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ScienceDirect
Forest Ecology and Management 254 (2008) 156–165
Forest Ecology and Management
www.elsevier.com/locate/foreco

Soil surface properties in Mediterranean mountain ecosystems: Effects of environmental factors and implications of management

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Received 23 January 2007; received in revised form 23 July 2007; accepted 31 July 2007

Abstract

Understanding soil processes is fundamental to the success of forest restoration programs. We compared different types of soils in Mediterranean mountain forests with respect to their edaphic environments and influence of vegetation cover and lithology. We then used this information to determine the suitability of current forest restoration programs in these ecosystems.

Twenty-four surface horizons in forest soils in two zones of contrasting lithology (calcareous and metamorphic) and different types of vegetation cover or management (shrubland, autochthonous forest and reforested forest) were sampled. A set of their essential soil properties were analysed and a series of parameters considered as indicators of surface soil processes was selected: aggregate size, structural stability, water repellency, mineralisation rate and fungal activity.

Results confirm that the lithological origins of soils determines the properties defined by the geochemical environment of soilscapes (texture, pH, exchange complex and free oxides), and does not much influence organic properties. On the other hand, the type of plant cover and management do not influence the geochemical properties of the soil decisively, but do maintain a relative control of organic properties, especially those that define their quality (C/N ratio).

The variability of surface properties is not well explained by environmental factors, and it is assumed that a large part may be related to the historical use of the soils.

The specificity of soilscapes implies differences in vulnerability to forest management: the surface horizons in siliceous environments are more vulnerable than calcareous environments. It is necessary to better characterize soil properties in these forests and accordingly re-evaluate forest restoration efforts with respect to them.

Keywords: Soil properties; Soil conservation; Soil ecology; Forest restoration; Ecosystem integrity

1. Introduction

Soil conservation is a priority feature of environmental policies in many countries and international organizations (Commission European, 2006). It is essential to the protection of ecosystem resources and affects a wide range of forest values, and this is reflected in the criteria defined by the Montreal processes that promote a common understanding of the sustainable management of forests (Hopmans et al., 2005).

Criteria for sustainability must consider ecosystem integrity and focus their attention on the conservation of soil functions. This is especially important in fragile ecosystems such as Mediterranean mountains (with unique environments due to complex topography, climate and historical use), where there is a high risk of intensifying desertification processes.

Some authors warn of a perceived gap between forest managers and soil ecologist and suggest a difference in perspective on soil recovery processes (Johnston and Crossley, 2002). From this perspective, it is important to identify soil functions that must be preserved in forest systems, such as the hydrological, carbon or nutrient cycles. In order to perform these functions, certain identifiable attributes must be maintained. To evaluate and, if necessary, monitor these attributes, parameters that can be used as indicators of soil conditions must be selected (Ramakrishna and Davidson, 1998).

The upper mineral soil horizons are the greatest ecosystem reservoir of organic matter and nutrients, and they influence or regulate most of the functional processes occurring throughout

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0378-1127/$ – see front matter © 2007 Elsevier B.V. All rights reserved.
doi:10.1016/j.foreco.2007.07.034

the ecosystem, such as nutrient cycling, moisture retention and erosion protection (Hopmans et al., 2005). Among the soil surface parameters that may be used as indicators, structural stability is one of the main factors controlling topsoil hydrology, crustability and erodibility (De Ploey, 1985). When soil structure breaks down, particle detachment and runoff are increased as infiltration and water retention are reduced. Poor soil structure, small aggregate size and low stability enhance surface sealing, reducing the infiltration rate and increasing the potential for soil erosion (Sarah, 2005). Soil aggregation, important in land degradation studies, is also a good indicator of ecosystem vulnerability (Cammeraat and Imeson, 1998).

Soil organic matter, its decomposition and the mineralisation of nutrients bound to this matter are important to forest ecosystem functioning. They contribute significantly to physical, chemical, hydrological and biological soil properties and these life-supporting processes are largely regulated by biological soil activity (Setälä et al., 2000). Therefore, organic matter is likely to be suitable as a surrogate indicator of the fertility of forest soils (Hopmans et al., 2005).

The hydrophobic behaviour (water repellency) of the uppermost mineral soil becomes a critical factor in water infiltration capacity, overland flow and soil loss (Sevink et al., 1989). The infiltration rate is diminished by natural water-repellent surfactants coating soil particles (Morley et al., 2005), components which under natural conditions are related to certain plant communities.

Thus, any action taken for soil recovery or ecosystem sustainability must consider the need for conserving soil functions, and include a set of parameters for their interpretation. Selection of parameters must not be considered as universal. Like ecosystems, soils are highly variable (diversity) in both components and functions. These variations are related to different environmental factors, such as topography, regional climate gradients, lithography and management. Different combinations of these factors give rise to different soilscapes and, above all, differences in the factors and attributes that control their basic functions (Amundson et al., 1994).

The restoration of plant cover by introducing forest species is one of the most widespread soil protection measures (Maestre and Cortina, 2004). But reforestation practices cause complex perturbations in the soil and impede, at least temporarily, the continuation of soil functions (Ballard, 2000). The intensity and duration of such perturbations depend on the characteristics of the soil, and soilscapes are vulnerable to forest practices to different extents (Francaviglia et al., 2004). In this work, we have distinguished Mediterranean mountain forest soilscapes which have different functions and differ in their vulnerability to forest management. We have assumed, under similar climatic conditions, the environmental factors that control soil properties and therefore, the main processes that take place in them, are lithology and the type of plant cover. These differences would have important implications for forest management.

Under these premises, the purpose of the work is, in the first place, to analyse the edaphic environment of two large Mediterranean mountain soilscapes, as a function of variables selected as indicators of surface processes in forest ecosystems. We also determine the control that the two factors (type of plant cover and lithology) exert on these attributes in order to better predict how soils function under different environmental conditions. Finally, an attempt was made to determine the effect that past action (40 years) taken in forests has had on soil attributes and properties, and the differences in vulnerability to forest management of the soilscapes.

2. Materials and methods

Twenty-four surface horizons were sampled (Ah morphological horizons) corresponding to mountain forest soil in the SE Iberian Peninsula (Fig. 1). For the selection of sampling sites, the primary criteria were lithological, assuming that contrasting edaphic environments are to be found on different kinds of lithological materials. The general criterion for site selection was to choose areas in good condition and with no appreciable perturbation. To find these unaltered areas, sampling was done in protected spaces that ensured controlled

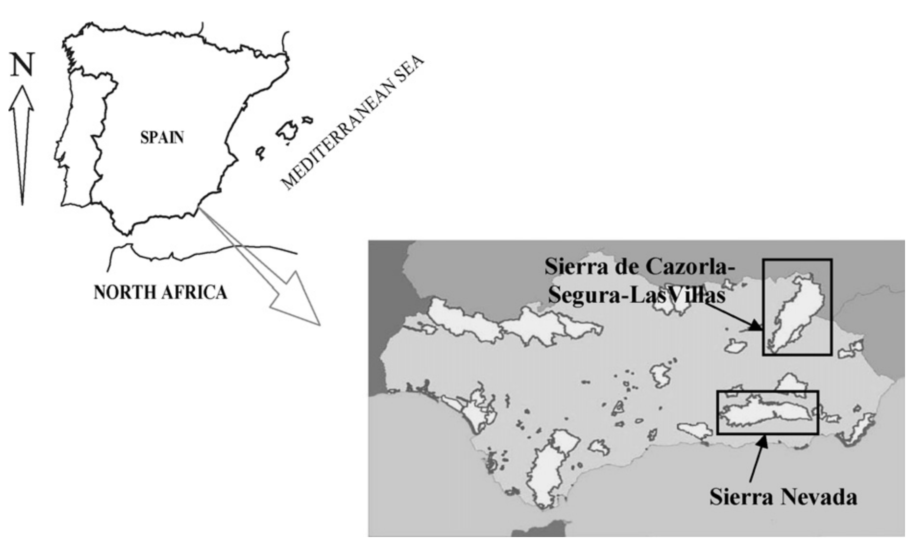

Fig. 1. Location of study area, Sierra Nevada National Park (samples SN) and Sierra de Cazorla-Segura-Las Villas Natural Park (samples SC).

management. It may therefore be pointed out that there have been no forest fires, nor harvesting, nor overgrazing, nor obvious signs of erosion beyond moderate laminar processes. Although, regarding site preparation in pine reforestation, even today, modification of the topography in preparing the terrain is clearly visible (terraced landscape).

Two protected areas were chosen for sampling: (1) the Sierra Nevada National Park, with a predominantly metamorphic lithology, and (2) the Sierra de Cazorla-Segura-Las Villas Natural Park, where carbonate sedimentary rock is predominant (Table 1). Later sampling locations were chosen within each sampling zone based on the vegetation/management type: native forests [NF: *Quercus* sp.], monodominant pine reforestation with trees about 40 years old [RF: *Pinus* sp.], and high-diversity shrublands [SH: different species of *Genista*, *Juniperus*, *Adenocarpus*, *Crataegus*, *Retama*, *Berberis*, *Ulex*, *Cistus* and *Erinacea*].

The climate in both zones is typically Mediterranean, with strong seasonal contrasts, xeric soil moisture and mesic soil temperature regimes (Soil Survey Staff, 1998). Mean annual rainfall is about 866 mm in the SC and 595 mm in the SN. The mean annual temperature is about 11.1 °C in the SC and 9.8 °C in the SN. The most relevant field data are shown in Table 1.

At each site selected, complete profiles were sampled and described. Subsampling was done for surface horizons, taking five samples distributed randomly in a 10 m^2 area around the location of the profile described. These field samples were mixed to form a single composite or bulk sample, which were then analysed in the laboratory (Petersen and Calvin, 1986).

Composite samples were passed through a 2 mm sieve and labelled the fine-earth fraction. The Methods of Soil Analysis of the American Society of Agronomy and Soil Science Society of America (Page et al., 1982; Klute, 1986) were followed for analyses. Particle-size distribution was determined by the pipette method after removal of organic matter with H_2O_2 and dispersion with Na-hexametaphosphate. Organic carbon content was determined by the Tyurin method using wet combustion with a mixture of $K_2Cr_2O_7$ and H_2SO_4 and titrating residual dichromate with ferrous sulphate. Organic nitrogen was mineralized with H_2SO_4 and selenium to NH_4SO_4, distilled in the form of NH_4OH and titrated with diluted H_2SO_4 by the Kjeldhal method. The pH (1:1 fine-earth:water suspension) was measured by the potentiometric method. Exchangeable bases

Table 1
Climatic parameters, soil type, vegetation, and parent material of the Mediterranean mountain soils studied

Soil	Altitude (m)	MAR[a] (mm)	MAT[b] (°C)	Soil type[c]	Vegetation type	Parent material
Sierra Nevada National Park						
SN-1	1800	603	9.6	Typic Haploxeroll	Quercus forest (*Quercus pyrenaica*)	Micaschists
SN-2	1850	614	9.3	Lithic Xerochrept	Pinus forest (*Pinus sylvestris*)	Micaschists
SN-3	1500	533	11.5	Dystric Xerochrept	Quercus forest (*Quercus pyrenaica*)	Micaschists
SN-4	2700	813	3.8	Lithic Cryumbrept	Shrubland (*Genista versicolor*, *Juniperus hemisphaerica*)	Micaschists
SN-5	1600	557	10.9	Dystric Xerochrept	Shrubland (*Ulex sp.*, *Cistus sp.*)	Micaschists
SN-6	1500	534	11.5	Typic Xerochrept	Quercus forest (*Quercus rotundifolia*)	Micaschists
SN-7	1320	492	12.6	Typic Xerochrept	Pinus forest (*Pinus sylvestris*)	Micaschists
SN-8	1800	603	9.6	Dystric Xerochrept	Shrubland (*Adenocarpus decorticans*, *Genista sp.*)	Micaschists
SN-9	2000	650	8.3	Dystric Xerochrept	Pinus forest (*Pinus sylvestris*)	Micaschists
SN-10	1650	568	10.5	Typic Xerochrept	Pinus forest (*Pinus sylvestris*)	Micaschists
SN-11	1820	608	9.5	Dystric Xerochrep	Shrubland (*Genista versicolor*, *Adenocarpus decorticans*)	Micaschists
SN-12	1620	561	10.7	Entic Ultic Haploxeroll	Quercus forest (*Quercus rotundifolia*)	Micaschists
Sierra de Cazorla-Segura-Las Villas Natural Park						
SC-1	1100	971	12.6	Lithic Xerorthent	Pinus forest (*Pinus pinaster*)	Limestones
SC-2	1400	1128	10.5	Ultic Haploxeroll	Quercus forest (*Quercus faginea*)	Calcareous sandstones
SC-3	1250	1050	11.5	Lithic Haploxeroll	Quercus forest (*Quercus rotundifolia*)	Limestones
SC-4	1650	1260	8.8	Lithic Xerochrept	Shrubland (*Erinacea anthyllis*, *Crataegus monogyna*)	Limestones
SC-5	1500	1181	9.8	Lithic Haploxeroll	Shrubland (*Ulex sp.*, *Retama sphaerocarpa*, *Thymus sp.*)	Calcareous Micaschists
SC-6	1150	997	12.2	Typic Xerochrept	Pinus forest (*Pinus pinaster*)	Marls
SC-7	700	760	15.4	Typic Xerochrept	Pinus forest (*Pinus pinaster*)	Clays
SC-8	1590	634	9.9	Lithic Xerochrept	Quercus forest (*Quercus rotundifolia*)	Dolomites
SC-9	1780	558	8.7	Lithic Haploxeroll	Shrubland (*Erinacea anthyllis*, *Berberis hispanica*)	Limestones
SC-10	1550	756	10.1	Typic Haploxeroll	Pinus forest (*Pinus pinaster*)	Dolomites
SC-11	1200	459	12.4	Lithic Xerochrept	Shrubland (*Juniperus thurifera*, *Ulex sp.*)	Limestones
SC-12	1410	634	11.0	Lithic Argixeroll	Quercus forest (*Quercus rotundifolia*)	Limestones

[a] MAR: mean annual rainfall.
[b] MAT: mean annual temperature.
[c] Soil Survey Staff, USDA (1998).

(Ca^{2+}, K^+ and Mg^{2+}) were extracted with 1N NH_4-acetate (pH 7) and determined by atomic absorption spectroscopy (AAS) and flame photometry. Cation exchange capacity (CEC) was determined with 1N Na-acetate at pH 8.2, and base saturation (V) using NH_4 and Na displacement solutions. Free oxides from iron and aluminium (Fe_d and Al_d) were extracted with citrate-dithionite by the Holmgren method and determined by AAS.

Samples also underwent organic matter sequential fractionation procedure (Duchaufour and Jacquin, 1975): particulate organic matter (POM) was removed by flotation in 1 M H_3PO_4 and centrifugation. Soil residue was subjected to successive extractions with 0.1 M $Na_4P_2O_7$ and 0.1 M NaOH. The total humic extract obtained was precipitated with HCl (pH 1) to separate insoluble humic acids (HA) from the soluble fulvic acids (FA), and the corresponding HA/FA ratio was found. The amounts of carbon in the above fractions were quantified by wet combustion (Page et al., 1982).

A Shimadzu UV-240 spectrophotometer was used to find the E_4/E_6 ratio by visible spectroscopy (absorbance at 465 nm and 665 nm of the visible spectra) and fungal activity (FAC) by derivatographic spectrometry, both in 0.2 mg C mL^{-1} solutions of HA in 0.1 M $NaHCO_3$ (Kononova, 1961).

Aggregate size distribution was determined by dry sieving of the fine-earth fraction, and mean weight diameter (MWD) was calculated as the sum of the products of (i) the mean diameter, x_i, of each size fraction and (ii) the proportion of the total sample weight, w_i, in the corresponding size fraction, where the summation is carried out over all n size fractions (Van Bavel, 1949). Structural stability (STA) was measured in dry aggregates. Resistance of aggregates to abrasion was determined by pouring weighed aggregates back into the dry pan, sliding them off the pan back into the feed bin of the rotatory, sieving and weighing again, and determining the changes in aggregate size distribution (Chepil, 1962). Low values indicate little change and, therefore, more aggregate stability. Water repellency–water drop penetration time (WDPT) was determined by timing how long it takes a drop of water to be absorbed by the soil (Savage et al., 1972), and the sample hydrophobicity was estimated.

Biodegradability of organic matter was studied by incubating whole soil samples. In vitro respiratory activity of soils was determined by estimating CO_2 released from soil samples moistened to 60% of their water holding capacity at 24 ± 1 °C, measured over a period of 40 days with a Carmhograph-12 gas analyzer (Almendros et al., 1990). Results are given in terms of total mineralisation coefficients (TMC), defined as the percentage of total C released during the incubation period.

Normality of the data was checked and the variables that did not meet this premise (WDPT, clay and Al_d) were logarithmically transformed. Means shown in the tables are for the original data, while transformed data were used for statistics. The method used to discriminate among the means was Fisher's least significant difference (LSD) test. A multifactor analysis of variance (multifactor ANOVA) was used to verify the influence of the environmental factors considered (parent material and vegetation type). Principal component (PCA) and correlation analyses were performed to find the relationship between the variables analysed. All statistical analyses were performed with the Statgraphics Plus v.4.1 software program (STSC, 1999).

3. Results and discussion

The variables analysed are grouped into three blocks, by whether they characterize (a) the geochemical environment, (b) the organic fraction, or (c) surface properties considered indicators of soil conditions exerting control on the outstanding surface processes: mean weight diameter (MWD), structural stability (STA), hydrophobicity (WDPT), total mineralisation coefficient (TMC), and fungal activity (FAC). The results are summarized in Table 2, where an LSD test is included to check for differences among the groups set up in the experimental design.

For this design, it was assumed that in Mediterranean environments, differences in the nature of the geological substrate would involve different edaphic environments. The results corroborate this, according to the properties found in horizons developing on metamorphic and calcareous materials are different. These differences are mainly in physico-chemical properties, and are less evident in the organic fraction and surface conditions.

Another assumption was plant formation would also condition the properties of surface horizons. The results show that this factor, observed in organic fraction properties and surface conditions, is less influential, and negligible in those that define the geochemical environment.

A PCA performed by blocks of variables demonstrates the existence of different edaphic environments and enables the soil properties in each to be characterized (Fig. 2). When the geochemical variables are considered, two factors are generated as responsible for 74% of the variance (factor 1:56%; factor 2:18%), and generating cluster data coinciding largely with their sampling zone. The axis of factor 1 shows that on calcareous substrates (SC) the surface horizons are more clayey, have a higher Ca^{2+} and K^+ content, and higher CEC, than on metamorphic substrates (SN). Factor 2 marks a more basic pH in the calcareous population and greater saturation, while higher content in iron and aluminium is found in soils on metamorphic substrates.

The influence of the parent material on soil properties has been described by other authors (i.e., Kooijman et al., 2005) who have found differences between lithologies that affect both physico-chemical properties (texture, pH and electrical conductivity), and organic properties (OC, C/N, fractionation of organic matter), and emphasize their importance, suggesting that the original material may modify local climatic conditions and, through their influence on soil properties, condition such important factors as water and nutrient availability. In this way they affect the productivity and functioning of ecosystems (Kooijman et al., 2005). However, our results did not show complete identification between type of lithological substrate and geochemical environment: three of the samples on a lime substrate appear together with the metamorphic substrate (Fig. 2). The three coincide with the highest precipitation

Table 2
Analytical data for the surface horizons of the soils studied

	Lithological environment[a]		Vegetation type[b]		
	SN	SC	NF	RF	SH
Samples	12	12	8	8	8
Geochemical properties					
Clay (g 100 g^{-1} soil)	10.3 (3.0) a	24.9 (10.3) b	15.1 (8.5) a	22.0 (15.7) a	16.5 (6.8) a
Sand (g 100 g^{-1} soil)	58.2 (6.4) a	41.2 (15.4) b	55.9 (9.8) a	38.3 (20.3) b	53.0 (6.3) a
pH	6.3 (0.6) a	7.4 (0.7) b	6.7 (0.7) a	7.3 (0.7) a	6.6 (1.0) a
Ca^{2+} (c mol c Kg^{-1})[c]	6.2 (4.6) a	31.7 (15.2) b	18.3 (12.8) a	21.8 (19.5) a	17.3 (19.8) a
Mg^{2+} (c mol c Kg^{-1})[c]	1.4 (0.8) a	2.9 (2.2) b	2.2 (1.0) a,b	3.4 (2.8) a	1.2 (0.6) b
K^{+} (c mol c Kg^{-1})[c]	0.3 (0.2) a	0.7 (0.4) b	0.5 (0.4) a	0.6 (0.5) a	0.4 (0.3) a
CEC (c mol c Kg^{-1})[c]	14.9 (5.7) a	24.8 (6.6) b	21.7 (6.9) a	20.4 (11.0) a	17.8 (6.1) a
Base saturation (*V*) (%)	53.9 (22.5) a	92.3 (13.8) b	80.2 (17.0) a	81.7 (19.9) a	60.2 (34.9) a
Free iron (Fe_d) (%)	3.1 (0.7) a	2.0 (1.1) b	2.3 (0.8) a	2.9 (1.2) a	2.4 (1.1) a
Free aluminium (Al_d) (%)	0.26 (0.13) a	0.18 (0.10) a	0.19 (0.1) a	0.18 (0.1) a	0.27 (0.2) a
Organic properties[d]					
OC (g 100 g^{-1} soil)	2.6 (1.5) a	3.3 (1.7) a	3.9 (1.6) a	2.5 (1.8) a	2.4 (1.3) a
N (g 100 g^{-1} soil)	0.19 (0.11) a	0.25 (0.11) a	0.29 (0.11) a	0.16 (0.09) b	0.21 (0.10) a,b
C/N	13.4 (3.1) a	13.0 (2.7) a	13.6 (1.5) a,b	15.0 (2.9) a	11.4 (2.9) b
POM [g C (100 g soil C)$^{-1}$]	7.5 (5.5) a	1.9 (1.0) b	4.3 (4.1) a	5.6 (7.2) a	4.4 (3.5) a
HA/FA	0.8 (0.3) a	0.7 (0.3) a	0.8 (0.4) a	0.7 (0.3) a	0.8 (0.4) a
E_4/E_6	4.6 (0.8) a	4.5 (0.7) a	4.5 (0.8) a	4.9 (1.1) a	4.3 (0.4) a
Surface properties[e]					
MWD	63.5 (15.9) a	71.8 (17.1) a	63.2 (14.2) a	76.0 (11.2) a	60.4 (19.6) a
STA	11.6 (6.0) a	8.6 (4.4) a	9.2 (4.4) a	10.8 (6.7) a	10.4 (5.6) a
WDPT (s)	2.4 (2.4) a	2.0 (3.0) a	4.3 (3.6) a	1.6 (1.2) a,b	0.7 (0.3) b
TMC [mg C (100 g soil C)$^{-1}$]	1.36 (0.69) a	1.19 (0.39) a	1.24 (0.58) a,b	1.65 (0.51) a	1.01 (0.44) b
FAC (Au)	0.013 (0.01) a	0.021 (0.01) a	0.022 (0.02) a	0.009 (0.01) b	0.020 (0.01) b

Includes means (standard deviation), and differences between groups (LSD test) significant to 95%.
[a] Lithological sets: SN: Sierra Nevada (metamorphic materials); SC: Sierra de Cazorla (calcareous materials).
[b] Vegetation types: NF: native forest; RF: reforested forest *(Pinus sp.)*; SH: shrubland.
[c] Exchange bases and cation exchange capacity.
[d] OC: total organic carbon; N: total organic nitrogen; POM: particulate organic matter; HA/FA: humic and fulvic acids ratio; E_4/E_6: visible spectroscopy ratio.
[e] MWD: mean weight diameter; STA: structural stability; WDPT: water drop penetration time; TMC: total mineralisation coefficient (40 days); FAC: fungal activity.

location (1000–1300 mm $year^{-1}$), and therefore they are washed with greater intensity, which would explain the drop in pH, loss of cations, and tendency to desaturation of the exchange complex, processes that justify their convergence with metamorphic soils.

The same PCA does not show clustering of type of plant cover for the geochemical variables, indicating that vegetation has a low influence on these types of variables.

The PCA performed with the set of organic fraction variables (Fig. 2) generates a first factor (41% of variance explained) which includes total organic carbon (OC), total nitrogen (N) and particulate organic matter (POM) as the components with the most weight. In the second factor (27% of variance) the C/N and E_4/E_6 ratios stand out. The lithological factor does not cause any clustering of samples, which are all widely dispersed. Although several authors have stated that geochemical conditions (pH, fertility, forms of iron/aluminium) influence formation of the type of humus (Oyonarte et al., 1994), in this study, the differences in geochemical environments do not seem to condition the soil carbon balances, nor do they direct their humification processes.

The type of plant cover does not show obvious clusters either, but interesting tendencies can be observed. In the horizons under native forest, the organic fraction is larger (higher OC and N content) and is better integrated with the mineral fraction (smaller POM fraction) than under reforested forests, according to its position with regard to the factor 1 axis. The samples from under shrubs have the most highly evolved organic component (low C/N and E_4/E_6 ratios with regard to factor 2), especially if we compare them to the samples under reforested forest.

3.1. Surface conditions: influence of environmental factors and relationship to other edaphic properties

Mean WDPT values (Table 2) are no longer than 5s, indicating absence of water repellency in soil (DeBano, 1981). The geochemical environment does not influence this parameter, although the type of vegetation does. The almost immediate absorption of water in horizons under shrubs, and the increase in absorption time of samples under native vegetation are noticeable.

These slightly hydrophobic conditions have been described in calcareous Mediterranean environments (Mataix-Solera and Doerr, 2004), although they contradict other studies that find strong water repellency under shrubs and pines in

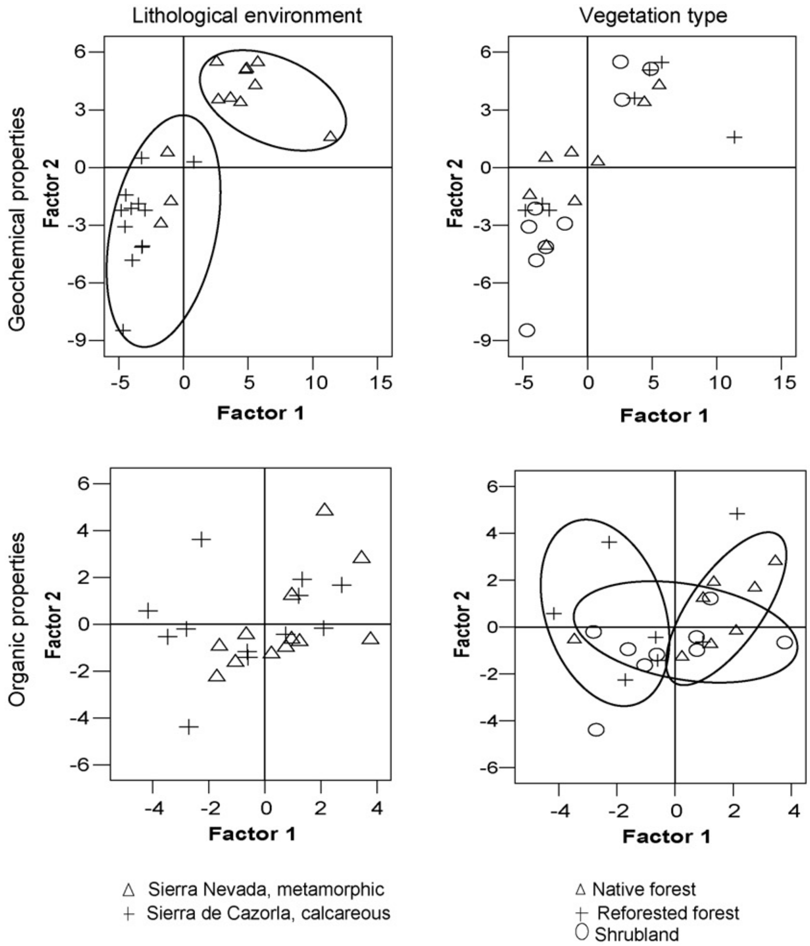

Table of Component Weights

	Factor 1	Factor 2
Sand	-0.8452	-0.1135
Clay	**0.8580**	0.1861
Ca^{2+}	**0.7628**	0.4956
K^{+}	**0.8440**	0.2129
Mg^{2+}	0.3423	**0.5216**
CEC	**0.8954**	0.1000
V	0.4957	**0.7932**
Fe_d	0.0181	**-0.7825**
Al_d	-0.0431	**-0.9086**
pH	0.4126	**0.8316**
Var. explained (%)	56.0	18.5

Table of Component Weights

	Factor 1	Factor 2
OC	**0.7830**	**0.5733**
POM	**-0.7193**	0.3376
AH/AF	0.4962	-0.0936
E_4/E_6	-0.1289	**0.8611**
N	**0.8554**	0.3299
C/N	0.0782	**0.7987**
Var. explained (%)	40.7	27.2

Fig. 2. PCA of physico-chemical and organic variables in surface horizons. 2D scatter plot of PCA and component weights, graphic of the classification of samples by lithology and vegetation type.

Mediterranean climates (Doerr et al., 1998). Cerdá and Doerr (2005) explain these differences based on soil acidity, and point out the alkalinity of calcareous soils as responsible for the low hydrophobicity. However, our results demonstrate that the geochemical conditions do not affect hydrophobicity of soil as no significant differences between soil groups were found (Table 2), nor were any correlation between WDPT and other edaphic properties such as pH or *V* (Tables 3 and 4).

This apparent contradiction could be due to the influence of these factors not being gradual and progressive, but only appearing when certain thresholds are surpassed. In our case, the mean pH, or degree of saturation, nears neutral (pH: 6.3–7.4) and the medium–high saturation base (*V*: 53.9–92.3%), explaining its similar hydrophobicity.

The influence of the type of vegetation on soil hydrophobicity is one of the factors most studied (Dekker et al., 2005). Our results confirm this relationship and show that native vegetation comprised of *Quercus* sp. is more repellent to water than reforestations of *Pinus* sp., and much more than repellency under shrubs (Table 2).

The spectroscopic behaviour of humic acids (HA), described by the presence of diagnostic valleys in the second derivative visible spectra, are related to the nature and processes of the organic soil fraction. Such a spectral pattern is typical of what is called P-type HA (Kumada and Hurst, 1967), characterized by the presence of microbial perylenequinonic pigments produced by certain species of soil microfungi (Almendros et al., 1985). The amount of fungal-derived pigments in the soil, labelled fungal activity (FAC), could be a valid indicator of the impact of land use on the structure of the soil microbial system and the mechanisms responsible for accumulation of HA.

The highest FAC are in calcareous environments (Table 2), showing their high variability (high standard deviation). The effect of the type of vegetation is seen in a significant decrease of fungic activity in horizons under reforested forest compared to natural forest and shrubs which have similar values. The presence of fungal-derived pigments in forest soils has been described in a wide variety of environments, from Mediterranean mountains similar to the subject of this study (Oyonarte et al., 1994) to semiarid environments (Aranda and Oyonarte, 2005) and temperate zones (Zancada et al., 2003), and tend to disappear in altered soils, which could be due to significant changes in the structure of microflora in the soil (Almendros et al., 2005). Thus, this decrease of fungic activity observed in soils under reforestation of *Pinus* sp. would indicate a negative impact of such management on the biological conditions of the soil, particularly in the structure of its microbial population.

Table 3
Coefficients correlation matrix between parameters defining surface conditions and edaphic variables in the SC population (calcareous environment, 12 samples)

	MWD	STA	WDPT	TCM	FAC
STA	0.6013*				
WDPT	−0.1919	0.0066			
TMC	0.4243	0.1525	0.5914*		
FAC	−0.3658	0.0381	0.1907	−0.5057	
Clay	0.5398	0.6636**	0.3112	0.6556*	−0.2332
Sand	−0.5172	−0.6885**	−0.1256	−0.4358	0.3206
Ca^{2+}	0.6488*	0.7578**	−0.2769	−0.0833	0.1897
Mg^{2+}	0.2194	0.3141	−0.0597	−0.0167	−0.2150
K^{+}	0.5889*	0.5656*	0.1219	0.3429	−0.0469
CEC	0.3979	0.6172*	0.2883	0.2519	0.2377
V	0.5827*	0.6084*	−0.3250	−0.1321	0.0328
Fe_d	0.2543	0.0807	0.1266	0.3534	−0.2168
Al_d	−0.1775	−0.4230	0.0162	0.1265	0.0026
pH	0.5021	0.5148	−0.4434	−0.1003	−0.1815
OC	0.0659	0.3824	0.2142	−0.2416	0.6891**
N	−0.2174	0.3061	0.1426	−0.5078	0.8908***
C/N	0.5351	0.2899	0.0448	0.2581	−0.1053
POM	−0.1250	0.0996	0.2522	−0.2395	0.5148
HA/FA	−0.3341	0.0377	−0.2699	−0.5605*	0.4714
E_4/E_6	0.4192	0.5677*	0.2331	0.3238	0.0622

Parameter abbreviations from Table 2.
* $P < 0.05$.
** $P < 0.01$.
*** $P < 0.001$.

Table 4
Coefficients correlation matrix between parameters defining surface conditions and edaphic variables in the SN population (metamorphic environment, 12 samples)

	MWD	STA	WDPT	TCM	FAC
STA	−0.6794*				
WDPT	−0.0580	0.4352			
TCM	0.6629**	−0.4548	0.0211		
FAC	−0.3896	0.4889	0.2018	−0.6709**	
Clay	−0.4800	0.1850	−0.1520	−0.4919	0.4798
Sand	0.6510*	−0.0659	0.1659	0.7241**	−0.3051
Ca^{2+}	0.2101	0.2298	0.6597*	−0.0361	0.3665
Mg^{2+}	0.2876	0.2609	0.5015	0.1422	0.2011
K^{+}	0.1406	0.1421	0.6641*	0.0588	0.2356
CEC	−0.3074	0.5641*	0.4801	−0.6202*	0.8502***
V	0.6245*	−0.1216	0.4263	0.5444	−0.274
Fe_d	−0.1907	−0.0957	−0.2756	−0.3509	−0.2366
Al_d	−0.7294**	0.3414	−0.1723	−0.7796**	0.6073*
pH	0.8223***	−0.4710	0.2440	0.6087*	−0.4051
OC	−0.2245	0.5543*	0.5455	−0.5462	0.7817**
N	−0.2863	0.5669*	0.5328	−0.5702*	0.8091**
C/N	0.2300	0.0526	0.2080	0.0246	0.1430
POM	0.4061	−0.2144	0.0050	0.8760***	−0.5709*
HA/FA	0.3552	−0.3131	−0.1414	−0.3454	0.2743
E_4/E_6	0.5502	−0.0852	0.2940	0.3976	−0.0144

Parameter abbreviations from Table 2.
* $P < 0.05$.
** $P < 0.01$.
*** $P < 0.001$.

The main differences found in the total mineralisation coefficient (TMC) also have to do with the type of vegetation (Table 2). Soils under shrubs communities have the lowest mean coefficient of mineralisation, and significantly lower than under *Pinus* sp. reforestation.

The TMC, interpreted as a risk indicator of soil organic carbon loss, allows establishing conditions of ecosystem fragility. Martins et al. (1991) also suggested that decline in OC is primarily due to rapid mineralisation of the coarse fraction of organic matter. This is especially true in our results, as we have established a very significant relationship between TMC and POM in the metamorphic environment (Table 4). A reverse relationship between the TMC and fungal activity would seem a logical indicator of soil stability, and in fact such behaviour is confirmed in the correlation analyses performed (Tables 3 and 4).

Soil structure results show that the mean size (MWD) and stability (STA) of the aggregates is somewhat better in calcareous environments, although the differences are not significant (Table 2). STA indicates the percentage of change in soil aggregates after perturbation, so a low STA should be interpreted as showing more resistance to change and, therefore, greater stability.

In the communities studied, it was found that, both the size and stability of structures are mainly conditioned by the geochemical variables and less by organic variables (Tables 3 and 4). The mechanisms that contribute to the structural stability of soil are diverse and complex, and one or the other prevails depending on a combination of factors as the geochemical environmental characteristics, the amount and the kind of organic matter, or the biological activity (Bronick and Lal, 2005). The coefficients correlation matrix found show that the factors that contribute to the formation and stability of structures are different for different geochemical environments. Stability in SC (Table 3) is strongly related to textural variables (clay and sand), exchange complex and pH, and the E_4/E_6 ratio as the only organic variable. On the other hand, SN (Table 4) is conditioned by OC and N, associated with the organic fraction, and the only geochemical variable it is related to is the CEC.

It has been pointed out that the type of aggregation is extremely important for soil management. Thus, Oades (1984) states that in those soils where organic matter is the main cementing agent, such as in SN, macroaggregation, which is much more vulnerable to management, is favoured. This is reflected in the SN correlation matrix data (Table 4), where an inverse relationship is observed between aggregate size and STA. On the contrary, this relationship disappears in SC (Table 3) where textural and chemical factors favour microaggregation. This is consistent with the mineralisation results, where a relationship is observed in aggregate size, stability and the coefficients of mineralisation of organic matter (Table 4). All of the above leads us to affirm that Mediterranean edaphic environments on metamorphic materials have fragile and less stable structures, making them more vulnerable to any perturbation or change in use.

Furthermore, a multifactor ANOVA was performed to determine the control that the environmental factors considered in the experimental design (geochemical environment and type

Table 5
Influence of the parent material and vegetation factors on analysed soil surface variables

	Geochemical factor		Vegetation factor		Interactions		Residual
	rvar	F	rvar	F	rvar	F	rvar
MWD	6.6	1.6	12.0	1.1	7.4	0.4	74.0
STA	38.4	13.8***	4.6	0.8	7.0	1.3	49.9
WDPT	2.7	0.9	38.5	6.2**	2.7	0.4	56.1
TMC	6.7	1.8	24.9	3.4*	2.6	0.4	65.8
FAC	17.8	6.3*	25.7	4.6*	5.8	1.0	50.6

Results of the multifactor ANOVA. rvar: relative variability (variability explained for each factor or its interaction corresponding to the percentage of the sum of squares of each factor with regard to the total sum of squares of the analysis). F statistic; significant at 95% (*), 99% (**) and 99.9% (***) confidence levels. Residual: variation component associated with other sources of variation.

of plant cover/management) exert on surface conditions. The results show (Table 5) that geochemical conditions, structural stability, and fungal activity (although not statistically significant) are factors important for controlling aggregate size. Plant cover type has control over soil water repellence, coefficient of mineralisation and fungal activity. The interaction of these factors does not influence the variables and is only seen in the stability of the aggregates. In any case, the effect is weak, and less than what each one of the factors would have alone.

As a prior analysis made with climate variables (mean annual rainfall and mean annual temperature) did not show any relationship with the variables analysed, the results seem to indicate that management and the history of land use may have modified their natural condition. In addition to management, another factor that can contribute to soil conditions is forest fires. This factor has been shown to contribute to modifying soil surface properties in ecosystems, increasing their spatial variability and altering their evolution over time (Cerdá and Doerr, 2005).

3.2. Effect of plant cover type (management) on surface properties

Fig. 3 is a graphic representation of the effect of interaction of environmental factors on soil surface conditions and some geochemical properties based on a multifactor ANOVA. This figure makes it possible to see specific changes introduced by reforestation compared to unmanaged plant cover and evaluate the vulnerability of each environment to each type of management.

It has been demonstrated that reforestation has a positive effect on surface horizon properties such as pH, *V*, and exchange K^+, especially compared to horizons under shrubs. This positive effect is seen in the increased pH and *V* in slightly acid environments, and a clear increase in exchange K^+ in calcareous environments, where this element is often blocked by other cations.

An increase in pH has been described as a result of perturbation in an acid soil environments in reforested forest compared to natural regeneration (Zheng et al., 2005), which might be related to site preparation that alter the original material, releasing cations into the edaphic medium. The effect on exchange cations may also be related to the efficiency of these plant formations in taking up cations from lower horizons and subsequently depositing them at the soil surface. The effect is especially noticeable in low-rainfall areas (Alfredsson et al., 1998) such as the Mediterranean region. In this sense, the behaviour of reforestation would be similar to native forests, and more effective than in shrubs (Fig. 3). The organic fraction observed in horizons under reforested forest does not significantly increase the total organic carbon content in the soil compared to forest or shrubs. Fig. 3 further shows that this parameter is similar in both geochemical environments.

The quality variables of organic matter show strong differences depending on the type of environment that is reforested. The significant increase in the light organic fraction not incorporated into the soil (POM) is only found in samples from metamorphic environments, while in calcareous environments the mean contents are even somewhat lower than in native forests or shrubs. This is probably a result of an increased rate breakdown of organic matter from intensified biological activity.

The C/N ratio behaves similarly, as seen in the strong slope of the line (Fig. 3) between the calcareous and metamorphic environments, showing that these latter environments are less resistant to change, which translates into a significant increase in the C/N ratio in the organic fraction.

The total mineralisation coefficient (TMC) and soil biological activity (FAC) showed the same behaviour, which is consistent with the relationships between the variables described above. In short-term laboratory incubation, the potentially mineralisable carbon values are essentially related to the amount of the labile organic carbon fraction. Higher respiration rates reflecting an accelerated mineralisation of OC, which are related to disturbed soils (Francaviglia et al., 2004). In the long-term, these could cause a loss of OC and less biological activity under reforested forests, especially in partially desaturated environments which have been shown to be more vulnerable.

The effect of reforestation on soil structure is observed in aggregate size and loss of structural stability values (Fig. 3). These two variables behave very differently depending on the geochemical environment. In calcareous media, the horizons under reforested forests are no different from other types of formations, while in metamorphic media, the decrease in aggregate size and loss of stability is very strong under this type of management.

The improvement in soil properties expected from reforestation has been described in many publications. Zheng

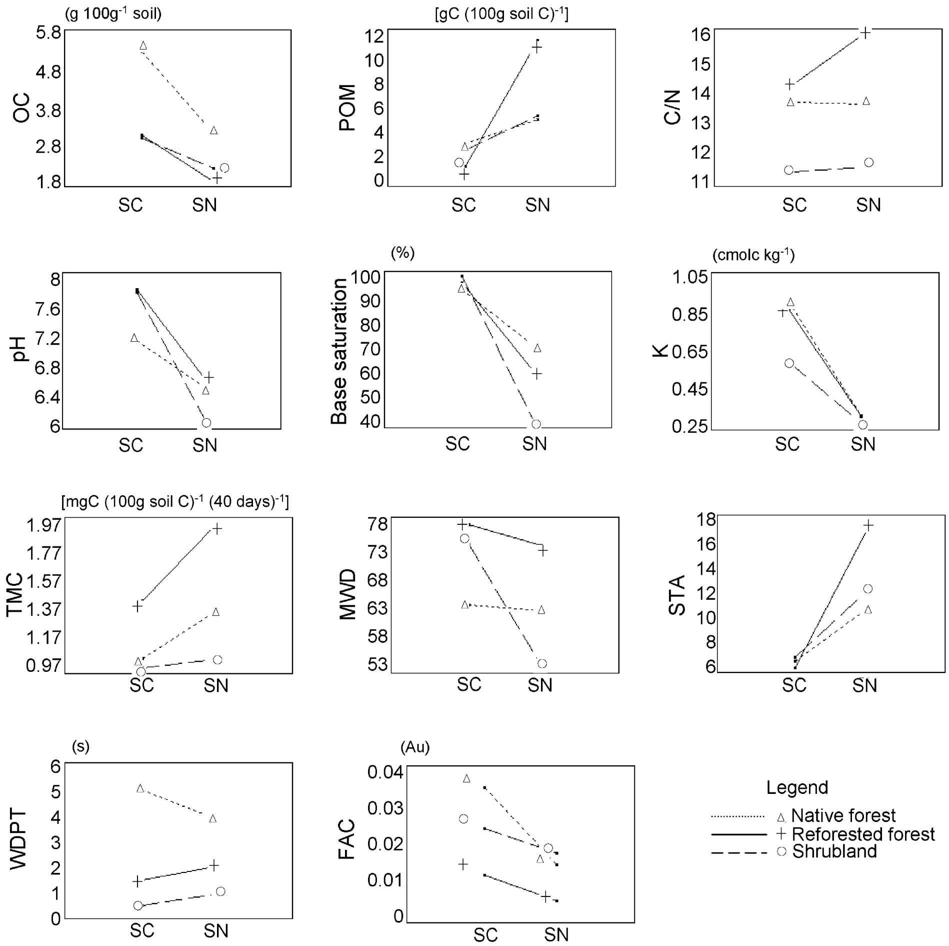

Fig. 3. Interplot graphic (extracted from multifactor ANOVA) for selected features of soil surface properties. Effect of the type of plant cover/management (selected variables, OC: total organic carbon; POM: particulate organic matter; C/N: carbon/nitrogen organic ratio; K^+: potassium exchange; MWD: mean weight diameter; STA: structural stability; WDPT: water drop penetration time; TMC: total mineralisation coefficient; FAC: fungal activity).

et al. (2005) found that although heavily disturbed soil conditions improve, plantations of certain species (e.g., *Pinus* sp.) are less able to recover the biological and physico-chemical properties of the soil than natural restoration. Maestre and Cortina (2004), in a review of the effect of *Pinus halepensis* plantations in the semiarid Mediterranean environment, demonstrate that the improvement in soil physico-chemical properties is very limited in most cases, and plantations rarely reach the values achieved in natural shrublands even after 40 years. These authors suggest that reforestation programmes should be re-evaluated, an idea which is supported by the result of our work.

4. Conclusions

In Mediterranean mountain ecosystems, the physico-chemical properties of surface horizons, which control the main soil functions, are conditioned by the lithological factor, giving rise to edaphic environments with differentiated functioning. These types of properties, however, are not sensitive to changes in plant cover. Nevertheless, the parameters of the organic fraction are mainly related to the type of vegetation, not to the edaphic environment.

The environmental factors that control soil surface conditions (or their indicators) critical to ecosystem functioning could not be clearly established. This implies the existence of some other factors, such as those related to the historically complex use of Mediterranean environments, that would justify the high variability in the parameters analysed.

The specificity of soil properties, marked by the influence of the lithological environment, implies differences in vulnerability of edaphic environments to forest management. Surface horizons in siliceous environments are much more fragile than in calcareous, and are more vulnerable to reforestation which involve a change in the type of plant cover. It thus appears necessary to include adequate characterisation of the landscapes affected and the boundaries of semi-natural geo-ecosystem resilience and resistance in reforestation programmes, adapting objectives and management techniques to them.

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