenarios. These results highlight the substantial role of water availability to enhance carbon assimilation through photosynthesis. While NEP in most of our scenarios was positive, indicating a carbon sink, in drier years NEP was often negative. Similarly, Pereira et al. (2007) found declines in NEP due to a drought event in a Mediterranean ecosystem in southern Portugal. This suggest that in water limited environments, such as the Mediterranean region, large-scale afforestation plans could have limited or negative effects on carbon sequestration for some climate scenarios. Notably, correlations between precipitation and NEP are higher under the warming scenario, reflecting a stronger dependency of carbon sequestration on water availability compared to the historic climate. This negative effect could substantially reduce the carbon sequestration efficiency of large-scale afforestation as a climate change mitigation choice in the Mediterranean basin where an increase in drought severity and frequency is forecasted in the future (Tramblay et al., 2020).

4.2. Water cycle

Changes to water fluxes due to land cover scenarios are significant while changes related to warming are non-significant. Afforestation decreases annual streamflow by up to 50% and doubles transpiration rates compared to the natural revegetation scenario. Consistent with our results, Nadal-Romero et al. (2016b) found similar runoff coefficients in an afforested experimental small catchment in the Central Spanish Pyrenees. In a global synthesis of afforestation programs, Farley et al. (2005) pointed out that in areas where runoff represents 30% of annual precipitation, afforestation may result in a significant decrease of 50% in runoff. The increase in vegetation density following land abandonment have been reported to reduce water yield in other Mediterranean mountain areas (García-Ruiz et al., 2011; López-Moreno et al., 2011). The higher LAI and the better access to deep water sources are two primary causes for the increase in transpiration (Engel et al., 2005; Khorchani et al., 2021a). This change in the partitioning of the precipitation could be highly conditioned by the annual rainfall regime. Farley et al. (2005) concluded that in high rainfall areas, the evaporation of the intercepted rainwater by the vegetation is the main driver of the increase in water loss, however in drier environments the ability of the vegetation to reach and use deeper water resources to sustain transpiration is the primary determinant in water yield changes.

Societal water availability in the Mediterranean region strongly depends on mountain areas (López-Moreno et al., 2008; García-Ruiz et al., 2011), which makes of the timing of the highest hydrological changes very important for a successful management of water resources. In this study, the highest changes to streamflow occur during the wet period between December and May, when human demands for water are lower. Greater wet season streamflow changes are likely related to generally higher transpiration rates during the wet season.

4.3. Linking water and carbon cycles

Afforestation decreases the average WUE to 1.4 g $C \cdot mm^{-1}$ compared to the natural revegetation scenario (1.7 g $C \cdot mm^{-1}$), although differences are only significant between PM and AFF2 scenarios. These WUE averages are comparable to global estimates obtained in Tang et al. (2014) from flux tower upscaling (1.71 g $C \cdot mm^{-1}$) and MODIS data (1.89 g $C \cdot mm^{-1}$), although slightly lower than estimates by Law et al. (2002) of WUE averages of 2.4 g $C \cdot mm^{-1}$ in evergreen coniferous forest and 3.2 g $C \cdot mm^{-1}$ for deciduous broadleaf forests, using the FLUXNET international network. In contrast to our results, Liu et al. (2015), revealed higher WUE in forests compared to shrublands in China. Differences between results obtained in these studies could be related to site specific characteristics related to vegetation type and climate since annual WUE is conditioned by inter-annual climate variability and latitude (Tang et al., 2014).

In this study, the increase in evapotranspiration under afforestation compared to natural revegetation is not compensated by similar increases in GPP, leading to a decline in WUE.

These differences are related to the physiological and structural characteristics of mature vegetation in the two scenarios. Relative to shrubs, mature trees lose a greater proportion of GPP to respiration in order to maintain their greater biomass. For shrubs, the reduced access to deep water, induced by a less developed rooting system than trees, limit water loss through evapotranspiration. In this Mediterranean region moisture conditions are one of the major drivers of ecosystem WUE (Liu et al., 2017). At monthly time scale, WUE is at minimum during the winter months and reaches maximum value in the summer months. This pattern is strongly attributed to the active period of plants and is consistent with transpiration and NEP patterns. Gains in water use efficiency under natural revegetation, relative to afforestation, occur primarily during the summer.

4.4. Management considerations

A conceptual model that summarizes the interactions between carbon and water cycles is presented in Figure 6.8. Although the relation between the two cycles is more complex than depicted here, depending on vegetation age, physical attributes and spatial expansion, this conceptualization provides a first order approximation that emphasizes potential tradeoffs. Recognizing that these tradeoffs exist is critically important for land management planning in Mediterranean mountain areas. Land managers will need to reconcile carbon gains and water losses to ensure the sustainability of ecosystem services supply. Afforestation is a key tool in the efforts to mitigate the increase in CO_2 concentrations in the atmosphere, however they could have significant consequences on the hydrological cycle. Our model based analysis of the Arnás catchment shows how these consequences can be quantified and illustrate that greater carbon assimilation rates can significantly decrease not only water yield but also water-use efficiency, or the gain in carbon per water lost. Results also show that impacts are not linear, with greater declines in the more intense afforestation scenarios. Targeted levels of afforestation will need to take these dynamics into account. A targeted optimum level of carbon sequestration should consider local physical and climatic particularities and management as these could significantly condition the interactions between carbon and water cycles. Our modeling approach is readily extendable to other sites. While this study relied on available streamflow data for model calibration, sites with similar geology can reliably use similar parameters without calibration (Tague et al., 2013).

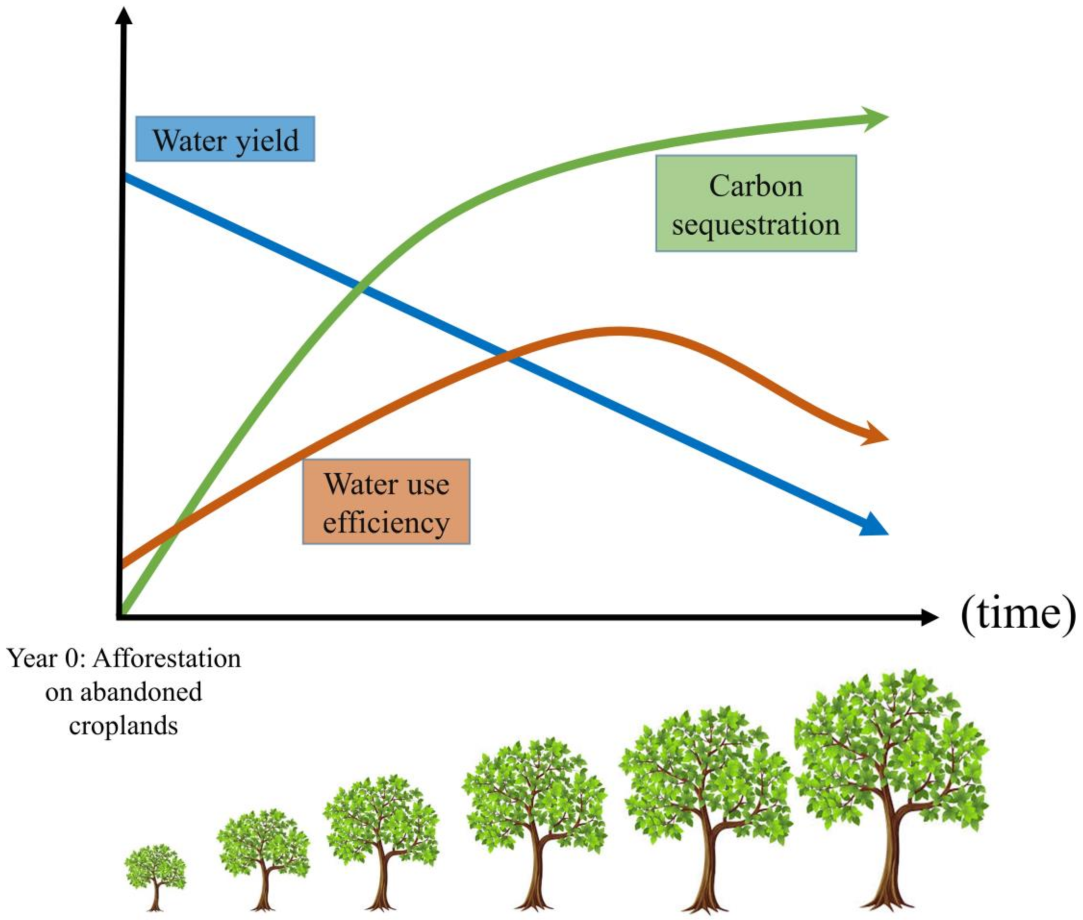

Figure 6.8. Conceptual model of the tradeoffs between carbon sequestration and water yield in Mediterranean mountain areas during afforestation of abandoned agricultural lands.

Results in this study also support studies that recommend forest density reduction in Mediterranean environments as an appropriate measure to ensure carbon sequestration without altering water availability (Varela et al., 2020). Our modeling approach could be used to determine the appropriate level of density reduction to maintain benefits from carbon sequestration programs while limiting consequences for water availability. One of the main objectives of afforestation programs in the Spanish Pyrenees was the economic benefit that these forests could generate through management and rotation cycles. However, decades later, most of afforested areas have not been exploited. How this land is managed may have substantial impacts on both water and carbon fluxes (Nadal-Romero et al., 2016b).

In this study, temperature increase does not significantly affect differences among scenarios. However, results do show a strong negative relationship between carbon sequestration and annual precipitation. Therefore, the efficiency of afforestation plans could significantly decrease in this and similar water limited environments, if precipitation declines. Similarly, in regions within the Central Spanish Pyrenees with lower precipitation, efficiency may decrease. Maximizing carbon sequestration gains for a given water budget will be an important consideration in strategies for mitigating climate change in the Mediterranean region. Although this study deepens insights on water costs of two possible restoration plans after cropland abandonment, more work is needed to identify optimum management plans ensuring sustainable water availability that are tailored to local catchments and changing climates. In that sense, future research in Mediterranean environments should focus on how much afforestation, an ecosystem can permit, without altering its water resources.

4.5. Model uncertainties and limitations

The results obtained in this study could be relevant for developing land management plans for the abandoned cropland areas in Mediterranean mountains, particularly for planning that tries to balance tradeoffs between enhancing carbon sequestration and maintaining water resources. However, extrapolating these results from local to regional scales will need to account for the diversity of geoclimatic and ecosystem conditions within Mediterranean environments. Further, models are always approximations of reality and uncertainty is an intrinsic property of all modeling tools (Orth et al., 2015). Uncertainty and error in model estimates arise from model inputs, calibration data and/or model structure (Moges et al., 2021). Therefore, generalizing model results to a wider spatial scale will need additional model evaluation across a range of sites to quantify this uncertainty. Further, evaluation of techniques to reduce uncertainty and error in inputs, particularly in model climatic inputs would increase confidence in results. Results from this study offer a general conceptual framework to guide the evaluation of water and carbon tradeoffs that will arise from land cover transformations in the Mediterranean mountains. This study also provides a first order approximation of the magnitude and direction of potential changes to water and carbon with different land management strategies and as such, motivates the need for additional analysis to support effective decision making on abandoned croplands.

5. Conclusions

This study presents a general assessment of tradeoffs between carbon sequestration and water resources following two restoration strategies after cropland abandonment in Mediterranean mountain areas during the first 50 years after cropland abandonment. Afforestation significantly increases carbon sequestration (NEP) compared to natural revegetation (shrub scenario before entry of trees). The increase is highly conditioned by vegetation physiological attributes, age and rainfall patterns. Annual carbon sequestration was significantly greater in wetter years, making of the efficiency of large-scale afforestation plans in Mediterranean environments highly dependent on water availability. On the other hand, afforestation significantly decreases water yield compared to natural revegetation restoration, primarily through increasing transpiration rates. Furthermore, WUE drastically decreases with afforestation. Carbon sequestration and water yield changes related to climate warming were small relative to these management effects. In this catchment, the beginning of the growing season is associated with a transition of the ecosystem from being a carbon source to carbon sink. The growing season is also associated with the highest changes to transpiration rates with both afforestration and natural revegetation, while the highest changes in streamflow occur during the wet period. These results contribute to a better understanding of the interactions between water and carbon cycles that can ultimately support adapting large-scale restoration programs to the particularities of Mediterranean mountain regions, which are providers of substantial ecosystem services for lowland areas.

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Chapter 7. Effects of cropland abandonment and afforestation on soil properties and soil redistribution in a small Mediterranean mountain catchment

This Chapter studies the effects of land use and land cover change, following the abandonment of agricultural lands, in the Central Spanish Pyrenees, on soil properties and soil redistribution rates. Physico-chemical soil properties and nutrients are measured in 61 soil samples and differences between land use and land covers are studied. Fallout ^{137}Cs was used to estimate soil redistribution rates.

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Abstract

In slopes of Mediterranean mid-mountain areas, land use and land cover changes linked to the abandonment of cropland activity affect soil quality, physico-chemical properties and soil redistribution; however, limited attention has been paid to this issue at catchment scale. This paper evaluates the effects of cropland abandonment and post-land abandonment management (through natural revegetation and afforestation) on soil properties and redistribution rates using fallout ^{137}Cs measurements in the Araguás catchment (Central Spanish Pyrenees). A total of 52 grid soil samples from the first 30 to 40 cm and 9 sectioned reference samples were collected across the catchment and their physico-chemical properties were analyzed. Fallout ^{137}Cs was measured in a 5 cm sectioned references samples and in bulk grid samples. ^{137}Cs inventories were used to estimate soil erosion and deposition rates across the catchment. Results show that land use and land cover changes and active restoration after cropland abandonment through afforestation had significant impacts on physico-chemical soil properties and soil nutrients. The highest erosion rates were recorded under sparsely vegetated sites in the badland area, while the lowest rates were found in the afforested area, but no differences were observed between the different uses and covers in soil redistribution rates due to a long history of human intervention through cultivation in steep slopes and afforestation practices. However, the recovery of the soil organic carbon and nitrogen stocks in afforested areas suggest that afforestation can reduce soil degradation at long-term scale. The information gained achieves a better understanding of soil nutrients and soil redistribution dynamics and provide knowledge for effective land management after cropland abandonment of agroecosystems in Mediterranean mountain areas.

1. Introduction

Mediterranean mid-mountain areas are sensitive agroecosystems prone to land degradation (García-Ruiz et al., 2013). Land use and land cover changes (LULCC) have been a continuous feature in these areas since the beginning of human civilization (García-Ruiz, 2010). Mountain slopes were cultivated with cereal crops, even on steep slopes, stony soils and under extreme climate conditions (Lasanta et al., 2020). However, during the second half of the 20th century, the farming systems in the Mediterranean mountains were progressively abandoned and a forest cover subsequently re-expanded through natural revegetation processes (passive restoration) or afforestation (active restoration) (García-Ruiz et al., 2020).

LULCC have been identified as major drivers of Global Change due to their impact on ecosystem services, such as water resources and soil (soil quality, soil degradation and soil organic carbon sequestration). In addition, there is an overall agreement that LULCC and climate change will be rapid and strong in Mediterranean mid-mountain areas (García-Ruiz et al., 2011; Lionello and Scarascia, 2018).

Natural revegetation processes and afforestation after cropland abandonment are different strategies to restore soil ecosystem services, such as nutrients, soil conservation and carbon sequestration (Bell et al., 2020). In general, natural revegetation after land abandonment results in a decline in water resources (Khorchani et al., 2020; 2021a; Peña-Angulo et al., 2021; Vicente-Serrano et al., 2021), a decrease in soil loss and sediment delivery (García-Ruiz and Lana-Renault, 2011; Nadal-Romero et al., 2013), and changes in the connectivity between sediment sources and channels (Llena et al., 2019), as well as a progressive improvement in soil characteristics (Navas et al., 2008; Lasanta et al., 2020).

Afforestation has been increasingly implemented around the world, and extensive afforestation programs were conducted by national forest services all over the Mediterranean region (i.e. Yaşar Korkanç, 2014). In Spain during the first half of the 20th century, afforestation plans were adopted to meet a number of socioeconomic and environmental needs: (i) create employment and diversify rural incomes, (ii) generate prime materials for paper industry, (iii) control the hydrological and geomorphological processes (particularly in the degraded areas due to past agricultural activities), and (iv) prevent flooding and reduce check dam siltation (Ortigosa et al., 1990). Recently, afforestation has been promoted also as a strategy to mitigate CO_2 emissions, as well as to prevent soil erosion, reverse land degradation and restore soil properties and quality (Romero-Díaz et al., 2010). In the Spanish Pyrenees, as in most Mediterranean mid-mountains, afforestation has been based mainly on several pine species. Pines (adapted to the ecological conditions of Mediterranean mid-mountain areas) are fast growing trees and in theory lead to a quick soil hydrologic restoration and the formation of a protective vegetation cover. In that sense, there is evidence that afforestation not only alters aboveground vegetation, but also leads to significant changes in soil properties and biochemical soil cycles, as well as soil conservation and soil erosion (Chirino et al., 2006; Cuesta et al., 2012; Nadal-Romero et al., 2016a). Likewise, afforestation also reduces water yields and soil loss (Andréassian, 2004; Khorchani et al., 2021b; Scorpio and Piegay, 2021), and modifies the connectivity between sediment sources and channels (Sanjuan et al., 2016).

Mediterranean soils are considered the most fragile part of the ecosystem due to the low organic matter content and the low rate formation, with thin and poorly developed soils (García-Ruiz et al., 2013). Thus, the impact of land degradation and soil erosion processes on Mediterranean ecosystems, as well as its role as source and sink of soil organic carbon (SOC), have received increasing attention during the last decade (Navas et al., 2012; Quijano et al., 2016; Romero-Díaz et al., 2017; Lizaga et al., 2019; Cerdà et al., 2021; Bell et al., 2021; Gaspar et al., 2021; Nadal-Romero et al., 2021). Since vegetation recovery differs from natural revegetation to afforestation, soil properties and soil conservation are expected to be different under these contrasting scenarios of passive and active restoration. However, debates concerning which strategy is better have been published (Guo and Gifford, 2002; Khorchani et al., 2022), and there is still no agreement in the scientific community.

To understand land degradation processes, soil redistribution processes need to be quantified to assess how LULCC and post-land abandoned practices affect soil loss in Mediterranean mid-mountain areas. Different methodologies have been used to estimate erosion rates worldwide (García-Ruiz et al., 2015), but few studies have used caesium-137 (^{137}Cs) to estimate and quantify soil erosion rates and soil redistribution at catchment scale (i.e., Porto et al., 2003; Navas et al., 2013; Lizaga et al., 2018, 2019), and only some examples have been published investigating the effects of land abandonment and post-land abandoned practices (passive and active restoration) on soil erosion using the proposed approach (i.e. Evrard et al., 2010; Gaspar and Navas, 2013; Navas et al., 2017). Since the first application of caesium-137 fallout in Mediterranean mountains (Navas and Walling, 1992), LULCC was identified as a main factor of soil mobilisation driving the source to sink paths of sediments (Quine et al., 1994). The role of rapid LULCC triggered by cropland abandonment on soil redistribution was first assessed at catchment scale using a transect based on ^{137}Cs approach (Navas et al., 2005), and detailed information of ^{137}Cs profiles allowed to interpret changes in soil properties and on the lateral transfer of soil and nutrients (Navas et al., 2012; Gaspar et al., 2019). Later, the potential of grid setups of ^{137}Cs measurements provided fundamental data to address the spatial patterns of soil redistribution and that of associated nutrients in catchments with dynamic LULCC after cropland abandonment (Navas et al., 2011; Gaspar et al., 2021). Grid ^{137}Cs estimates in complex catchments with intricate mosaic of land use and land covers (LULC) allowed identification of which of them acted as sinks or sources of soil particles and soil dynamics (Navas et al., 2014). Due to the magnitude of LULCC affecting Mediterranean mountain agroecosystems further insights on the effect of such changes was assessed in medium size catchments by Lizaga et al. (2018) were ^{137}Cs derived soil redistribution was linked to afforestation, revegetation processes and agricultural practices with clear impacts on soil properties after cropland abandonment (Lizaga et al., 2019).

The main objective of this study is to assess the effects of cropland abandonment and post-land abandonment management (through natural revegetation and afforestation) on soil properties and soil nutrients, and quantify soil redistribution rates using fallout ^{137}Cs measurements at catchment scale. The specific objectives are to (i) establish the reference inventories of ^{137}Cs and soil nutrients in soils of representative undisturbed Mediterranean vegetation cover, (ii) assess at catchment scale the spatial patterns of soil properties and soil nutrients, soil organic and inorganic carbon and total nitrogen related to LULCC, and (iii) to compare the reference inventories with the values of ^{137}Cs inventories at the sampling points at different LULC, and establish the areas where loss and gain of ^{137}Cs fallout has occurred within the catchment, in relation to cropland abandonment, afforestation practices and soil redistribution.

We start from the following research hypotheses: (i) LULCC after cropland abandonment and post-land abandonment practices have a significant impact on soil nutrients and soil redistribution rates, and (ii) afforestation can accelerate the recovery of specific soil properties and reduce soil degradation after cropland abandonment compared to natural revegetation. The study was carried out in a small experimental mountain catchment located in the Central Pyrenees (NE Spain) with four different LULC: (i) badland areas and sparsely vegetated areas as a consequence of intense geomorphological and degradation processes, (ii) pasturelands (present agricultural activity), (iii) shrubs area growing in old croplands; and (iv) afforested areas with *Pinus nigra* and *Pinus sylvestris*. Previous studies have been carried out in the Araguás catchment related to hydrological dynamics (Nadal-Romero et al., 2016b, 2018), erosion processes (Nadal-Romero et al., 2015) and the effects of afforestation and land use changes in soil properties and SOC dynamics (Nadal-Romero et al., 2016a, 2016c). However, in the Araguás catchment, no information has been published so far, on the effects of LULCC and post-land abandonment management at catchment scale (through natural revegetation and afforestation) on soil properties, soil nutrients, and soil redistribution, neither in the soil redistribution using fallout ^{137}Cs measurements.

2. Materials and methods

2.1. The Araguás catchment

The Araguás catchment (0.45 km^2) is a north-south small experimental catchment, with altitudes between 780 and 1100 m a.s.l. (Figure 7.1A, 1B, 1C). The present landscape is a complex mosaic in which alternate: (i) afforested areas, that were previously cultivated with cereal crops in terraced fields and were afforested in the late 1960s with Black pine (*P. nigra*) and Scots pine (*P. sylvestris*) (Figure 7.1D), (ii) dense and open shrub areas that were also cultivated and underwent a process of natural plant colonization (natural revegetation) with *Genista scorpius*, *Juniperus communis*, *Rosa gr. canina* and *Buxus sempervirens* (Figure 7.1E), (iii) small agricultural areas characterized by permanent pasturelands for grazing but not currently tilled (Figure 7.1E), and (iv) badland areas characterized by sparsely vegetated areas and extreme geomorphological dynamics (Figure 7.1F).

Two different lithologies are presented in the catchment (Figure 7.1C): Eocene marls and Eocene turbidites (flysch deposits consisting of bedded thin layers of sandstones and marls), in the lower and upper part of the catchment, respectively. The soils are stony and shallow, resulting from centuries of cultivation, and are classified as Calcaric Leptic Regosols following the WRB taxonomy (IUSS Working Group WRB, 2015).

Climate in the area is sub-Mediterranean with oceanic and continental influences and mean annual temperature is 10 ºC (ranging between -15 ºC and > 30 ºC) and mean annual precipitation is 800 mm (oscillating between 500 and 1000 mm).

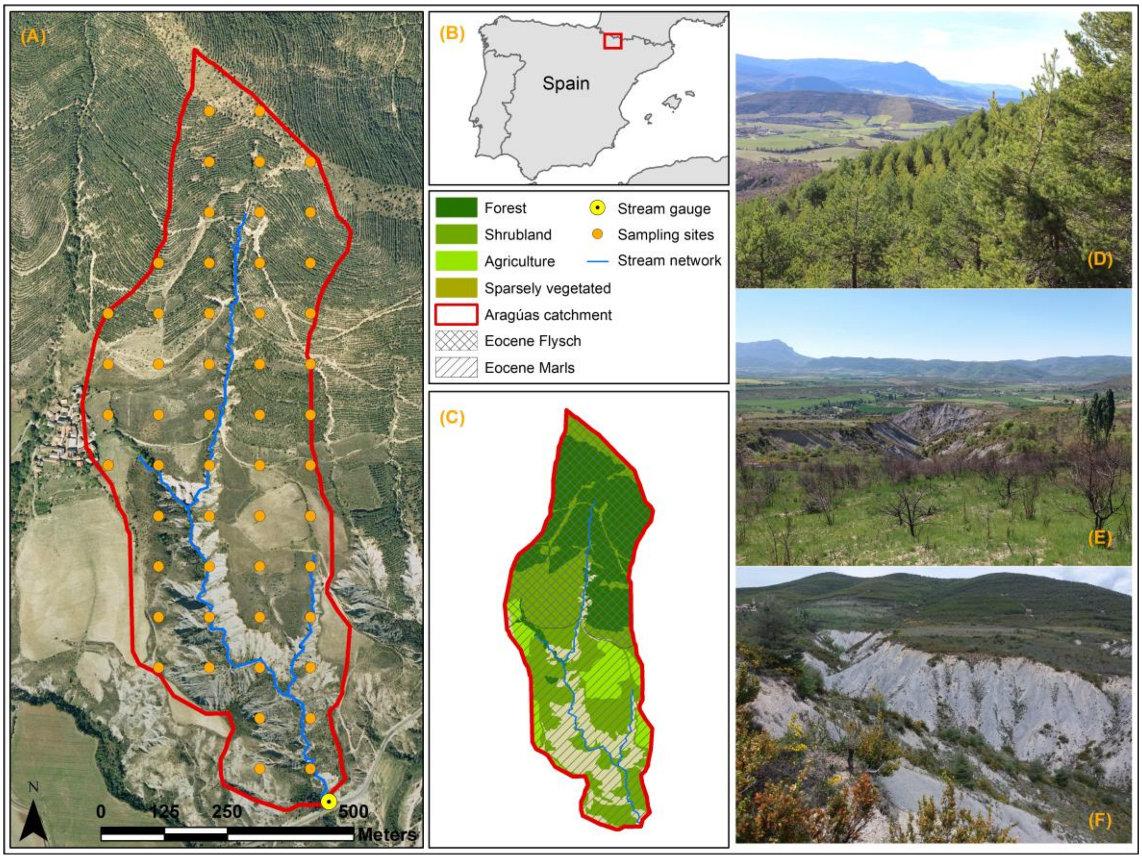

*Figure 7.1. The Araguás catchment. A: the sampling network composed by a grid of 100 * 100 m; B: localization of the catchment in Spain; C: land cover map using the aerial photography PNOA 2018; D: afforestation in the upper part of the catchment; E: natural revegetation in the abandoned croplands; F: eroded marls around the main stream and pasturelands in the upper part of the slopes.*

2.2. Soil sampling design

In 2019 a total of 52 bulk core soil samples were collected (Figure 7.1A). A steel core tube was used to collect two replicates of bulk soil samples at each sampling point from the surface until a depth varying from 30 to 40 cm depending on the local soil thickness. The sampling points were distributed proportionally across the catchment surface using a 100 x 100 m grid with a sampling density of 1.2 sample·ha^{-1} (see Figure 7.1A).

In order to establish the local reference inventory for the experimental catchment, nine core samples were collected as reference sites in flat undisturbed locations under stable soil conditions, where neither erosion or deposition was expected to have occurred during the last decades. Sampling was done using core tubes on a depth varying from 40 to 45 cm. The soil cores were sectioned in 5 cm intervals in order to study the vertical distribution of ^{137}Cs (see Figure 7.2).

2.3. Soil analysis

The two soil cores from each sampling site were open and air-dried in the laboratory then mixed, homogenized and sieved to ≤ 2 mm. The following physico-chemical soil properties were determined in the laboratory at the Pyrenean Institute of Ecology (IPE-CSIC) and at the Experimental Station of Aula Dei (EEAD-CSIC): (i) electrical conductivity (EC) and pH were measured in a deionized water suspension (1:2.5) using a pH meter and a conductivity meter; (ii) particle size analysis was carried out with a Beckman Coulter LS 13 320 laser diffraction particle size analyser (Beckman Coulter Inc., 2011) after oxidizing the organic matter by pre-treating the soil with H_2O_2 (10%) in a boiling water bath at 80ºC and adding 2 ml of solution of a dispersing agent (40% sodium hexametaphosphate to avoid grain flocculation) (Murray, 2002); (iii) total carbon (TC), soil organic carbon (SOC) and total nitrogen (TN) were measured by dry combustion in an elemental analyser (LECO CNS 928, Leco Corporation); (iv) organic matter (OM) was calculated using the van Bemmelen factor, using as universal conversion factor, assuming that organic matter contains 58% organic carbon; (v) CN ratio was calculated using SOC and TN; (vi) $CaCO_3$ (%) was determined through the Bernard Calcimeter; (vii) SOC and TN stocks were calculated by multiplying each value by the respective soil sampling thickness (m) and bulk density.

The methodology followed for ^{137}Cs analysis has been widely described in the literature (i.e. Walling and Quine, 1991; Navas and Walling, 1992; Navas et al., 2005, 2008, 2011; among others). Measurements of ^{137}Cs mass activity were performed using a high resolution, hyperpure germanium, coaxial gamma-ray detector of the Experimental Station of Aula Dei (EEAD-CSIC, Spain) coupled to an amplifier and multichannel analyser (Canberra Xtra, Canberra industries, Inc. USA). The content of ^{137}Cs was expressed as a concentration or massic activity (Bq kg^{-1}). Estimates of soil redistribution rates (Mg ha^{-1} yr^{-1}) derived from ^{137}Cs inventories (Bq m^{-2}) were obtained by applying the conversion model reported by Soto and Navas (2004a, 2004b, 2008) for uncultivated and cultivated soils. These models compare the measured inventory with the local reference inventory and determine the erosion or deposition rates relative to the reference inventory.

2.4. Data analysis

As the assumption of normal distribution per factor when checked by the Shapiro Wilk normality test was met for most parameters, parametric tests were used to monitor differences between LULC and soil redistribution. The homogeneity of variance using Levene's test was also carried out. R. Pearson's correlation coefficients were used to assess the relationships between the different physico-chemical soil properties and ^{137}Cs values. A one-way analysis of variance (ANOVA) and the Tukey Post-Hoc tests (when the F test was significant) were performed to assess differences between LULC and to evaluate if erosion and deposition rates were different in function of land uses. In all cases, we considered differences to be statistically significant at $p < 0.05$.

In addition, a Principal Component Analysis (PCA) was performed to analyse the relationships between ^{137}Cs and main physical and chemical soil properties. The sampling adequacy of variables was analysed by the Kaiser-Meyer-Olking measure (> 0.50) and by Bartlett's test of sphericity (<0.005). The selection of the main components was based on the latent root criterion with eigen-values > 1.0. All statistical analyses were carried out using R 3.4.3.

Finally, an ordinary kriging with constant trend was selected to model the spatial distribution of soil properties at catchment scale. All the output maps and interpolations were performed using ESRI ArcGis software.

3. Results

3.1. Soil properties in the reference sites

Figure 7.2 shows the mean physical and chemical soil properties at the nine reference sites (see Supplementary Table 7.1). In general, soils in the undisturbed areas were shallow and poorly developed. Coarse fraction was homogeneously distributed through the soil profile. Most soil samples had a clay (34% of the samples) and silt loam (32% of the samples) texture, with silt fraction predominating in all depths (except > 40 cm). The mean values of silt, clay and sand were 41.2 ± 13%, 31.4 ± 13% and 27.4 ± 10% respectively. Soils were alkaline (pH 8.29 ± 0.21). The mean SOC content and SOC stock in the reference 5 cm intervals were 1.6% and 8.8 kg m^{-2} respectively. The mean total SOC stocks considering the complete soil profile was 72.6 kg m^{-2}. Total nitrogen was 0.16 ± 0.07% and TN stock considering the complete profile was 7.6 ± 1.5 kg m^{-2}. The reference ^{137}Cs inventory for the Araguás catchment was 1854.0 ± 101 Bq m^{-2}.

The reference profiles showed an exponential decrease of ^{137}Cs, SOC, TN from the surface to the deepest layers (Figure 7.2). The silt, SIC and coarse fraction were distributed relatively uniform with depth, and had no significant differences between the top and deep soil layers. ^{137}Cs massic activity was undetectable below 30 cm (Figure 7.2).

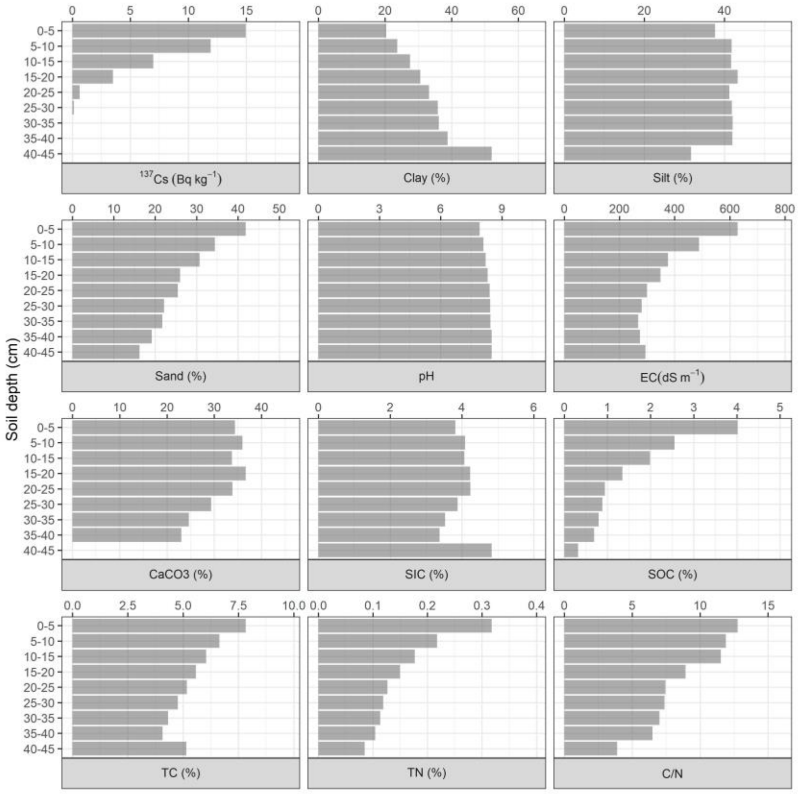

Figure 7.2. Depth distribution of the ^{137}Cs, SOC, SIC, TN, OM, soil texture, soil fine fraction and total SOC and N inventories in the reference site. Note: SOC, soil organic carbon; SIC, soil inorganic carbon; TN, total nitrogen.

3.2. Soil properties in the Araguás catchment and LULCC

All the samples had a silt-loam texture with a mean value of 63% of silt content, ranging between 57 and 76% (Figure 7.3). The content of the coarse material varied between 5 and 43%, with a mean value of 22% (Supplementary Table 7.1). Soils are alkaline with a mean pH value of 8.4 and a high carbonate content (mean value 38.9%). The mean contents of SOC and SIC were 1.1 ± 0.4% and 4.7 ± 0.8% respectively. Mean TN content was 0.1%, with maximum values below 0.3% (Figure 7.3). CN ratio oscillated between 4.5 and 12.4 with a mean value of 8.3 ± 2.1. Only in the afforested sites, the CN ratio was generally higher than 10, considering this value optimal for the best incorporation rate of the organic matter into the soil profile. Mean total SOC and TN stocks were 54.5 ± 17.3 kg m^{-2} and 6.8 ± 2.1 kg m^{-2}.

Table 7.1. Basic statistics of the soil properties in grid samples under the different land uses and land covers in the Araguás catchment. SOM: Soil Organic Matter, SOC: Soil Organic Carbon, TN: Total Nitrogen.

		Stonies (%)	Clay (%)	Silt (%)	Sand (%)	SOM (%)	SOC (%)	TN (%)	CN ratio	^{137}Cs activity (Bq kg^{-1})	^{137}Cs inventory (Bq m^{-2})
Afforestation n = 17	Median	24.3	17.5	64.0	20.6	2.3	1.3	0.1	10.7	3.8	1847.0
	Mean	24.0	17.1	64.0	18.9	2.4	1.4	0.1	10.3	3.2	1333.6
	SD	7.1	2.2	4.9	6.1	0.6	0.3	0.0	1.5	2.7	1088.4
	Max	35.3	20.9	72.4	28.8	3.5	2.0	0.2	12.4	7.1	2915.5
	Min	12.3	13.4	56.5	8.8	1.5	0.8	0.1	6.7	0.0	0.0
	CV	0.3	0.1	0.1	0.3	0.2	0.2	0.2	0.2	0.8	0.8
Shrublands n =20	Median	19.9	18.6	68.4	12.9	2.0	1.2	0.1	8.2	1.2	504.2
	Mean	22.1	19.5	67.8	12.7	1.9	1.1	0.1	7.7	1.9	939.9
	SD	7.3	2.8	3.0	4.8	0.5	0.3	0.0	1.8	2.0	1033.3
	Max	43.3	26.6	71.9	23.6	2.5	1.4	0.2	11.0	5.4	2544.0
	Min	9.7	15.7	59.9	6.2	0.7	0.4	0.1	4.5	0.0	0.0
	CV	0.3	0.1	0.0	0.4	0.3	0.3	0.2	0.2	1.1	1.1
Grasslands n = 5	Median	25.4	20.3	64.2	16.1	1.8	1.0	0.1	6.7	3.1	1733.3
	Mean	25.9	20.8	64.2	15.0	1.8	1.0	0.1	7.1	2.1	1100.0
	SD	2.7	2.2	3.3	4.5	0.7	0.4	0.1	1.1	2.0	1006.9
	Max	29.3	23.4	68.8	19.3	2.9	1.7	0.3	8.4	3.9	1942.4
	Min	22.3	18.1	60.4	8.6	0.9	0.5	0.1	5.9	0.0	0.0
	CV	0.1	0.1	0.1	0.3	0.4	0.4	0.5	0.2	0.9	0.9
Sparsely vegetated areas n = 10	Median	14.6	20.0	73.6	6.8	1.1	0.7	0.1	6.2	0.0	0.0
	Mean	15.4	20.0	72.4	7.6	1.3	0.8	0.1	6.5	0.9	526.7
	SD	6.2	2.4	3.6	3.7	0.5	0.3	0.0	1.4	1.5	871.1
	Max	25.2	24.0	75.6	16.8	2.3	1.3	0.2	9.6	3.4	2185.5
	Min	4.6	15.8	64.3	4.3	0.8	0.5	0.1	5.0	0.0	0.0
	CV	0.4	0.1	0.1	0.5	0.4	0.4	0.3	0.2	1.6	1.7

The mass activity of ^{137}Cs in the catchment ranged from below detection limit to 7.1 Bq kg^{-1} (only 14% of the samples had a massic activity higher than 5 Bq kg^{-1}) (Figure 7.3); and the inventories of ^{137}Cs varied from 0 to 2915.6 Bq m^{-2} (Supplementary Table 7.1). Significant differences were found between LULC (Figure 7.3). Silt and clay values were significantly lower in the afforested areas (Table 7.1). In these afforested areas, ^{137}Cs (excluding points below detection limit, Supplementtray Figure 7.1), SOC, CN ratio and sand content were higher than in shrubs and sparsely vegetated areas. pH values were higher in the sparsely vegetated areas. No differences were observed in EC values, SIC, $CaCO_3$, and TN between the different LULC (Figure 7.3 and Table 7.1).

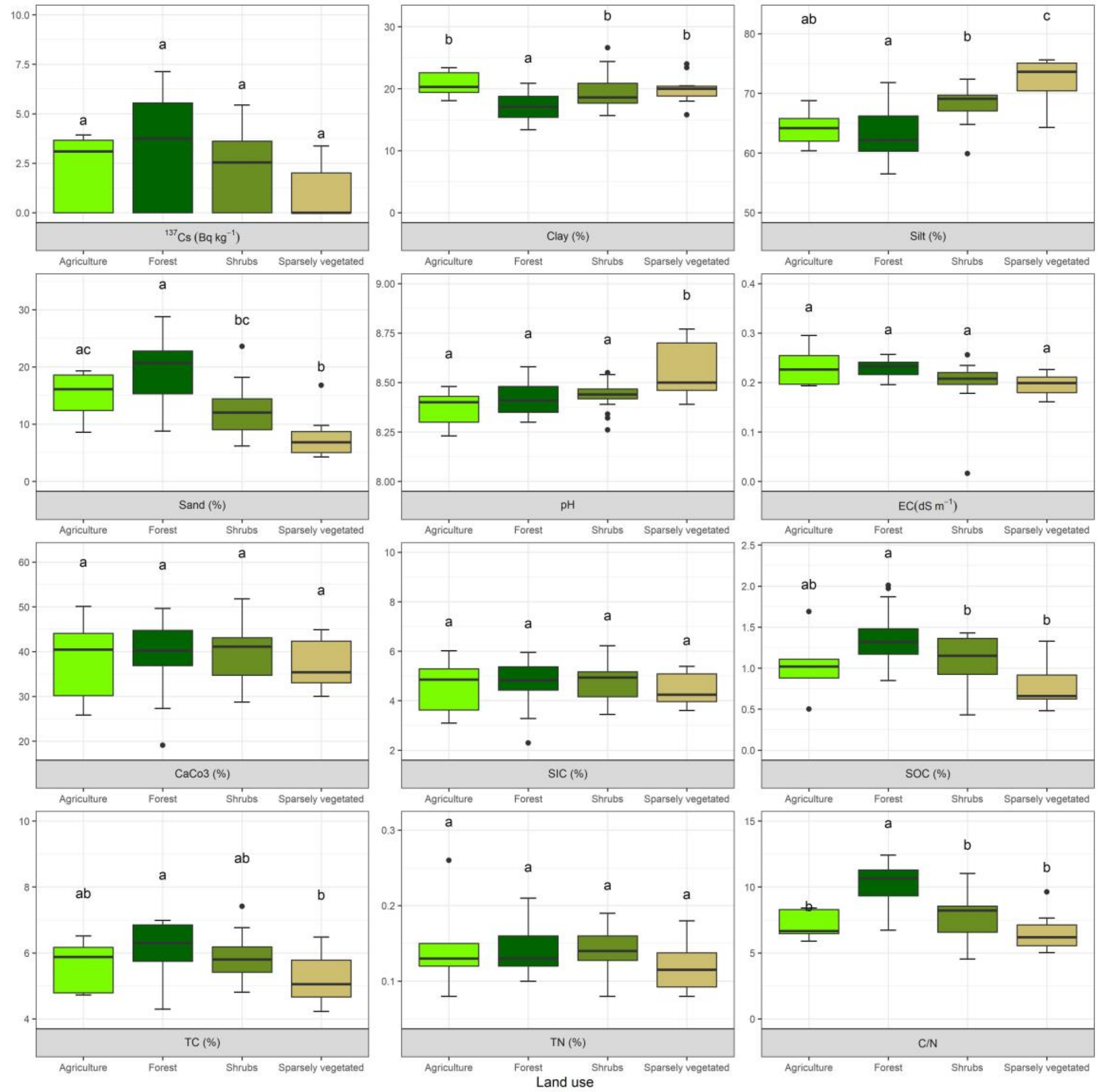

Figure 7.3. Boxplot with land uses classification and soil properties: Forest (afforestation), Shrubs (natural revegetation), Agriculture (grassland), Sparsely vegetated (badlands). SOC, soil organic carbon; SIC, soil inorganic carbon; TN, total nitrogen. Note, significant differences were found between land uses in ^{137}Cs when excluding values below detection limits (Supplementray Figure 7.1).

Figure 7.4 shows the spatial distribution of the interpolated soils properties and ^{137}Cs massic activity. In general, the soil properties were highly variable across the catchment, and the spatial pattern was not so clear for most of the properties (see Figure 7.4). Relatively high clay and silt content were recorded in the lower part of the catchment.

High SOC, TC, TN values were found at the upper part of the catchment, related to the presence of the afforested areas (Table 7.1). Whereas lower values were found in the lower part of the catchment, close to the ravine, related to the presence of sparsely vegetated areas and badlands. The pH and $CaCO_3$ did not show any clear distribution pattern in the catchment. It should be highlighted that the spatial pattern of nutrients showed in general good agreement with that of ^{137}Cs activity (Figure 7.4).

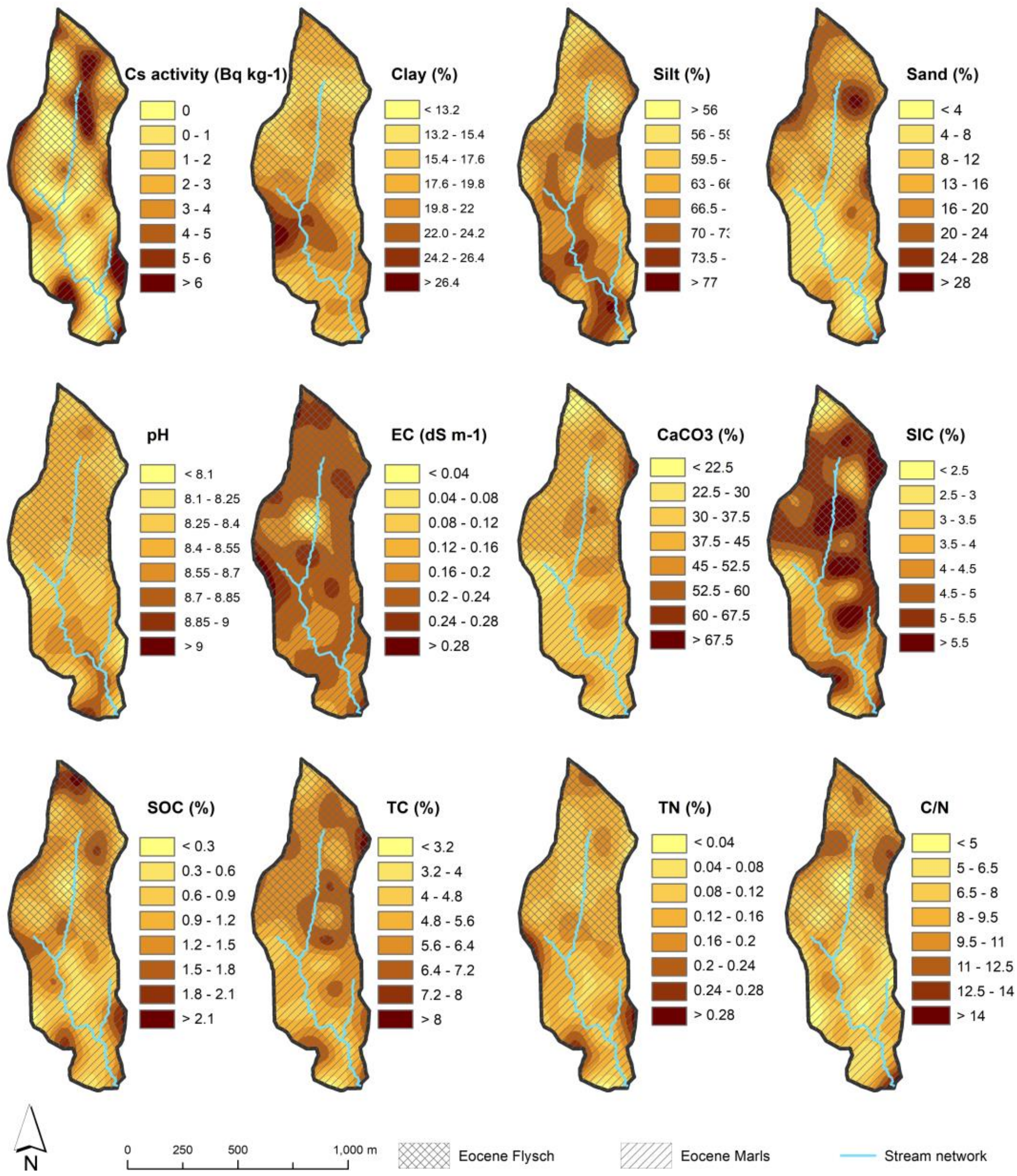

Figure 7.4. Spatial distribution of ^{137}Cs activity, clay, silt, sand, pH, EC, $CaCO_3$, SIC, SOC, TC, TN and CN ratio in the Araguás catchment produced by an ordinary kriging. Lithology and the stream gauge is also shown. Note: EC, electrical conductivity; $CaCO_3$, carbonate content; SIC, soil inorganic carbon; SOC, soil organic carbon; TC, total carbon; TN, total nitrogen.

Figure 7.5 shows the correlation between soil properties considering all the LULC. ^{137}Cs activity was directly and significantly correlated with sand, EC, SOC, TN and CN ratio, and inversely correlated with silt and pH. SOC was directly correlated with ^{137}Cs activity, sand EC, TN and CN, and inversely correlated with silt, pH, $CaCO_3$ and SIC. Similar patterns to SOC values were observed with TN.

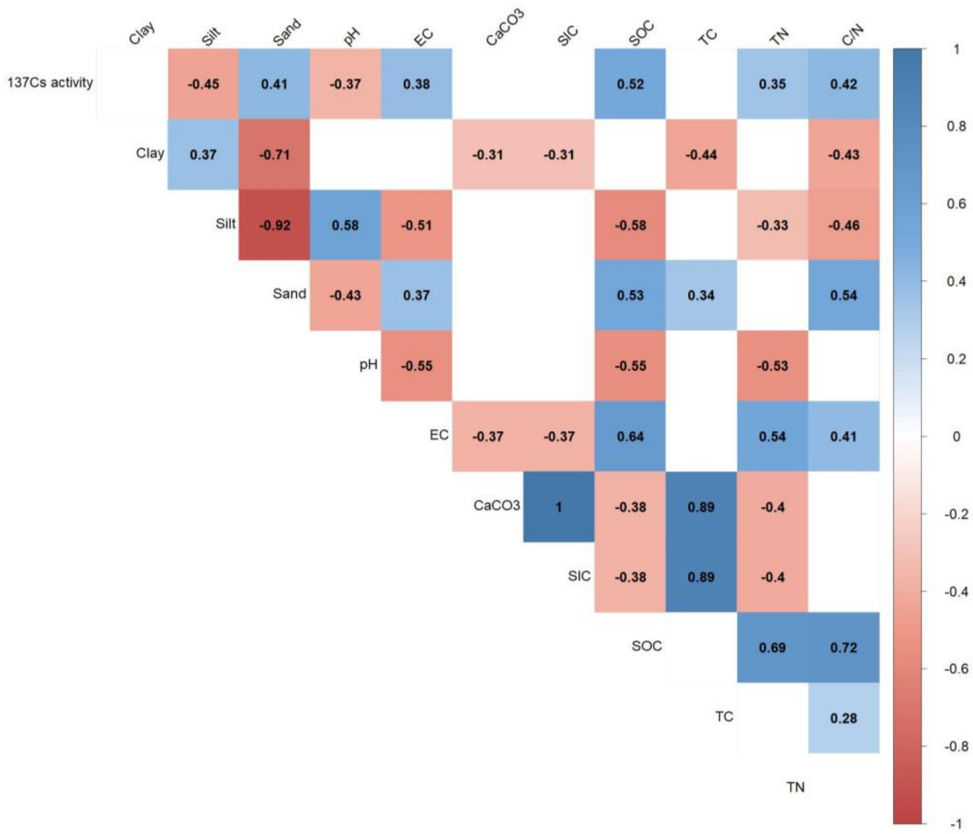

Figure 7.5. Correlation matrix of the main soil properties in the Aragúas catchment. Note: EC, electrical conductivity; $CaCO_3$, carbonate content; SIC, soil inorganic carbon; SOC, soil organic carbon; TC, total carbon; TN, total nitrogen.

Figure 7.6 shows the results of the PCA analysis and the PCA scores in the plane of PC1 and PC2. Values are large and positive for SOC, sand, EC, CN ratio and ^{137}Cs activity, and large and negative for silt and pH, which explain 41.7% of the variance. The second component explained 23.5% of the variance: it has large and positive eigenvector values for $CaCO_3$ and TC, and negative for clay and TN. Although no strong discrimination is evident, there are some differences between LULC. Forest sites are on the positive side of PC1, clearly separated from sparsely vegetated sites, and with large values of SOC, sand and CN ratios. The sites of agricultural use and shrubs are clustered together close to the centre of both components. Sparsely vegetated sites are partially distinguishable, on the negative side of both components with high silt and pH values.

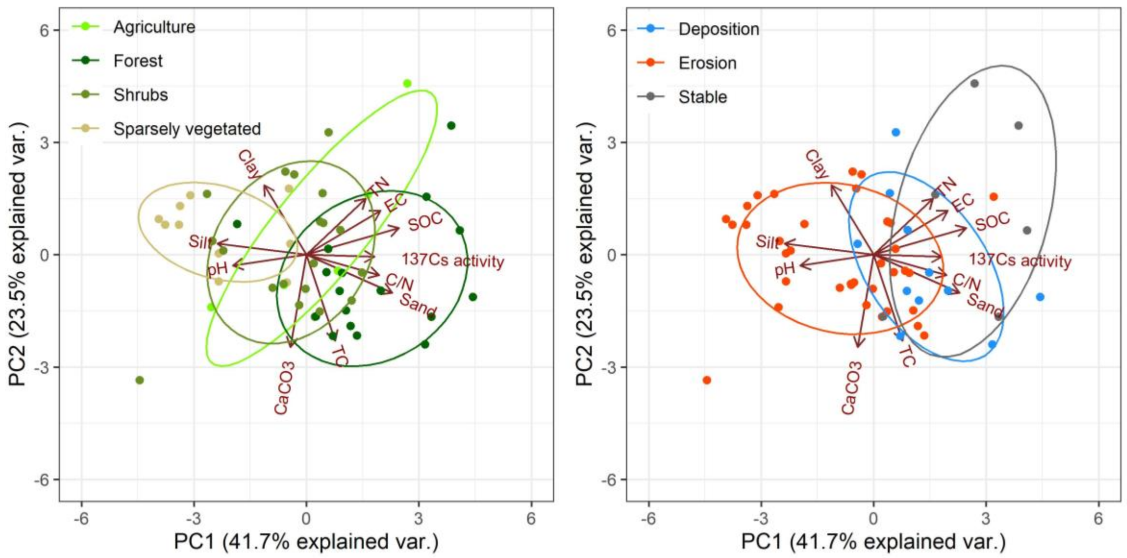

Figure 7.6. Dispersion diagram plots and principal component loadings of the two first components from a PCA for the complete set of soil samples. LULC and deposition, erosion and stable sites were also represented in the plots. Note: EC, electrical conductivity; $CaCO_3$, carbonate content; SIC, soil inorganic carbon; SOC, soil organic carbon; TC, total carbon; TN, total nitrogen. Forest (afforestation), Shrubs (natural revegetation), Agriculture (grassland), Sparsely vegetated (badlands).

3.3. The effects of soil redistribution on soil properties

Considering that the reference ^{137}Cs inventory was 1854 ± 101 Bq m^{-2}, samples with ^{137}Cs inventories higher than 1955 Bq m^{-2} were considered deposition sites, inventories lower than 1753 Bq m^{-2} were considered erosion sites, while inventories with deviations lower than 5.4% from the reference inventory were considered stable sites. Most soil samples (n=35, 67%) had lower values than the ^{137}Cs reference inventory and were identified as eroded points. Eleven samples (21%) had values higher than the reference inventory and were identified as depositional sites. Only 6 samples showed values that fall within the stability range (see Table 7.2).

The mean values of SOC and TN in the reference inventories were 72.6 and 7.6 kg m^{-2} respectively. Regarding SOC, most soil samples had lower values than the reference inventory, and only 17% of the soil samples had higher values than the SOC reference inventory, most of them located in the afforested area (Table 7.2). A general loss of TN was also recorded in the Araguás catchment: 28% of the points showed higher values than the reference sites, most of them located in the shrubland area.

Table 7.2. Mean values of soil properties at erosion, deposition and stable sites under the different land uses and land cover in the Araguás catchment. SOM: Soil Organic Matter, SOC: Soil Organic Carbon, TN: Total Nitrogen. Forest (afforestation), Shrubs (natural revegetation), Agriculture (grassland), Sparsely vegetated (badlands).

		n	Stonies (%)	Clay (%)	Silt (%)	Sand (%)	SOM (%)	SOC (%)	TN (%)	CN ratio	^{137}Cs activity (Bq kg−1)	^{137}Cs inventory (Bq m−2)
Erosion n=35	Forest	8	22.4	17.7	65.3	17	2.2	1.3	0.1	9.8	0.8	346.1
	Shrubs	15	22.2	19.4	68.4	12.2	1.7	1	0.1	7.7	1	471.9
	Agriculture	3	24.2	20	65.6	14.4	1.4	0.8	0.1	7	1	577.8
	Sparsely vegetated	9	15.5	19.9	72.4	7.7	1.3	0.7	0.1	6.6	0.7	342.4
Deposition n=11	Forest	5	26.07	16.4	63.8	19.8	2.4	1.4	0.1	10.2	6.3	2486.3
	Shrubs	5	21.8	19.7	66	14.3	2.2	1.3	0.2	7.8	4.5	2344
	Agriculture	-	-	-	-	-	-	-	-	-	-	-
	Sparsely vegetated	1	14.5	30.5	72.2	7.3	1.9	1.1	0.2	6	3.4	2185.5
Stable n=6	Forest	4	24.5	16.6	62.9	21.5	2.9	1.7	0.1	11.3	4.2	1867.7
	Shrubs	-	-	-	-	-	-	-	-	-	-	-
	Agriculture	2	28.4	21.9	62.3	15.8	2.4	1.4	0.2	7.4	3.8	1883.3
	Sparsely vegetated	-	-	-	-	-	-	-	-	-	-	-

Comparing deposition and erosion sites, SOC and TN showed significant differences, with lower values in the eroded sites. Otherwise, soil properties were similar and no significant differences were observed (Figure 7.7, Table 7.2). On the other hand, comparing erosion and stable sites, significant differences were observed for soil properties, SOC, SIC and TN.

The spatial distribution of ^{137}Cs, SOC and TN, showed high spatial variability in the Araguás catchment with similar patterns for the three variables (Figure 7.8). Areas of ^{137}Cs gain generally coincide with areas of high SOC accumulation pointing to a recuperation of soil layers particularly in the afforested part of the catchment (Figure 7.8).

The sparsely vegetated areas recorded the highest soil erosion rates (121.9 Mg ha^{-1} yr^{-1}) while shrublands recorded the highest deposition rates (62.7 Mg ha^{-1} yr^{-1}). Mean soil erosion and deposition values were 68.6 Mg ha^{-1} yr^{-1} and 17.7 Mg ha^{-1} yr^{-1}, respectively. Most of the sparsely vegetated and pastureland sites experienced soil erosion with a mean value of 68 Mg ha^{-1} yr^{-1} (ranging between -121.9 and 4.4 Mg ha^{-1} yr^{-1}) and 44 Mg ha^{-1} yr^{-1} (ranging between -110.0 and 8.4 Mg ha^{-1} yr^{-1}) respectively. Soil deposition was recorded in 45% of shrublands and afforested sites, with a mean value of 33 Mg ha^{-1} yr^{-1} in shrubland and 3.6 Mg ha^{-1} yr^{-1} in afforested sites. Soil erosion was recorded in 75% and 47% of the sampling points of shrublands and afforested sites with mean values of 64.2 Mg ha^{-1} yr^{-1} and 58.5 Mg ha^{-1} yr^{-1} respectively. However, no significant differences in soil redistribution rates were found between the different LULC.

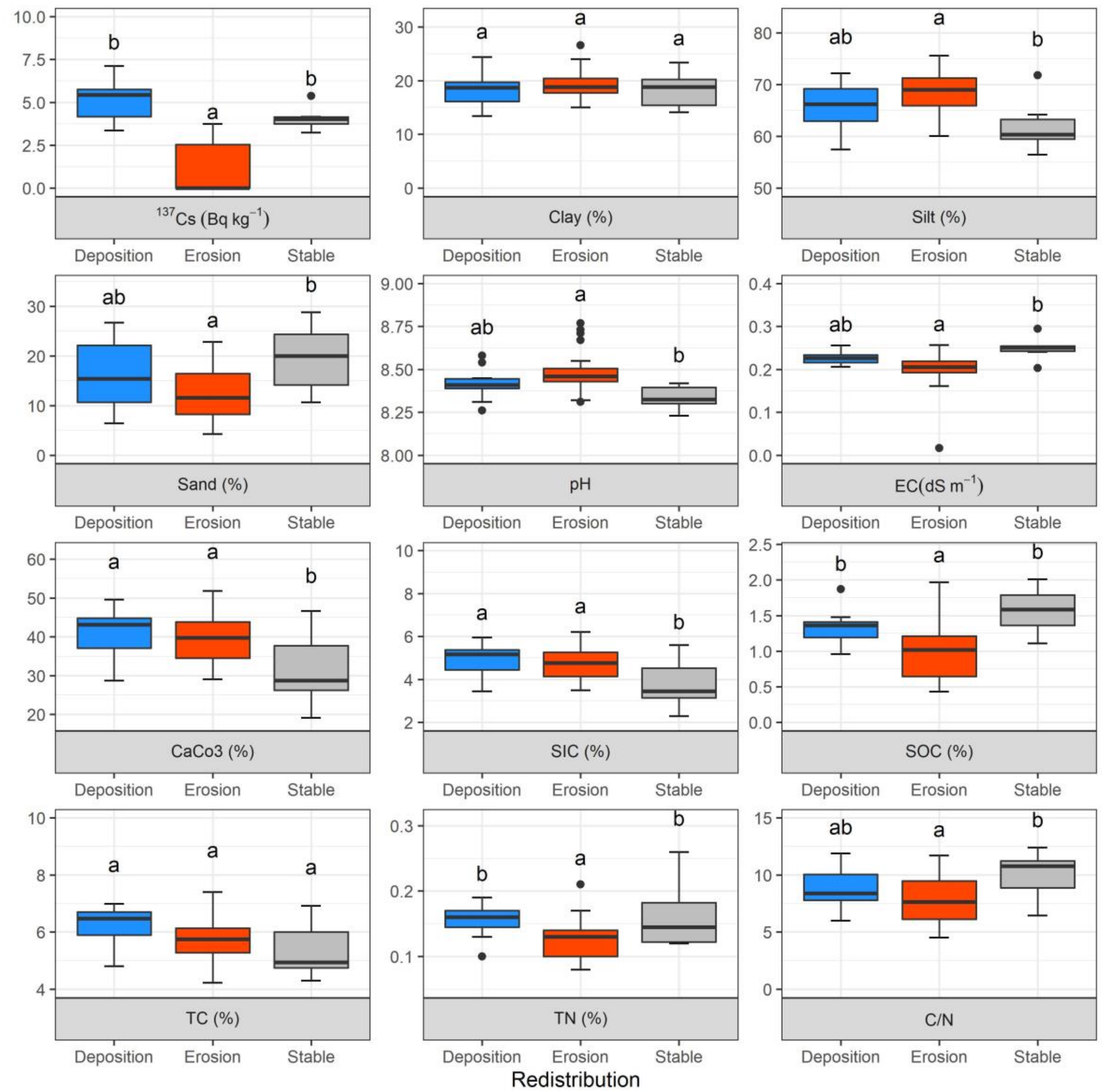

Figure 7.7. Box plot of erosion, deposition, and stable zones with soil properties values. Note: EC, electrical conductivity; $CaCO_3$, carbonate content; SIC, soil inorganic carbon; SOC, soil organic carbon; TC, total carbon; TN, total nitrogen.

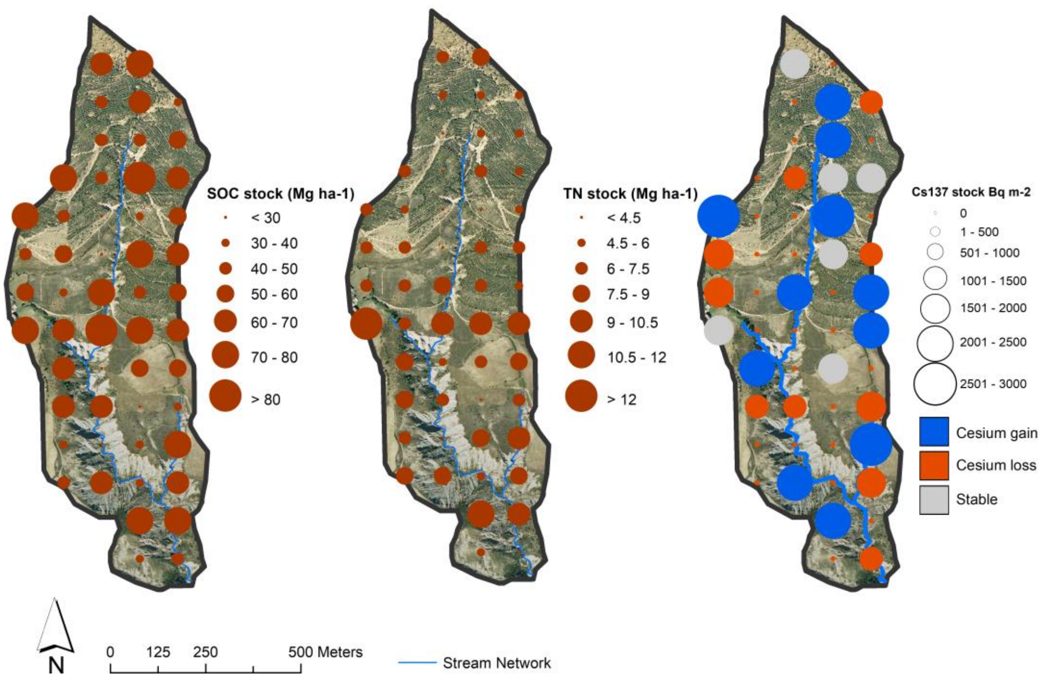

Figure 7.8. Spatial distribution of ^{137}Cs, SOC and SON inventories, showing positive (gain) and negative (loss) of SOC, SON and ^{137}Cs.

4. Discussion

The legacy of the historic LULCC, and the distribution LULC across the Araguás catchment is a key factor in the spatial patterns of soil properties, soil nutrients and soil redistribution processes.

The depth distribution of ^{137}Cs in the reference profiles followed the typical pattern (exponential decline) of undisturbed areas in mountain regions (see Walling and Quine, 1995; Navas et al., 2005; Gaspar and Navas, 2013; Quijano et al., 2016). Most of the ^{137}Cs was found in the upper part of the soil (above 15 cm), and the reference inventory falls within the range (1475-2288 Bq m^{-2}) estimated by Legarda et al. (2011) and Caro et al. (2013) for the Iberian Peninsula, although presents lower values than the local reference inventory of around 4500 Bq m^{-2} obtained in the neighbouring Aisa Valley by Navas et al. (2005, 2017) due to its higher annual rainfall and the radionuclide decay. An exponential decrease was also observed with SOC values, confirming that the soil remains undisturbed after ^{137}Cs deposition. It is interesting to note the retention of ^{137}Cs in the first topsoil layers at the reference sites, that is attributed to firm bounds of ^{137}Cs to organic matter, as indicated by its significant correlation with ^{137}Cs mass activity (Kim et al., 2006; Gaspar et al., 2013).

Since soil changes and forest soils can take a long-time to build up, the effects of afforestation after cropland abandonment can be sometimes limited because the nature of the soil has not changed (Maestre and Cortina, 2004; Iroumé and Palacios, 2013). In the Araguás catchment, the interpolation of the physical and chemical soil properties showed a large variation, and significant differences between LULC were only observed under mean contents of SOC, ^{137}Cs and CN ratio. Higher values were recorded in afforested sites compared with shrublands and sparsely vegetated areas, although no differences were observed with pasturelands. These results suggest that LULCC is one of the principal factors affecting the variation of soil properties, as has been found in previous studies (Navas et al., 2008; Lizaga et al., 2019, among others). For instance, Cuesta et al. (2012) indicated that afforestation practices can accelerate the recovery of some soil properties of abandoned croplands in comparison with secondary succession, but these effects are noticeable at long-term scale.

The results of this study show that soils in the Araguás catchment after LULCC are prone to degradation, as indicated by a generalized loss of ^{137}Cs, SOC and TN compared to the inventories of the reference sites: 67%, 87% and 71% of the points were affected by ^{137}Cs, SOC and TN loss respectively. Although the presence of ^{137}Cs is associated with the presence of clay (Forkapic et al., 2017), the depth distribution of clay with enrichment at the deeper layers in our soil profiles besides the relatively low clay content (< 27%), and the limited range of variations in the Araguás catchment (92% of the samples show a clay content between 15 and 25%), may explain the lack of correlation of both variables, also observed in other studies (Gaspar et al., 2021). On the other hand, the highest activity of ^{137}Cs in the afforested area is related with the strong positive relationships between ^{137}Cs and SOC (Figure 7.5), confirming that ^{137}Cs remain strongly fixed to the organic matter (i.e., Gaspar et al., 2021). Most of the points with SOC accumulation stocks were recorded in the afforested sites, as a consequence of constant SOC accumulation in the organic and mineral horizons due to litter deposition and stabilization processes. On the other hand, most of the TN stock accumulation was recorded in the shrubland and pastureland areas mainly associated with the higher presence of leguminous vegetation with high N fixing capacity.

The ^{137}Cs results in the Araguás catchment suggest that erosion and sedimentation processes were active in the last 50-60 years showing erosion but also deposition. Soil loss was associated with all LULC (mean value 69 Mg ha^{-1} yr^{-1}), with more critical values in badland areas (maximum value of 122 Mg ha^{-1} yr^{-1}). Furthermore, the lowest values of ^{137}Cs and SOC were found under the sparsely vegetated sites in the badland areas. These results can be related to the intense geomorphological dynamics in these areas, that besides physico-chemical weathering is mainly conditioned by erosion processes and the high hydrological and sedimentological dynamics recorded in the catchment (Nadal-Romero and Regüés, 2010). Scientific literature shows that erosion rates in badlands are greater than in other landforms and LULC (Nadal-Romero and García-Ruiz, 2018): in humid Mediterranean badlands high sediment yield, exceeding 500 Mg ha^{-1} were recorded (Brochot, 1993), and most of the studies concluded that annual sediment yield resulted mainly from a small number of maximum events (Regüés et al., 2000; Mathys et al., 2005). In the dryer conditions of the Moroccan Rif the highest rates of soil loss were also recorded with ^{137}Cs in badlands in comparison to other land uses (Sadiki et al., 2007). These results, confirm the values obtained through the continuous monitoring of the Araguás catchment with an average sediment yield of 153 Mg ha^{-1} yr^{-1} over the entire study area, or 575 Mg ha^{-1} yr^{-1} from badland areas.

The long history of human activity through agricultural practices and active restoration practices (afforestation) have determined the intense degradation values recorded in the Araguás catchment. Soil erosion during the cultivation and after land abandonment was severe in steep slopes in mountain areas (Navas et al., 2017). In these areas, a period of intense erosion is usually observed after cropland abandonment (García-Ruiz and Lana-Renault, 2011): (i) lands were abandoned after the harvest, with high percentage of bare soil, coinciding with intense rainfall events enhancing crust development, high runoff values and soil erosion processes (Navas et al., 2008; García-Ruiz et al., 1995); (ii) terraces and stone walls were abandoned and no longer maintained (Moreno de las Heras et al., 2019); (iii) most of the practices used in afforestation increase soil erosion (Nunes et al., 2011); (iv) poor (low soil organic matter level after centuries on human activities) and bare soils in the abandoned areas, and the slow growth of the coniferous plantation limited the re-growth of grassland and shrubs that could protect the soils during the first years (de Wit and Brouwer, 1997).

Afforestation in the Araguás catchment was carried out in the highest and steepest areas, that in addition were the less accessible and consequently were the first abandoned. The early abandonment of the area and the afforestation at the end of the 1960’s likely contributed to changes in soil properties and intense soil redistribution processes.

Extensive afforestation was carried out after cropland abandonment in an attempt to control land degradation and soil erosion processes (Sorriso-Valvo et al., 1995; Boix-Fayos et al., 2007; Nunes et al., 2011). However, soil operations for afforestation planting have a direct effect on soil loss and can fully truncate the soil profile as well as on soil properties. Such effects occur even during the first years of tree development because the soil surface remains unprotected, accelerating erosion processes (Chaparro and Esteve, 1995; de Wit and Brouwer, 1997; Romero-Díaz et al., 2010). Thus, although the lowest values were recorded in the afforested sites, no significant differences in soil redistribution rates were recorded between shrubs and afforested areas, contrarily to other results recorded in Mediterranean mountain areas (Navas et al., 2005, 2011). We hypothesized that high soil erosion processes occurred in the afforested area, previous to restoration due to centuries of cultivation and also due to afforested practices. In that sense, literature confirm this hypothesis: De Wit and Brouwer (1997) observed an increase in soil erosion processes at least during the 14 years after planting; Chaparro and Esteve (1995) observed an increase in the geomorphic activity even 20 years after planting using terracing and heavy machinery; and Nunes et al. (2011) showed that erosion in afforested plots was very high in comparison with other LULC.

In the Araguás catchment, afforestation was installed in low fertile soils, and during the first years after restoration soil erosion was high. However, certain changes in soil properties, such as the increase in soil organic carbon and soil organic matters values and soil aggregation (Nadal-Romero et al., 2009, 2016c) suggest a decrease in soil erosion processes in the afforested area in the last decades. Important changes have been observed related to organic matter quality and aggregation: higher aggregation and high carbon accumulation in macroaggregates (> 5mm) (Nadal-Romero et al., 2016c) and higher lignin content (Campo et al., 2019). Similar results were described in Boix-Fayos et al. (2007) and LaManna et al. (2021). In the Ijuez river (Central Spanish Pyrenees), farmland abandonment and afforestation resulted in a rapid recovery of vegetation cover, which produce a significant decrease in the area affected by erosion, declining sediment yield and also modifying the connectivity between the hillslope and channel (Gómez-Villar et al., 2014; Sanjuán et al., 2016).

Radiotracers approaches offer a considerable potential to study soil erosion processes and quantify soil redistribution rates (i.e. Navas et al., 2017). The radioactive fallout of Caesium-137 has been successfully applied in Mediterranean mountain environments for documenting not only soil erosion but also the mobilisation of nutrients associated to soil particles (Navas et al., 2012: Gaspar et al; 2021), which is fundamental to understand changes of organic matter quality (Nadal-Romero et al., 2016c). As it has been shown in this manuscript, this technique provides medium-term spatially distributed soil redistribution rates, representing annual values for the last 55-60 years, that has been an intense period of LULCC in mountain areas after cropland abandonment. Future studies using soil erosion modeling (i.e. WATEM/SEDEM) and ^{137}Cs information for the validation of distributed soil erosion (i.e. Alatorre et al., 2012; Quijano et al., 2016) will allow simulating soil redistribution under past, current and hypothetical future LULC scenarios, based on revegetation processes conditions in Mediterranean mid-mountain areas. These expected results would support the development of soil conservation strategies which may help to mitigate soil degradation processes after cropland abandonment in a context of Global Change.

5. Conclusions

This study provides novel information on the effects of land use and land cover changes in Mediterranean mid-mountain areas on physico-chemical soil properties, soil nutrients and soil redistribution processes.

The following conclusions can be made:

(i) Land use and land cover changes during the last decades after cropland abandonment affect the physical and chemical soil properties and soil redistribution processes.

(ii) LULC are key factor in controlling the depth distribution of the radioisotope.

(iii) This study also demonstrated the potential of ^{137}Cs analysis to quantify and spatialize information on soil redistribution rates due to LULCC during the last decades.

(iv) LULCC and active restoration through afforestation have significant impacts on physico-chemical soil properties and soil nutrients. Contrary to our hypothesis no differences were observed between the different LULCC in soil redistribution rates due to a long history of human intervention through cultivation in steep slopes and afforestation practices. However, the recovery of the soil organic carbon and nitrogen stocks in afforested areas suggest that afforestation can reduce soil degradation after cropland abandonment at long-term scale.

(v) The highest erosion rates were recorded under sparsely vegetated sites in the badland area, while the lowest rates were found in the afforested area. However, the observed differences were not significant, suggesting that soil redistribution in the catchment is not attributed only to LULCC but also to the land use legacy of the catchment and the techniques used during the active restoration.

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Chapter 8. Discussions and general synthesis

This Chapter presents a coherent global synthesis of the main findings from this PhD dissertation. The main research results from the five previous studies are presented and discussed, highlighting the main used research approaches and a general synthesis on the management of abandoned croplands is pointed out.

1. Discussions

The present PhD dissertation addresses land cover changes following cropland abandonment in the Central Spanish Pyrenees and its effects on ecosystem processes. The implications of these land cover transformations as well as climate change on water resources, vegetation dynamics and soil properties and redistribution are studied in three study areas representative of Mediterranean mountains. The studied land cover changes represent the main restoration trajectories in abandoned cropland areas in the Central Spanish Pyrenees. These trajectories combine natural revegetation (passive restoration) and active restoration practices, through shrub clearing and afforestation programs. Changes to water and carbon fluxes and the interaction between hydrological and vegetation dynamics are studied at different spatial and temporal scales. Further, the repercussions of vegetation change on soil properties and soil redistribution rates are estimated.

1.1. Proposed model implementation

Our approach relied on the application of the Regional Hydro-Ecological Simulation System (RHESSys) which showed good suitability to model land cover and climate change at local to regional spatial scales around the world (Martin et al., 2017; Saksa et al., 2020; Zierl et al., 2007) including the Pyrenees (López-Moreno et al., 2014; Morán-Tejeda et al., 2015; Zabalza-Martínez et al., 2018). RHESSys reproduced well the observed streamflow data from the Arnás catchment, during the calibration process, and showed good sensitivity to changes in land cover and climate in the Central Spanish Pyrenees.

This PhD dissertation presents the first application of an ecohydrological model to simulate active management practices (shrub clearing and afforestation) after cropland abandonment in the Mediterranean mountains. Previous studies were carried out comparing different catchments or different plots within the same catchment (García-Ruiz et al., 2010; Nadal-Romero et al., 2013, 2018, 2019; Lopéz-Vicente et al., 2017; Lana-Renault et al., 2018). The advantage of the used modeling approach is studying different land management trajectories under the same catchment conditions. This implementation permitted to detect the net effect of vegetation and climate change and offered a good opportunity to test different management and climate trajectories in well monitored study areas.

The modeling approach permitted an upscaling from local to regional spatial scale using parameters from the calibration of RHESSys in the Arnás catchment following Tague et al. (2013). Model calibration and validation usually rely on available hydrological data, however at regional spatial scale this information is often unavailable due to the complexity of monitoring large watersheds particularly in complex mountainous terrain. Our model based approach is readily extendable to other regions in the Central Spanish Pyrenees and could use the same calibration parameters in sites with similar geology, vegetation and climate conditions.

The present PhD dissertation used RHESSys for the first time to model the secondary succession process in abandoned croplands. Contrarily to other approaches used to model the natural revegetation process following land abandonment in Mediterranean mountains (e.g. Markov chains, CLUE-S model, etc.) (i.e. Pueyo and Begueria, 2007), RHESSys was not developed to explicitly model ecological succession. However, the proposed approach permitted an estimation of the rate of succession between grass, shrub and forest stages and spatially distributed the changes over the study area. The obtained results from this model implementation provided important insights to understand the effects of the delay in managing abandoned croplands and evaluate the efficiency of the shrub clearing practice for projected secondary succession scenarios.

The selected study areas are good examples of the land cover legacy in the Spanish Pyrenees and are similar to many of the upslope regions of Mediterranean mountains, that are both locations of land abandonment and contribute to downslope water supply (Lasanta-Martínez et al., 2005; Lasanta and Vicente-Serrano, 2007). On the other hand, the studied scenarios relied on realistic trajectories of land cover and climate change to quantify changes to ecohydrological dynamics. These trajectories were based on adopted practices from the Spanish regional administrations, to manage abandoned cropland areas and from projected climate change scenarios for the region. We caution that the climate change scenarios, in this PhD dissertation, do not consider changes in precipitation, given the strong spatial and seasonal variability of these over the Pyrenees (OPCC, 2018) and the high uncertainty toward future projections in the Mediterranean region (IPCC, 2021). The representativeness of the study areas and the realistic land cover and climate trajectories enhance the generalization of the obtained results to Mediterranean mountain environments. Results could be of great importance in elaborating sustainable management plans for the abandoned cropland areas in Mediterranean mountains.

1.2. Changes to ecohydrological dynamics

Our modeling results showed that natural revegetation significantly reduces streamflow and increases evapotranspiration. The decrease in streamflow reached 19% within 30 years of vegetation succession in the Estarrún watershed. While the estimated decadal change in streamflow from the Arnás catchment ranged around 6%·decade^{-1} over the first 30 years after abandonment. These estimates are consistent with previous studies in the Central Spanish Pyrenees attributing the decline in runoff to the increase in the vegetation cover (Beguería et al., 2003; García-Ruiz et al., 2011; López-Moreno et al., 2014; Morán-Tejeda et al., 2014, 2010). The spread of tree cover increases the interception of rainwater and the transpiration from the denser vegetation cover, all of which contribute to higher evapotranspiration leading to declines in water yield (Brown et al., 2005; Malmer et al., 2010). In the same line, evapotranspiration increased by 34% within 30 years of natural revegetation in the Estarrún watershed, while in the Arnás catchment transpiration increase reached 3.3%·decade^{-1}.

Changes to water fluxes due to afforestation are higher compared to natural revegetation. The decrease in streamflow reached up to 53%, while transpiration increased up to 23% compared to natural revegetation in the Arnás catchment, 50 years after management. These changes are related to the morphological attributes of the vegetation. The higher Leaf Area Index (LAI), the denser vegetation biomass and the deeper rooting system of forest compared to shrubs are reflected in their higher water consumption, a behavior that is widely recognizable in hydrological studies (Bosch and Hewlett, 1982; Peel, 2009; Sahin and Hall, 1996) and consisting with our modeling results.

On the other hand, density reduction due to forest management or natural disturbance (fire, mortality, etc.), are generally related to increases in water yield and decreases in evapotranspiration (Callegari et al., 2003; Lasanta et al., 2015; Serengil et al., 2007; Tague et al., 2019). The shrub clearing strategy as it affects vegetation density resulted in changes in water yield in Arnás and Estarrún. Due to shrub clearing, annual streamflow increased up to 24% and evapotranspiration decreased to almost 8% in 10 years in the Arnás catchment, while in the Estarrún watershed, water yield and evapotranspiration increased and decreased respectively by up to 6% in the 5 years following shrub clearing. These results are overall consistent with other reported estimates of post management effects on annual water yield and evapotranspiration (Dore et al., 2012; Dung et al., 2012;

Koch et al., 2010; Lane and Mackay, 2001). Similar to most biomass removal practices, shrub clearing reduce canopy interception making more rainwater reach the surface (Mazza et al., 2011) and reduce water loss through evapotranspiration (Sun et al., 2015). The good results of shrub clearing in reducing forest fires (Lasanta et al., 2018) without increasing soil erosion (Nadal-Romero et al., 2018) together with its hydrological benefits makes of this management practice appropriate for abandoned croplands in Mediterranean mountains.

However, the magnitude of the changes varied substantially depending on the managed area and the intensity of intervention. The higher the managed area or intervention intensity, the higher the changes to streamflow and evapotranspiration. For example, in the Arnás catchment, the highest shrub clearing intensities generated the highest annual streamflow and the lowest evapotranspiration rates compared to natural revegetation, while the hydrological changes related to the intense afforestation scenario bypassed those of the low afforestation scenario. This is generally accepted in scientific literature, for instance, Bosch and Hewlett (1982) and Tague et al. (2019) associated the magnitude of the post management hydrological response to the harvested area and the thinning intensity and Yurtseven et al. (2018) qualified as “evident” the lower runoff response to lower timber harvest.

The timing of highest changes to hydrological fluxes is as important as its magnitude. Knowledge on the timing of an expected change could enhance the management of water resources and infrastructure (e.g. dams, surface irrigation, etc.). Our model based approach, revealed important insights on the timing of post management hydrological changes. At short term, the highest changes to water yield due to shrub clearing were found during the first three years after intervention. However, at long term, changes linked to afforestation were only significant during the first 30 years after plantation. This timing is highly linked to vegetation age and physiological attributes as was previously reported in other studies (Scott and Prinsloo, 2008; van Dijk and Keenan, 2007).

Delaying the management of abandoned cropland areas is also a key issue that could have important repercussions on post management response. The total manageable area and the efficiency and/or feasibility of management practices are among the most problematic aspect of this delay. For instance, the target area for shrub clearing in the Estarrún watershed could decline to the half and changes to streamflow by 66% within 30 years of vegetation succession. Declines in the total available area for shrub clearing and its hydrological impacts would limit its efficiency as a low cost management practice in abandoned cropland areas. Therefore, land managers would have to adopt more complex and expensive management strategies in these areas (e.g. forest thinning).

This PhD dissertation also treated post management vegetation response, as changes to water dynamics are in part linked to changes in vegetation water use. The study of the interactions between water and vegetation dynamics stimulated the interest to investigate the tradeoffs between carbon sequestration and water yield in afforestation plans to control soil erosion in Mediterranean mountains. The analysis of the Net Ecosystem Production (NEP) following a low and an intense afforestation plan compared to natural revegetation in the Arnás catchment permitted to quantify these tradeoffs. Afforestation significantly increased annual carbon sequestration in the catchment (up to 112 g $C \cdot m^{-2} \cdot yr^{-1}$) compared to natural revegetation (-27 g $C \cdot m^{-2} \cdot yr^{-1}$). This carbon sequestration rate is within ranges of other studies in the Mediterranean region (Valentini et al., 2000). The role of trees as carbon sink in global carbon analysis is well recognized (Harris et al., 2021; Pan et al., 2011). Afforestation programs may accelerate carbon uptake by living-biomass (Nave et al., 2019) and enhance its accumulation in soil layers (Nave et al., 2018). The first 10 years of management coincide with the highest levels of carbon sequestration by the young tree stands, while after 10 years, the carbon sequestration decreases substantially and stabilize. This pattern is related to a higher photosynthetic activity during the first years after plantation enhanced by vegetation growth and a lower growth rate with mature vegetation (the photosynthetic activity is more restricted to the maintenance of the living biomass).

However, the highest carbon sequestration rates under afforestation scenarios were associated with the lowest water use efficiency (1.4 g $C \cdot mm^{-1}$) compared to natural revegetation. Mature trees, as well as their high capacity for carbon sequestration through the photosynthesis process, generate important water loss through evapotranspiration. Moisture conditions is one of the main drivers of ecosystem water use efficiency (Liu et al., 2017). The deeper and developed rooting system of trees compared to shrubs enhances their access to water, contributing to a higher photosynthetic activity and evapotranspiration. The tradeoffs between carbon sequestration and water supply are, therefore, an important factor to consider in the efforts to mitigate climate change and soil erosion particularly in areas of limited water availability as the Mediterranean region.

Climate variability and change plays a key role in the interactions between vegetation and water resources. Although the warming scenarios did not show significant differences to historic climate scenarios as might be expected given high vapor pressure and a potentially longer growing season, the increase in temperature clearly affected water fluxes and vegetation response to management plans. In our model based approach, the increase in temperature did not affect the duration of the growing season while increased evapotranspiration and soil saturation deficit, all of which contributing to lower water yield. However, in all cases, changes related to climate were overcome by those attributed to vegetation change resulting from the adopted management strategy. Further, precipitation was determinant of the magnitude of the changes to water and vegetation dynamics. For example, our results suggest that annual precipitation could limit annual NEP in dry environments, while it could enhance the effects of shrub clearing in wetter environments than the Central Spanish Pyrenees.

1.3. Changes to soil properties and redistribution rates

As well as its effects on water regimes, vegetation change and restoration practices also affect the soil. For this reason, this PhD dissertation also addresses changes in soil properties, nutrients and redistribution related to LULCC following cropland abandonment.

Different methodologies have been used in the literature to estimate soil erosion rates and land degradation (García-Ruiz et al., 2015). The use of radioisotope methods is a newly developing tool in soil erosion studies (Navas et al., 2005): radioisotope surveys can provide true erosion rates, including intermediate deposition areas. Different authors have highlighted positive and negative aspects related to this method: for example, Boardman (2006) suggested that the assumptions on which it is based are debatable being a time-consuming method; Parsons and Foster (2011) argued that the loss of ^{137}Cs is not necessarily proportional to the real loss of soil. Others indicated the difficulty to establish a reference site, especially in Mediterranean mountain areas where human impact has been intense (García-Ruiz et al., 2015). In this case, fallout ^{137}Cs was used to study soil redistribution in the Aragúas catchment. Since its first application in Mediterranean landscapes (Navas and Walling, 1992), the grid measurement of ^{137}Cs fallout has shown a good suitability to study the spatial patterns of soil redistribution and that of associated nutrients in catchments with dynamic LULCC after land abandonment (Navas et al., 2011; Gaspar et al., 2021), and many studies have reaffirmed the usefulness and accuracy of this method for soil erosion studies (i.e., Mabit et al., 2013).

First, a reference ^{137}Cs inventory was established in undisturbed reference sites in the catchment and permitted to classify the sampling sites in stable, erosion or deposition sites. The estimated reference inventory falls within ranges of previous studies in the Iberian Peninsula (Legarda et al., 2011; Caro et al., 2013), which confirms the suitability of fallout ^{137}Cs for the study of the spatial redistribution of soil and nutrients in the Araguás catchment. The LULCC related to land abandonment and active restoration, through afforestation, significantly affected soil nutrients and its physical and chemical properties. Afforestation showed the highest inventories of ^{137}Cs and SOC indicating a higher recuperation of soils compared to shrubs, agriculture and sparsely vegetated areas. This is consistent with Cuesta et al. (2012) pointing out that afforestation can accelerate the recovery of soils in abandoned croplands compared to natural revegetation.

On the other hand, the analysis of soil redistribution in the Araguás catchment showed a generalized loss of ^{137}Cs pointing to an important degradation process during the last decades. The highest values of erosion rates were found in the sparsely vegetated areas, however the differences between LULC types were non-significant, suggesting that the generalized soil erosion was not only attributed to LULCC but also to the legacy of the historic human activity in the catchment (agricultural activities, afforestation practices, etc.).

2. General synthesis

The management of abandoned cropland areas in Mediterranean mountains is a complex issue. The complexity of this management stems from the interactions between ecosystem components. Changes to the vegetation cover affect the soil and water resources, therefore the adopted management plans should take into account these interactions to ensure the desired benefits without affecting the sustainability of ecosystem services. To face the complexity of management, efforts should be united between policy makers, scientists and stakeholders to elaborate appropriate land management plans. Independently of the management practices implemented (passive or active restoration), the adopted strategy would have effects on water resources, vegetation and soils. The desired effects or the "gains" from these strategies are mostly evident in controlling forest fires, soil erosion or enhancing water yield. However, their secondary or undesired effects on water resources, vegetation and soils are the ones that need to be adequately understood and taken into account for a sustainable policy making in these areas. A policy making that would be conditioned by several challenging issues during the next few decades. Challenges related to climate change in one of the most sensitive regions to its impacts, to the socioeconomic particularities of the population and to the uncertainty toward future projections (climatic, economic, demographic, etc.) among others. All of which suggest that the efficiency of land management plans should be evaluated not only at short term but also at long term. Therefore, a continuous evaluation of the adopted strategies should be implemented to guarantee the desired efficient management.

On the other hand, the complexity of the management should not hide the great potential of the abandoned croplands in mitigating climate change and generating socioeconomic benefits. A potential that should be also taken into consideration by policy makers given that the process of land abandonment is expected to continue during the next decades and may extend to other regions due to socioeconomic and climatic circumstances.

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Chapter 9. General conclusions and future work

This Chapter outlines the main conclusions from this PhD dissertation. A short presentation of future research work is also presented to address limitations of the present PhD dissertation and enhance the understanding of the post abandonment changes to water, vegetation and soil resources.

1. General conclusions

This PhD dissertation addresses the main effects of the land cover changes related to the abandonment of croplands and its subsequent restoration in the Central Spanish Pyrenees. The conducted studies addressed effects on the main ecosystem components: water resources, vegetation, and soil. Although results particularly focused on changes to water resources, given the importance of these in Mediterranean environments, changes to vegetation and soil were also studied. The main conclusions can be drawn as follows:

1. RHESSys is a suitable tool for modeling the effects of land cover changes linked to the abandonment of croplands in the Mediterranean mountains. The model showed good sensitivity to changes in the vegetation and climate and was successfully implemented to model vegetation succession in the Estarrún watershed and spatially distribute the changes over the study area

2. Natural revegetation (passive restoration) affects water resources in the Central Spanish Pyrenees by reducing streamflow and increasing evapotranspiration. The process is projected to continue in the future given the continuity of the process of land abandonment in the Mediterranean region.

3. The delay in managing abandoned croplands in the Central Spanish Pyrenees could have important repercussions on water supply for lowland areas. Delaying the management would boost the vegetation succession process to reach tree stage and consequently limit the efficiency of the shrub clearing as a low-cost restoration practice.

4. Shrub clearing (active restoration) could be an effective low-cost management practice to reduce the effects of natural revegetation on water resources. Shrub clearing increases streamflow and reduces evapotranspiration, however its effects are limited by the clearing intensity, the total cleared area, and climate variability. The highest changes to water fluxes occur during the first three years after shrub clearing and subsequently decrease with vegetation recovery.

5. Afforestation plans (active restoration) affect water resources by reducing streamflow and increasing evapotranspiration. The changes are significant during the first 30 years after management and increase with the total afforested area. These changes are related to vegetation maturity and physiological attributes. The higher LAI, the more developed biomass, and the deeper roots of trees boost its water consumption compared to shrubs.

6. Afforestation boosts ecosystem carbon uptake and its accumulation in soil layers, however it is associated with the lowest water yield and water use efficiency compared to natural revegetation. A sustainable management strategy for the abandoned croplands in the Central Spanish Pyrenees should maximize carbon sequestration without affecting the water supply.

7. Changes related to the warming climate were overcome by those linked to vegetation change, however, they clearly reduced streamflow, increased evapotranspiration, and conditioned the magnitude of the ecohydrological changes.

8. Annual precipitation is a key factor in the effects of vegetation change on water resources and carbon sequestration. In wetter climates, changes to water fluxes related to shrub clearing could be higher. While in water-limited environments, carbon sequestration from afforestation plans could be limited.

9. LULCC resulting from land abandonment and active restoration plans (afforestation) affected soil properties, nutrients and, redistribution. However, soil erosion processes are not only attributed to these changes but also to the long history of human activity and the techniques used during the afforestation of the catchment.

10. Active restoration through afforestation increased SOC and enhanced soil recuperation compared to passive restoration through natural revegetation, however, tree plantation could have been associated with important mass movements that eliminated all the deposited ^{137}Cs.

2. Conclusiones generales

Esta tesis doctoral aborda los principales efectos de los cambios en la cubierta del suelo relacionados con el abandono de las tierras de cultivo y su posterior restauración en el Pirineo Central español. Los estudios realizados abordaron los efectos sobre los principales componentes del ecosistema: recursos hídricos, vegetación y suelo. Aunque los resultados se centraron especialmente en los cambios en los recursos hídricos, dada la importancia de estos en los entornos mediterráneos, los cambios en la vegetación y el suelo también fueron estudiados. Las principales conclusiones son:

1. RHESSys es una herramienta adecuada para modelar los efectos de los cambios en la cubierta del suelo relacionados con el abandono de las tierras de cultivo en las montañas mediterráneas. El modelo mostró buena sensibilidad a los cambios en la vegetación y el clima y se implementó con éxito para modelizar la sucesión de la vegetación en la cuenca del Estarrún y distribuir espacialmente los cambios en el área de estudio.

2. La revegetación natural (restauración pasiva) afecta a los recursos hídricos del Pirineo Central español al reducir los caudales y aumentar la evapotranspiración. Se prevé que el proceso continúe en el futuro dada la continuidad del proceso de abandono de tierras en la región mediterránea.

3. El retraso en la gestión de las tierras de cultivo abandonadas en los Pirineos centrales españoles podría tener importantes repercusiones en el suministro de agua para las zonas bajas. Retrasar el manejo impulsaría el proceso de sucesión vegetal para alcanzar la fase de bosque y, en consecuencia, limitaría la eficiencia de los desbroces de matorral como práctica de restauración de bajo coste.

4. Los desbroces de matorral (restauración activa) podrían ser una práctica de gestión eficaz y de bajo coste para reducir los efectos de la revegetación natural en los recursos hídricos. Los desbroces aumentan los caudales y reducen la evapotranspiración; sin embargo, sus efectos están limitados por la intensidad de desbroce, el área total desbrozada y la variabilidad climática. Los mayores cambios en los flujos de agua ocurren durante los primeros tres años después de la intervención y posteriormente disminuyen con la recuperación de la vegetación.

5. Las repoblaciones forestales (restauración activa) afectan a los recursos hídricos al reducir los caudales y aumentar la evapotranspiración. Los cambios son significativos

durante los primeros 30 años después de la gestión y aumentan con el área total forestada. Estos cambios están relacionados con la madurez de la vegetación y sus atributos fisiológicos. El mayor LAI, la más desarrollada biomasa y las raíces más profundas de los árboles impulsan su consumo de agua en comparación con los matorrales.

6. La repoblación forestal aumenta el secuestro de carbono del ecosistema y su acumulación en las capas del suelo; sin embargo, la repoblación forestal disminuye los recursos hídricos y la eficiencia en uso del agua en comparación con la revegetación natural. Una estrategia de gestión sostenible para las tierras de cultivo abandonadas en los Pirineos centrales españoles debería maximizar el secuestro de carbono sin afectar el suministro de agua.

7. Los cambios relacionados con el calentamiento climático fueron menores que los vinculados con el cambio de vegetación. No obstante, claramente redujeron el caudal, aumentaron la evapotranspiración y condicionaron la magnitud de los cambios ecohidrológicos.

8. La precipitación anual es un factor clave en los efectos del cambio de vegetación sobre los recursos hídricos y el secuestro de carbono. En climas más húmedos, los cambios en los flujos de agua relacionados con los desbroces de matorral podrían ser mayores. En entornos con escasez de agua, el secuestro de carbono relacionado con los planes de repoblación forestal podría ser limitado.

9. La LULCC resultante del abandono de la tierra y los planes de restauración activa (repoblación forestal) afectaron a las propiedades del suelo, sus nutrientes y su redistribución. Sin embargo, los procesos de erosión del suelo no solo se atribuyen a estos cambios, sino también a la larga historia de la actividad humana y las técnicas utilizadas durante la repoblación forestal de la cuenca.

10. La restauración activa mediante repoblaciones forestales aumentó el SOC y mejoró la recuperación del suelo en comparación con la restauración pasiva mediante la revegetación natural; sin embargo, la plantación de árboles podría haber estado asociada con importantes procesos de remoción del suelo que facilitarón su erosión, lo que llevaría a la eliminación de todo el ^{137}C depositado.

3. Future work

This PhD dissertation provides important insights on the response of water, vegetation and soil, to the different restoration strategies in abandoned cropland areas in the Central Spanish Pyrenees. These results could help develop efficient land management plans for these areas that ensure both ecosystem services and ecosystem sustainability. However, more work is needed to address all literature gaps on the effects of land cover changes, following cropland abandonment in Mediterranean mountains, on provisioning and regulation ecosystem services. The learned research tools and methodologies in this PhD dissertation and the available data record from the monitoring of several experimental catchments in the Central Spanish Pyrenees, could contribute to important research studies in the future to fill these literature gaps. Below are highlighted some of the research studies that could be conducted in continuation to this PhD dissertation.

1. Setting up sediment delivery model WATEM/SEDEM in the Araguás catchment taking advantage from the estimated soil redistribution rates for the calibration process. This study would help evaluate several land management and climate change scenarios and their effects on soil erosion and deposition.

2. Use RHESSys model to define optimum active management plans ensuring water supply and maximizing carbon sequestration. When, where and how much to manage are the main questions this study would try to answer.

3. Use RHESSys to model other adopted forest management practices in areas where the natural revegetation process is more advanced. The Estarrún watershed is a good example for this study given where the vegetation succession process is reaching forest stage and the adoption of shrub clearing would not be possible in the next decades.

4. As it has been shown in this PhD dissertation that the secondary succession process could be very advanced toward forest stage in some abandoned areas in the Central Spanish Pyrenees, the shrub clearing practice would be unadoptable in these areas. In this case forest management practices through, thinning or forest clearing (removing understory shrub cover) could be a good alternative to manage abandoned areas. RHESSys model can be used to model forest management and study its effects on the ecohydrological processes.

5. Use RHESSys to model forest fires in a natural forest in the Central Spanish Pyrenees (the San Salvador catchment). As well as it ecohydrological component RHESSys model includes a fire spread and fire effect component that could help evaluate forest fire risks on ecohydrological dynamics. The fire component could be used to study the effect of management practices on fire spread (shrub clearing, thinning, forest clearing).

6. Include a soil redistribution component to RHESSys model based on knowledge from the use of WATEM/SEDEM model. RHESSys is a process based distributed model that calculates a wide range of ecosystem variables, however it still doesn't estimate soil transport across catchments.

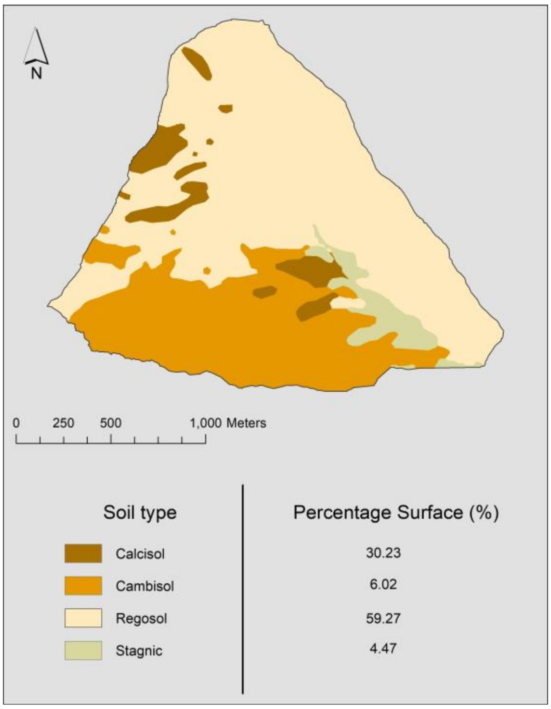
7. Use the recent climate projections from the sixth IPCC report to model the effects of climate change on the ecohydrological dynamics in abandoned cropland areas in Mediterranean mountains. In this PhD dissertation, the effects of temperature rise on the hydrological dynamics were small at local scale compared to those of vegetation change. For this reason, setting RHESSys in a representative regional river basin could enhance the understanding of climate change impacts on water resources. The Aragón river basin could be a good representative study area for this modeling exercise

Appendix

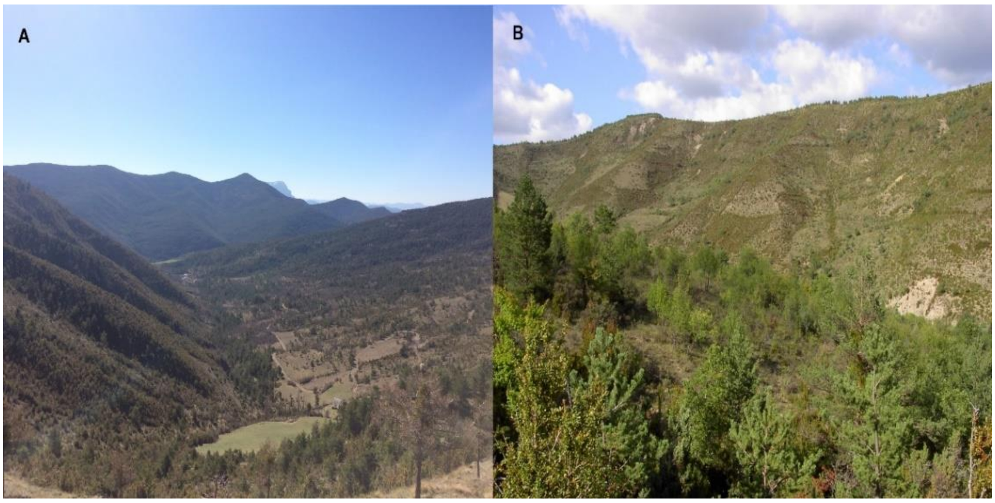
This section presents the supplementary material for the different chapters, the authors contribution in the studies, and information on publication sources (based on the *Journal Citation Reports 2020*). Author contribution to other research studies and national and international conferences are also listed.

1. Supplementary materials

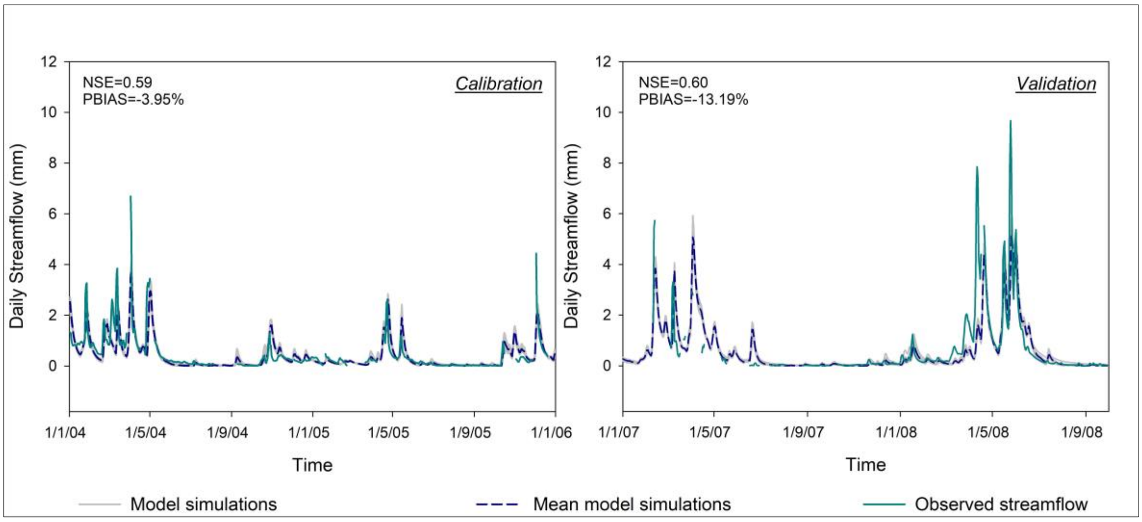
Supplementary materials for Chapter 3



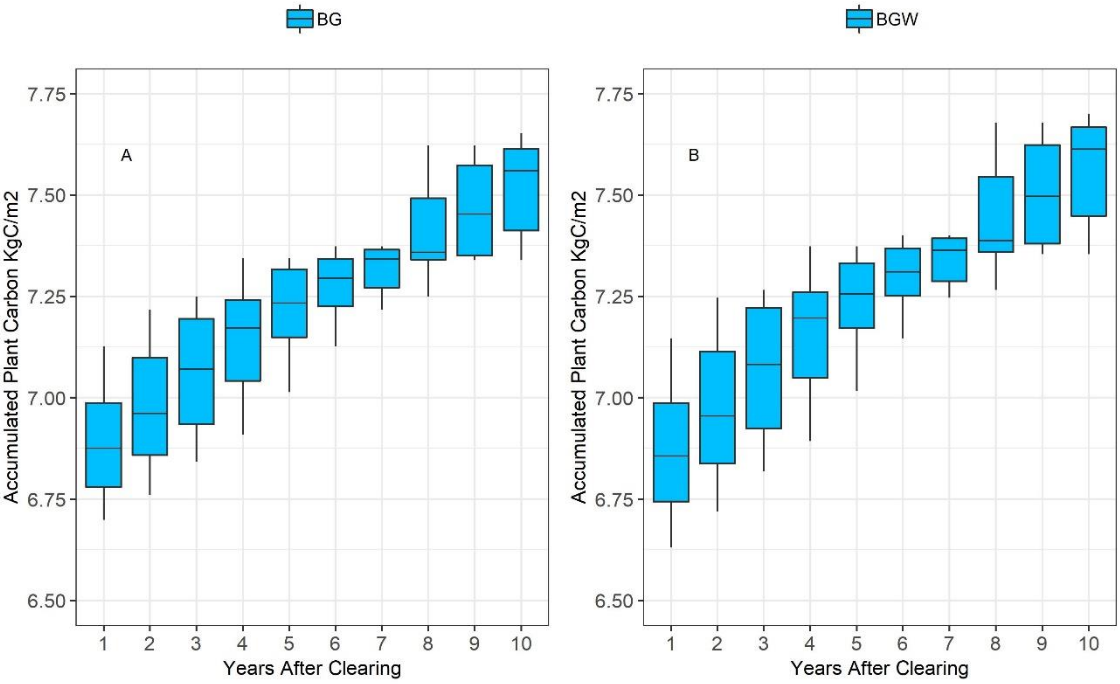
Supplementary Figure 3.1. The reclassified soil map of the Arnás catchment following Seeger et al. (2006) and their correspondent percentage of the study area.



Supplementary Figure 3.2. Photos of the passive management situation of the study area and the natural revegetation process that colonized the abandoned agricultural field. In the bottom of the photo A appear one of the last pasture fields in the study area that could simulate the conditions after shrub clearing. In the photo B we can observe the north-facing slope of the catchment colonized by forest species (mainly pines) however in the south-facing slopes the process is slower.



Supplementary Figure 3.3. Daily modeled streamflow and daily observed streamflow for the calibration (left plot) and validation (right plot) periods. NSE: The Nash-Sutcliffe model efficiency coefficient. PBIAS: Percent Bias.



Supplementary Figure 3.4. Accumulated plant carbon of the baseline scenarios during the 10 years' recovery period. (A: no warming, B: warming). BG: baseline no warming, BGW: baseline warming (Dynamic mode).

Supplementary Table 3.1. The top 9 set of parameters selected in the calibration of RHESSys. The model performance for the validation period is also indicated.

Soil parameters						Calibration statistics		Validation statistics	
m	k	po	pa	gw1	gw2	NSE	PBIAS	NSE	PBIAS
0.11	167.01	0.82	2.98	0.03	0.88	0.56	-0.48	0.61	-9.39
0.06	633.74	0.82	2.58	0.03	0.96	0.56	-0.49	0.59	-9.49
0.09	684.44	0.73	2.99	0.05	0.16	0.6	2.84	0.57	-11.03
0.08	944.07	0.73	2.86	0.03	0.88	0.58	7.49	0.59	-14.66
0.08	961.99	0.74	2.33	0.03	0.56	0.6	8.77	0.58	-18.74
0.07	645.67	0.72	2.74	0.05	0.99	0.58	3.48	0.59	-11
0.10	428.25	0.85	2.30	0.02	0.78	0.56	4.16	0.6	-13.71
0.13	209.83	0.82	2.84	0.03	0.66	0.57	2.7	0.58	-12.28
0.11	136.25	0.81	2.18	0.01	0.96	0.6	7.04	0.59	-18.43

Supplementary materials for Chapter 4

Supplementary Table 4.1. Comparison between the different projections and management scenarios on Streamflow (STR). p value of the TukeyHSD test at 95% of significance level. VS0, VS10, VS20 and VS30: represent vegetation succession projections. SC0, SC10, SC20 and SC30: represent shrub clearing projections.

STR	VS0	VS10	VS20	VS30	SC0	SC10	SC20	SC30
SC30	*$P < 0.05$*	0.97	1	1	*$P < 0.05$*	0.76	1	
SC20	*$P < 0.05$*	0.97	1	1	*$P < 0.05$*	0.76		
SC10	0.48	1	0.57	0.54	*$P < 0.05$*			
SC0	0.93	*$P < 0.05$*	*$P < 0.05$*	*$P < 0.05$*				
VS30	*$P < 0.05$*	0.87	1					
VS20	*$P < 0.05$*	0.89						
VS10	0.17							
VS0								

Supplementary Table 4.2. Comparison between the different projections and management scenarios on Evapotranspiration (ET). p value of the TukeyHSD test at 95% of significance level. VS0, VS10, VS20 and VS30: represent vegetation succession projections. SC0, SC10, SC20 and SC30: represent shrub clearing projections.

ET	VS0	VS10	VS20	VS30	SC0	SC10	SC20	SC30
SC30	*$P < 0.05$*	0.10	1	0.99	*$P < 0.05$*	*$P < 0.05$*	1	
SC20	*$P < 0.05$*	0.12	1	0.99	*$P < 0.05$*	*$P < 0.05$*		
SC10	*$P < 0.05$*	0.82	*$P < 0.05$*	*$P < 0.05$*	*$P < 0.05$*			
SC0	0.13	*$P < 0.05$*	*$P < 0.05$*	*$P < 0.05$*				
VS30	*$P < 0.05$*	*$P < 0.05$*	1					
VS20	*$P < 0.05$*	*$P < 0.05$*						
VS10	*$P < 0.05$*							
VS0								

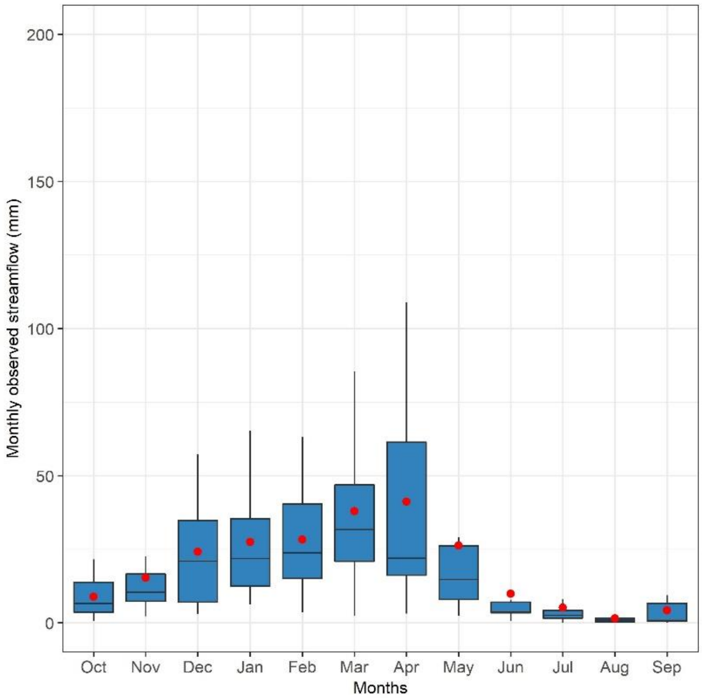
Supplementary materials for Chapter 5

Supplementary Table 5.1. TukeyHD test of the differences in annual streamflow and transpiration between management and climate scenarios. 95% level is used for the significance of the p-value of the TukeyHD test. Scenario 1 (PM and PM.W: Passive management), Scenario 2 (AFF1 and AFF1.W: Active management), Scenario 3 (AFF2 and AFF2.W: Active management). PM.W, AFF1.W and AFF2.W represent warming climate scenarios.

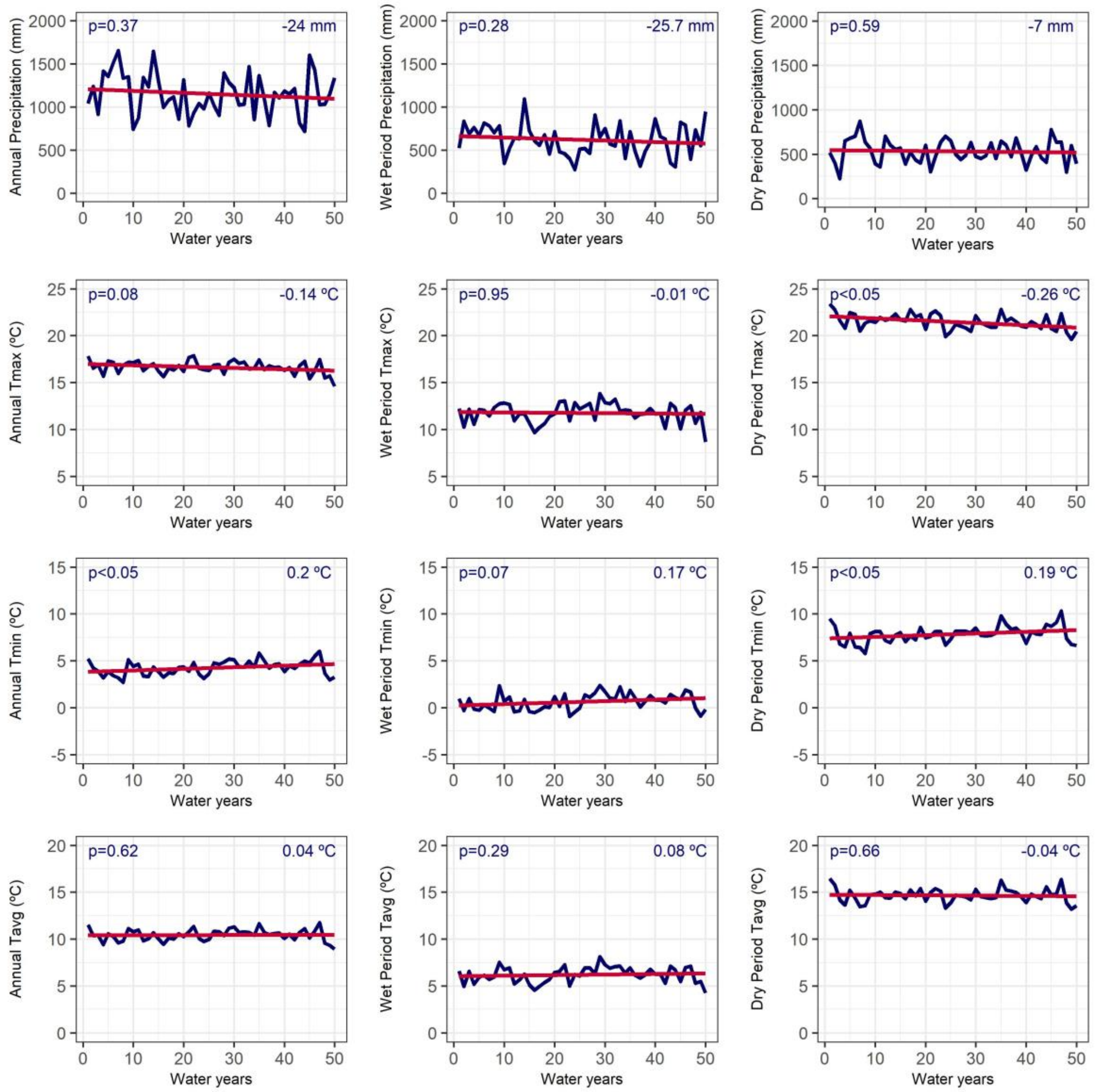
	Comparison	p value	Significance
Annual Streamflow	PM vs. AFF1	$p<0.05$	Yes
	PM vs. AFF2	$p<0.05$	Yes
	AFF1 vs. AFF2	$p<0.05$	Yes
	PM vs. PM.W	P = 0.99	No
	AFF1 vs. AFF1.W	P = 0.99	No
	AFF2 vs. AFF2.W	P = 0.99	No
Annual Transpiration	PM vs. AFF1	$p<0.05$	Yes
	PM vs. AFF2	$p<0.05$	Yes
	AFF1 vs. AFF2	$p<0.05$	Yes
	PM vs. PM.W	P = 0.99	No
	AFF1 vs. AFF1.W	P = 0.99	No
	AFF2 vs. AFF2.W	P = 0.99	No

Supplementary Table 5.2. TukeyHD test of the differences in annual vegetation biomass, Leaf Area Index and rooting depth between management and climate scenarios. 95% level is used for the significance of the p-value of the TukeyHD test. Scenario 1 (PM and PM.W: Passive management), Scenario 2 (AFF1 and AFF1.W: Active management), Scenario 3 (AFF2 and AFF2.W: Active management). PM.W, AFF1.W and AFF2.W represent warming climate scenarios.

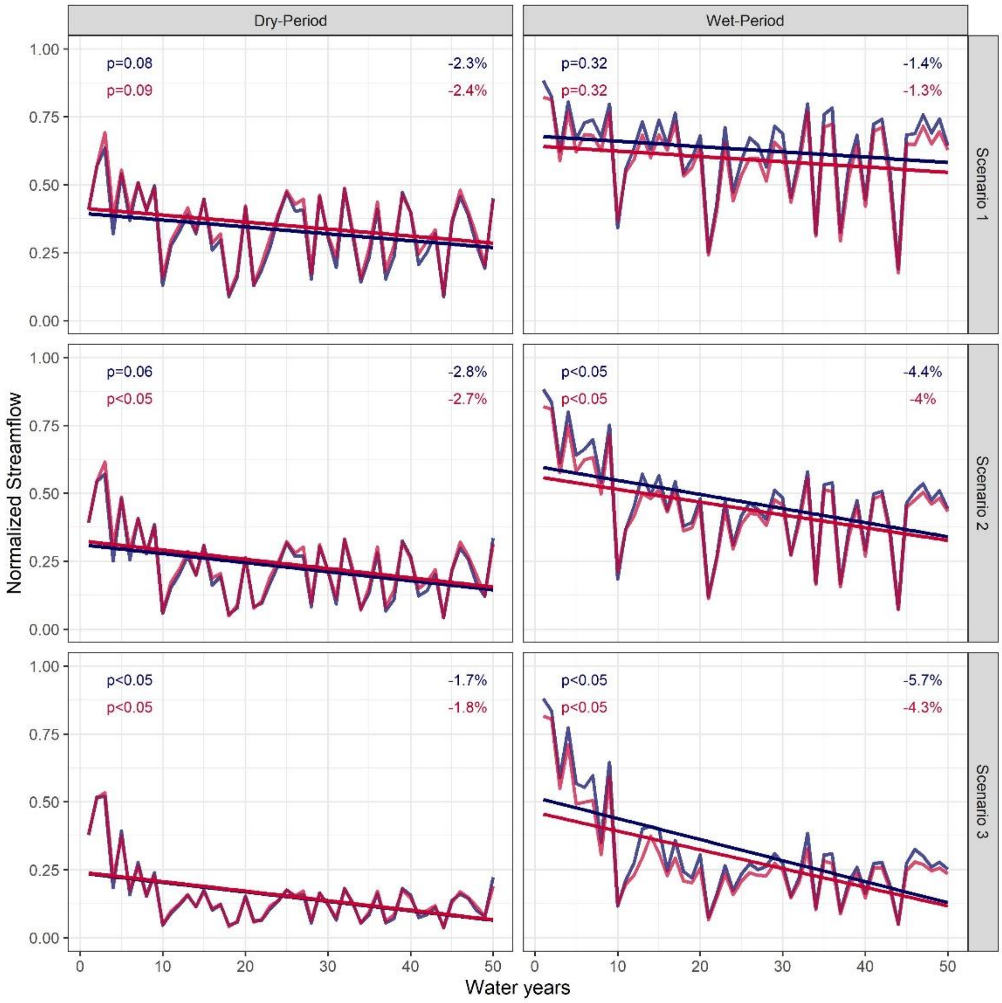
	Comparison	p value	Significance
Average annual Vegetation biomass	PM vs. AFF2	p<0.05	Yes
	PM vs. AFF1	p<0.05	Yes
	AFF1 vs. AFF2	p<0.05	Yes
	PM vs. PM.W	P = 0.95	No
	AFF1 vs. AFF1.W	P = 0.99	No
	AFF2 vs. AFF2.W	P = 0.99	No
Average annual Leaf Area Index	PM vs. AFF2	p<0.05	Yes
	PM vs. AFF1	p<0.05	Yes
	AFF1 vs. AFF2	p<0.05	Yes
	PM vs. PM.W	P = 0.99	No
	AFF1 vs. AFF1.W	P = 0.99	No
	AFF2 vs. AFF2.W	P = 0.41	No
Average annual rooting depth	PM vs. AFF2	P = 0.63	No
	PM vs. AFF1	P = 0.41	No
	AFF1 vs. AFF2	P = 0.85	No
	PM vs. PM.W	P = 0.75	No
	AFF1 vs. AFF1.W	P = 0.93	No
	AFF2 vs. AFF2.W	P = 0.99	No



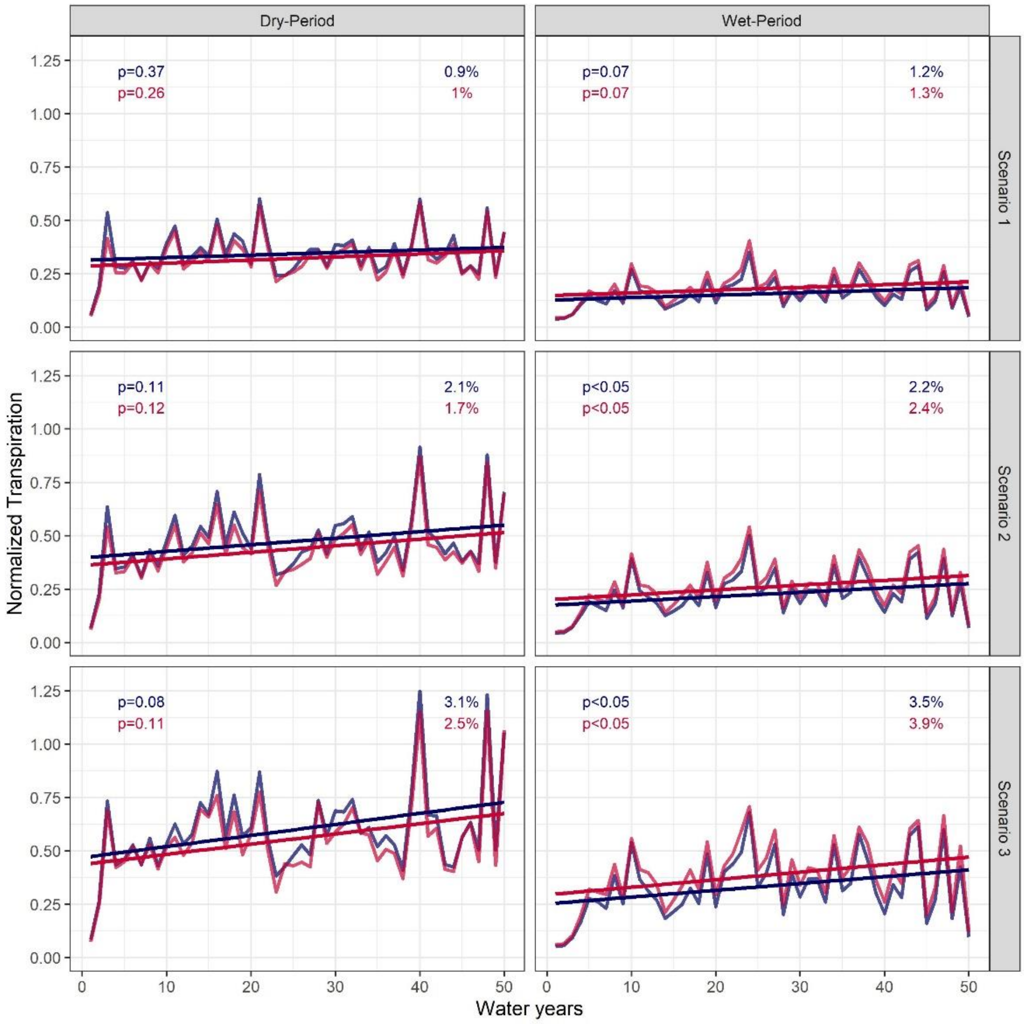
Supplementary Figure 5.1. Distribution of monthly observed streamflow in the Arnás catchment between 2000 and 2010. Median value (the line in the middle of boxes), 1st and 3rd quartiles (lower and upper box boundaries respectively), whiskers (hanging bars). Red points correspond to monthly averages.



Supplementary Figure 5.2. Annual and seasonal trends and changes in precipitation, maximum, minimum and average temperature (Tmax, Tmin and Tavg respectively). 95% level is used for the significance of the p-value of the Mann Kendall test. Values in the top right corner of plots indicate the magnitude of change per decade.



Supplementary Figure 5.3. Trends and magnitude of change in normalized streamflow in the wet (December to May) and dry periods (June to November) under the different management and climate scenarios. Scenario 1 (PM and PM.W: Passive management), Scenario 2 (AFF1 and AFF1.W: Active management), Scenario 3 (AFF2 and AFF2.W: Active management). Blue and red colors correspond to the no warming and warming scenarios respectively. 95% level is used for the significance of the p-value of the Mann Kendall test. Note that percentages indicate the magnitude of change per decade.



Supplementary Figure 5.4. Trends and magnitude of change in normalized transpiration in the wet (December to May) and dry periods (June to November) under the different management and climate scenarios. Scenario 1 (PM and PM.W: Passive management), Scenario 2 (AFF1 and AFF1.W: Active management), Scenario 3 (AFF2 and AFF2.W: Active management). Blue and red colors correspond to the no warming and warming scenarios respectively. 95% level is used for the significance of the p-value of the Mann Kendall test. Note that percentages indicate the magnitude of change per decade.

Supplementary materials for Chapter 6

Supplementary Table 6.1. TukeyHD test of the differences in Net Ecosystem Production (NEP) between land cover and climate scenarios. 95% level is used for the significance of the p-value of the TukeyHD test. PM and PM.W: No management, natural revegetation. AFF1 and AFF1.W: Active management. low afforestation, AFF2 and AFF2.W: Active management, intense afforestation. PM.W, AFF1.W and AFF2.W represent warming climate scenarios.

	Comparison	p value	Significance
Annual NEP	PM vs. AFF1	p<0.05	Yes
	PM vs. AFF2	p<0.05	Yes
	AFF1 vs. AFF2	p<0.05	Yes
	PM vs. PM.W	p=0.33	No
	AFF1 vs. AFF1.W	p=0.53	No
	AFF2 vs. AFF2.W	p=0.92	No

Supplementary Table 6.2. TukeyHD test of the differences in annual streamflow, transpiration and evaporation between land cover and climate scenarios. 95% level is used for the significance of the p-value of the TukeyHD test. PM and PM.W: No management, natural revegetation. AFF1 and AFF1.W: Active management. low afforestation, AFF2 and AFF2.W: Active management, intense afforestation. PM.W, AFF1.W and AFF2.W represent warming climate scenarios.

	Comparison	p value	Significance
Annual Streamflow	PM vs. AFF1	p<0.05	Yes
	PM vs. AFF2	p<0.05	Yes
	AFF1 vs. AFF2	p<0.05	Yes
	PM vs. PM.W	p>0.95	No
	AFF1 vs. AFF1.W	p>0.95	No
	AFF2 vs. AFF2.W	p>0.95	No
Annual Transpiration	PM vs. AFF1	p<0.05	Yes
	PM vs. AFF2	p<0.05	Yes
	AFF1 vs. AFF2	p<0.05	Yes
	PM vs. PM.W	p>0.95	No
	AFF1 vs. AFF1.W	p>0.95	No
	AFF2 vs. AFF2.W	p>0.95	No
Annual Evaporation	PM vs. AFF1	p<0.05	Yes
	PM vs. AFF2	p<0.05	Yes
	AFF1 vs. AFF2	p<0.05	Yes
	PM vs. PM.W	p>0.95	No
	AFF1 vs. AFF1.W	p>0.95	No
	AFF2 vs. AFF2.W	p>0.95	No

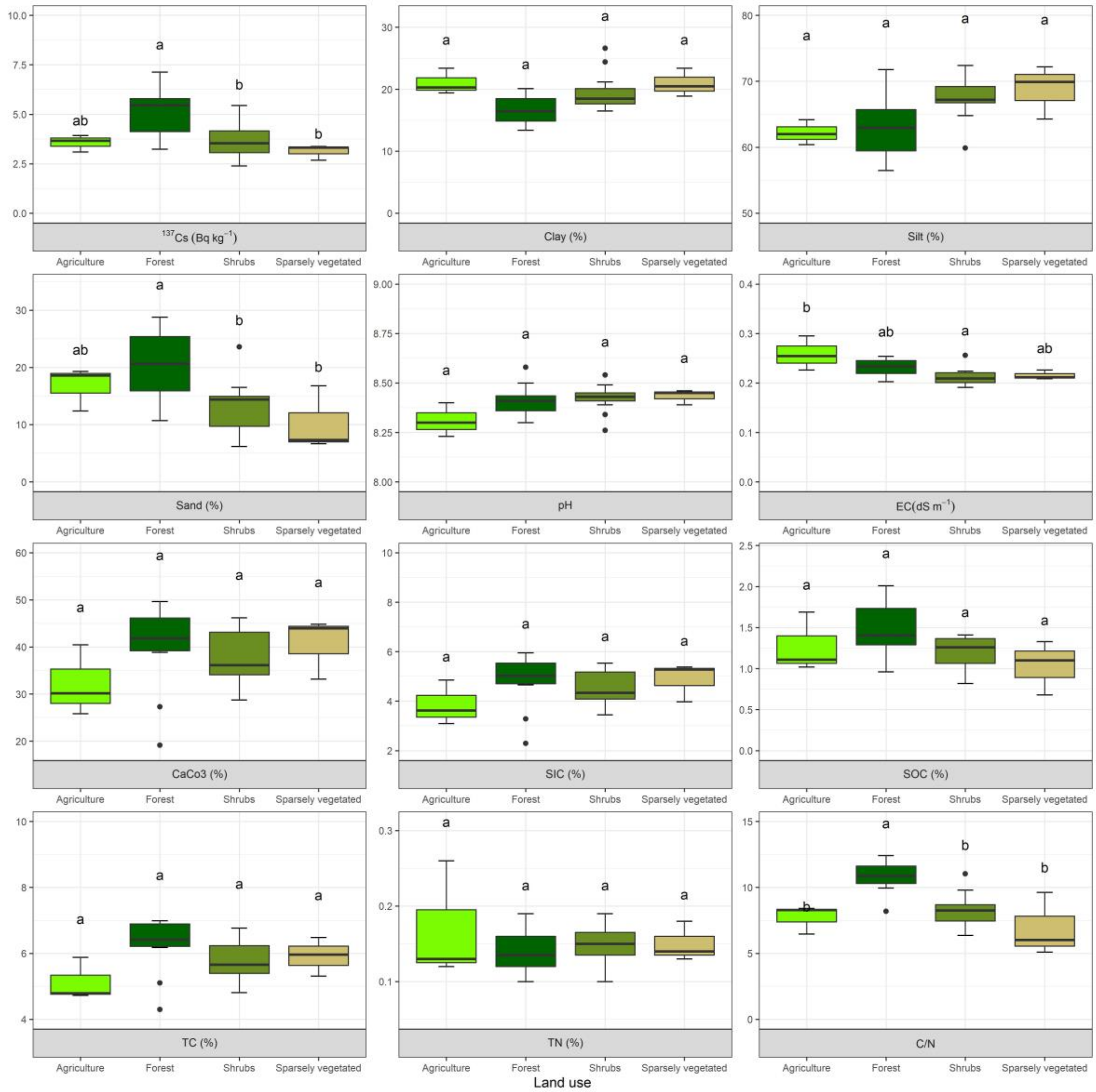
Supplementary Table 6.3. TukeyHD test of the differences in annual Water-Use Efficiency (WUE) between land cover and climate scenarios. 95% level is used for the significance of the p-value of the TukeyHD test. PM and PM.W: No management, natural revegetation. AFF1 and AFF1.W: Active management. low afforestation, AFF2 and AFF2.W: Active management, intense afforestation. PM.W, AFF1.W and AFF2.W represent warming climate scenarios.

	Comparison	p value	Significance
Annual WUE	PM vs. AFF1	p=0.31	No
	PM vs. AFF2	p<0.05	Yes
	AFF1 vs. AFF2	p=0.22	No
	PM vs. PM.W	p>0.95	No
	AFF1 vs. AFF1.W	p>0.95	No
	AFF2 vs. AFF2.W	p=0.84	No

Supplementary materials for Chapter 7

Supplementary Table 7.1. Basic statistics of the different soil properties in the Araguás catchment in the reference and grid sites. SOM: Soil Organic Matter, SOC: Soil Organic Carbon, TN: Total Nitrogen.

		Stonies (%)	Clay (%)	Silt (%)	Sand (%)	SOM (%)	SOC (%)	TN (%)	CN ratio	^{137}Cs activity (Bq kg^{-1})	^{137}Cs inventory (Bq m^{-2})
Reference sites	Median	24.5	30.6	34.4	26.4	1.8	1.1	0.1	8.4	1.5	1884.4
	Mean	23.6	31.4	41.2	27.4	2.8	1.6	0.2	9.0	4.6	1854.0
	SD	15.0	12.8	13.0	9.7	2.0	1.2	0.1	3.0	5.7	131.4
	Max	58.7	58.3	63.2	56.8	9.8	5.7	0.4	18.0	18.9	2009.8
	Min	0.3	13.5	21.1	13.4	0.5	0.3	0.1	3.9	0.0	1641.2
	CV	0.6	0.4	0.3	0.4	0.7	0.7	0.4	0.3	1.2	0.1
Grid Sites	Median	20.7	18.8	67.4	13.5	2.0	1.1	0.1	8.3	2.5	1166.5
	Mean	21.8	18.9	67.1	14.0	1.9	1.1	0.1	8.3	2.2	1004.6
	SD	7.4	2.7	4.9	6.4	0.7	0.4	0.0	2.1	2.3	1032.6
	Max	43.3	26.6	75.6	28.8	3.5	2.0	0.3	12.4	7.1	2915.5
	Min	4.6	13.4	56.5	4.3	0.7	0.4	0.1	4.5	0.0	0.0
	CV	0.3	0.1	0.1	0.5	0.3	0.3	0.3	0.3	1.1	1.0



Supplementary Figure 7.1. Boxplot with land uses classification and soil properties: Forest (afforestation), Shrubs (natural revegetation), Agriculture (grassland), Sparsely vegetated (badlands). Note: For ^{137}Cs activity boxplots samples below detection limits are excluded. SOC, soil organic carbon; SIC, soil inorganic carbon; TN, total nitrogen.

2. Research contributions and publication sources

<table>
<tr><th colspan="6">Publication sources and authors contributions</th></tr>
<tr><td colspan="6">Khorchani, M., Nadal-Romero, E., Tague, C., Lasanta, T., Zabalza, J., Lana-Renault, N., Domínguez-Castro, F., Choate, J., 2020. Effects of active and passive land use management after cropland abandonment on water and vegetation dynamics in the Central Spanish Pyrenees. Science of the Total Environment 717, 137160 https://doi.org/10.1016/j.scitotenv.2020.137160.</td></tr>
<tr><td colspan="6">The PhD student, Makki Khorchani, is the responsible of model set up, calibration, validation, analysis of results, writing and revisions. Nadal-Romero, E., Tague, C., Lasanta, T., Lana-Renault, N. and Domínguez-Castro, F. contributed in research designing, analysis of the results, writing and revision of manuscripts. Zabalza, J. and Choate, J. contributed in model set up and simulations.</td></tr>
<tr><th>ISSN</th><th>IF</th><th>5 years IF</th><th>JCR® category</th><th>Ranking in category</th><th>Category quartile</th></tr>
<tr><td>0048-9697</td><td>7.963</td><td>7.842</td><td>Environmental Sciences</td><td>25/274</td><td>Q1</td></tr>
</table>

<table>
<tr><th colspan="6">Publication sources and authors contributions</th></tr>
<tr><td colspan="6">Khorchani, M., Nadal-Romero, E., Lasanta, T., Tague, C., 2021. Effects of vegetation succession and shrub clearing after land abandonment on the hydrological dynamics in the Central Spanish Pyrenees. Catena 204, 105374. https://doi.org/10.1016/j.catena.2021.105374</td></tr>
<tr><td colspan="6">The PhD student, Makki Khorchani, is the responsible of model set up, analysis of results, writing and revisions. Nadal-Romero, E., Lasanta, T. and Tague, C. contributed in research designing, analysis of the results, writing and revision of manuscripts.</td></tr>
<tr><th>ISSN</th><th>IF</th><th>5 years IF</th><th>JCR® category</th><th>Ranking in category</th><th>Category quartile</th></tr>
<tr><td>0341-8162</td><td>5.198</td><td>5.594</td><td>Water Resources</td><td>12/98</td><td>Q1</td></tr>
</table>

<table>
<tr><th colspan="6">Publication sources and authors contributions</th></tr>
<tr><td colspan="6">Khorchani, M., Nadal-Romero, E., Lasanta, T., Tague, C., 2021. Natural revegetation and afforestation in abandoned cropland areas: Hydrological trends and changes in Mediterranean mountains. Hydrological Processes 35, e14191. https://doi.org/10.1002/hyp.14191</td></tr>
<tr><td colspan="6">The PhD student, Makki Khorchani, is the responsible of model set up, analysis of results, writing and revisions. Nadal-Romero, E., Lasanta, T. and Tague, C. contributed in research designing, analysis of the results, writing and revision of manuscripts.</td></tr>
<tr><th>ISSN</th><th>IF</th><th>5 years IF</th><th>JCR® category</th><th>Ranking in category</th><th>Quartile</th></tr>
<tr><td>0885-6087</td><td>3.565</td><td>4.129</td><td>Water Resources</td><td>26/98</td><td>Q2</td></tr>
</table>

Publication sources and authors contributions					
Khorchani, M., Nadal-Romero, E., Lasanta, T., Tague, C., 2022. Carbon sequestration and water yield tradeoffs following restoration of abandoned agricultural lands in Mediterranean mountains. **Environmental Research** 112203. https://doi.org/10.1016/j.envres.2021.112203					
The PhD student, Makki Khorchani, is the responsible of model set up, analysis of results, writing and revisions. Nadal-Romero, E., Lasanta, T. and Tague, C. contributed in research designing, analysis of the results, writing and revision of manuscripts.					
ISSN	**IF**	**5 years IF**	**JCR® category**	**Ranking in category**	**Category quartile**
0013-9351	6.498	6.824	Environmental Sciences	36/274	Q1

Publication sources and authors contributions					
Khorchani, M., Gaspar, L., Nadal-Romero, E., Arnáez, J., Lasanta, T., Navas, A. To be submitted. Effects of cropland abandonment and afforestation on soil properties and redistribution in a small Mediterranean mountain catchment.					
The PhD student, Makki Khorchani, is the responsible of field work, laboratory analysis, analysis of results and writing. Nadal-Romero E., Lasanta, T., Navas, A., Gaspar, L. and Arnáez J. contributed in research designing, field and laboratory work, analysis of results and writing.					
ISSN	**IF**	**5 years IF**	**JCR® category**	**Ranking in category**	**Category quartile**
-	-	-	-	-	-

3. Coauthored publications

Research under review

1. Montorio, R., Nadal-Romero, E., Cammeraat, E.L.H., Pérez-Cabello, F., **Khorchani, M.**, Badía-Villas, D., García-Martín, A. **2021**. Modelling soil organic carbon fractions using visible-near infrared spectroscopy of soil density fractions under different land uses. Under review in **Geoderma.** IF 6.114 **Q1 D1** (3/38 Soil Science)
2. Lasanta, T., Cortijos-López, M., Errea, M.P., **Khorchani, M.**, Nadal-Romero, E. **2021**. An environmental management experience to control forest fires in the mid-mountain Mediterranean area: shrub clearing to generate mosaic landscapes. Under review in **Land Use Policy**. IF 5.398 **Q1** (23/125 Environmental Sciences)

Published research

1. **Khorchani, M.**, Nadal-Romero, E., Lasanta, T., Tague, C., **2022.** Carbon sequestration and water yield tradeoffs following restoration of abandoned agricultural lands in Mediterranean mountains. **Environmental Research** 112203.

https://doi.org/10.1016/j.envres.2021.112203 IF 6.498 **Q1** (36/274 Environmental Sciences)

2. **Khorchani, M.**, Nadal-Romero, E., Lasanta, T., Tague, C., **2021**. Natural revegetation and afforestation in abandoned cropland areas: Hydrological trends and changes in Mediterranean mountains. **Hydrological Processes**, e14191. https://doi.org/10.1002/hyp.14191 IF 3.565 **Q2** (26/98 Water resources)
3. **Khorchani, M.**, Nadal-Romero, E., Lasanta, T., Tague, C., **2021**. Effects of vegetation succession and shrub clearing after land abandonment on the hydrological dynamics in the Central Spanish Pyrenees. **Catena** 204, 105374. https://doi.org/10.1016/j.catena.2021.105374 IF 5.198 **Q1** (12/98 Water Resources)
4. Nadal-Romero, E., Juez, C., **Khorchani, M.**, Peña-Angulo, D., Lana-Renault, N., Regüés. D., Lasanta, T., García-Ruiz, JM. **2021**. Impacts of land abandonment on flood mitigation in Mediterranean mountain areas. The Handbook of Environmental Chemistry Book series. **Springer** (Book chapter)
5. Juez, C., Peña-Angulo, D., **Khorchani, M.**, Regüés, D., Nadal-Romero, E., **2021**. 20-Years of hindsight into hydrological dynamics of a mountain forest catchment in the Central Spanish Pyrenees. **Science of the Total Environment,** 766, 142610. https://doi.org/10.1016/j.scitotenv.2020.142610 IF 7.963 **Q1 D1** (25/274 Environmental Sciences)
6. Lasanta, T., Nadal-Romero, E., **Khorchani, M.**, Romero-Díaz, A., **2021**. A review of abandoned lands in Spain: from local landscapes to global management strategies. **Geographical Research Letters**, 47(2), 477-521. https://doi.org/10.18172/cig.4755
7. **Khorchani M.**, Nadal-Romero E., Lasanta T., Zabalza J., Tague C., Lana-Renault N. Domínguez-Castro F., Choate J. **2020**. Effects of active and passive land use management after cropland abandonment on water and vegetation dynamics in the Central Spanish Pyrenees. **Science of the Total Environment**, 717, 137160. https://doi.org/10.1016/j.scitotenv.2020.137160 IF 7.963 **Q1 D1** (25/274 Environmental Sciences)
8. Lasanta, T, Sánchez-Navarrete, P., Medrano-Moreno, L.M., **Khorchani, M.**, Nadal-Romero, E., **2020**. Soil quality and soil organic carbon storage in abandoned agricultural lands: effects of revegetation processes in a Mediterranean mid-mountain area. **Land Degradation & Development**, 31, 2830-2845. https://doi.org/10.1002/ldr.3655 IF 4.977 **Q1** (8/37 Soil Science)

9. Nadal-Romero, E., **Khorchani, M.**, Lasanta, T., García-Ruiz, J. M. **2019**. Runoff and Solute Outputs under Different Land Uses: Long-Term Results from a Mediterranean Mountain Experimental Station. **Water**, 11, 976. https://doi.org/10.3390/w11050976 IF 3.103 **Q2** (39/98 Water Resources)

10. Navarro-Serrano, F., López-Moreno, J. I., Azorin-Molina, C., Buisán, S., Domínguez-Castro, F., Sanmiguel-Vallelado, A., Alonso-González E., **Khorchani, M. 2019**. Air temperature measurements using autonomous self-recording dataloggers in mountainous and snow covered areas. **Atmospheric Research**, 224, 168-179. https://doi.org/10.1016/j.atmosres.2019.03.034 IF 5.369 **Q1** (16/94 Meteorology & Atmospheric Sciences)

11. Peña-Angulo D., **Khorchani M.**, Nadal-Romero E., Errea P., Martínez-Arnáiz M., Lasanta T. **2019**. Factors explaining the diversity of land cover in abandoned fields in a Mediterranean mountain area. **Catena**, 181, 104064. https://doi.org/10.1016/j.catena.2019.05.010 IF 5.198 **Q1** (12/98 Water Resources)

12. **Khorchani M.**, Vicente-Serrano SM., Azorín-Molina C., García M., Martín-Hernández N., Peña-Gallardo M., El Kenawy A., Domínguez-Castro F. **2018**. Trends in LST over the peninsular Spain as derived from the AVHRR imagery data. **Global and Planetary Change**, 166, 75-93. https://doi.org/10.1016/j.gloplacha.2018.04.006 IF 5.114 **Q1** (24/199 Geosciences, Multidisciplinary)

13. **Khorchani M.**, Martín-Hernández N., Vicente-Serrano S-M., Azorín-Molina C., García M., Domínguez-Durán M.A., Reig F., Peña-Gallardo M., Domínguez-Castro F. **2018**. Average annual and seasonal Land Surface Temperature, Spanish Peninsular. **Journal of Maps**, 14, 465-475. https://doi.org/10.1080/17445647.2018.1500316 IF 2.709 **Q2** (40/85 Geography)

14. Lana-Renault N., López-Vicente M., Nadal-Romero E., Ojanguren R., Llorente J.A., Errea P., Regüés D., Ruiz-Flaño P., **Khorchani M.**, Arnáez J., Pascual N. **2018**. Catchment based hydrology under post farmland abandonment scenarios. **Geographical Research Letters**, 44, 503-534. https://doi.org/10.18172/cig.3475

15. Lasanta T., **Khorchani M**., Pérez-Cabello F., Errea P., Sáenz-Blanco R., Nadal-Romero E. **2018**. Clearing shrubland and extensive livestock farming: Active prevention to control wildfires in the Mediterranean mountains. **Journal of Environmental Management**, 227, 256-266. https://doi.org/10.1016/j.jenvman.2018.08.104. IF 6.789 **Q1** (34/274 Environmental Sciences)

4. Contributions to national and international conferences

1. Long-term abandonment of agro-ecosystems in Mediterranean mid-mountain areas: environmental consequences. (SERE2021, Alicante, Spain) Nadal-Romero, E., **Khorchani, M.**, Cortijos-López, M., Sánchez-Navarrete, P., Errea, P. and Lasanta, T. (Online presentation)
2. Shrub clearing to limit the negative effects of the natural revegetation in Mediterranean mountains. (COLORURAL2020, Valladolid, Spain). **Khorchani M.**, Sánchez-Navarrete P., Nadal-Romero E., Lasanta T. (Online presentation)
3. Shrub clearing as Active Management strategy to control land abandonment in the Central Spanish Pyrenees: The effects and the limits (EGU2020, Vienna, Austria). **Khorchani M.**, Nadal-Romero E., Lasanta T. (Online presentation)
4. Effects of land management after cropland abandonment on soil organic carbon stocks and soil quality in a sub-Mediterranean mountain area: the role of passive and active (shrub clearing and afforestation) practices (EGU2019, Vienna, Austria). Nadal-Romero E., Sánchez-Navarrete P., **Khorchani M.**, Lasanta T. (Poster)
5. Effects of afforestation programs on catchment hydrological response and water production among the 2050 horizon. (SER Europe2018, Reykjavic, Iceland) Nadal-Romero, Estela, **Khorchani, M.**, Lana-Renault, N., Zabalza, J., Tague, C., Choate, J., Regüés, D., Lasanta, T. (Oral presentation)
6. Effects of land use changes on water production among the 2050 horizon in the Central Spanish Pyrenees (EGU2018, Vienna, Austria). **Khorchani M.**, Zabalza J., Tague C., Vicente-Serrano S., Lana-Renault N., Domínguez-Castro F., Choate J., Nadal-Romero E. and Lasanta T. (Poster)
7. Shrub clearings and extensive livestock: an action of the administration to control the fires in the Mediterranean mountain (TERRAENVISION2018, Barcelona, Spain) **Khorchani M.,** Sáenz-Blanco R., Nadal-Romero E. and Lasanta T. (Poster)
8. Effects of afforestation and natural reforestation of abandoned croplands on water yield in the Central Spanish Pyrenees. (ECODESERT2019, Almería, Spain) **Khorchani Makki**, Nadal-Romero E., Lasanta T., Zabalza J. and Tague C. (Oral Presentation)
9. Effects of land use changes and land management after cropland abandonment in soil organic carbon stocks in a sub-mediterranean mountain area (Iberian System).

(ECODESERT2019, Almería, Spain). Sánchez-Navarrete P., **Khorchani M.**, Nadal-Romero E. and Lasanta T. (Oral presentation)

10. Changes in soil organic carbon stocks in the subalpine belt in the Borau Valley as a consequence of revegetation processes (EGU2018, Vienna, Austria). Otal-Laín I., Nadal Romero E., Lasanta T., Sánchez-Navarrete P., **Khorchani M.**, Errea P. and Cammeraat E. (Poster)

11. Changes in catchment hydrological connectivity under different post farmland abandonment scenarios. (EGU2018, Vienna, Austria) Lana-Renault N., López-Vicente M., Nadal-Romero E., Ojanguren R., Llorente J. A., Errea P., Regüés D., Ruiz-Flaño P., **Khorchani M.**, Arnáez J. and Pascual N. (Poster)

12. Effects of shrub clearing in soil organic carbon stocks in a sub-mediterranean mountain area. (EGU2018, Vienna, Austria). Sánchez-Navarrete P., **Khorchani M.**, Nadal-Romero E. and Lasanta T. (Poster)