SOIL Discuss., 2, 187–215, 2015
www.soil-discuss.net/2/187/2015/
doi:10.5194/soild-2-187-2015

SOIL Discussions Open Access

Ecological soil quality affected by land use and management on semi-arid Crete

J. P. van Leeuwen[1], D. Moraetis[2], G. J. Lair[3,4], J. Bloem[5], N. P. Nikolaidis[6], L. Hemerik[1], and P. C. de Ruiter[1,7]

[1]Biometris, Wageningen University, P.O. Box 16, 6700 AA Wageningen, the Netherlands
[2]College of Science, Earth Science Department, Sultan Qaboos University, P.O. Box 36, Muscat 123, Oman
[3]Institute of Soil Research, University of Natural Resources and Life Sciences (BOKU), Peter-Jordan-Straße 82, 1190 Vienna, Austria
[4]Institute of Ecology, University of Innsbruck, Sternwartestraße 15, 6020 Innsbruck, Austria
[5]Alterra, Wageningen University and Research Centre, P.O. Box 47, 6700 AA Wageningen, the Netherlands
[6]Department of Environmental Engineering, Technical University of Crete, Polytechnioupolis, 73100 Chania, Greece
[7]Institute for Biodiversity and Ecosystem Dynamics (IBED), Faculty of Science, University of Amsterdam, P.O. Box 94248, 1090 GE Amsterdam, the Netherlands

Received: 6 February 2015 – Accepted: 24 February 2015 – Published: 3 March 2015

Correspondence to: J. P. van Leeuwen (jeroen.vanleeuwen@wur.nl)

Published by Copernicus Publications on behalf of the European Geosciences Union.

Abstract

Land use and soil management practice can have strong effects on soil quality, defined in terms of soil fertility, carbon sequestration and conservation of biodiversity. In this study, we investigate whether ecological soil quality parameters are adequate to assess soil quality under harsh conditions, and are able to reflect different land uses and intensities of soil management practices.

We selected three sites as main representatives for the dominant types of land use in the region: an intensively cultivated olive orchard (annually tilled), an extensively used olive orchard (not tilled) and a heavily grazed pasture site in the Koiliaris catchment (Crete/Greece). Soil quality was analysed using an ecosystem approach, studying soil biological properties such as soil organism biomass and activity, and taxonomic diversity of soil microarthropods, in connection to abiotic soil parameters, including soil organic matter contents, and soil aggregate stability.

The intensively cultivated olive orchard had a much lower aggregate water stability than the extensive olive orchard and the pasture. Contents of soil organic C and N were higher in the extensively used olive orchard than in the intensively cultivated orchard, with intermediate concentrations in the pasture. This was mainly caused by the highest input of organic matter, combined with the lowest organic matter decomposition rate. Soil organism biomasses in all sites were relatively low compared to values reported from less harsh systems, while microarthropod richness was highest in the pasture compared to both the intensive and extensive olive orchards.

From the present results we conclude that microarthropod taxonomic richness is a very useful indicator for ecological soil quality, because it is not only able to separate harsh sites from other systems, but it is also sensitive enough to show differences between land management practices under harsh conditions. Microbial biomass and especially microarthropod biomass were much lower in our harsh study sites than reported from less affected areas, and have therefore also potential as biological indicators for degradation.

SOILD
2, 187–215, 2015

1 Introduction

Soils provide a wide array of ecosystem services, such as the provision of food, feed and fibre, carbon storage and sequestration, hydrological regulation, and contaminant attenuation (Costanza et al., 1997). Soil quality can be defined as the ability of the soil to provide these services. In large areas in the Mediterranean region, soil quality is adversely affected by overgrazing and overharvesting of natural vegetation, ultimately leading to soil degradation, erosion, and desertification (Milgroom et al., 2007). Such losses in soil quality in semi-arid regions impose a severe and increasing risk for the local populations, because climate predictions indicate decreasing precipitation in the near future for the Mediterranean region (Chartzoulakis and Psarras, 2005).

In order to understand the interrelationships between land use and soil quality, the Critical Zone Observatory (CZO) network was established across the USA and Europe (Anderson et al., 2008). The CZO network is an internationally coordinated interdisciplinary research effort, including chemical, physical, and biological processes that govern soil ecosystem services.

As part of the CZO research effort, the European Commission has provided funding for a large multi-disciplinary research project: Soil Transformations in European Catchments (SoilTrEC) (Bernasconi et al., 2011; Menon et al., 2014). The European CZOs represent different stages in the soil life, including sites along soil formation gradients (Austria, Switzerland, Iceland), along a soil degradation gradient (Greece), along a lithology gradient (Czech Republic), and of agricultural sites differing in soil management (Austria, Iceland) (Banwart et al., 2011; Menon et al., 2014).

The aim of the present study is to investigate soil quality at the Koiliaris CZO sites in Crete (Greece) that are considered to be at risk of potential soil degradation and desertification. Koiliaris CZO is meant to be representative for the soils in the Mediterranean region impacted by a strong climatic gradient, steep upland slopes, and anthropogenic intensification, which make these soils sensitive to degradation. The sites in the Koiliaris CZO (Crete, Greece) include three dominant land use types: an intensively

SOILD
2, 187–215, 2015

cultivated olive orchard, an extensively used olive orchard, and a pasture site. The intensively cultivated olive orchard represents the conventional practice including tillage, litter removal and fertilisation, while at the extensively used orchard these measures are not applied. The pasture site represents a former cropland now used as grazed area for about 70 years.

Loss of soil fertility and soil degradation have mostly been approached from an abiotic perspective, with emphasis on soil structure (Celik, 2005), water erosion (Kosmas et al., 1997), nutrient cycling (Solomon et al., 2000), and organic matter dynamics (Wu and Tiessen, 2002). Here, we look in particular at biological soil quality parameters, in addition to the abiotic parameters. The role of soil organisms has received less attention in assessments of land degradation and desertification, although the importance of biology in soil quality and fertility is more and more acknowledged (Ashford et al., 2013; Brussaard et al., 1997, 2007; Buchan et al., 2013; Cole et al., 2006; De Deyn et al., 2003; Holtkamp et al., 2008; Hunt et al., 1987; Moore, 1994; Oades, 1993; Setälä and Huhta, 1991; Wardle et al., 2004). Up till now, soil quality assessments from an ecological perspective have mostly been carried out in soils that were not prone to severe losses of soil in terms of degradation, erosion and desertification, while focusing on soil quality for sustainable agricultural fertility or for habitat preservation of biodiversity in natural ecosystems (Bending et al., 2004; Birkhofer et al., 2008; Carpenter-Boggs et al., 2000; Doran and Zeiss, 2000; van Leeuwen et al., 2015). In this study we investigate whether ecological soil quality parameters are also adequate to assess soil quality under harsh conditions, and are able to reflect different land uses and intensities of soil management practices.

The present paper investigates soil quality with an emphasis on biological parameters in relation to abiotic properties, i.e. we look at soil structure, soil organic matter, nutrient availability, and soil as a habitat for species-rich communities. These soil properties are all considered to be important aspects of soil quality and are inextricably linked. For example, soil structure affects soil organic matter decomposition and the biological habitat function of the soil, soil organic matter is the most important resource

SOILD
2, 187–215, 2015

J. P. van Leeuwen et al.

for the soil food web, and soil organisms play a role in soil structure formation, organic matter decomposition and nutrient mineralisation (van Leeuwen et al., 2015). Soil physical and chemical measurements included soil aggregate size distribution, soil organic matter contents and quality, nitrogen (N) content, and soil pH. Soil biological measurements included the presence and abundance of microbes (bacteria, fungi) and soil fauna (protozoa, nematodes and microarthropods), representing the main taxonomic groups and trophic levels in the soil food web. In addition we measured the taxonomic richness and diversity within the group of microarthropods.

2 Methods

2.1 Site description

Crete represents Mediterranean soils under imminent threat of desertification. This is characterized by loss of vegetation, inducing water erosion, and subsequently loss of soil (Tsiafouli et al., 2005), which will be intensified by the predicted desiccation for the region over the next century (Chartzoulakis and Psarras, 2005). Deforestation, extensive grazing, and human-induced fires on the island over centuries have caused the native evergreen woodlands to be replaced by shrub-degradation formations, mainly in the southern and eastern parts of the island, covering now 57.6 % of Crete. Besides these rangelands, other agricultural practises including olive and orange orchards (15.6 %) and heterogeneous agricultural areas (25.4 %) are dominating (Nikolaidis et al., 2010). Olive orchards are considered as the least impacting agricultural activity in Mediterranean areas considering soil erosion rates, compared to vineyards and cereals (Kosmas et al., 1997). The Koiliaris CZO is situated 25 km east from the city of Chania, Crete, Greece (Moraetis et al., 2014). The total watershed area is 130 km^2 and the main supply of water originates in the White Mountains. The main outcropping lithology includes thick bedded limestone, metamorphic rock, neogene limestone, and alluvium sediments. Samples were taken at three land use types (Table 1). Site I was

Discussion Paper | Discussion Paper | Discussion Paper | Discussion Paper |

SOILD

2, 187–215, 2015

Ecological soil quality affected by land use and management

J. P. van Leeuwen et al.

an intensively cultivated olive orchard (20 year old trees) where tillage (once a year to facilitate harvesting), litter removal (to be used as fodder for livestock), and fertilisation were applied, at 20 m above sea level (a.s.l.) on alluvium sediments in a floodplain. Site E was an extensively used, 600 year old terrace (no tillage, litter removal or fertilisation) with olive trees on a steep slope at 465 m a.s.l., while site P was formed by a 600 year old terrace, formerly utilised as cropland (until 1940), with permanent grassland and sparse tree/shrub cover at 1065 m a.s.l., currently used as grazed pasture (see Table 1 for site characteristics). Sites E and P were both situated on soils developed on bedded limestone.

2.2 Sampling scheme

Samples were taken in May 2010. In each sampling site, three plots were selected in which all measurements were carried out; the plots were 10–20 m apart. In each plot, mixed soil samples (ca. 1 kg) were taken from the edge of a soil profile pit of about 1 m wide for microbial (bacteria, fungi), microfaunal (protozoa, nematodes) and SOM characterization, and by use of a 5 cm diameter corer for the mesofauna (enchytraeids and microarthropods). All samples were taken from the topsoil (0–10 cm), biologically the most active layer (Ekelund et al., 2001; Miura et al., 2008).

2.3 Soil analyses

Particle size distribution (clay content), soil pH, and calcium content were determined as described in van Leeuwen et al. (2015). Soil structure was experimentally approached by measuring the water stability of aggregates (1–3 mm in diameter), using a standard wet sieving procedure modified after Yoder (1936). Water stable aggregates (WSA) were calculated by the mass of aggregates remaining on the 1 mm sieve after wet sieving and subtracting the mass of sand <1 mm from this aggregate size fraction (e.g. Kercheva et al., 2011). WSA indicates the suitability of soil for agricultural production.

SOILD
2, 187–215, 2015

Total carbon (TOC) and nitrogen (TN) contents, hot-water-extractable carbon (HWC), potentially mineralisable nitrogen (PMN), and C and N mineralisation rates were determined as described in van Leeuwen et al. (2015).

Soil biological measurements included the presence and abundance of the major taxonomic groups of soil organisms: microbes (bacteria, fungi) and soil fauna (protozoa, nematodes and microarthropods). Within these taxonomic groups we defined "trophic groups" based on diet and life-history traits, following the method of Moore et al. (1988). Abundances were transformed into estimates of biomass based on body-size information, and expressed in units of kilograms of carbon per hectare for the 0–10 cm top soil layer. The laboratory techniques used to analyse the biological parameters are described in van Leeuwen et al. (2015).

Regarding the taxonomic species richness in the microarthropods we used three metrics, i.e. the absolute number of taxa present, the Shannon diversity index (H), and the Pielou evenness index (J). For the Shannon diversity index (H) we used the following formula:

$$H = -\sum_{i=1}^{N} (p_i \cdot \ln(p_i)),$$

in which p_i is the fraction of the total biomass present in species i, i.e. the relative biomass of species i, and N is the total number of taxa present. For the Pielou evenness index (J) we used the formula

$$J = \frac{H}{\ln(N)},$$

in which H represents the Shannon diversity index, and N the total number of taxa present.

SOILD
2, 187–215, 2015

2.4 Statistics

Differences in soil physicochemical and biological properties were tested with an ANOVA for repeated measures (rmANOVA), with the replicates within a site taken as repeated measures from the same object. We tested correlations between soil parameters with Pearson's correlation test. All data were log-transformed to obtain homogeneity of variances. Statistical analyses were carried out using SPSS (20.0.0) and *R* (2.15.2; R Core Team, 2012).

3 Results

3.1 Soil physicochemical measurements

To quantify soil structure, we measured the water stability of soil aggregates (WSA).The intensively cultivated olive orchard had a significantly lower WSA than the extensively used olive orchard and pasture ($p = 0.005$, Fig. 1a).

Dynamics of soil organic matter and N cycling are biologically mediated soil quality indicators. Total organic carbon (TOC, Fig. 1b) and total nitrogen (TN, Fig. 1c) were both greatest in the extensively used orchard, smallest in the intensively cultivated orchard and intermediate in the pasture ($p = 0.04$ and $p = 0.003$, respectively, Table 2). As a result, TOC and TN were strongly positively correlated with each other (Pearson correlation test, $r = 0.97$, $p < 0.001$). The pool of labile C, measured as HWC, showed the same differences as TOC and TN, and was smallest in the intensively cultivated orchard ($p = 0.045$). No differences in PMN ($p = 0.475$) and the total C : N ratio of the soil (calculated as TOC : TN) were found (Table 2). The C : N ratio of the labile organic matter (calculated as HWC : PMN), however, was larger in the extensively used olive orchard than in the two other sites ($p = 0.042$). C mineralisation rate and especially N mineralisation rate were greatest in the pasture site ($p = 0.048$ and $p = 0.011$, respectively, Table 2).

SOILD

2, 187–215, 2015

Ecological soil quality affected by land use and management

J. P. van Leeuwen et al.

To identify the relation of abiotic soil parameters with soil structure formation, we tested the correlations of WSA with TOC, HWC and clay content. WSA was positively correlated with TOC and HWC ($r = 0.89$, $p = 0.001$, and $r = 0.80$, $p = 0.012$, respectively) and clay content ($r = 0.82$, $p = 0.007$).

3.2 Soil biological measurements

Based on presence-absence data of the soil organisms, we constructed soil food web diagrams for the three sites (Fig. 2). These diagrams of the three sites were highly similar and most of the trophic groups were present in all sites. A few trophic groups were missing in some of the sites: Symphyla and fungivore mites were both missing in the intensively cultivated orchards, herbivore collembolans were missing in the extensively used orchards, whereas nematovore mites were only present in the extensively used orchards (Table 3).

Analysis of the soil community as a whole showed the following statistically significant differences between the sites. Total soil biomass was greatest in the intensively cultivated olive orchard, followed by the pasture and smallest in the extensively used olive orchard ($p = 0.024$, Table 3). Bacterial biomass was greatest in the intensively cultivated olive orchard, followed by the pasture and smallest in the extensively used olive orchard ($p = 0.003$, Fig. 1c). Fungal biomass followed the same trend, but here differences were not statistically significant ($p = 0.095$, Fig. 1c). Bacterial activity (measured as labelled thymidine (Thy) and leucine (Leu) incorporation rates) showed the same pattern, and was smallest in the extensively used orchard ($p = 0.026$ and $p = 0.014$, respectively, Table 3). The ratio of fungal to bacterial biomass is indicative for C sequestration and disturbance, where a higher ratio indicates a higher C sequestration and lower disturbance. The ratio did not differ statistically significantly between the sites, although the data indicated a greater ratio in the extensively used orchard compared to the intensively cultivated orchard (Table 3). Soil pH is known to influence microbial activity, but it was not significantly correlated with Thy and Leu ($p = 0.070$ and $p = 0.141$, respectively). To identify the role of microbial biomass in soil structure for-

SOILD
2, 187–215, 2015

mation, we tested the correlations of WSA with bacterial and fungal biomass. WSA was negatively correlated with fungal biomass ($r = -0.70$, $p = 0.035$) and bacterial biomass ($r = -0.87$, $p = 0.002$).

Total nematode biomass did not differ between the sites. We only found a smaller biomass of herbivorous nematodes in the extensively used olive orchard, while we found no differences in the other groups of nematodes, nor in biomass of protozoa (amoebae and flagellates), between the sites (Table 3).

Microarthropod abundance varied from 7000 m^{-2} in the intensively cultivated orchard to over 200 000 individuals m^{-2} in the pasture, but did not differ statistically significantly among the three sites. The relative dominance of omnivorous mites in the intensively cultivated olive orchard (95 ± 5 % of total microarthropod abundance were omnivorous mites) was remarkable, compared to both extensively used olive orchard (68 ± 32 %) and pasture (67±3 %). Although total microarthropod biomass did not differ significantly between sites (Fig. 3b), many trophic groups within the microarthropods showed a similar pattern towards greater biomass in the pasture compared to the olive orchards. For predaceous mites this difference was significant ($p = 0.016$), while for fungivorous Collembola this was nearly the case ($p = 0.057$). When looking at the microarthropod taxa present in the three sites, the data showed a nested pattern, with a progressive species loss from the pasture towards the intensively cultivated olive orchard, and not so much of a species turnover (Appendix A). That is, the taxa present in the intensively cultivated olive orchard are mostly also present in the extensively used orchard and in the pasture, with additional taxa present at the latter sites.

In total, 49 taxa of microarthropods were found in the study sites (Appendix A). Only one taxon (omnivorous prostigmatid mites from the family Alicorhagiidae) was found exclusively in the intensively cultivated olive orchard. Seven taxa were exclusively found in the extensively used olive orchards, whereas 13 taxa were only found in the pasture sites. As a result, soil microarthropods had a greater taxa richness (on average 28 taxa present) in the pasture compared to the intensively cultivated (7 taxa) and the extensively used (15 taxa) olive orchards ($p = 0.037$, Fig. 3c). When including evenness in the diversity measures, no statistically significant differences were found however, not in the Shannon diversity index nor the Pielou evenness index.

4 Discussion

The aim of the present study was to investigate ecological soil quality in southern European soils that are at risk of potential soil degradation and desertification. In addition, we identified whether the currently used ecological soil quality parameters are adequate to assess soil quality under harsh conditions.

4.1 Soil aggregate formation, soil organic matter, and soil nutrient cycling

Soil aggregate formation is an important index for soil quality. The intensively cultivated olive orchard had a much lower aggregate water stability than the extensively used olive orchard and the pasture. This is consistent with literature, which shows that tillage negatively affects soil aggregate stability (Beare et al., 1994). Soil structure (aggregate stability) was strongly positively correlated to C content and clay content in our study, which is also consistent with literature (Six et al., 2006; Wright et al., 2007). In contrast to our expectation, we found a negative correlation between fungal biomass and aggregate stability. Several studies have shown that fungal biomass and activity enhance aggregate stability (Beare et al., 1997; Bossuyt et al., 2001). Both hyphae and exudates produced by fungi (polysaccharides) are assumed to serve as bonding material (De Gryze et al., 2005). Fungal products, compared to bacterially derived products, are more chemically resistant to decay and preferentially protected from decomposition through interactions with clay and soil aggregates (Simpson et al., 2004). The negative correlation resulted from the extensively used orchard, which had the highest aggregate stability, and the lowest bacterial and fungal biomass and activity. We think that the low water availability limited microbial activity mostly at the extensively used orchard. The limited water availability simultaneously caused physical changes such as swelling

SOILD
2, 187–215, 2015
Ecological soil quality affected by land use and management
J. P. van Leeuwen et al.

and shrinking of the clay-rich soils. Physical factors therefore might have been more important than microbial factors in the build-up and stability of soil aggregates in this system.

All parameters related to soil organic matter contents, such as TOC, TN and HWC showed highest values at the extensively used olive orchard, while C and N mineralisation rates were both highest at the pasture. The TOC and TN contents in our study were in the same order of magnitude as contents reported from less harsh environments (Culman et al., 2010; Holtkamp et al., 2011). The lower C and N contents in the intensively cultivated orchard might have been due to leaf litter removal and soil tillage in this site, in combination with the lower clay content. The absence of these activities in the extensively used orchard may have led to an accumulation of plant and olive residues in a relatively undisturbed upper soil horizon, resulting in relatively high amounts of organic C and N. The litter of olive trees is lignin-rich (30.4 %), with a high C : N ratio (33.0) and is therefore thought to be difficult to decompose (Canali and Benedetti, 2006; Gallardo and Merino, 1993). This substrate generally favours slow fungal over fast bacterial activity, because fungi are assumed to be better able to degrade lignin-rich substances (Bossuyt et al., 2001). We found indications for a higher fungal to bacterial biomass ratio in the extensively used olive orchard, although differences were not statistically significant. In addition to substrate quality, soil pH is known to affect microbial activity; higher pH is thought to enhance bacterial activity (Bååth and Anderson, 2003) and to decrease the ratio of fungal to bacterial activity (Blagodatskaya and Anderson, 1998). We did indeed find the lowest bacterial activity in the extensively used orchard, which also had the lowest pH (5.4, in comparison with 5.9 at the pasture and 6.9 at the intensively cultivated orchard).

4.2 Soil as habitat for soil organisms

All microbial parameters, i.e. the biomasses of bacteria and fungi and the two indicators for microbial activity, showed statistically significant minimum values at the extensively used olive orchard. The microbial biomass in the extensively used olive orchard was

SOILD
2, 187–215, 2015

with 37 kg C ha^{-1} much lower than reported from less harsh environments (de Ruiter et al., 1993; Holtkamp et al., 2008), while the pasture (72 kg C ha^{-1}), and especially the intensively cultivated olive orchard (111 kg C ha^{-1}), reached values that are closer to values reported in literature. For example, Holtkamp et al. (2008) report 60–100 kg microbial C ha^{-1} for fields in transition from arable field to heathland in the Netherlands, while de Ruiter et al. (1993) report similar biomasses from arable fields in the Netherlands (90–100 kg C ha^{-1}) and prairie soil in the USA (150 kg C ha^{-1}), but much higher values for arable fields in USA (400–550 kg C ha^{-1}) and Sweden (900–1300 kg C ha^{-1}) (all values correspond to 0–10 cm soil depth). Other studies provide higher microbial biomass levels ranging from 300 to 1300 kg C ha^{-1} based on chloroform fumigation methods (e.g. Culman et al., 2010; Schnürer and Rosswall, 1987; Schröter et al., 2003), but values found using this method are not directly comparable to microscopic counting (Martens, 1995). The intensity of agricultural management at the intensively cultivated olive orchard including tillage and fertilisation, as well as soil pH and leaf litter composition, led to the expectation that we would find a lower fungal to bacterial biomass ratio, as indicator for C sequestration and disturbance, compared with the extensively used orchard and pasture. A lower ratio indicates a lower C sequestration and higher disturbance. We indeed found indications for a lower ratio in the intensively cultivated olive orchard, but the differences were not significant.

4.3 Microarthropod biomass and diversity

Total soil microarthropod biomass and taxonomic richness within the soil microarthropods have been proposed as biological soil quality indicators (Gardi et al., 2009; Parisi et al., 2005), but the suitability of these parameters has not yet been tested on soils under harsh conditions. Total microarthropod biomass in our systems, especially in the intensively cultivated olive orchard, was with 0.06–0.72 kg C ha^{-1} lower than biomasses of 0.5–3.8 kg C ha^{-1} reported from less harsh arable systems (de Ruiter et al., 1993; Holtkamp et al., 2011). In our sites, microarthropod taxonomic richness strongly increased along with microarthropod biomass from the intensively cultivated olive orchard, to the extensively used olive orchard, to the pasture. Microarthropod taxa richness was higher in our study than reported from semi-arid croplands in central Spain (Kautz et al., 2006), comparable to the values found by Tsiafouli et al. (2005) in pine forests in Greece, and in the lower range of the values found on farms in Iceland and Austria (van Leeuwen et al., 2015). The higher richness we found in the pasture, compared to the olive orchards, confirms findings in Mediterranean Spain showing the highest richness in Oribatid mite communities in pastures and forests in comparison with cropland (Arroyo and Iturrondobeitia, 2006). This pattern of increasing microarthropod biomass and taxonomic richness could be related to a lower disturbance of the topsoil in the pasture, for which the microarthropods are known to be very sensitive (Wardle, 1995), but could also be related to soil moisture availability. Soil moisture availability in our sites increased with elevation. This was caused by the increasing average precipitation and decreasing average temperature (Table 1), hence decreasing evaporation, leading to a high soil moisture content in the pasture as compared to the olive orchards. Also Tsiafouli et al. (2005) reports an increasing species richness and diversity of soil microarthropods with an increase in water availability in an experimental setup in pine forests in Greece.

We found statistically significant differences in taxa richness, but not in the Shannon diversity index (SDI), nor in Pielou evenness. Kautz et al. (2006) finds comparably low SDI values in croplands in central Spain, despite a lower taxa richness and microarthropod abundance. It appears that taxonomic richness of microarthropods is able to differentiate between land management practices, hence is useful as an indicator of ecological soil quality, whereas the SDI may separate harsh sites from other sites, but is not sensitive enough to detect differences between different land management practices under harsh conditions.

SOILD
2, 187–215, 2015
Ecological soil quality affected by land use and management
J. P. van Leeuwen et al.

4.4 Limitations

The sites discussed in this study are part of a coherent set of CZOs throughout Europe, ranging from a soil formation cycle to soils at risk of degradation as presented in this paper. The aim of the present study was to investigate ecological soil quality in southern European soils that are at risk of potential soil degradation and desertification, via an integral approach including physical, chemical and biological soil processes. The presently chosen design did not allow to pronounce upon land use effects in a generalised way, however, as the study included information from only one example per land use type. As these single examples of land use types were measured at various plots, we could statistically test differences between sites, but we could not generalise our results to an interpretation in terms of land use. Generalisation over land use type was also constrained by other potentially important factors that may have played a role in the observed differences between the sites, such as temperature, moisture availability, and clay content. A second reason to treat our results with caution is that the measurements were carried out at one particular moment, i.e. May 2010. Therefore we lack information regarding variability in time and/or effects of seasons. For future research aiming at improved generalisation of the results of these study sites, accounting for temporal and spatial variability by extending the sampling design is recommended.

5 Conclusions

The present study investigated ecological soil quality through an integral approach including physical, chemical and biological parameters in soils under relatively harsh conditions, varying in land use type. The most sensitive soil parameter seemed to be the microarthropod richness: taxa richness increased from the intensively cultivated olive orchard to the extensively used orchard to the pasture. This confirmed the use of this parameter as indicator for ecological soil quality. Microbial biomass and especially microarthropod biomass showed lower values in our harsh study sites than reported

SOILD
2, 187–215, 2015

J. P. van Leeuwen et al.

from less affected areas. In this way they may also be suitable as ecological indicators for soil degradation. The ratio of fungal to bacterial biomass, which is frequently proposed as indicator for C sequestration and disturbance, did not show a clear pattern in our study, probably because at our sites many factors may have affected this ratio, such as tillage, pH and leaf litter composition, and might therefore be less suitable as indicator for soil quality under harsh conditions.

Acknowledgements. This research was funded by the EU FP7-ENV-2009 Project SoilTrEC "Soil Transformations in European Catchments" (Contract no. 244118). This research was also supported by the research program KB IV "Innovative scientific research for sustainable green and blue environment" funded by the Netherlands Ministry of Economic Affairs, Agriculture and Innovation, and carried out by Wageningen University and Research Centre.

We thank An Vos, Meint Veninga, Popko Bolhuis and Tamas Salanki for technical assistance. Wim Dimmers, Gerard Jagers and Jeffrey Newton are acknowledged for counting and identifying microarthropods.

References

Anderson, S. P., Bales, R. C., and Duffy, C. J.: Critical zone observatories: Building a network to advance interdisciplinary study of earth surface processes, Mineral. Mag., 72, 7–10, 2008.

Arroyo, J. and Iturrondobeitia, J. C.: Differences in the diversity of oribatid mite communities in forests and agrosystems lands, Eur. J. Soil Biol., 42, 259–269, doi:10.1016/j.ejsobi.2006.01.002, 2006.

Ashford, O. S., Foster, W. A., Turner, B. L., Sayer, E. J., Sutcliffe, L., and Tanner, E. V. J.: Litter manipulation and the soil arthropod community in a lowland tropical rainforest, Soil Biol. Biochem., 62, 5–12, doi:10.1016/j.soilbio.2013.03.001, 2013.

Bååth, E. and Anderson, T.-H.: Comparison of soil fungal/bacterial ratios in a ph gradient using physiological and plfa-based techniques, Soil Biol. Biochem., 35, 955–963, 2003.

Banwart, S., Bernasconi, S. M., Bloem, J., Blum, W., Brandao, M., Brantley, S., Chabaux, F., Duffy, C., Kram, P., Lair, G., Lundin, L., Nikolaidis, N., Novak, M., Panagos, P., Ragnarsdottir, K. V., Reynolds, B., Rousseva, S., de Ruiter, P., van Gaans, P., van Riemsdijk, W., White, T.,

SOILD
2, 187–215, 2015

and Zhang, B.: Soil processes and functions in Critical Zone Observatories: Hypotheses and experimental design, Vadose Zone J., 10, 974–987, doi:10.2136/vzj2010.0136, 2011.
Beare, M., Hu, S., Coleman, D., and Hendrix, P.: Influences of mycelial fungi on soil aggregation and organic matter storage in conventional and no-tillage soils, Appl. Soil Ecol., 5, 211–219, 1997.
Beare, M. H., Hendrix, P. F., and Coleman, D. C.: Water-stable aggregates and organic matter fractions in conventional- and no-tillage soils, Soil Sci. Soc. Am. J., 58, 777–786, doi:10.2136/sssaj1994.03615995005800030020x, 1994.
Bending, G. D., Turner, M. K., Rayns, F., Marx, M.-C., and Wood, M.: Microbial and biochemical soil quality indicators and their potential for differentiating areas under contrasting agricultural management regimes, Soil Biol. Biochem., 36, 1785–1792, doi:10.1016/j.soilbio.2004.04.035, 2004.
Bernasconi, S. M., Bauder, A., Bourdon, B., Brunner, I., Bünemann, E., Chris, I., Derungs, N., Edwards, P., Farinotti, D., and Frey, B.: Chemical and biological gradients along the Damma glacier soil chronosequence, Switzerland, Vadose Zone J., 10, 867–883, 2011.
Birkhofer, K., Bezemer, T. M., Bloem, J., Bonkowski, M., Christensen, S., Dubois, D., Ekelund, F., Fließbach, A., Gunst, L., Hedlund, K., Mäder, P., Mikola, J., Robin, C., Setälä, H., Tatin-Froux, F., van der Putten, W. H., and Scheu, S.: Long-term organic farming fosters below and aboveground biota: Implications for soil quality, biological control and productivity, Soil Biol. Biochem., 40, 2297–2308, 2008.
Blagodatskaya, E. V. and Anderson, T.-H.: Interactive effects of ph and substrate quality on the fungal-to-bacterial ratio and qCO_2 of microbial communities in forest soils, Soil Biol. Biochem., 30, 1269–1274, 1998.
Bossuyt, H., Denef, K., Six, J., Frey, S. D., Merckx, R., and Paustian, K.: Influence of microbial populations and residue quality on aggregate stability, Appl. Soil Ecol., 16, 195–208, 2001.
Brussaard, L., Behan-Pelletier, V. M., Bignell, D. E., Brown, V. K., Didden, W., Folgarait, P., Fragoso, C., Freckman, D. W., Gupta, V. V. S. R., Hattori, T., Hawksworth, D. L., Klopatek, C., Lavelle, P., Malloch, D. W., Rusek, J., Söderström, B., Tiedje, J. M., and Virginia, R. A.: Biodiversity and ecosystem functioning in soil, Ambio, 26, 563–570, 1997.
Brussaard, L., de Ruiter, P. C., and Brown, G. G.: Soil biodiversity for agricultural sustainability, Agr. Ecosyst. Environ., 121, 233–244, doi:10.1016/j.agee.2006.12.013, 2007.

SOILD
2, 187–215, 2015
Ecological soil quality affected by land use and management
J. P. van Leeuwen et al.

Buchan, D., Gebremikael, M. T., Ameloot, N., Sleutel, S., and De Neve, S.: The effect of free-living nematodes on nitrogen mineralisation in undisturbed and disturbed soil cores, Soil Biol. Biochem., 60, 142–155, doi:10.1016/j.soilbio.2013.01.022, 2013.

Canali, S. and Benedetti, A.: Soil nitrogen mineralization, in: Microbiological methods for assessing soil quality, edited by: Bloem, J., Hopkins, D. W., and Benedetti, A., CABI Publishing, Wallingford, UK, 127–135, 2006.

Carpenter-Boggs, L., Kennedy, A. C., and Reganold, J. P.: Organic and biodynamic management effects on soil biology, Soil Sci. Soc. Am. J., 64, 1651–1659, doi:10.2136/sssaj2000.6451651x, 2000.

Celik, I.: Land-use effects on organic matter and physical properties of soil in a southern Mediterranean highland of Turkey, Soil Till. Res., 83, 270–277, 2005.

Chartzoulakis, K. and Psarras, G.: Global change effects on crop photosynthesis and production in Mediterranean: The case of Crete, Greece, Agr. Ecosyst. Environ., 106, 147–157, 2005.

Cole, L., Bradford, M. A., Shaw, P. J. A., and Bardgett, R. D.: The abundance, richness and functional role of soil meso- and macrofauna in temperate grassland—a case study, Appl. Soil Ecol., 33, 186–198, doi:10.1016/j.apsoil.2005.11.003, 2006.

Costanza, R., d'Arge, R., de Groot, R., Farber, S., Grasso, M., Hannon, B., Limburg, K., Naeem, S., O'neill, R. V., Paruelo, J., Raskin, R. G., Sutton, P., and van den Belt, M.: The value of the world's ecosystem services and natural capital, Nature, 387, 253–260, 1997.

Culman, S. W., DuPont, S. T., Glover, J. D., Buckley, D. H., Fick, G. W., Ferris, H., and Crews, T. E.: Long-term impacts of high-input annual cropping and unfertilized perennial grass production on soil properties and belowground food webs in Kansas, USA, Agr. Ecosyst. Environ., 137, 13–24, doi:10.1016/j.agee.2009.11.008, 2010.

De Deyn, G. B., Raaijmakers, C. E., Zoomer, H. R., Berg, M. P., de Ruiter, P. C., Verhoef, H. A., Bezemer, T. M., and van der Putten, W. H.: Soil invertebrate fauna enhances grassland succession and diversity, Nature, 422, 711–713, 2003.

De Gryze, S., Six, J., Brits, C., and Merckx, R.: A quantification of short-term macroaggregate dynamics: Influences of wheat residue input and texture, Soil Biol. Biochem., 37, 55–66, doi:10.1016/j.soilbio.2004.07.024, 2005.

de Ruiter, P. C., van Veen, J. A., Moore, J. C., Brussaard, L., and Hunt, H. W.: Calculation of nitrogen mineralization in soil food webs, Plant Soil, 157, 263–273, 1993.

Doran, J. W. and Zeiss, M. R.: Soil health and sustainability: Managing the biotic component of soil quality, Appl. Soil Ecol., 15, 3–11, doi:10.1016/S0929-1393(00)00067-6, 2000.
Ekelund, F., Rønn, R., and Christensen, S.: Distribution with depth of protozoa, bacteria and fungi in soil profiles from three Danish forest sites, Soil Biol. Biochem., 33, 475–481, doi:10.1016/S0038-0717(00)00188-7, 2001.
Gallardo, A. and Merino, J.: Leaf decomposition in two Mediterranean ecosystems of southwest Spain: Influence of substrate quality, Ecology, 74, 152–161, 1993.
Gardi, C., Montanarella, L., Arrouays, D., Bispo, A., Lemanceau, P., Jolivet, C., Mulder, C., Ranjard, L., Römbke, J., Rutgers, M., and Menta, C.: Soil biodiversity monitoring in Europe: Ongoing activities and challenges, Eur. J. Soil Sci., 60, 807–819, 2009.
Holtkamp, R., Kardol, P., van der Wal, A., Dekker, S. C., van der Putten, W. H., and de Ruiter, P. C.: Soil food web structure during ecosystem development after land abandonment, Appl. Soil Ecol., 39, 23–34, 2008.
Holtkamp, R., van der Wal, A., Kardol, P., van der Putten, W. H., de Ruiter, P. C., and Dekker, S. C.: Modelling C and N mineralisation in soil food webs during secondary succession on ex-arable land, Soil Biol. Biochem., 43, 251–260, doi:10.1016/j.soilbio.2010.10.004, 2011.
Hunt, H. W., Coleman, D. C., Ingham, E. R., Ingham, R. E., Elliot, E. T., Moore, J. C., Rose, S. L., Reid, C. P. P., and Morley, C. R.: The detrital food web in a shortgrass prairie, Biol. Fert. Soils, 3, 57–68, 1987.
Kautz, T., López-Fando, C., and Ellmer, F.: Abundance and biodiversity of soil microarthropods as influenced by different types of organic manure in a long-term field experiment in central Spain, Appl. Soil Ecol., 33, 278–285, doi:10.1016/j.apsoil.2005.10.003, 2006.
Kosmas, C., Danalatos, N., Cammeraat, L. H., Chabart, M., Diamantopoulos, J., Farand, R., Gutierrez, L., Jacob, A., Marques, H., Martinez-Fernandez, J., Mizara, A., Moustakas, N., Nicolau, J. M., Oliveros, C., Pinna, G., Puddu, R., Puigdefabregas, J., Roxo, M., Simao, A., Stamou, G., Tomasi, N., Usai, D., and Vacca, A.: The effect of land use on runoff and soil erosion rates under Mediterranean conditions, Catena, 29, 45–59, doi:10.1016/S0341-8162(96)00062-8, 1997.
Martens, R.: Current methods for measuring microbial biomass C in soil: Potentials and limitations, Biol. Fert. Soils, 19, 87–99, 1995.
Menon, M., Rousseva, S., Nikolaidis, N. P., van Gaans, P., Panagos, P., de Souza, D. M., Ragnarsdottir, K. V., Lair, G. J., Weng, L., Bloem, J., Kram, P., Novak, M., Davidsdottir, B., Gisladottir, G., Robinson, D. A., Reynolds, B., White, T., Lundin, L., Zhang, B., Duffy, C.,

SOILD
2, 187–215, 2015

Bernasconi, S. M., de Ruiter, P., Blum, W. E., and Banwart, S. A.: Soiltrec: A global initiative on critical zone research and integration, Environ. Sci. Pollut. Res. Int., 21, 3191–3195, doi:10.1007/s11356-013-2346-x, 2014.

Milgroom, J., Soriano, M. A., Garrido, J. M., Gómez, J. A., and Fereres, E.: The influence of a shift from conventional to organic olive farming on soil management and erosion risk in southern Spain, Renew. Agr. Food Syst., 22, 1–10, doi:10.1017/S1742170507001500, 2007.

Miura, F., Nakamoto, T., Kaneda, S., Okano, S., Nakajima, M., and Murakami, T.: Dynamics of soil biota at different depths under two contrasting tillage practices, Soil Biol. Biochem., 40, 406–414, 2008.

Moore, J. C.: Impact of agricultural practices on soil food web structure: Theory and application, Agr. Ecosyst. Environ., 51, 239–247, 1994.

Moore, J. C., Walter, D. E., and Hunt, H. W.: Arthropod regulation of micro-and mesobiota in below-ground detrital food webs, Ann. Rev. Entomol., 33, 419–435, 1988.

Moraetis, D., Paranychianakis, N. V., Nikolaidis, N. P., Banwart, S. A., Rousseva, S., Kercheva, M., Nenov, M., Shishkov, T., de Ruiter, P., Bloem, J., Blum, W. E. H., Lair, G. J., van Gaans, P., and Verheul, M.: Sediment provenance, soil development, and carbon content in fluvial and manmade terraces at Koiliaris River Critical Zone Observatory, J. Soils Sediments, 15, 347–364, 2014.

Nikolaidis, N. P., Stamati, F., Schnoor, J., Moraetis, D., and Kotronakis, M.: Hydrologic and soil science in a Mediterranean Critical Zone Observatory: Koiliaris river basin, EGU General Assembly, Vienna, p. 9496, 2010.

Oades, J. M.: The role of biology in the formation, stabilization and degradation of soil structure, Geoderma, 56, 377–400, doi:10.1016/0016-7061(93)90123-3, 1993.

Parisi, V., Menta, C., Gardi, C., Jacomini, C., and Mozzanica, E.: Microarthropod communities as a tool to assess soil quality and biodiversity: A new approach in Italy, Agr. Ecosyst. Environ., 105, 323–333, 2005.

R Core Team: R: A language and environment for statistical computing, R foundation for Statistical Computing, Vienna, Austria, 2012.

Schnürer, J. and Rosswall, T.: Mineralization of nitrogen from ^{15}N labelled fungi, soil microbial biomass and roots and its uptake by barley plants, Plant Soil, 102, 71–78, 1987.

Schröter, D., Wolters, V., and de Ruiter, P. C.: C and N mineralisation in the decomposer food webs of a European forest transect, Oikos, 102, 294–308, 2003.

SOILD
2, 187–215, 2015

Ecological soil quality affected by land use and management

J. P. van Leeuwen et al.

Setälä, H. and Huhta, V.: Soil fauna increase *Betula pendula* growth: Laboratory experiments with coniferous forest floor, Ecology, 72, 665–671, 1991.

Simpson, R. T., Frey, S. D., Six, J., and Thiet, R. K.: Preferential accumulation of microbial carbon in aggregate structures of no-tillage soils, Soil Sci. Soc. Am. J., 68, 1249–1255, 2004.

Six, J., Frey, S. D., Thiet, R. K., and Batten, K. M.: Bacterial and fungal contributions to carbon sequestration in agroecosystems, Soil Sci. Soc. Am. J., 70, 555, doi:10.2136/sssaj2004.0347, 2006.

Solomon, D., Lehmann, J., and Zech, W.: Land use effects on soil organic matter properties of chromic luvisols in semi-arid northern Tanzania: Carbon, nitrogen, lignin and carbohydrates, Agr. Ecosyst. Environ., 78, 203–213, doi:10.1016/S0167-8809(99)00126-7, 2000.

Tsiafouli, M. A., Kallimanis, A. S., Katana, E., Stamou, G. P., and Sgardelis, S. P.: Responses of soil microarthropods to experimental short-term manipulations of soil moisture, Appl. Soil Ecol., 29, 17–26, doi:10.1016/j.apsoil.2004.10.002, 2005.

van Leeuwen, J. P., Lehtinen, T., Lair, G. J., Bloem, J., Hemerik, L., Ragnarsdóttir, K. V., Gísladóttir, G., Newton, J. S., and de Ruiter, P. C.: An ecosystem approach to assess soil quality in organically and conventionally managed farms in Iceland and Austria, SOIL, 1, 83–101, doi:10.5194/soil-1-83-2015, 2015.

Wardle, D. A.: Impacts of disturbance on detritus food webs in agro-ecosystems of contrasting tillage and weed management practices, in: Advances in ecological research, edited by: Begon, M., and Fitter, A. H., Academic Press, 105–185, 1995.

Wardle, D. A., Bardgett, R. D., Klironomos, J. N., Setälä, H., van der Putten, W. H., and Wall, D. H.: Ecological linkages between aboveground and belowground biota, Science, 304, 1629–1633, 2004.

Wright, S. F., Green, V. S., and Cavigelli, M. A.: Glomalin in aggregate size classes from three different farming systems, Soil Till. Research, 94, 546–549, doi:10.1016/j.still.2006.08.003, 2007.

Wu, R. and Tiessen, H.: Effect of land use on soil degradation in alpine grassland soil, China, Soil Sci. Soc. Am. J., 66, 1648–1655, 2002.

Yoder, R. E.: A direct method of aggregate analysis of soils and a study of the physical nature of erosion losses, Agronomy J., 28, 337–351, 1936.

Table 1. Characteristics of the Koiliaris Critical Zone Observatory (CZO) at the three different sites (I = intensively cultivated olive orchard, E = extensively used olive orchard, P = pasture).

Site	I	E	P
Land use type	Intensive olive orchard	Extensive olive orchard	Pasture
Tillage	yes	no	no
Fertilization	yes	no	no
Litter removal	yes	no	no
Grazing pressure	Not grazed	Grazed	Heavily grazed
Elevation	20 m	465 m	1065 m
Average rainfall	567 mm	915 mm	1335 mm
Average temp.	19 °C	18 °C	14 °C

SOILD
2, 187–215, 2015

Table 2. Soil physicochemical properties and biologically mediated processes at three different sites in the Koiliaris Critical Zone Observatory (Crete, GR) (I = intensively cultivated olive orchard, E = extensively used olive orchard, P = pasture). Values represent mean and standard deviation (between brackets). The *p* values represent significance levels from an ANOVA for repeated measures, where the superscript letters denote statistically significant differences between sites, and number of ** denotes statistical significance level (*: $p < 0.05$, **: $p < 0.01$). All measurements were done in the topsoil (0–10 cm).

Site	I	E	P	rmANOVA (*p* value)
Soil moisture (%)	4.09 (2.21)[a]	9.83 (1.26)[b]	14.47 (0.55)[c]	0.006 **
pH-H_2O	6.9 (0.96)	5.4 (0.41)	5.9 (0.11)	0.056
Clay content (%)	5.1 (0.43)[a]	26.6 (10.5)[b]	24.6 (9.2)[b]	0.002**
$CaCO_3$ ($g\,kg^{-1}$)	22.2 (18.2)	1.78 (0.59)	1.39 (0.25)	0.086
WSA (%)[1]	38.4 (5.4)[a]	77.0 (7.4)[b]	67.1 (5.4)[b]	0.005**
TOC ($kg\,ha^{-1}$)[2]	21 670 (2662)[a]	59 926 (8444)[c]	39 991 (6319)[b]	0.004**
HWC ($kg\,ha^{-1}$)[3]	390 (112)[a]	952 (406)[b]	700 (28)[b]	0.045*
Total N ($kg\,ha^{-1}$)	1557 (249)[a]	4246 (363)[c]	2843 (421)[b]	0.003**
PMN ($kg\,ha^{-1}$)[4]	81.26 (22.97)	66.73 (47.43)	101.8 (20.38)	0.475
TOC : Total N	13.98 (0.54)	14.14 (2.02)	14.07 (0.58)	0.994
HWC : PMN	4.80 (0.14)[a]	20.29 (13.48)[b]	7.03 (1.18)[a]	0.042*
C min ($kg\,ha^{-1}\,r^{-1}$)[5]	2526 (1131)[a]	2418 (103)[a]	2818 (1080)[b]	0.048*
N min ($kg\,ha^{-1}\,yr^{-1}$)[6]	24.34 (18.31)[a]	54.11 (20.62)[a]	172.9 (93.00)[b]	0.011*

[1] Percentage of water stable aggregates of 1–3 mm; [2] Total soil organic carbon; [3] Hot-water-extractable carbon; [4] Potential mineralisable nitrogen; [5] Carbon mineralisation rate; [6] Nitrogen mineralisation rate

SOILD
2, 187–215, 2015

Table 3. Biological parameters at the three different sites in the Koiliaris Critical Zone Observatory (Crete, GR) (I = intensively cultivated olive orchard, E = extensively used olive orchard, P = pasture): biomasses (in $kg\,C\,ha^{-1}$) of the trophic and taxonomic groups in the soil food webs, bacterial activity and microarthropod diversity. Values represent mean and standard deviation (between brackets), nd: not detected. The p values represent significance levels from an ANOVA for repeated measures, where the superscript letters denote statistically significant differences between sites, and number of ** denotes statistical significance level (*: $p < 0.05$, **: $p < 0.01$). All measurements were done in the topsoil (0–10 cm).

Site	I	E	P	rmANOVA (p value)
Bacteria	63.31 (11.95)[b]	14.60 (3.79)[a]	38.17 (5.83)[b]	0.003**
Thy ($pmol\,g^{-1}\,h^{-1}$)[1]	5.52 (3.56)[b]	0.77 (0.46)[a]	7.27 (0.98)[b]	0.026*
Leu ($pmol\,g^{-1}\,h^{-1}$)[2]	229.3 (89.78)[b]	86.26 (21.22)[a]	294.4 (45.02)[b]	0.014*
Fungi	50.19 (10.82)	24.94 (11.73)	34.70 (10.35)	0.095
Fungal : bacterial biomass ratio	0.79 (0.05)	1.75 (0.87)	0.91 (0.23)	0.121
Flagellates	0.65 (0.32)	0.51 (0.16)	0.32 (0.26)	0.52
Amoebae	23.34 (22.30)	6.39 (4.10)	9.45 (3.22)	0.587
Fungivore Nematodes	0.05 (0.02)	0.007 (0.006)	0.04 (0.02)	0.148
Bacterivore Nematodes	0.10 (0.03)	0.03 (0.005)	0.06 (0.06)	0.241
Herbivore Nematodes	0.18 (0.02)[b]	0.01 (0.006)[a]	0.28 (0.12)[b]	0.015*
Omnivore Nematodes	0.23 (0.17)	0.12 (0.04)	0.10 (0.14)	0.620
Predaceous Nematodes	0.09 (0.09)	0.03 (0.03)	0.08 (0.06)	0.564
Total nematode biomass	0.65 (0.22)	0.20 (0.06)	0.57 (0.33)	0.152
Enchytraeids	0.003 (0.006)	0.03 (0.06)	0.003 (0.006)	0.545
Herbivore mites	0.0002 (0.0003)	0.0003 (0.0003)	0.0008 (0.0007)	0.329
Herbofungivore mites	0.0001 (0.0001)	0.002 (0.001)	0.008 (0.007)	0.152
Fungivore mites	nd	0.003 (0.004)	0.02 (0.02)	0.174
Nematovore mites	nd	0.002 (0.001)	nd	0.112
Omnivore mites	0.05 (0.03)	0.17 (0.24)	0.31 (0.22)	0.374
Predaceous mites	0.0005 (0.0008)[a]	0.003 (0.004)[a]	0.08 (0.04)[b]	0.016*
Herbivore collembola	0.001 (0.002)	nd	0.001 (0.002)	0.689
Herbofungivore collembola	0.002 (0.003)	0.006 (0.008)	0.05 (0.07)	0.426
Fungivore collembola	0.0006 (0.001)	0.01 (0.02)	0.14 (0.10)	0.057
Symphyla	nd	0.03 (0.06)	0.03 (0.03)	0.418
Total biomass	137.0 (16.99)[b]	46.20 (6.01)[a]	82.98 (10.63)[ab]	0.024*
Total microarthropod biomass	0.06 (0.02)	0.23 (0.21)	0.62 (0.47)	0.133
Microarthropod taxa richness	6.67 (2.52)[a]	15.33 (7.51)[a]	27.67 (1.53)[b]	0.037*
Shannon H index	1.25 (0.50)	1.35 (0.77)	2.44 (0.18)	0.281
Pielou evenness J	1.52 (0.34)	1.12 (0.51)	1.69 (0.12)	0.322

[1] bacterial activity: Thymidine incorporation rate; [2] bacterial activity: Leucine incorporation rate

SOILD
2, 187–215, 2015

Table A1. Biomasses (kg C ha^{-1}) of the microarthropod taxa in the soil food web at three different sites on Crete, Greece (I = intensively cultivated olive orchard, E = extensively used olive orchard, P = pasture). Trophic groups: omnivorous mites (Ommi), fungivorous mites (Fumi), nematovorous mites (Nemi), predatory mites (Prmi), herbivorous mites (Hemi), herbofungivorous mites (HFmi), herbofungivorous collembolans (HFco), fungivorous collembolans (Fuco) and Symphylans (Symp). Numbers represent mean and standard deviation (between brackets), measured in the topsoil (0–10 cm).

Site		I	E	P
Acari				
Astigmata	Ommi	0.0003 (0.0005)	0.0025 (0.0044)	0.0021 (0.0037)
Tyrophagus	Ommi	0.0018 (0.0031)	0.2417 (0.3952)	
Mesostigmata	Ommi			0.0007 (0.0012)
Epicriopsis	Fumi		0.0001 (0.0002)	
Hypoaspis	Prmi	0.0005 (0.0008)	0.0005 (0.0008)	0.0087 (0.0076)
Leioseius	Prmi			0.0069 (0.0088)
Macrocheles	Prmi			0.0045 (0.0079)
Pachylaelaps	Prmi			0.0045 (0.0079)
Rhodacaridae	Prmi			0.0110 (0.0129)
Zercon	Nemi		0.0009 (0.0008)	
Oribatida	Ommi		0.0213 (0.0097)	0.0400 (0.0542)
Aphelacarus acarinus	HFmi			
Brachychthoniidae	HFmi	0.0001 (0.0001)	0.0005 (0.0008)	0.0014 (0.0016)
Ceratozetidae	Ommi		0.0008 (0.0014)	0.0103 (0.0104)
Cosmochthonius	Hemi	0.0002 (0.0003)	0.0003 (0.0002)	0.0008 (0.0007)
Damaeidae	Fumi		0.0003 (0.0003)	0.0005 (0.0009)
Hermanniella	HFmi			0.0005 (0.0009)
Licnodamaeus pulcherrimus	HFmi		0.0011 (0.0008)	0.0002 (0.0003)
Mycobatidae	HFmi			0.0024 (0.0034)
Nanhermanniidae	HFmi		0.0005 (0.0009)	
Oppiidae	Ommi		0.0005 (0.0009)	0.0200 (0.0271)
Oribatellidae	HFmi			0.0008 (0.0006)
Pelopsidae	Fumi			0.0003 (0.0006)
Rhinoppia	Fumi		0.0021 (0.0030)	
Tectocepheus	Ommi			0.0275 (0.0110)
Prostigmata				
Alicorhagiidae	Ommi	0.0136 (0.0058)		
Erythraeidae	Prmi		0.0005 (0.0009)	
Eupodes	Ommi		0.0013 (0.0023)	0.0080 (0.0121)
Microtydeus	Ommi	0.0090 (0.0020)	0.0125 (0.0084)	0.0942 (0.0837)
Nanorchestes	Ommi			0.0119 (0.0154)
Paratydeidae	Prmi		0.0015 (0.0026)	0.0287 (0.0151)
Pyemotes	Prmi			0.0072 (0.0068)
Pygmephorus	Fumi		0.0010 (0.0010)	0.0177 (0.0188)

SOILD
2, 187–215, 2015

Table A1. Continued.

Rhagidia	Prmi		0.0010 (0.0009)	0.0041 (0.0042)
Scutacarus	Ommi		0.0041 (0.0064)	0.0212 (0.0200)
Speleorchestes	Ommi	0.0013 (0.0012)		0.0182 (0.0225)
Stigmaeidae	Prmi			0.0014 (0.0024)
Tarsonemus	Ommi	0.0006 (0.001)	0.0018 (0.0030)	0.2529 (0.1300)
Tydeidae	Ommi	0.0516 (0.0439)	0.0003 (0.0004)	0.0032 (0.0038)
Collembola				
Entomobryomorpha				
Lepidocyrtus	HFco		0.0011 (0.0087)	0.0188 (0.0214)
Lepidocyrtus lignorum	HFco		0.0050 (0.0087)	
Poduromorpha				
Friesea	Fuco			0.0096 (0.0167)
Hypogastrura	Fuco		0.0046 (0.0030)	
Mesaphorura	Fuco			
Onychiuridae	HFco	0.0017 (0.0030)		0.0289 (0.0501)
Onychiurus	Fuco	0.0006 (0.001)		0.0967 (0.0670)
Paratullbergia	Fuco		0.0079 (0.0136)	0.0084 (0.0110)
Symphypleona				
Sminthuridae	Heco	0.0011 (0.0020)		0.0014 (0.0024)
Protura	Fuco		0.0005 (0.0009)	0.0251 (0.0287)
Symphyla	Symp		0.0336 (0.0582)	0.0250 (0.0284)

SOILD
2, 187–215, 2015

Ecological soil quality affected by land use and management

J. P. van Leeuwen et al.

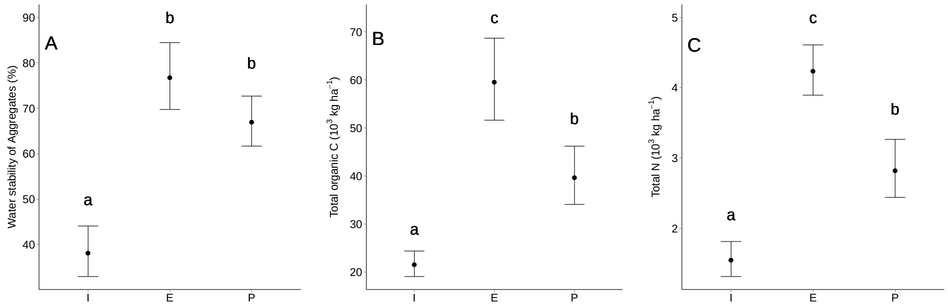

Figure 1. Water stability of aggregates of 1–3 mm (%) **(A)**, total organic C in 10^3 kg ha^{-1} **(B)**, and total N in 10^3 kg ha^{-1} **(C)**, for the three land use types in the Koiliaris Critical Zone Observatory (Crete, GR) (I = intensively cultivated olive orchard, E = extensively used olive orchard, P = pasture), points represent back transformed means, error bars depict standard deviations, and small letters in the graphs (a–c) represent significant differences between sites.

SOILD
2, 187–215, 2015

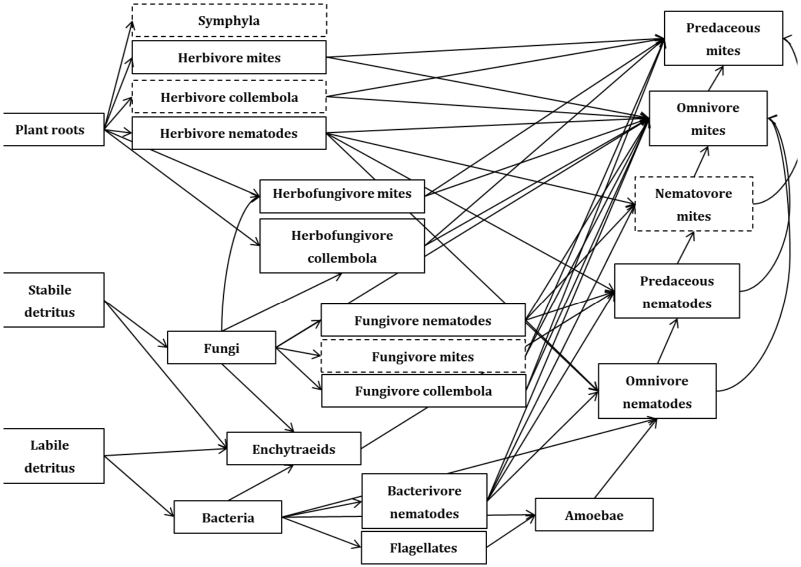

Figure 2. Soil food web diagram representative for all three land use types in the Koiliaris Critical Zone Observatory (Crete, GR). Boxes represent the presence of trophic groups in the soil food web, arrows represent feeding interactions based on diet information (the arrow points from the group eaten to the group that eats). Groups with drawn boxes were present at all sites, groups with dashed boxes were only present at some sites.

SOILD
2, 187–215, 2015

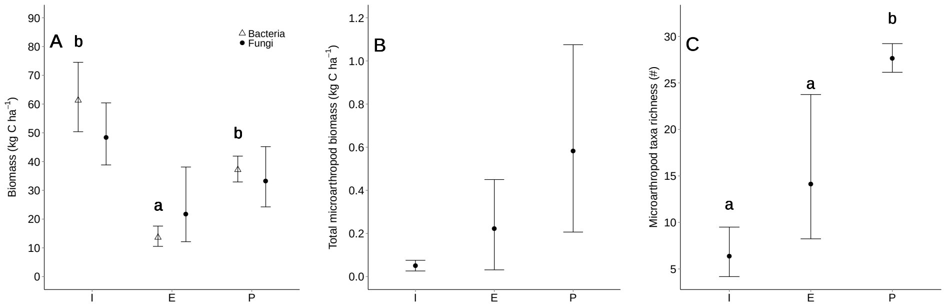

Figure 3. Bacterial (open triangles) and fungal biomass (closed points) in kg C ha^{-1} **(A)**, total microarthropod biomasses in kg C ha^{-1} **(B)**, and microarthropod taxonomic richness **(C)**, for the three land use types in the Koiliaris Critical Zone Observatory (Crete, GR) (I = intensively cultivated olive orchard, E = extensively used olive orchard, P = pasture), points represent back transformed means, error bars depict standard deviations, and small letters in the graphs (a–c) represent significant differences between sites.

SOILD
2, 187–215, 2015

Ecological soil quality affected by land use and management

J. P. van Leeuwen et al.