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Land-use changes and carbon sequestration through the twentieth century in a Mediterranean mountain ecosystem: Implications for land management

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ABSTRACT

Ecosystems in the western Mediterranean basin have undergone intense changes in land use throughout the centuries, resulting in areas with severe alterations. Today, most these areas have become sensitive to human activity, prone to profound changes in land-use configuration and ecosystem services. A consensus exists amongst stakeholders that ecosystem services must be preserved but managerial strategies that help to preserve them while ensuring sustainability are often inadequate. To provide a basis for measuring implications of land-use change on carbon sequestration services, changes in land use and associated carbon sequestration potential throughout the 20th century in a rural area at the foothills of the Sierra Nevada range (SE Spain) were explored. We found that forest systems replaced dryland farming and pastures from the middle of the century onwards as a result of agricultural abandonment and afforestation programs. The area has always acted as a carbon sink with sequestration rates ranging from 28,961 t CO_2 year^{-1} in 1921 to 60,635 t CO_2 year^{-1} in 1995, mirroring changes in land use. Conversion from pastures to woodland, for example, accounted for an increase in carbon sequestration above 30,000 t CO_2 year^{-1} by the end of the century. However, intensive deforestation would imply a decrease of approximately 66% of the bulk CO_2 fixed. In our study area, woodland conservation is essential to maintain the ecosystem services that underlie carbon sequestration. Our essay could inspire policymakers to better achieve goals of increasing carbon sequestration rates and sustainability within protected areas.

1. Introduction

Ecosystems in the western Mediterranean basin have undergone intense changes in land use over the past several centuries (Puigdefábregas and Mendizábal, 1998; Blondel, 2006). The expansion of dryland farming that was practiced until the beginning of the 20th century almost completely degraded vegetation in many areas while grazing and selective logging practices disturbed others (Brandt and Thornes, 1996; Latorre et al., 2001). In the second half of the century socioeconomic forces triggered the abandonment of farmland and rural life (Garcia Ruiz et al., 1996; Debussche et al., 1999; Lasanta-Martinez et al., 2005). This occurred in conjunction with intensive forestry policies that expanded forested land in mountainous areas (Kaul, 1970; Scarascia-Mugnozza et al., 2000; Poyatos et al., 2003; Falcucci et al., 2007). Currently, existing Mediterranean woodlands face various threats such as deforestation, man-made fires, and urban/industrial development (Bussotti and Ferretti, 1998; Scarascia-Mugnozza et al., 2000; Palahi et al., 2008).

Land-use change always impacts local environments, but the dynamics of these changes have become a driving force of potentially global consequences (Foley et al., 2005). Changes in land use enable humans to increase resource appropriation, but also of potentially undermine the capacity of ecosystems to provide services. Therefore, quantifying the magnitude of land-use change is essential to estimate its consequences on ecosystem services. Carbon sequestration is one such example of an ecosystem service that is dependent on land-use change (Metzger et al., 2006; Schulp et al., 2008). Most terrestrial ecosystems act as net carbon sink, fixing more CO_2 than they release back into the atmosphere through autotrophic and heterotrophic respiration (Schimel, 1995), particularly forests, which are important components in the global C budget because of the large quantities of biomass stored above and belowground, thereby regulating atmospheric CO_2 concentrations and, hence, the climate (Fahey et al., 2010). Forest conversion to other uses releases C to the atmosphere and influence the

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provision of services underlying carbon sequestration (Feddema et al., 2005; Metzger et al., 2006; Schulp et al., 2008) since different ecosystems differ in potential rates of carbon sequestration. For instance, the conversion from forests to croplands or vice versa has a strong bearing on carbon budgets (Silver et al., 2000; Niu and Duiker, 2006; Sharma and Rai, 2007; Don et al., 2009).

Estimating carbon sequestration associated to land use is particularly important at the regional level where managers and policymakers alike must make informed decisions to better assess the implications of land-use changes (Feng, 2005; Yin et al., 2007). Moreover, knowing how much and where this service is localized may ease management decisions (Janssens et al., 2005) since estimates of vegetation units can serve as a basis to model implications of land-use changes on carbon sequestration (Millennium Ecosystem Assessment, 2005).

In this study, land-use changes and associated carbon sequestration that occurred through the 20th century in a rural area of SE Spain are explored. As in many regions around the Mediterranean basin, this particular area has historically experienced important land-use changes and is an example of changes that occurred in SE Spain in the last fifty years, i.e., reduction of dry farming, increase in woodlands, and agriculture intensification. The economy within the area relies to a great extent on agriculture and subsidies and barely profits from natural resource values (Vidal et al., unpublished). Although biodiversity within the area is exceptional in terms of endemic species and forest cover (Molero Mesa et al., 1992), deforestation related to intensive agriculture may threaten it.

Carbon sequestration potential as an ecosystem service could foster not only woodland conservation but also promote sustainable rural development. To assess the evolution of this potential, plant cover and land use taken from local cadastres and forest surveys in 1921, 1947, and 1995 were recorded while potential carbon sequestration for each land-use type was calculated from published sequestration rates. Methods traditionally intended for regional scales and based upon biomass increments (Rodriguez-Murillo, 1997; IPCC, 2006) could not be applied here because consecutive data for the sample sites used in this study were lacking. Assessments of C sequestration are available for a wide range of environments and scales, yet little work has been carried out at regional scales. First, because research conducted in experimental areas (e.g., plots), though very reliable, restricts to relatively uniform, representative land areas of up to several hundred meters in length (Moncrieff et al., 2000; Baldocchi et al., 2001). Second, because large-scale models (Janssens et al., 2003) may suffer from inaccuracy due to oversimplified land-use categories. The regional scale approach applied here may be valid for managerial purposes as it provides insights linking land-use changes with carbon sequestration.

2. Methods

We first carried out a land-use classification, then determined carbon sequestration rates for the different land-use classes, later scaled up carbon sequestration rates, and finally integrated total carbon sequestration of the different land-use units.

2.1. Description of the study area

The study area includes the Abla and Abrucena municipalities (lat 37° N, long 2° W), small villages within the Nacimiento river valley, Almería Province (SE Spain). The area covers approximately 13,000 ha between the Sierra de los Filabres range to the north and the Sierra Nevada range to the south (from 750 m to 2500 m elevation), both of which frame the Nacimiento valley (Fig. 1). Soil type mostly consists of eutric cambisol developed over micaschist bedrock. The Sierra Nevada range hosts exceptional biodiversity (Molero Mesa et al., 1992) that is protected at the regional, national, and European scales and is considered a Biosphere Reserve by UNESCO. Approximately 60% of the study area is protected by way of legal safeguards in one way or another.

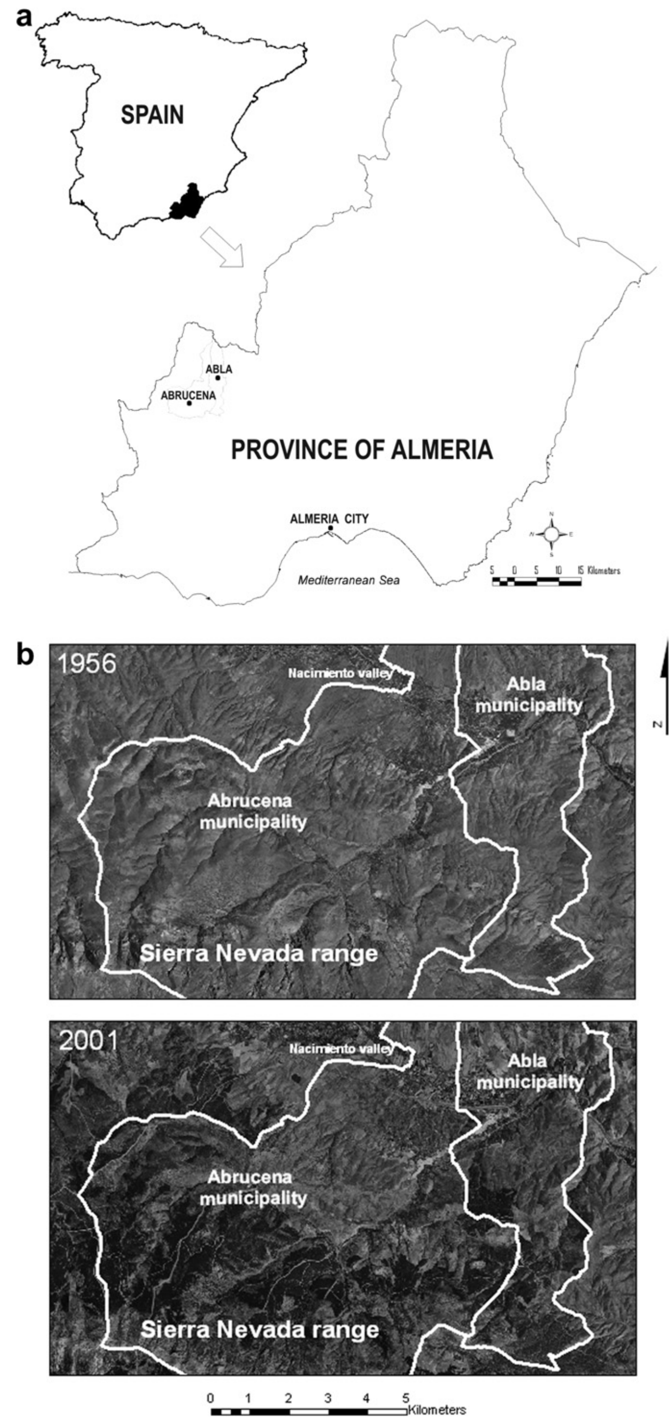

Fig. 1. Location map of the study area (a) and growth of afforested areas from 1956 to 2001 as shown by ortophotos of the respective years (b). Note deforested areas in 1956 and afforestation by pines in darker areas in 2001. Source: modified from Red de Información Ambiental (Junta de Andalucía).

The landscape within the N and S borders experience rugged and steep terrain with peaks reaching from 2200 m to 2500 m in elevation. The climate is typical Mediterranean, with a marked dry season and irregular precipitation throughout the year. It is characterized by moderately low temperatures in winter while being mild in summer. Two climatic zones can be distinguished: an alpine zone with a relative high precipitation rate (from 500 mm to 700 mm year^{-1}) and cold weather (annual mean temperature <10 °C) and a lowland zone that experiences semiarid conditions (from 300 mm to 500 mm year^{-1}; Red de Información Ambiental de Andalucía, 1961–1990) and milder temperatures (mean annual temperature from 12 °C to 13 °C). Vegetation has been modeled by a long history of anthropogenic activity but more intensively within the last century by way of forest fires, logging, extensive pine afforestation, and dryland subsistence farming and terracing, leading to semi-natural agro-ecosystems and forests.

Land above 2000 m in elevation is currently dominated by common juniper (*Juniperus communis*) and yellow broom (*Genista versicolor*). Disturbances to this community lead to a grassland–scrubland ecosystem dominated by tor-grass (*Festuca indigesta*) and sierra thyme (*Thymus serpylloides*). Primary forest patches are remnants of pine afforestation that occurred in the last sixty years as well as regeneration of native Holm Oak forests. Pine forests occur mostly within the 750–2000 m elevation range with Aleppo pine occurring at lower elevations, maritime and black pines at mid-elevations, and Scots pine higher up. Holm Oak forests dominated by *Quercus ilex* and accompanied by the shrubs retama (*Retama sphaerocarpa*) and silver broom (*Adenocarpus decorticans*) occur in the 900–2000 m range. Degradation of this community leads to a shrubland ecosystem consisting of retama, silver broom, *Genista* spp., and *Artemisia barrelieri*. A plant community consisting primarily of tussock grasses (*Stipa tenacissima*, *Brachypodium retusum*) interspersed with shrubs such as albaida (*Anthyllis cytisoides*) dominates at low elevations and those under more xeric conditions (Valle et al., 2003). Dryland farmed almond trees and irrigated olive orchards grow on terraces and rolling hills. In the fertile lowlands, fruit trees, cereal, and vegetable crops dominate (MAPA, 2000).

2.2. Land-use changes

We classified the territory into seven land-use categories based upon rankings reported in local historical cadastres and the National Forest Survey. Categories were established according to the dominant species or land use and included cereal crops (primarily barley, wheat, and oats), olive groves (*Olea europaea*), almond orchards (*Prunus dulcis*), vineyards (*Vitis vinifera*), pine forests (primarily Aleppo pine, *Pinus halepensis;* European black pine, *Pinus nigra;* maritime pine, *Pinus pinaster;* Scots pine, *Pinus sylvestris*), Holm Oak forests (*Q. ilex*), and grassland–shrubland (*Stipa* spp., *Genista* spp., *A. cytisoides*). All seven land-use categories accounted for more than 98% of the study area. The remaining 2%, including urban areas and vegetable crops, was discarded due to the inherent variability of these units.

Surface area per land-use category in the early and mid-twentieth century was obtained from local historical cadastre sheets recorded in 1921 and 1947 (Archivo Histórico Provincial de Almería). For the late twentieth century, surface area was obtained from the latest National Forest Survey available (IFN2, MMA 2001) that was carried out in 1995. Dimensions of the two municipalities did not vary substantially in the last century. Agricultural land uses at the end of the twentieth century were obtained from local and regional statistics Institutes (Cámara de Almería and Instituto de Estadística de Andalucía). Land use described in cadastres and the National Forest Survey roughly matched, making the two sources comparable along the years.

2.3. Carbon sequestration

Carbon sequestration rates reported for similar ecosystem type dominated by the same plant species were used and scaled up to estimate the amount of carbon sequestered by each land-use type (Table 1). The *Web of Science* database (ISI-Thomson) was applied to search for the keywords *net ecosystem exchange, carbon sequestration*, *carbon flux*, *carbon fixation*, and *carbon capture*, as well as the desired land-use type (e.g., cereal crops, almond orchards, etc.). Whenever possible, selected papers reported on data from Mediterranean systems. Moreover, papers that reported on carbon *Net Ecosystem Exchange* (*NEE*) (i.e., the net balance between carbon fixation and emission fluxes for a period of at least one year) were focused on. Unfortunately, NEE rates for certain land-use types used in this study were not found, and estimations had to be carried out from *Net Primary Productivity* (*NPP*) rates that did not consider heterotrophic carbon emissions. Chiesi et al. (2005), however, modeled an NEE/NPP ratio of 0.645 ± 0.087 for Mediterranean forests in central Italy (42° N). This ratio was used to obtain NEE rates from reported NPP as latitude and climate are similar.

Carbon sequestration rates were obtained for each land-use type and were then applied to corresponding surface areas to obtain the amount of carbon that can be sequestered yearly by a particular land type. Rates were averaged when more than one sequestration rate was found for a given land-use type, so carbon sequestration data are presented as means ± standard error throughout. For our mixed grassland–shrubland land-use type, sequestration rates for grasslands and shrublands were averaged. CO_2 sequestration within a given land use was eventually obtained via simple stoichiometry, and total CO_2 sequestration in the study area by summing the CO_2 sequestration from each land-use type.

3. Results

3.1. Cereal crops

A sizable decrease in cereal crops was documented in 1921 (5016 ha), and from 1947 (4434 ha) to 1995, when barely 7 ha remained for cereal crop production (Fig. 2). Wheat crops were reported to sequester 1.85 to 2.45 t C ha^{-1} annually in Germany (Anthoni et al., 2004) and 0.63 t C ha^{-1} year^{-1} in Belgium (Moureaux et al., 2008). Since no rates for cereal crops at southern latitudes were found in our review, and most of the crops grown within the sample sites used in this study are barley and wheat, the rates were averaged in which a net sequestration rate of 1.64 ± 0.54 t C ha^{-1} year^{-1} (Table 2) was obtained for cereal crops. Wheat productivity in Almería Province is much lower than it is in the aforementioned studies. This is reflected in grain yield; while yields in the German and Belgian sites were approximately 8.1 ± 0.7 t ha^{-1}, the average yield in Almería was 1.2 ± 0.1 t ha^{-1} (Consejería de Agricultura y Pesca, 2000–2006). Therefore, assuming proportionality between reported NEE and grain yield, the carbon sequestration rate of the cereal crops grown within the study area would be 0.25 ± 0.05 t C ha^{-1} year^{-1}. By taking into account the surface area of this land use over a period of a century, cereal crops would have sequestered 1254 ± 145 t C year^{-1} in 1921, 1102 ± 138 t C year^{-1} in 1947, and 1.8 ± 0.2 t C year^{-1} in 1995 (Fig. 3).

3.2. Woody cultures

The olive groves surface area remained for the most part constant between 1921 (326 ha) and 1947 (363 ha) but increased to a great extent in the second half of the century, to 548 ha in 1995. For olive orchards, Sofo et al. (2005) estimated an NPP of 1.67 t C ha^{-1} year^{-1} in Italy, which would be equivalent to an NEE of 1.07 ± 0.14 t C ha^{-1} year^{-1} when assuming the NEE/NPP ratio reported by Chiesi et al. (2005). In a grove in southern Spain, Testi et al. (2008) calculated an NEE of 2.8 t C ha^{-1} year^{-1}, but the olive

Table 1
Key characteristics of reviewed publications and reported carbon sequestration rates for the primary species dominating each land-use type.

Key species	Reference	Location	Ecosystem	Density (trees ha^{-1})	Rainfall (mm)	Latitude	C sequestration (t $year^{-1}$ ha^{-1})	Data
Triticum aestivum	Anthoni et al., 2004	Germany	Wheat crop	–	–	51° N	1.85–2.45	NEE
	Moureaux et al., 2008	N Belgium	Wheat crop	–	800	50° N	0.63	NEE
Olea europaea	Testi et al., 2008	S Spain	Olive grove	408	555	38° N	2.80	NEE
	Sofo et al., 2005	S Italy	Olive grove	156	–	40° N	1.67	NPP
Prunus dulcis	Esparza et al., 1999	California, USA	Almond orchard	–	–	38° N	7	NPP
Vitis vinifera	Evrendilek et al., 2005	S Turkey	Vineyard	650	647	37° N	- 2.27 ± 1.14	NEE
Pinus halepensis	Grunzweig et al. 2007	Israel	Pine stand	360	270	31° N	0.99	NPP
Pinus nigra	Evrendilek et al., 2006	Turkey	Forest	300	800	37° N	1.57 ± 0.18	NEE
Pinus pinaster	Berbigier et al., 2001	SW France	Pine stand	500	930	44° N	5.7 ± 0.8	NEE
Pinus sylvestris	Valentini et al., 2000	Netherlands	Pine Stand	446	786	52° N	2.10	NEE
	Zha et al., 2004	Finland	Pine stand	1176	724	62° N	1.58 ± 0.22	NEE
	Bravo et al., 2008	NE Spain	Pine stand	600	800	42° N	2.26 ± 0.32	NPP
Quercus ilex	Valentini et al., 2000	Italy	Forest	–	500	41° N	6.6	NEE
	Allard et al., 2008	S France	Forest	–	907	43° N	2.78 ± 0.48	NEE
Stipa krylovii	Li et al., 2005	NE Mongolia	Grassland	–	196	47° N	0.41	NEE
Artemisia tridentata	He and Zhang, 2003	Nevada, USA	Scrubland	–	–	36° N	1.06 ± 0.04	NEE
Adenostoma fasciculatum	Luo et al., 2007	S California, USA	Shrubland	–	349	33° N	0.52	NEE

Positive values represent ecosystem carbon sinks while negative values represent ecosystem carbon source; NEE: net ecosystem exchange; NPP: net primary productivity.

stand was denser than in the case of the olive grove used in this study (408 *vs.* 150 trees ha^{-1}, respectively). By scaling the latter NEE to the tree density of our olive groves, a NEE of 1.03 t C ha^{-1} $year^{-1}$ was obtained. By averaging the Sofo and Testi NEEs, a carbon sequestration rate of 1.06 ± 0.06 t C ha^{-1} $year^{-1}$ was then obtained. By applying this rate to the olive grove surface area of the sample site used in this study, a continuous increase in carbon sequestration potential was found from 295 ± 62 t C in 1921, 329 ± 69 in 1947, and 496 ± 105 t C in 1995.

The area of almond orchards more than doubled from 1921 (368 ha) to 1947 (772 ha) and tripled from 1947 to 1995 (2032 ha). Esparza et al. (1999) calculated an NPP of 7 t C ha^{-1} $year^{-1}$ in Californian orchards where intensive farming practices are applied. Almond trees in Abla and Abrucena produce much less than those in California, which is reflected in almond production. While yields in California were approximately 1.6 t ha^{-1} (Almond Board of California, 2006), yields in Almería averaged 0.260 t ha^{-1} (Consejería de Agricultura y Pesca, 2001–2005). Therefore, assuming proportionality between NPP and almond production, the NPP of the orchards within the sample sites used for this study would be 1.14 t C ha^{-1} $year^{-1}$. Furthermore, by applying the aforementioned NEE/NPP ratio developed by Chiesi et al. (2005), a sequestration rate of 0.735 ± 0.1 t C ha^{-1} $year^{-1}$ was obtained, which would amount to a potential carbon sequestration of 270 ± 21, 567 ± 45, and 1494 ± 117 t $year^{-1}$ in 1921, 1947, and 1995, respectively.

Vineyard surface area decreased towards the end of century from 126 ha in 1921, 108 ha in 1947, and 18 ha in 1995. Evrendilek et al. (2005) reported a net ecosystem emission of 2.3 ± 1.1 t C ha^{-1} $year^{-1}$ for a Turkish vineyard (lat 37° N). Since rainfall and plant density in the Turkish site were similar to those in the area under examination for this study, this rate was applied to the vineyard within the sample site where a carbon emission

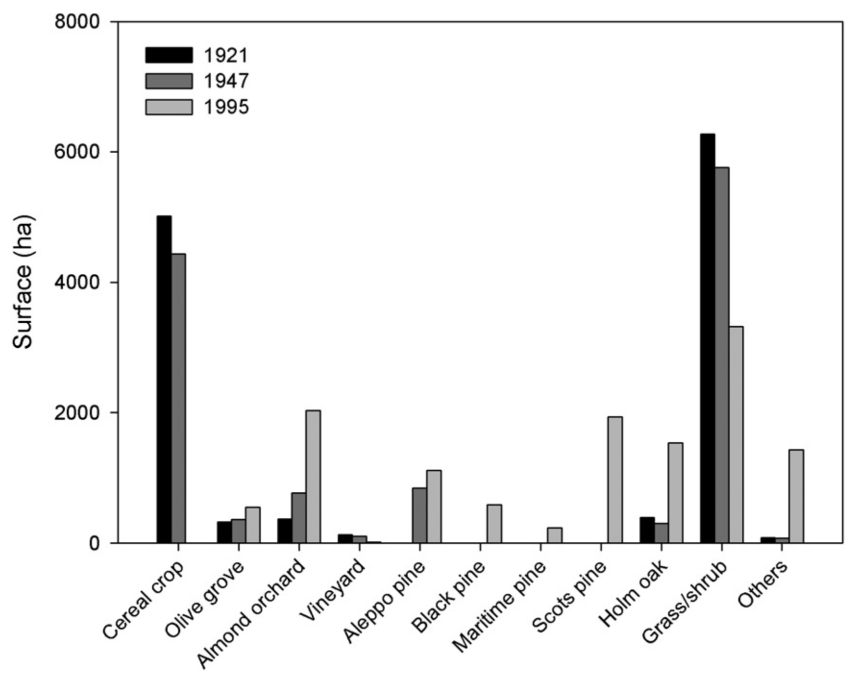

Fig. 2. Surface area (ha) of each land use defined in the study region for the years 1921, 1947, and 1995.

Table 2
Adopted carbon sequestration rate (±1 SE) for each land-use type.

Land use	Main species	Carbon sequestration rate (t C $year^{-1}$)
Cereal crops	*Hordeum* sp., *Triticum* sp.	0.25 ± 0.05
Olive groves	*Olea europaea*	1.07 ± 0.06
Almond orchards	*Prunus dulcis*	0.74 ± 0.1
Vineyards	*Vitis vinifera*	−2.27 ± 1.14
Pine forests	*Pinus halepensis*	0.65 ± 0.09
	Pinus nigra	1.57 ± 0.18
	Pinus pinaster	5.7 ± 0.8
	Pinus sylvestris	1.48 ± 0.23
Holm oak forests	*Quercus ilex*	4.05 ± 1.30
Grassland–shrubland	*Genista* sp., *Stipa* spp.	0.76 ± 0.17[a]

[a] Obtained by averaging grassland and shrubland rates.

ranging from 286 ± 83 t C $year^{-1}$, 245 ± 71 t C $year^{-1}$, and 41 ± 12 t C $year^{-1}$ for the years 1921, 1947, and 1995, respectively, was obtained.

3.3. Pine forests

Although pine forests were nonexistent in 1921, forestry activity that was initiated towards the middle of the century established 844 ha of pineland by 1947. Pine plantations intensified from the middle of the century onwards. By 1995, pine forests covered an overall surface area of 3872 ha in which Scots pine was the most abundant species.

For Scots pine forests, Zha et al. (2004) reported an NEE of 1.58 ± 0.22 t C ha^{-1} $year^{-1}$ in Finland (lat 62° N). Similarly, Valentini et al. (2000) reported an NEE of 2.10 t C ha^{-1} $year^{-1}$ in the Netherlands (lat 52° N). In northern Spain (lat 42° N), Bravo et al. (2008) calculated an NPP of 2.26 ± 0.32 t C ha^{-1} $year^{-1}$, which would convert to an NEE of 1.476 ± 0.231 t C ha^{-1} $year^{-1}$ when applying the NEE/NPP ratio developed by Chiesi et al. (2005). Given that latitude is the most appropriate scaling factor to determine the NEE of a mature forest (Valentini et al., 2000), the Spanish NEE rate was taken for this study. By taking this NEE rate into account and applying it to the surface area of Scots pine forests in 1995 (1936 ha), this land-use unit would have sequestered 2858 ± 258 t C $year^{-1}$.

In an Aleppo pine forest near the Negev Desert in Israel (lat 31° N, 270 mm $year^{-1}$ annual precipitation), Grunzweig et al. (2007) calculated an NPP of 0.99 t C ha^{-1} $year^{-1}$, equivalent to an NEE of 0.645 ± 0.087 t C ha^{-1} $year^{-1}$ when applying the NEE/NPP ratio (Chiesi et al., 2005). Given the comparable rainfall and proximity in latitude between the Israelis site and the sample site used in this study, the former rate was applied to the latter surface area (1116 ha). A potential carbon sequestration rate of 720 ± 56 t $year^{-1}$ was then calculated.

In a Mediterranean black pine forest located in Turkey with an elevation of 1550 m and an annual rainfall of 800 mm, Evrendilek et al. (2006) estimated an NEE rate of 1.57 ± 0.18 t C ha^{-1} $year^{-1}$. Tree density in the area under examination for this study was close to that of the Turkish site. Given the similarities in latitude, climate, and tree density, the reported rate was applied to the 587 ha sample site in which a carbon sequestration rate of 922 ± 61 t C $year^{-1}$ was calculated.

In a maritime pine forest in Bordeaux (France), Berbigier et al. (2001) calculated a NEE rate of 5.7 ± 0.8 t C ha^{-1} $year^{-1}$. It was estimated that this pine forest (233 ha) would have sequestered 1328 ± 108 t C $year^{-1}$ in 1995.

No information is available concerning what specific pine species dominated in the year 1947. Due to this, NEE rates reported on the aforementioned pine species were averaged (i.e., 2.35 ± 0.60 t C ha^{-1} $year^{-1}$). It was estimated that the 844 ha pine forests present in year 1947 would have sequestered 1983 ± 292 t C $year^{-1}$.

3.4. Holm oak forests

Holm Oak forests were scarce in 1921 and 1947, covering less than 400 ha each year. However, its surface area considerably increased in the second half of the century. More than 1500 ha of oak forests were present in 1995. For this particular oak species, Valentini et al. (2000) reported a net NEE rate of 6.6 t C ha^{-1} $year^{-1}$ in central Italy. Allard et al. (2008) found that a typical Mediterranean forest in

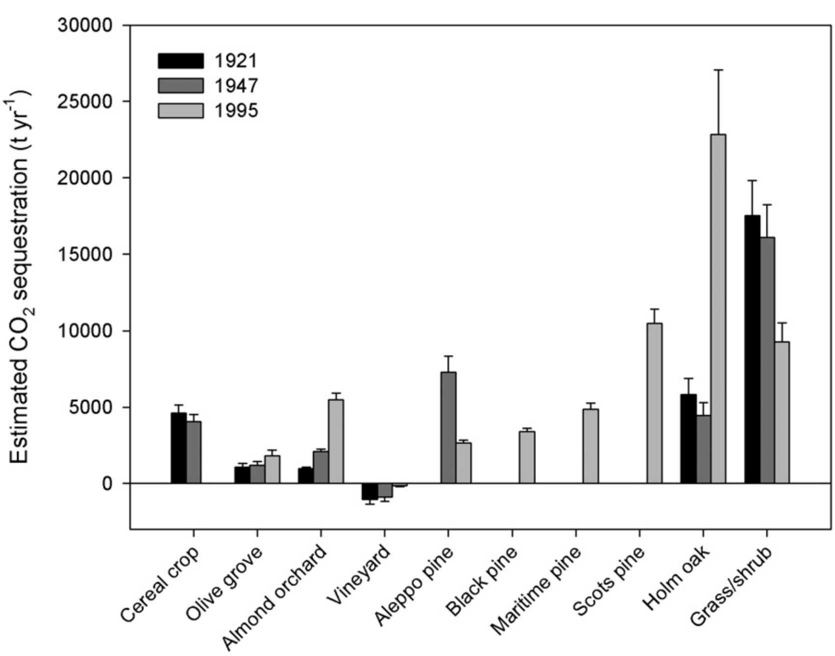

Fig. 3. Estimated CO_2 sequestration ±SE for each land-use type in the study area through the twentieth century.

Montpellier, France, sequestered 2.78 ± 0.48 t C ha^{-1} $year^{-1}$ on average. Given the similarity in climate and latitude between the aforementioned studies and the sample sites, the above NEE rates were averaged (4.05 ± 1.3 t C ha^{-1} $year^{-1}$) and applied to the surface area of the site under examination for this study. The carbon sequestration rate of oak woodland would thus have changed from 1585 ± 294 t C $year^{-1}$ and 1216 ± 226 t C $year^{-1}$ in 1921 and 1947, respectively, to 6226 ± 1156 t C $year^{-1}$ in 1995.

3.5. Grassland–shrubland

Grasslands and shrublands decreased in the second half of the century. Their surface area in 1921 (6269 ha) and 1947 (5762 ha) were approximately double than that of 1995 (3321 ha). No data exists concerning the NEE rate for the grassland–shrubland land-use type of this study, but there exists some reports on grasslands and shrublands in other regions of the world. Li et al. (2005) reported a NEE of 0.41 t C ha^{-1} $year^{-1}$ in an arid steppe in Mongolia. In California, Luo et al. (2007) calculated a NEE rate of 0.52 t C ha^{-1} $year^{-1}$ in a semiarid shrubland. Similarly, Wohlfahrt et al. (2008) reported a NEE rate of 1.06 ± 0.04 t C ha^{-1} $year^{-1}$ for a scrubland in the Mojave Desert. Since grass species and low shrubs occur in interspersed mixtures in the study area, the above rates were averaged (0.76 ± 0.17 t C ha^{-1} $year^{-1}$). The reduction in grassland–shrubland total surface area reflected the reduction in carbon sequestration, which would have reduced from 4780 ± 630 t C $year^{-1}$ and 4394 ± 579 t C $year^{-1}$ in 1921 and 1947, respectively, to 2532 ± 334 t C $year^{-1}$ in 1995.

3.6. Total carbon sequestration in the area

Pooling all land types together, the total amount of carbon sequestered in the study area would have been 7898 ± 713 t $year^{-1}$ in 1921, 9346 ± 817 t $year^{-1}$ in 1947, and 16,537 ± 1274 t $year^{-1}$ in 1995. This means a continuous increase of C capture potential: 15% from 1921 to 1947, 43% from 1947 to 1995, and a total increase of 52% from 1921 to 1995.

4. Discussion

4.1. Land-use changes

Intense changes in land use took place over the twentieth century in the Sierra Nevada range (SE Spain) with important consequences for carbon sequestration. Two land-use types (grassland–shrubland and cereal crops) accounted for more than 80% of the study area in the first half of the century, reflecting a subsistence economy based upon 1) extensive sheep and goat grazing in pastureland (Barroso and Lázaro, 1999) and, more notably, 2) extensive dryland farming of cereals in terrace and lowland terrain (Ortiz Ocaña, 2002). Oak woodland in the first quarter of the century represented as little as 3% of the study area. However, a sizable increase in woodland area occurred towards the middle of the century due to afforestation initiatives that were implemented around that time.

The most important land-use change took place in the second half of the twentieth century. Dryland farming was progressively abandoned as it was elsewhere in the Mediterranean basin (Brandt and Thornes, 1996; Puigdefábregas and Mendizábal, 1998). By the end of the century, cereals crops covered 0.05% of the total surface area in comparison to 35–40% of the surface area before 1947. Moreover, grassland–shrubland was less abundant in 1995 than in the first half of the century, showing a 48% decrease. This was likely due to woodlands being established in abandoned grassland, shrubland, and terraces. Intensive pine plantation initiatives that started mid-century onwards were intended for timber production and the protection against soil erosion (Allue Andrade et al., 1970) and, therefore, took place in unproductive terraces, grassland, and lowland shrubland. Regeneration of Holm oak forests likely took place in shrubland–grassland areas after grazing cessation.

4.2. Carbon sequestration

Changes in land use mirrored potential carbon sequestration. The amount of carbon potentially sequestered in 1995 more than doubled that of 1921, with a net increase of more than 8500 t C $year^{-1}$ towards the end of the twentieth century. Holm Oak and pine forests were the two land-use types that sequestered the most carbon overall (2.7 t $year^{-1}$ ha^{-1} on average), with cereal crops being the lowest. It can therefore be deduced that the modest presence of forests in 1921, when compared to the latter part of the century, and the low sequestration potential of the vast areas of cereal crops were together responsible for the low carbon sequestration rates found in the first half of the previous century. Grasslands and shrublands, despite possessing one of the lowest carbon sequestration rates, accounted for 60% of the fixed carbon within the study area in 1921, mostly due to their dominance at that time. It was during the second half of the century when cereal crops, grasslands, and shrublands were replaced by Holm Oak and pine afforestation initiatives that forests themselves became responsible for the bulk of carbon sequestration (i.e., forests accounted for 23%, 37%, and 73% of carbon sequestration in 1921, 1947, and 1995, respectively).

4.3. Management implications

The quantity of CO_2 sequestered in the area under study would amount via stoichiometry to 28,961 ± 2614 t $year^{-1}$ in 1921, and increased to 34,269 ± 2996 t $year^{-1}$ in 1947 and 60,635 ± 4671 t $year^{-1}$ in 1995. These figures somewhat exceed anthropogenic CO_2 emissions reported for the experimental area at the end of the last century (9790 t $year^{-1}$, Inventario de Emisiones a la Atmósfera 2004, Andalusia Regional Govt.), which outlines the important role the area plays as a CO_2 sink. However, potential CO_2 sequestration of the area calculated for 1995 would be altered if extreme changes occurred in upcoming years. In the most extreme case, CO_2 sequestration would decrease to 20,547 t $year^{-1}$ (i.e., a 66% reduction) if the woodland were totally cut down and replaced by grassland/shrubland, but it would increase to 84,229 t $year^{-1}$ (i.e., a 39% increment) if grasslands and shrublands were converted to forests either through secondary succession or forest restoration. Thus, given the substantial contribution of woodlands to carbon sequestration, their conservation must be encouraged as a means to counter atmospheric CO_2 emissions (FAO, 2006; Bonan, 2008; Canadell and Raupach, 2008).

Intense land exploitation by human activity has notably reduced woodland surface area worldwide (FAO, 2006). This is true for the Mediterranean basin as it is elsewhere (Mota et al., 1996; Bussotti and Ferretti, 1998). In this sense, the restoration of degraded forests is a means to help offset atmospheric CO_2 emissions (Silver et al., 2000; Grunzweig et al., 2007). In our area, despite that woodland surface increased notably in the last century thanks to pine plantations, efforts should now focus on reestablishing holm oak forests more than expanding pine afforestations. Restoration of native holm oak forests can increase CO_2 sequestration while preserving the biodiversity of native Mediterranean forests unlike pine afforestations (Santos et al., 2006), which are not native to the region and are prone to fire (Valle et al., 2003). This way, both carbon sequestration and biodiversity conservation would be included in the managerial strategy of this rural area, thus ensuring maintenance of ecosystem services related to native forests.

4.4. Value of carbon sequestration

Society and markets have rarely appreciated the value underlying ecosystem services (Costanza et al., 1997). However, some appraisal strategies can ensure proper ecosystem service maintenance, rural life sustainability, and biodiversity conservation (Plummer, 2009). In the area under study, estimating the value underlying carbon sequestration may reinforce arguments in favor of forest conservation as well as contributing to global sustainability. One way to estimate the economic value underlying carbon sequestration is based on the CO_2 stock exchange (Sandor et al., 2002; Scott et al., 2004). One ton of CO_2 is quoted at € 13.09 in the European Union Emission Trading (averaged monthly value for the year 2009). The economic value of carbon sequestration in the area under study would, therefore, be 793,718 € $year^{-1}$ in 1995. In the most extreme cases, the economic value of CO_2 sequestration would decrease to 215,160 € $year^{-1}$ if the woodland were totally cut down and converted to shrubland–grassland, but it would increase to 1,102,558 € $year^{-1}$ if the shrubland–grassland were converted back to forests. These latter amounts may be considerable in contributing to ecosystem service maintenance and woodland conservation if reverted back to local municipalities as payment for environmental services, i.e., subsidizes and incentives to the local society and stakeholders to preserve the services local ecosystems provide (Engel et al., 2008; Fisher et al., 2008; Turpie et al., 2008).

4.5. Uncertainty and sources of errors

As we based our assessment on data found elsewhere, the lack of monitoring sites in our study area makes our quantification inherently coarse. However, it is worth noting that our approach is meaningful in relative terms, as it allows comparing carbon sequestration trends associated to land-use changes. This information may be applicable at the regional level, even if there are no monitoring sites. Moreover, we provided the most reliable estimations by using, mostly, averaged NEE rates that took into account carbon emissions due to autotrophic and heterotrophic respiration, and by considering as detailed land-use types as possible.

The largest error likely relies on almond and olive orchards, where we had to estimate NEE from NPP and almond production, and scaled data to our plant density. Errors associated to other estimations are presumably lower since stand characteristics and latitude of literature roughly matched ours. The carbon sequestration in forests is strongly age dependent (Schulp et al., 2008), yet we have no means to date our stands in 1947. Pine cultures initiated around the middle of the century, but no exact years are known, so it is possible that forest were young in 1947, therefore we could have overestimated carbon sequestration of the *ca.* 1200 ha of woodland in 1947. Carbon stored in biomass and soils were not investigated here either because of the lack of data, yet we are ware that these two compartments are of great importance for carbon balance.

Overall, despite these uncertainties, the exercise shown here (1) makes more evident the value of services provided by ecosystems; (2) establishes at least a first approach to the relative magnitude of these services; and (3) stimulates further research (Costanza et al., 1997).

5. Conclusion

Here we show that a woody area located in SE Spain acts as a carbon sink that captures more CO_2 than it releases into the atmosphere. Agricultural abandonment and forest restoration that took place in the 20th century more than doubled the carbon sequestration potential seen at the beginning of the century. In this sense, woodland conservation is essential to maintain the ecosystem services that underlie carbon sequestration. Payments for such services may restore the underlying economic value back to the local community, thus contributing to conservation while achieving rural sustainability.

This assessment can help policymakers in rural municipalities and protected areas to make better informed decisions regarding land-use changes in which goals of higher carbon sequestration rates and sustainability can be achieved. The value of services provided by the ecosystem becomes more evident through this exercise, which constitutes a first approach to understand the relative magnitude of these services.

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