Università Politecnica delle Marche

Department of Agricultural, Food and Environmental Sciences

Scientific field: AGR/02 - Agronomy and field crops

Ph.D. IN AGRICULTURAL, FOOD AND ENVIRONMENTAL SCIENCES

XV EDITION (2013-2016)

Soil CO_2 emissions and C stock as Ecosystem Services: a comparison between transhumant and conventional farming systems

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ACADEMIC YEAR 2015/16

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General Introduction

1. Preface

Ecosystems and Mankind

Probably the first reference to the linkages between mankind and ecosystems are in date back at least to ancient Greek philosopher Plato (c. 400 BC) who reported that: *"The loss of timber had denuded the hills and plains surrounding Athens and caused massive soil erosion."* (Williams, 2003). The concept of the relation between mankind and ecosystems changed and evolved through the centuries until 1864 where George Perkins Marsh (1801-1882) proposed a modern theoretical framework that challenged the idea that Earth's natural resources are unbounded by pointing out changes in soil fertility (Marsh, 1864). The first concern of Marsh was to highlight the lack of natural resources useful to mankind development and the fragile equilibrium that allow the nature to support the core function of the human life. Within the hypothesis developed by Marsh, the reason behind the collapse of many ancient civilizations where an imperfect balance between economy and ecology. However, just from the second half of XIX century the scientific community started to investigate the linkages between the nature functions and the development of economic systems. The term "*environmental services*" was introduced in a 1970 report of the Study of Critical Environmental Problems (MIT, 1970) which included insect pollination, fisheries, climate regulation and flood control. In following years, variations of the term were used, but eventually "*Ecosystem Services*" became the standard in scientific literature after the publication of the Millennium Ecosystem Assessment (MA, 2005) which defined them as "*The benefits people obtain from ecosystems*".

Transhumant Systems

Transhumance is defined as the seasonal migration of flocks, heard and shepherds from summer to winter pastures and vice-versa. The term derives from the combination of the Latin terms: "*trans*" and "*humus*" which respectively means "across" and "ground". Transhumance developed in Europe, Asia, Australia, Africa and South America with many cultural and technological variations especially between the continents. Transhumance is different from the practice of nomadism since the winter and summer pastures are fixed and are not influenced by other factors (e.g. seasonal trends). Traditionally, the herded animals were moved along specific drove paths (e.g. "*Cañadas*" in Spain or "*Tratturi*" in Italy) even for hundreds of kilometers. Nowadays traditional transhumance disappeared in favor of smaller distance and vertical animal movements between lowland and upland pastures (Caballero et al., 2009). Many studies highlighted how the disappearance of the drove systems of the traditional transhumance *per se* lead to a decrement of biodiversity and many other Ecosystem Services linked to them (e.g. Azcárate et al., 2013; Juncal et al., 2014; López-Santiago et al., 2014 Russo et al., 2014;)

In Italy, transhumance was practiced with sheep and its system was structured since the Middle Age in the central and southern Apennines, with an organized pastoral society from Abruzzo to Apulia regions. It was also practiced within the former Papal state including Emilia-Romagna, Marche, Umbría and Lazio regions. Nowadays, a meat-oriented sheep transhumant system is still present in a large scale grazing system 4000Km^{-2} area of Central Italy (including Lazio, Umbria, Marche and Northern Abruzzo). The flocks (composed by 500-5000 sheep) graze upland pastures from June to the beginning of autumn reaching the highest pastures (up to 2400m a.s.l.) during July and August. In autumn-winter flocks are progressively transferred to the Adriatic coast where they mainly graze alfalfa (*Medicago sativa*) fields (Caballero et al., 2009).

2. Aims and structure of the thesis

The scientific literature suggest that transhumant systems have the potential to deliver a wide array of Ecosystem services (Karatassiou et al., 2015; López-Santiago et al., 2014; Oteros-Rozas et al., 2014). On the other hand, as pointed out by the MA (2005), researches on Ecosystem Services need take in consideration many factors like biodiversity relation or tradeoffs between the services.

For these reasons, in agreement with the Millennium Assessment, this thesis aims to provide a frame of a picture that include the Ecosystem Services provided by sheep transhumant systems in Central Italy. The thesis mainly focuses on the comparison of Soil C dynamics of transhumant with conventional farming system along a transect (from the coastal area to the mountain area of Central Italy) following the seasonal movements of the flocks. The present researches are included in a larger and ambitious project that envisage the assessment of other Ecosystem Services like Food (a parallel PhD thesis concerning lamb meat quality provided by the same system), the inclusion of bio-indicators (carabidae monitoring), plant biodiversity (interpreted as the basis of the Ecosystem Services) and the further development of indicators that will be able to estimate the cultural Ecosystem services (e.g. stakeholders landscape perception).

The body of the thesis includes three articles based on researches realized during the period of PhD:

1. A literature Review entitled: "**Ecosystem Services (ES) provided by livestock grazing systems: a review of trends and approaches**". This chapter deals with the ES approach that has become a popular instrument for the assessment and valuation of ecosystems and their functions. It is focused on grazing lands and their potential to deliver ESs depending on management practices and intensity. The paper reviews the trends and approaches used in the analysis of some relevant ESs provided by grazing systems in line with the Millennium Ecosystem Assessment (MA) framework principles. The main findings of this review that included 62 papers are that i) some

papers misunderstood the concept of ESs as defined by the MA (e.g. lack of anthropocentric vision); ii) 34% of the papers dealt only with 1 ES neglecting the need for the multisectoral approach suggested by the MA; iii) only a few papers included stakeholder (SH) involvement to improve local decision-making processes; iv) cultural ESs were poorly studied despite being considered the most relevant for local and general SHs; v) SH awareness of the well-being provided by ESs in grazing systems could foster both agri-environmental schemes and the willingness to pay for these services.

2. A full research paper entitled: **"Soil respiration dynamics of wheat and alfalfa under Mediterranean conditions"**. Within the second chapter are reported soil respiration dynamics and the soil C stock of alfalfa fields (used by sheep transhumant systems as winter pastures) in comparison with wheat fields (that represent one of the main crops of Mediterranean climate). The main objective of this research are i) to measure annual variation in soil respiration, ii) to study the effect of the two major drivers (Soil T and SWC) of this variations and iii) assess the soil C stock. Despite this study provides novel insight into the drivers of soil respiration and contributes to enlarge the current limited scientific knowledge, further analyses are needed to fully understand the factors that regulate soil CO_2 effluxes under Mediterranean conditions.

3. A third research paper entitled: **"Soil respiration dynamics and Carbon Stock of mountain agricultural systems in Central Italy"**. In this chapter are presented the results of seasonal soil respiration dynamics in the snow-free period (from April to November) in Central Italy uplands. The research was conducted in three experimental fields composed by a portion of grasslands (used by sheep transhumant farm as summer pasture) and two adjacent arable fields cultivated with spelt and lentil. The main objectives of this research were to investigate changes generated from a land cover change from (grasslands

to croplands) in a temperate mountain area. In particular, seasonal variation in soil respiration was measured as well as soil quality indicators like soil C stock or Humification rates. This paper provides new data concerning C cycle dynamics within a context previously not described in international literature.

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Chapter I: Ecosystem Services (ES) provided by livestock grazing systems: a review of trends and approaches

1. Introduction

Although the first references to the concept of 'ecosystem functions, services and values' date back to around the 1960s, the number of scientific papers concerning Ecosystem Services (ES) has grown exponentially in the last few decades (de Groot et al., 2002) and particularly after the Millennium Ecosystem Assessment (MA) was published (Fisher et al., 2009). The Millennium Ecosystem Assessment (Alcamo et al., 2003; MA, 2005) represents one of the most extensive and widely accepted studies on the links between human well-being and the world's ecosystems. The MA defines the ecosystem as *"a dynamic complex of plant, animal (including humans), and microorganism communities and the non-living environment interacting as a functional unit"* and ecosystem services as "the benefits people obtain from ecosystems*"*.

The MA identified four groups of ES: (i) Supporting: services necessary for the production of all other ES (e.g. soil formation and nutrient cycling) whose impact on people is either indirect or occurs over a very long time; (ii) Provisioning: products obtained from ecosystems, such as food and fresh water; (iii) Regulating: benefits obtained from the regulation of ecosystem processes, such as climate and disease regulation; (iv) Cultural: non-material benefits people obtain from ecosystems through spiritual enrichment, cognitive development, reflection, recreation and aesthetic experiences. A second key study concerning ES is The Economics of Ecosystems and Biodiversity (TEEB, 2010) which defines ES as *'the direct and indirect contributions of ecosystems to human well-being'*, and separates the concepts of services and benefits (welfare gains generated by ES), considering supporting services merely as ecological processes and not strictly as ES.

If on the one hand every ecosystem is able to produce a large number of ES (Alcamo et al., 2003; MA, 2005), on the other, ecosystems may also produce Ecosystem Disservices that are harmful or detrimental to human well-being (von Döhren and Haase, 2015). Thus, the term “ecosystem service” is anthropocentric and is intended to have a positive sense. This vision is one of the recurring critiques of the concept of ES and, according to Schröter et al. (2014), the ES concept is not meant to replace biocentric arguments but it groups together a wide variety of anthropocentric arguments for the protection and sustainable human use of ecosystems. Schröter et al. (2014) also counter-argued six other main critiques to the ES concept derived from scientific literature.

Ecosystem Services are spatial-scale and time-scale dependent, and there is a risk that spatial scale mismatches between ecological processes and decision-making will occur. For this reason, the need for an integrated approach which also takes into account the local knowledge of stakeholders (SHs) is a key requirement in assessing ES (Alcamo et al., 2003; MA, 2005; Reed, 2008).

According to the MA (2003) and TEEB (2010), ecosystems and biodiversity are closely related concepts although the latter is not strictly considered as an ES but rather as a source or a regulator of the former (Harrison et al., 2014). The knowledge gap regarding both the links and the difficulties in understanding the relationships between ES and biodiversity has been highlighted by many authors (e.g. Jax and Heink, 2015; Sircely and Naeem, 2012; Harrison et al., 2014).

Livestock systems occupy about a third of the planet’s ice-free terrestrial surface and represent an important source of income, or may even be essential, for the survival of vulnerable human communities. In these systems, grazing lands could deliver a large and differentiated number of ES (Porqueddu et al., 2016; Tarráson et al., 2016). These services in turn are dependent on different management practices (Fischer et al., 2010; Steiner et al., 2014), such as different grazing regimes (Ford et al., 2012).

The aim of this paper is to review the current trends and approaches used in the analysis of some relevant ES provided by livestock grazing systems under different site, climate and management conditions. The results of the review will be used to derive recommendations for research activities in the analysis of ES.

2. Grazing systems: classification criteria and terminology

To date no unique classification of livestock systems is available (Robinson et al., 2011). Broadly defined, livestock systems are a subset of farming systems (Ruthenberg, 1980), in which livestock contributes more than 10 percent to total farm output (Seré and Steinfeld, 1996), with similar enterprise patterns, livelihoods and resource base (Dixon et al., 2001).

A more livestock-oriented classification of farming systems was developed by Seré and Steinfeld (1996) for 'solely livestock production systems', dividing them into grassland-based (LG) and landless (LL) systems. In LG and LL systems the dry matter fed to animals is 10 percent higher and lower than the feed produced on the farm, respectively, and in which annual average stocking rates are below and above 10 standard livestock units per hectare of agricultural land, respectively. An interactive map of their distribution is provided by the FAO (2016) Global Livestock Production and Health Atlas (GLiPHA).

Ecosystem classification is performed according to the various fields of research. Biomes are the most basic units that ecology uses to describe global patterns of ecosystem form, process and biodiversity (Ellis and Ramankutty, 2008). Historically, biomes were identified and mapped according to general differences in vegetation types associated with regional variations in climate (Matthews, 1983; Olson et al., 2001). Further classifications dealing with potential land uses for agriculture in a geographical context are agro-ecological zones devised by the

FAO, which have been widely applied at global, regional and national levels (FAO, 2011; FAO and IIASA, 2007). Considering the first classification, only some biomes provide the necessary conditions for livestock systems (e.g. tundra, taiga, steppe, savanna). In the second classification, ecological zones are divided based on the length of the grazing period and potential evapotranspiration.

In an attempt to relate livestock systems and agroecosystems, land use types emerge. 'Rangelands' include land on which the indigenous vegetation (climax or subclimax) is predominantly grasses, grass-like plants, forbs or shrubs that are grazed or have the potential to be grazed, and which are used as a natural ecosystem for the production of grazing livestock and wildlife (natural grasslands, savannas, shrublands, many deserts, steppes, tundras, alpine communities and marshes). 'Grazing lands' extend the potential land use from natural compositions to any vegetated land that is grazed or has the potential to be grazed by animals (domestic and wild) (Allen et al., 2011). This term is all-inclusive and covers all kinds and types of land that can be grazed (rangelands and artificial pastures).

Within both the ecosystem and the agro-ecological classifications, a set of terms is in use to distinguish between systems and management practices. The applied terms mainly reflect the relationship between the exploitation of land and vegetation type, such as pastoralism (land-use systems in which grasslands and shrublands are exploited through grazing) and silvopastoralism (land-use systems and practices in which trees and pastures are deliberately integrated with livestock components). While the first and second terms relate directly to the management practice, agroforestry (land-use systems or practices in which trees are deliberately integrated with crops and/or animals on the same land management unit) merely indicates a relationship between forestry and agriculture on a territorial unit.

In the context of this review, grazing systems include the production systems in which grazing is one of the main management practices adopted across the grazing lands.

3. Links between biodiversity and ecosystem services

Biodiversity is the variability between living organisms and includes diversity within and among species and ecosystems. It is the source of many goods and services, such as food and genetic resources, and changes in biodiversity can influence the supply of ES (Alcamo et al., 2003). Subsequently the MA (2005) defined biodiversity as a necessary condition for the delivery of all ES and, in most cases, a greater level of biodiversity is associated with a larger or more dependable supply of ES.

According to the MA (2005) biodiversity is both a response variable affected by global change drivers (e.g. climate or land use change) and a factor modifying ecosystem processes and services and indirectly, human well-being (e.g. health or freedom of choice and action). Changes in human well-being may lead to the modification of management practices with direct effects on ecosystem processes and biodiversity (Fig. 3.1).

Although the MA describes a unilateral relationship between biodiversity and ES, some authors consider biodiversity as a service in its own right; for example, as the basis of nature-based tourism (van Wilgen et al., 2008), while others consider biodiversity and ES as synonyms (Mace et al., 2012).

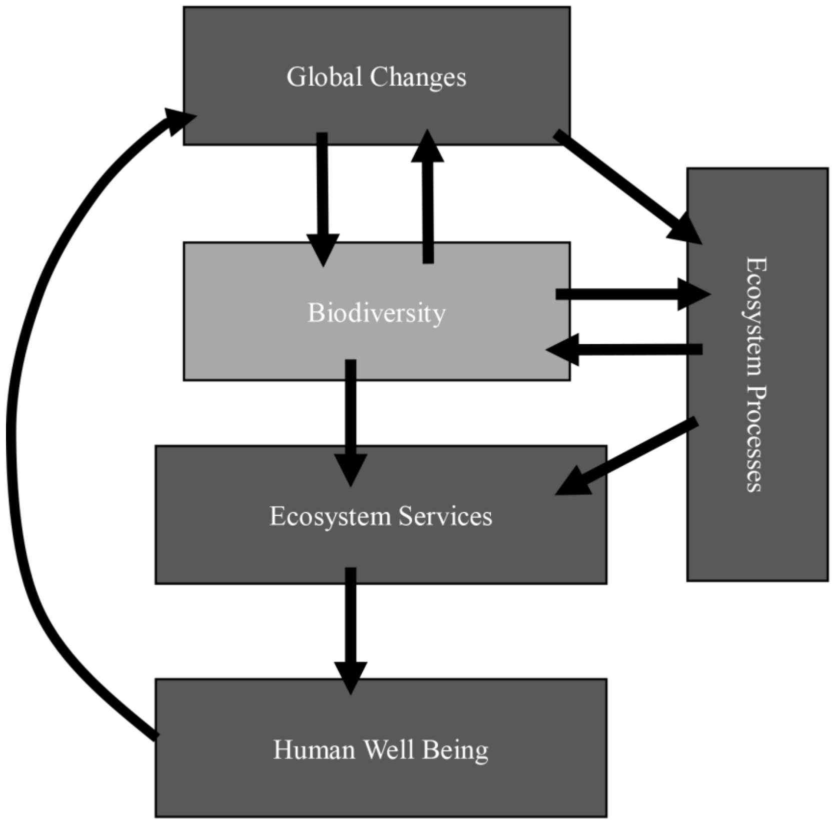

Figure 3.1. Interrelations between biodiversity, ecosystem functioning and ES (modified from MA, 2005).

Habitat provisioning is one of the main ecosystem services linking the effects of livestock grazing to the biodiversity of the host ecosystem (FAO, 2014). Habitat services arise from the direct interaction of animals with their environments, and are hence related to land management practices, especially in grazing systems. Unlike the MA (2003, 2005), the TEEB (2010) considers habitat services as a separate category. In accordance with these documents, this review considers habitat services within supporting services, because of their interconnected nature, as well as their shared roles in underpinning the delivery of other services.

4. Bibliographical search and analysis criteria

This review is based on the ES provided by grazing systems as categorized and found to be prominent by the FAO (2014) (Table 3.1). Among these, the ES relevant to the expertise and background of the authors were analysed in detail: Primary Production, Habitat for Species, Food, Land Degradation Prevention, Water Quality Regulation, Regulation of Water Flows, Climate Regulation, Moderation of Extreme Events, Natural (Landscape) Heritage.

Papers dealing with ES were selected in January 2016 using the Web of Science™ (WoS) selecting 'topic' as the search option. The basic string *"ecosystem service*" and ("grassland*" or "rangeland*" or "shrubland*" or "scrubland*") and "grazing"* was used as input in the 'field search' ('basic search') starting from 2004 as the 'timespan'. In order to select papers for each analysed ES, specific search terms were added to the basic string according to the keywords (Table 3.1) included in the FAO report (2014). The search terms are reported in detail in each ES section. The following papers were excluded from the review: (i) papers dealing with ES not analysed, (ii) reviews, editorials and meta-analyses, (iii) papers not adopting the MA framework, and (iv) those not directly analysing the ES for which they were extracted.

Table 4.1. Papers dealing with ES provided by grazing systems returned by the basic string in the Web of Science (WoS) and after selection according to the review criteria. Each paper can deal with more than one ES (WoS: 155 papers; eligible for the review: 62 papers).

ES Group	Ecosystem service	Description	Papers extracted in the WoS[1]	Papers responding to analysis criteria[2]	
			number	number	%
Supporting	Maintenance of soil structure and fertility	Nutrient cycling on farm and across landscapes, soil formation	12	not analysed	not analysed
	Primary production	Improving vegetation growth/cover	72	39	63
	Habitat services *(as part of supporting services)*				
	Maintenance of life cycles of species	Habitat for species, especially migratory species	78	35	56
	Habitat connectivity	Seed dispersal in guts and coats	2	not analysed	not analysed
	Maintenance of genetic diversity	Gene pool protection and conservation	0	0	0
Provisioning	Food	Meat, milk, eggs, honey, wool, leather, hides, skins, etc.	12	6	10
	Fertilizer	Manure and urine for fertilizer	9	not analysed	not analysed
	Fuel	Manure and CH_4 for energy, manure biogas, etc.	11	not analysed	not analysed
	Power	Draught animal power	0	0	0
	Genetic resources	Basis for breed improvement and medicinal purposes	10	not analysed	not analysed
	Biotechnical/Medicinal resources	Lab. animals, test-organisms, biochemical products	0	0	0
Regulating	Waste recycling and conversion of non-human edible feed	Recycling of crop residues, household waste, swill, primary vegetation consumption	1	not analysed	not analysed
	Land degradation and erosion prevention	Maintenance of vegetation cover	26	10	16
	Water quality regulation/purification	Water purification/filtering in soils	8	5	8

	Regulation of water flows	Natural drainage and drought prevention, influence of vegetation on rainfall, timing/magnitude of runoff/flooding	44	15	24
	Climate regulation	Soil C sequestration, GHG mitigation	60	31	50
	Moderation of extreme events	Avalanche and fire control	19	4	6
	Pollination	Yield/seed quality in crops and natural vegetation; genetic diversity	17	not analysed	not analysed
	Biological control and animal/human disease regulation	Destruction of habitats of pest and disease vectors; yields	3	0	0
Cultural	Opportunities for recreation	Eco/agro-tourism, sports, shows and other recreational activities involving specific animal breeds	50	not analysed	not analysed
	Knowledge systems and educational values	Traditional and formal knowledge about the breed, the grazing and socio-cultural systems of the area	23	not analysed	not analysed
	Cultural and historic heritage	Presence of the breed in the area helps to maintain elements of the local and/or culture that are valued as part of local heritage; cultural identity	21	not analysed	not analysed
	Inspiration for culture, art and design	Traditional art /handicraft; fashion; cultural, intellectual and spiritual enrichment and inspiration; pet animals, advertising	12	not analysed	not analysed
	Natural (Landscape) heritage	Values associated with landscape as shaped by animals themselves or as a part of landscape, e.g. aesthetic values, sense of place, inspiration	39	4	6
	Spiritual and religious experience	Values related to religious rituals, human life-cycle such as religious ceremonies, funerals or weddings	0	0	0

1: 155 papers extracted in the WoS;
2: 62 papers responding to analysis criteria

5. General trends

The query returned a total of 155 papers (Table 3.1) with an increasing trend starting from 2010 (Fig. 4.1). The multiple occurrence of different ES within single papers results in a total of 529 findings, within the 155 papers. Most of the papers dealt in particular with supporting (mostly on primary production and maintenance of life cycles of species), regulating (in particular climate and water flow regulation) and cultural ES (recreation opportunities and landscape). Only a few papers dealt with provisioning services and, surprisingly, very few with food. According to the review criteria, 62 papers (147 findings) were eligible for the analysis. The predominance of papers dealing with Primary Production (63% of the papers), Habitat for Species (55%) and Climate Regulation (50%) was confirmed. Natural (Landscape) Heritage apparently assessed in 25% of the papers (WoS) proved to be analysed as a cultural ES in only 6% of the papers. Many of the WoS findings derived from mentions of the term 'ecosystem service' in the text (e.g. just in the introduction or conclusions) or from a mere mention of ES (e.g. food or landscape) and were not analysed by the authors.

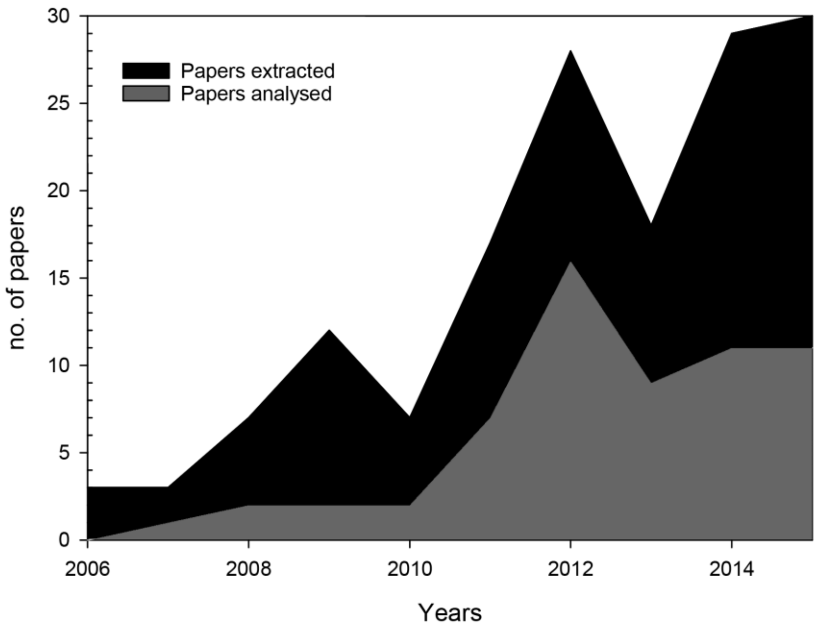

Figure 5.1. Number of papers extracted in the Web of ScienceTM and analysed according to the review criteria.

Despite the fact that the MA (2005) recommended the implementation of a multisectoral approach to fully evaluate changes in ES, their interactions, trade-offs and impact on people, 34% and 23% of the 62 papers analysed just one or two ES, respectively (Fig. 4.2). Only 11 percent of the papers dealt simultaneously with more than five ES with respect to the MA principles.

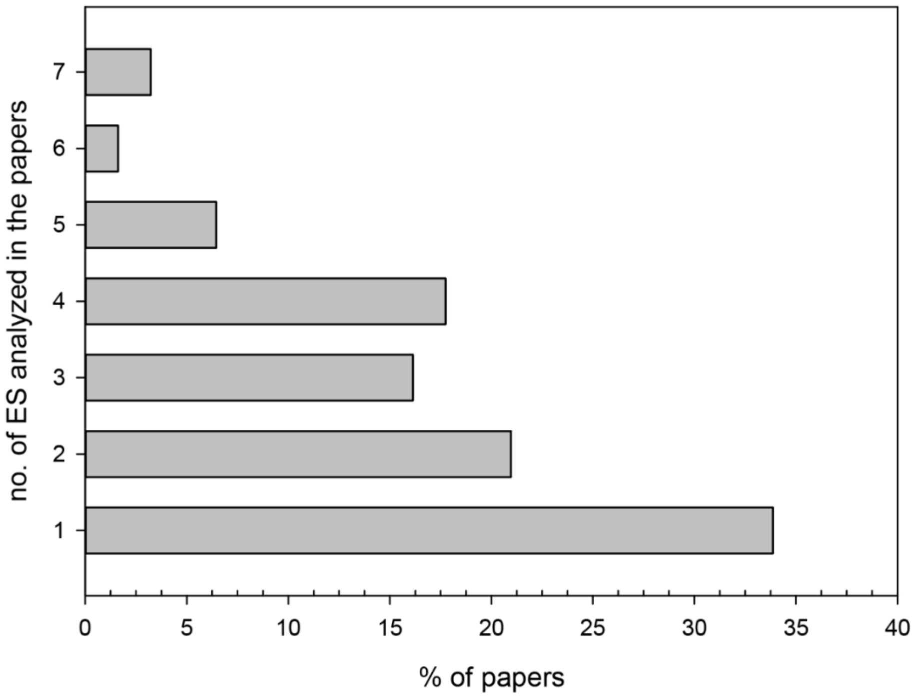

Figure 5.2. Percentage of papers eligible for the review (n=62) dealing with one or more Ecosystem Services.

6. Trends and approaches in ES analysis

Primary production (PP) is a fundamental service defined in the MA (2003) as the assimilation (gross) or accumulation (net) of energy and nutrients by green plants. Maintaining or enhancing the productive capacity and resilience of grazing land ecosystems is critical for the continued support of livelihoods and the ES that benefit society at large (Teague et al., 2015). According to the MA, the authors considered PP as a provisioning ES when harvested and sold outside the commercial fields or as a supporting ES if basic feed for wild or domestic animals.

Papers were extracted according to the additional string *("primary production" or "vegetation growth" or "vegetation cover" or "vegetation" or "NPP" or "net primary production")*. The search returned 72 papers but only 39 met the analysis criteria (Tab. 3.1). Primary production was mainly analysed with Habitat for Species or Climate Regulation ES (Fig. 5.1). Contrary to expectations, only 5 papers analysed the relationship between PP and Food ES (e.g. meat and milk quality and yield). The lack of studies that analyse PP together with other ES limits the assessment of trade-offs, synergies and relationships in response to diverse site-specific, land use and management conditions. Examples are the analysis of the effect of vegetation cover on soil fertility, soil loss or runoff (Giese et al., 2013; van Oudenhoven et al., 2015).

Primary production was mainly assessed as above-ground biomass, often in combination with other characteristics (e.g., mainly the below-ground component, but also litter, vegetation cover, herbage nutritive value, etc.) in several rangeland ecosystems, under different site, climate and management conditions. Different methods and approaches were used for assessing PP including direct field-based surveys (e.g. Oñatibia et al., 2015) at different spatial and temporal scales. Field-plot experiments assessed the effects of different types of management (e.g. mowing, grazing and undisturbed or abandonment) and intensities on PP in short but also long-term (e.g. Marriot et al., 2010) monitoring. In these studies, plot dimensions varied from a minimum of 0.45 (Marriot et al., 2010) up to 170 ha (Medina-Roldán et al., 2012) in designs with 2-4 replicates and included enough heterogeneity to reduce pseudo-replication effects. In other cases, landscape scale was applied to take into account management or site conditions on farms or along transects. In multiple zonal grazing land along climatic and management gradients (e.g. Medina-Roldán et al., 2012; Sasaki et al., 2012), transects were used to assess the effects of grazing on the PP (e.g. above- and below-ground and litter biomass, C : N : P stoichiometry). To overcome the limits to account for the spatial and temporal variation of the field-based methods, remote sensing and simulation models were used to assess and monitor grazing land dynamics and their ability to provide ES according to management strategies. Sant et al. (2014) used high-resolution imagery as enhanced ground samples to assess the vegetation cover as an indicator of range condition in order to develop improved management prescriptions. On other sites, several authors (e.g. Schaldach et al., 2013; Teague et al., 2015) used models both for the simulation of ecological, environmental and economic effects under various combinations of changing livestock, climate and management conditions and for an integrated analysis of land cover changes. The results of the simulation approach proved to be useful tools, allowing for a more complete analysis of the impact of different types of management when integrated with field data. The ES method is recognized as a suitable tool (e.g. Schaldach et al., 2013) to support climate adaptation strategies integrating both ecological and socio-economic aspects. Nevertheless, none of the analysed papers highlighted the inclusion of a participatory process or stakeholder involvement as performed by Tarrasón et al. (2016).

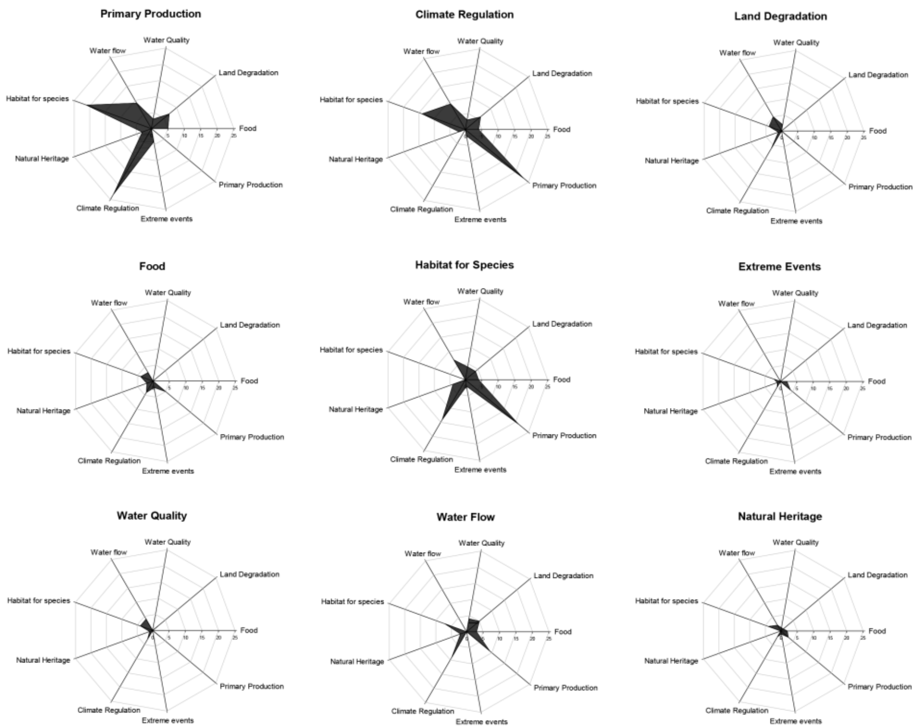

Figure 6.1. Multisectoral approach in the 62 papers eligible for the review reporting the number of papers in which the Ecosystem Services are analysed in combination with each of the others. For example, Primary Production is analysed in 23 papers in combination with Climate Regulation and in 21 papers in combination with Habitat for Species.

Habitat services (HSs) facilitate the life cycles of animals and plants, prevent the occurrence of less valuable ecological states through the encroachment of bush and/or invasive species, and conserve the wildlife and protected areas found in coevolved landscapes (FAO, 2014). The most important clusters of HSs provided by livestock are those that support the maintenance of species life cycles and those related to the connection of habitats (FAO, 2014). Papers were extracted according to the additional criteria *("species" or "habitat" or "life cycle")* and returned 78 papers, of which only 34 fulfilled the review criteria and were hence analysed (Tab. 3.1). As previously pointed out, this ES was mainly analysed in combination with PP and with Climate Regulation (Fig. 5.1); 10 out of 34 papers analysed HSs alone, disregarding the need for the multisectoral approach suggested by the Millennium Assessment.

Plants were the most studied factor (e.g. Duru et al., 2015) followed by pollinator (e.g. Cole et al., 2015) and non-pollinator (e.g. Cole et al., 2012) insects. The literature review highlighted that HS assessment methods are species (animals, plants, etc.) and mainly spatial-scale dependent. Transect surveys were mostly used for species sampling at different scales (e.g. for Carabidae diversity, Cole et al., 2012). Random sampling was used in field-plot experiments (e.g. Boughton et al., 2013) or at farm level for assessing species diversity/abundance and vegetation cover. The point-quadrat method (e.g. Klumpp and Soussana, 2009) and the abundance/dominance method (Fontana et al., 2014) were generally used for the vegetation survey. The most frequent indicators used for both plant and animal species were the following: species richness (e.g. Duru et al., 2013), abundance (e.g. Stein et al., 2014), Shannon diversity (e.g. Fontana et al., 2014), evenness (e.g. Cole et al., 2015) and Simpson index (e.g. Franzén and Nilsson, 2008). A GIS, sometimes in combination with remote sensing technologies, was used to analyse land use (e.g. Fontana et al., 2014) or to identify scenarios of biodiversity trajectories (e.g. Lindborg et al., 2009). Modelling was used to simulate vegetation dynamics, also in relation to other ES (e.g. cattle grazing and elk hunting, Hussain and Tschirhart, 2013), and to identify scenarios related to climate change (e.g. Peringer et al., 2013) or to land use management options (e.g. for biodiversity conservation, Lindborg et al., 2009). Others researchers used SH involvement to provide supporting tools for the sustainable management of grazing land (Fisher et al., 2011).

Food and other livestock related products (FPs) in grazed ecosystems include provision of high-protein meat and dairy products along with leather and other by-products of livestock production (Steiner et al., 2014). From this analysis, the extraction string *("meat" or "milk" or "honey" or "wool" or "leather" or "hide" or "skin" or* *"wax")* revealed 12 publications, only 6 of which were developed in the ES framework (Fig. 5.1 and Tab. 3.1). Despite the fact that livestock production is clearly related to the forage characteristics of the pastures (e.g. yields, quality, species diversity, plant active compounds) (Lieber et al., 2014) the papers extracted integrated Food and other ES for a more holistic analysis. In particular, using different methodologies, they analysed the effects of several scenarios (e.g. climate change, policies, management) on the provision of FPs. Koniak et al. (2011) addressed issues related to honey production and developed a mathematical model which predicts the dynamics of multiple services in response to management scenarios (grazing, fire and their combination), mediated by vegetation changes. In this paper, the potential contribution to honey production was combined with other ES from different groups, despite their different nature, into one 'ES basket'. Bernués et al. (2014) combined FPs with other ES in a focus group with stakeholders to discuss the effects of different scenarios generated by contrasting policies on product quality linked to the territory. Dong et al. (2012 and 2014) use the emergy approach to calculate the performance of several ES under different systems and scenarios to support local resource management and larger-scale environmental resource decision-making.

Land degradation and soil erosion (LD) are not seen just as a loss of soil and fertility but also as a deterioration of balanced ecosystems and the loss of ES (Nachtergaele et al., 2011). The additional string *("land degradation" or "erosion" or "cover crop*" or "vegetation cover")* returned 26 papers. According to the analysis criteria, only 10 of these were eligible for this review (Table 3.1). The ES was assessed in combination with many other ES (Fig. 5.1) rather than as a single and separate ES. This approach aimed not only to produce tools and a framework to support the stakeholders' decision-making processes regarding land management but also to identify new economic instruments (e.g. Payments for Ecosystem Services) designed to enhance the flow of ecosystem services that support livelihoods in rangeland systems (Reed et al., 2015).

For example, a participatory methodological framework was used to identify features of LD and links with other ES provisions (Tarrasón et al., 2016). This study designed a four-step methodological framework to integrate local and scientific knowledge within a participatory assessment of land degradation in a pastoral system. Field visits, in-depth interviews with key informants and farmers produced information that was integrated with scientific knowledge validated by focus groups and then used in a state-and-transition conceptual model. Field data on cover vegetation and plot life forms were used in thematic working groups with different SHs to discuss the results of the previous phases and to develop adaptive management options to maintain or improve ES. The same model was validated by Miller et al. (2011) with field studies conducted in a semi-arid grassland ecosystem in the USA, quantifying structural and functional attributes related to the states and processes represented in the model. Moreover, a wind erosion simulation model was used to investigate the effects of measured biophysical attributes on predicted rates of wind-driven soil movement at plot scale. A global scale research was performed by Petz et al. (2014a) using a combined approach of literature review, data and models (e.g. 'IMAGE-USLE') to study the interactions between input data, livestock density and ES in order to strengthen and optimize the choices of local stakeholders for the future management of the area.

Water quality regulation/purification (WQ) is an ES that is linked directly to human welfare. Ecosystems can be a source of impurities in fresh water but can also help to filter and decompose organic wastes introduced into inland waters (Alcamo et al., 2003). The additional extraction string *("water quality" or "water regulation" or "water purification" or "water filtering in soil")* returned 8 papers of which 5 were eligible for the analysis (Table 3.1; Fig. 5.1). Like the previous ES, mainly a holistic approach combining many ES was applied by the authors involving stakeholders to explore the relationship between land management and ES. For example, Fisher et al. (2011) explored the variation in ES delivery resulting from different management practices in UK wetlands. In particular, the role of species-led (both animals and plants) management on biodiversity was investigated. In a following step, a consultation with SHs and experts was carried out through workshops and meetings to elaborate specific details of management impact on ES, including hydrology. Three categories of key ES (and disservices) were identified and linked to the range of management. These results are particularly relevant for the drafting of management plans that should carefully balance the effects of management practices. One example in this sense was provided by Van Horn et al. (2012) who suggested taking into account grazing-related effects on some ES, such as water-quality parameters like turbidity and temperature. Lamarque et al. (2014) applied a role game in which farmers were faced with changes in ES under climatic and socio-economic scenarios and prompted to plan for the future and to take land management decisions as deemed necessary. The results demonstrated that farmers were not aware of the potential effects of their activities on nitrate leaching and that feedback loops between ES and land management decisions could favour more sustainable ES management.

Regulation of water flows (WF) in MA (2003) deals with the timing and magnitude of runoff, flooding, and aquifer recharge which can be strongly influenced by changes in land cover, including alterations in the water storage potential of the system. The addition of specific search terms *("water" or "natural drainage" or "drought prevention" or "runoff" or "rainfall" or "flooding")* to the basic string produced 44 papers. Fifteen of these (Table 3.1) analysed WF as an ecosystem service in grazing systems. This ES was mainly analysed in combination with PP and with Climate Regulation (Fig. 5.1).

The analysis revealed different scale approaches used in the papers. A large scale analysis of WF with WQ as previously described in WQ was carried out by Fisher et al. (2011). At catchment scale, Petz et al. (2014b) evaluated alternative land management scenarios with SH involvement by mapping and modelling multiple ES, including water supply. Other authors used the InVEST model to assess water supply. At catchment scale, Pan et al. (2015) studied the effects of spatial/temporal variation and the effects of land use change on water supply. The input variables and parameters for InVEST were land use/cover and the territorial characteristics derived by a digital elevation model (DEM). Field experiments were conducted by Ford et al. (2012) to estimate ES from grasslands in three replicated experimental blocks, each containing three 10×10 m plots, identifying different management treatments (different grazing animals and stocking rates, un-grazed). Soil/vegetation characteristics and invertebrates were analysed to assess the effects of management on WF. Inauen et al. (2013) studied the effect on the water balance of the grazing reduction in four alpine grassland types and the consequent provisioning of fresh water and the potential for hydroelectric power production. Lysimeters were used in field experiments under free-air CO_2 enrichment, to solve the issue of the hydrological water balance.

Climate regulation (CR) influences the climate both locally and globally. For example, at local level, changes in land cover can affect both temperature and precipitation, while on a global scale ecosystems play an important role in climate regulation by either net sequestration or net emissions of greenhouse gases. This ES is receiving increasing attention since the effects of climate change over the next century are expected to affect, directly and indirectly, all types of ecosystems and ES (MA, 2005). The extraction string *("climate" or "soil carbon" or "greenhouse gas*" or "GHG" or "CO_2" or "CH_4" or "N_2O")* provided 60 papers and 31 (50% of the total number of analysed papers) were eligible for the analysis (Tab. 3.1). As stated previously, this ES was mainly analysed in combination with Primary Production (23 papers) and Habitat for Species (15 papers) (Fig. 5.1).

The literature review shows that a thorough analysis of soil C pool and CO_2 fluxes has been performed while other GHG, such as CH_4 and N_2O have been less investigated. Different approaches and scales were adopted by the authors. Field-scale experiments were conducted by Ford et al. (2012) in fixed sand dune grasslands in the UK, in order to investigate C stock from soil, roots, litter and shoots under different management systems. Marriott et al. (2010) investigated soil total C and N in pastures under different management options. A landscape scale was used for assessing N- and C-cycling in grazed and non-grazed upland grassland in northern England (Medina-Roldán et al., 2012). Transect analysis was carried out by Farley et al. (2013) to examine soil and above-ground C in 8 sites in Ecuador. Klumpp and Soussana (2009) extracted monoliths from two contrasted long-term field treatments (high vs. low grazing disturbance) and exposed to both low and high (simulated grazing) disturbance during a 2-year experiment. Subsequently, a mathematical framework was used to predict changes in C fluxes after grazing disturbance.

Predictive models for grassland dynamics were used by Peringer et al. (2013) in woodland pastures while Scheiter et al. (2015) used a dynamic vegetation model to project how climate change and fire management might influence future vegetation in northern Australian savannas. Carbon fluxes from natural grasslands under different grazing pressures were assessed by dynamic carbon models (Dong et al., 2012). Koniak et al. (2011) applied a mathematical model to study the relationships between C retention in woody plants and other ES. Concerning the expected progressive increment of CO_2 concentration in the atmosphere, experiments related to changes in botanical composition in grassland were carried out by Newton et al. (2014) and Inauen et al. (2013) using the Free Air Carbon Dioxide Enrichment (FACE) technique. To identify the most desirable management options for lowland wet grassland, Fisher et al. (2011) analysed the management plans and annual reports for 22 UK reserves. Services and disservices including GHG fluxes were used in SH meetings as support tools for discussion and awareness.

The need to focus on the long-term protection of rangelands to conserve soil carbon rather than focusing on annual fluxes and management initiatives was underlined by Booker et al. (2013). Their study illustrates the misalignment between policies targeting vegetation management for enhanced carbon uptake and carbon dynamics on arid United States rangelands. Carbon uptake on arid and semi-arid rangelands is most often controlled by abiotic factors which are not easily changed by the management of grazing or by vegetation. For this reason, the authors conclude that the current carbon policy, as exemplified by carbon credit exchange or offsets, could result in a net increase in emissions.

Moderation of extreme events (EE) is mainly referred to the ability of livestock grazing to prevent avalanches and wildfires (FAO, 2014). The additional string *("avalanche*" or "fire" or "extreme event*")* produced 19 papers, 4 of which were eligible for this review (Table 3.1; Fig. 5.1). The extracted papers dealt only with 'fire', thus highlighting a lack of studies on other events.

Rather than being considered as an 'extreme event', fire is analysed by the excluded papers as a management tool to enhance other ES (e.g. habitat provisioning, prevention of wildfires, etc.). For example, Joubert et al. (2014) investigated the effect of annual burning on plant species richness, composition and turnover in three firebreak types under different cattle grazing levels. Boughton et al. (2013) conducted an 8-year split-plot experiment studying the effect of season of burn on plant composition in semi-natural grassland in Florida (USA) where, in addition to prescribed winter burns, natural historical wildfires occur in abandoned ranchlands. The response of vegetation disturbance was studied by Hancock and Legg (2012) with prescribed fire management in pine forests and ericaceous heathlands in the UK.

In the analysis of EE as an ecosystem service, Wessels et al. (2011) compared tree canopy cover and height distributions between areas of contrasting management in the Lowveld savanna with LiDAR; survey-based choice experiments where SHs focused on the prevention of forest fires was a key ES delivered by grazing agroecosystems. Bernués et al. (2014) tried to elucidate the socio-cultural and economic value of some ES (e.g. forest fire events, habitat for species, aesthetic and recreational value of the landscape) delivered by mountain agroecosystems in northeast Spain by identifying SH willingness to pay for their provision. Focus groups and survey-based stated preference methods were combined to identify the effects on ES of three different scenarios deriving from different policies and to test the willingness to pay for ES compared to the current EU agri-environmental payments.

Natural (landscape) Heritage (NH) is mentioned in the MA among cultural services and includes values as shaped by the animals themselves or as part of the landscape (e.g. aesthetic values) (FAO, 2014). Natural heritage was analysed in this sense alone in the bibliography review. The additional string *("landscape" or "aesthetic" or "inspiration")* produced 39 publications of which only 4 analysed the landscape as a cultural service (Table 3.1; Fig. 5.1). In the other papers the landscape was considered: (i) for the effects that it could have on biodiversity (e.g. Cole et al., 2015; Kearns and Oliveras, 2009; Lindborg et al., 2009; Littlewood et al., 2012; Sanderson et al., 2007); (ii) as support for improving or maintaining other ES (e.g. Lavorel et al., 2011, 2015; Schaldach et al., 2013); (iii) as an assessment scale for other ES (e.g. Hussain and Tschirhart, 2013; Peringer et al., 2013; Kimoto et al., 2012); (iv) for the effects that different drivers had on it without directly analysing the consequences on its cultural value (e.g. Cousins et al., 2015; Lamarque et al., 2014; Schaich et al., 2015). The limited number of papers dealing with landscape as a cultural ES has to be related to the difficulty in measuring this aspect and to the few currently available indicators (Feld et al., 2009; TEEB, 2010).

The study of Ford et al. (2012), previously seen in WF and CR ecosystem services, also analysed the cultural services provided by different grazing regimes, concluding that aesthetic appreciation is greater in extensively grazed than in ungrazed grassland. Fontana et al. (2014) analysed the effects of management changes of larch grasslands in the Italian Alps (abandonment and intensification vs. traditional management) on valuable cultural ES (scenic beauty and traditional healing plants). They conducted a phyto-sociological study on plots randomly selected by using GIS. For each plant species recorded, three out of eight plant traits were chosen explicitly for their relevance for ES provision: flower colour, high diversity of pollination agents and the occurrence of edible or healing value for traditional meals and medicines. The provision of scenic beauty and other ES was associated with specific management systems to be addressed when planning future subsidies as well as with specific financial support for a traditional agroforestry system. Bernués et al. (2014), as previously seen in EE, and Tarrasón et al. (2016) also analysed the effects of different policy and management scenarios on NH, in order to involve local and general stakeholders in a collective reflection on the ES generated by grazing systems and to define policies and shared adaptive strategies.

7. Concluding remarks

The extraction criteria used for the bibliographic review produced a limited number of papers. Ecosystem Service was the divide term between a vast literature and minimal results. Indeed, if some other terms had been added to the basic string other results would have been obtained. For example, by adding *or "good*"* to the basic string the total number of papers for 'Food and other livestock related products' would increase from 12 to 38. This fact highlights that many authors did not analyse food as an ES. Similar considerations could be stated for the other ES analysed.

Although the Millennium Ecosystem Assessment has been the most widely accepted ES assessment framework since 2003, the analysis of the extracted papers highlighted misunderstandings concerning the concept of ES. One clear example is the confusion concerning biodiversity, which in several papers, contrary to the MA, is considered as an ES *per se* (e.g. Lindborg et al., 2009). Not all the analysed papers understand or accept the anthropocentric vision of the ES framework. For example, some authors propose biocentric solutions to reverse the inner dynamics of systems without taking into account SH opinions or needs (e.g. Bai et al., 2012; Cole et al., 2015).

The need to examine the supply and condition of each ES as well as the trade-offs (e.g. Marriot et al., 2010; Oñatibia et al., 2015) and interactions between them as requested by the MA (2003) was applied in many of the analysed papers (e.g. Koniak et al., 2011; van Oudenhoven et al., 2015; Petz et al., 2014a). Management and development options should take into account the internal dynamics of systems, biophysical components, but also socio-economics, socio-cultural and institutional features (Caballero and Fernández-Santos, 2009). Only a few authors integrated a multi-stakeholder approach in the analysis of ES and their interactions (e.g. Bernués et al., 2014; Petz et al., 2014b; Tarrasón et al., 2016). Many tools which are commonly used in scientific activity, such as mathematical models, indicators and biophysical data, were adopted by authors to engage the SHs. In other papers, future scenarios were generated from scientific data in order to facilitate discussion with and between SHs. The effects of different management options on SH well-being were discussed using ES as the basis for favouring more sustainable practices (e.g. Lamarque et al., 2014). The need for SH involvement emerged in some papers that underlined how the ES concept was not familiar to SHs (e.g. Bernués et al., 2014; Tarrasón et al., 2016) and was often confused, for example with the responsibility of humans to preserve nature. Other authors (e.g. Lindborg et al., 2009) emphasized how the SHs and their knowledge inclusion is needed in order to improve the effectiveness of local decision-making processes. The integration of local and scientific knowledge generates hybrid knowledge thereby encouraging the participation of local SHs in the decision-making processes. This allowed the identification of adaptive strategies for key services to be maintained in future (Lamarque et al., 2014; Francioni et al., 2014), for example through the implementation of *in-situ* experiments on native pasture management (Tarrasón et al., 2016).

In the analysed literature, cultural ES were poorly studied despite these being considered the most relevant for local and general SHs (Bernués et al., 2014), thereby limiting the ES framework to agricultural-related aspects. A better SH awareness of the well-being provided by ES in livestock grazing systems could foster agri-environmental schemes and the willingness to pay for these services. Many papers analysed and proposed different management options for improving the provision of ES (e.g. Cole et al., 2015) but did not analyse the effects on the Natural Heritage (e.g. landscape aesthetic value) that could be relevant in the policy making process (Bulte et al., 2008) and, for instance, in the definition of Payments for Ecosystem Services. Compensation and market-related policies have gained prominence to encourage farmers, policy makers and land managers to change their behaviour and may be a mechanism to align potentially opposing interests, for example in the area of wildlife management or biodiversity conservation.

8. References

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Chapter II: Soil respiration dynamics of wheat and alfalfa under Mediterranean conditions

1. Introduction

Soil respiration (R_S) is the releasing of CO_2 from the soil to the atmosphere and it can be partitioned in autotrophic (respiration form plant roots) and heterotopic (R_H) components (soil decomposers activities). Many scientific efforts have been made to separate R_S in its components since they are essential to understand ecosystems CO_2 exchanges between soil and the atmosphere in relation to soil use changes (Bond-Lamberty, Wang, and Gower 2004). Soil respiration is either a good indicator of soil quality (Karlen et al., 1997) but a poor indicator of C stock changes (Ryan and Law, 2005) while global warming has the potential to enhance C losses due to R_S and R_H increment (Bond-Lamberty et al., 2004). Loss of soil C from agricultural practices are likely to have a significant effect on atmospheric CO_2 concentration (Smith 2008) and land degradation (Ryan et al., 2008). In this vision, land use changes and management practices represent two of the main driving forces behind soil C pools and CO_2 effluxes (Erol et al., 2016; Guo and Gifford, 2002).

Changes in vegetation resulting from human activities or global environmental change have the potential to modify the soil CO_2 fluxes to the atmosphere (Lal, 2004; Raich and Tufekciogul, 2000). Many studies agree that the conversion of permanent vegetation in croplands can lead to a decrease in soil carbon stock (Fan et al., 2015; Haghdoost et al., 2013; Lal, 2004) and *vice versa* (Guo and Gifford, 2003; Morris et al., 2007; Smith 2008). According to Stocker et al. (2014) land use and cover changes led to global annual CO_2 efflux rates of 1.4±0.8 Pg C between 1980 and 1989, 1.6±0.8 Pg C between 1990 and 1999, and 0.9±0.8 Pg C between 2002 and 2011. Predictions of land use change effects in relation to climate dynamics tends to be difficult (Rounsevell et al., 2006) especially because the former are likely to occur on a regional scale (Miraglia et al., 2009; Pielke et al., 2002; Verburg et al., 2010; Vleeshouwers and Verhagen, 2002). Several studies proposed Q_{10} (sensitivity of soil respiration to temperature) as a metric for ecosystems comparison (Davidson et al., 2006; Fan et al., 2015; Kishimoto-Mo et al., 2015) despite its limitations in terms of seasonal fluctuations (Janssens and Pilegaard, 2003; Schindlbacher et al., 2009) or other temperature-sensitive processes (Davidson et al., 2006; Tuomi et al., 2008).

Mediterranean climate is characterised by warm, wet winters and dry and hot summers. It is the typical climate of the Mediterranean basin but it is present in many other areas of the world (e.g. California, Australia etc.). Mediterranean is one of the most vulnerable region to global climate change (Giannakopoulos et al., 2009; Giorgi and Lionello, 2008) especially for the expected impacts on water availability (Bangash et al., 2013), exchange of air masses (Cros, 2004), human health (Poumadère et al., 2005), fauna (Filipe et al., 2013; Lawrence et al., 2010), crop yields (Olesen and Bindi, 2002; Ludwig et al., 2006; Pmpadopoulou et al., 2015) and food in general (Miraglia et al., 2009). Within Mediterranean climate there are still uncertainties and contrasting results linked to soil respiration dynamics and their interactions with soil temperature (T), soil water content (SWC) (e.g. Feiziene et al., 2015) and other factors like soil microbial population (Lai et al., 2012) or soil oxygenation (Ryan and Law, 2005). In addition to the little amount of studies investigating soil respirations in arid and semiarid soils (Almagro et al., 2009; Maestre and Cortina, 2003) the few data available in this climate areas mainly address forest ecosystems (e.g. Casals et al., 2000; Díaz-Pinés et al., 2014; Rey et al., 2002). In this perspective, in two of the main land-uses (alfalfa meadows and winter cereals) in plain and hilly Mediterranean areas Central Italy, our main objectives were i) to measure annual variation in soil respiration (R_S and R_H), ii) to study the effect of the two major drivers (Soil T and SWC) of this variations and iii) assess the C stock. The study provides novel insight into the drivers of soil respiration, which should be taken into account in view of global warming.

2. Materials and Methods

2.1. Study sites description

The study sites were located in the plain and hill areas of Macerata province (Marche Region, central Italy) where the typical farming system is based on the rotation between winter cereals (e.g. wheat, barley), summer crops (mainly, maize or sunflower) and alfalfa as a cover crop. All the crops are rain-fed with the exception of maize that in plain areas usually is irrigated.

At the beginning of November 2014, in a representative area of the mentioned cropping system, two study sites with two contrasting crops were identified: i) in the alluvial plains (43°22'20.2"N 13°35'26.9"E; 28 m a.s.l.), two adjoined fields of alfalfa (AP) and wheat (WP); ii) in the hilly area (43°20'40.9"N 13°36'19.5"E; 120 m a.s.l.), two adjoined fields of alfalfa (AH) and wheat (TH). The spatial distance between the two study sites (AP-WP and AH-WH) is approximately 3 km (Figure 2.1).

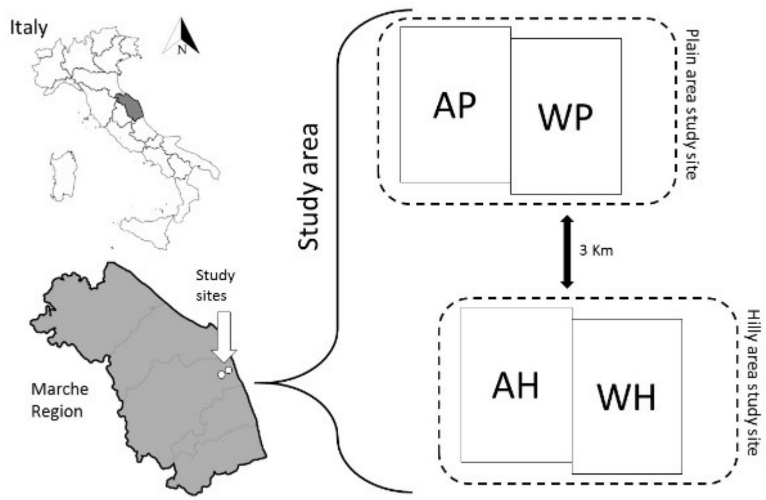

Figure 2.1. Study area and study sites. Fields. A= alfalfa; W= wheat; P= Plain area study site; H= Hilly area study site.

In the study areas, wheat cultivation normally requires a deep soil tillage (e.g. ploughing 0.3-0.4 m depth), a fertilization with N_2O (150-200 kg ha^{-1}), P_2O_5 (30-50 kg ha^{-1}), K_2O (0-10 kg ha^{-1}) and sowing with a rate of 230 kg ha^{-1} at 40-50 mm depth at the beginning of November. Harvesting is usually performed by the end of June with an average production of 4.5 t ha^{-1}. The soil is again tilled by the end of September to prepare the seedbed for the next crop. Alfalfa is usually sowed in spring, after a deep soil tillage (0.3-0.4 m depth) performed in the previous summer and followed by two minor soil tillage before the sowing. Alfalfa does not require particular treatments once implanted. Mowing is performed from two to four times per year with an expected production decrement over the years. In the study area alfalfa meadows last typically for 3-4 years or even more (up to 10 years) when they are used by transhumant flocks during the lowland grazing activities (Budimir et al., 2016). The climate is Mediterranean and during the study period (from January to December 2015) an average annual precipitation of 908.6 mm and a mean annual temperature of 15.9 °C were recorded. The mean air temperature ranged from 7.3 °C (February) to 28.0 °C (July). The rainiest month was October (190.4 mm) and the most dry was July (2.4 mm) (Figure 2.2).

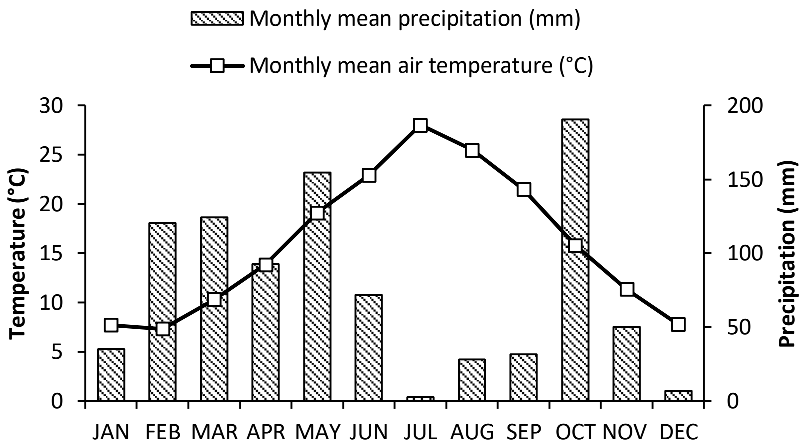

Figure 2.2. Mean precipitation and air temperature during the experimental period (January-December 2015). Data were provided by the Agrometeorological Extension Service of Marche Region (ASSAM).

The soils in the study sites are classified as Inceptisol according to USDA (Soil Survey Staff, 2014) and their basic physicochemical characteristics are shown in Table 2.1.

Table Errore. Per applicare 0 al testo da visualizzare in questo punto, utilizzare la scheda Home.**.1. Soil basic physicochemical characteristics (mean ± standard error) of the different fields in the two study sites.**

Field	**pH**	**Sand (%)**	**Silt (%)**	**Clay (%)**	**SDB ($g\ cm^{-3}$)**	**WP (%)**	**FC (%)**	**TOC ($g\ kg^{-1}$)**	**Tot. N ($g\ kg^{-1}$)**	**C/N**
WP	8.26±0.02	23.67±1.40	46.83±1.85	29.50±0.96	1.64±0.09	19.19±0.28	27.27±0.61	11.17±0.12	1.22±0,03	8.73±0.12
AP	8.08±0,07	18.30±0.26	46.20±0.40	35.50±0.44	1.57±0.06	19.81±1.02	28.68±0.84	13.60±0.79	1.38±0,08	9.43±0.42
WH	8.39±0.03	8.17±1.17	48.87±1.19	42.97±0.12	1.46±0.05	19.30±0.28	30.25±0.71	6.80±0.20	0.80±0,05	6.33±0,06
AH	8.25±0.03	12.13±1.72	47.73±2.56	40.13±2.41	1.64±0.06	21.16±2.1	29.72±2.14	8.13±0.76	0.92±0,08	7.93±0.45

Note: Data were obtained from the analysis of 5 subsamples per field collected in the first 0-30 cm layer: Soil Bulk density (SBD) was assessed by cylinder method; Total Organic Carbon (TOC) by Springer-Klee method; Total Nitrogen by Kjeldahl method.

2.2. Measurements description

Soil respiration (R_S) efflux was measured in situ using a portable, closed chamber, soil respiration system (EGM-4 with SRC-1, PP-Systems, Hitchin, UK) with a measurement time of 120 s. At the beginning of November 2014 six polyvinyl chloride (PVC) collars per field (100 mm inner diameter and 100 mm long, with perforated walls in the first 50 mm) were inserted into the soil at a depth of 9 cm. In order to assess the soil heterotrophic respiration (R_H), at each field three of the six collars were placed on a root exclusion subplot where soil was isolated with a PVC cylinder (0.4 m diameter, 0.4 m high) opened at both ends, following the method described by Alberti et al. (2010). During each CO_2 efflux measurement, the SRC-1 chamber was fitted to a PVC collar. Measurements started in January 2015 and ended in December 2015 with a frequency from 2 to 3 times per month depending on the weather variability and the management practices (N=25 CO_2 measurements per field). Soil respiration was always measured between 8:30 and 12:00 am standard time in order to avoid effluxes fluctuation (Almagro et al., 2009; Fan et al., 2015; Xu and Qi, 2001). Soil temperature (T) was measured at each plot at the same time of CO_2 efflux measurement using the build-in temperature probe EMG-4 at 10 cm depth. Soil water content (SWC) was assessed using the oven dry method (105 °C to a constant weight) on soil samples collected in the top 20 cm layer.

2.3. Data analysis

One-way ANOVA with repeated measures (Proc GLM, SAS) and LSD test for the pairwise comparisons were used to test differences for R_S, R_H, Soil T and SWC of the studied field plots. Regression analysis was used to examine the relationship between soil respirations (R_S and R_H), SWC and soil T (Davidson et al., 1998). Equations are as follows:

For R_S/R_H and soil T: $y = a\, e^{bx}$ (1)

For R_S/R_H and SWC: $y = a + bx$ (2)

Where y is the measured R_S or R_H (μmol CO_2 m^{-2} s^{-1}), b is the coefficient (soil T in the exponential and SWC in the linear equations) and x are the constant coefficients. The former simple empirical models were obtained using thresholds values through a step-by-step procedure removing one measurement date at time in order to distinguish the drought period from the rest. Q_{10} was calculated from the differences in the respiration rate of R_S and R_H at 10 °C interval using the exponential regression model with the following formulae:

$$R_{10} = a\, e^{10b} \quad (3)$$

$$Q_{10} = (e^{10b}) \quad (4)$$

Where R_{10} is the basal rate of respiration at the reference temperature of 10 °C (Davidson et al., 2006; Kishimoto-Mo et al., 2015 and Xu and Baldocchi, 2004). Finally, soil C stock at 0.3 m depth was estimated with the following formulae:

TOC (g C/kg soil) x SDB (kg soil/ha) x depth (0.3m) (5)

Statistical analyses were performed with SAS/Studio® software, Version 3.5 of the SAS System for Windows.

3. Results

3.1. Spatial and temporal dynamics of soil respiration, temperature and water content

Soil Water Content (SWC), soil Temperature (T), soil respiration (R_S) and its heterotrophic component (R_H) showed different spatial and temporal dynamics with variability between study sites, crops and date of assessment (Figure 3.1). In agreement with studies conducted in Mediterranean climates (e.g. Almagro et al., 2009; Lai et al., 2012; Rey et al., 2002) the highest values of T recorded during the summer period coincided with the lowest SWC values (Figure 3.1 a-b-c-d). The annual mean soil T at 0.1 m depth was 16.32, 15.63, 17.07 and 16.54 °C in AP, WP, AH and WH, respectively. From January to December, SWC at 0.2 m depth ranged from 11.21 to 42.66 (AP), from 9.64 to 40.06 (WP), from 9.96 to 42.0 (AH), from 13.90 to 40.64 (WH). Overall, despite both soil T and SWC showed some significant differences within date of measurements, the dynamics tends to be very similar in both study sites suggesting that these differences are not ascribable to the different management practices (Figure 3.1 a-b-c-d).

The annual mean R_S rates were 2.48, 1.33, 2.03 and 1.49 µmol CO_2 m^{-2} s^{-1} in AP, WP, AH and WH, respectively. Due to the different management practices the R_S peaks of the two crops (wheat and alfalfa) showed a temporal variability and were recorded in different periods of the year: AP showed two peaks of 5.28 µmol CO_2 m^{-2} s^{-1} on May and June respectively while a single peak of R_S was observed for WP (2.88 µmol CO_2 m^{-2} s^{-1}) in April. The alfalfa field of the hilly site (AH) showed a peak of 4.04 µmol CO_2 m^{-2} s^{-1} recorded in June and two peaks of WH (3.35 and 3.28 µmol CO_2 m^{-2} s^{-1}) were observed respectively in April and May. The annual mean R_H rates were 1.19, 0.96, 1.03 and 0.80 µmol CO_2 m^{-2} s^{-1} in in AP, WP, AH and WH, respectively. The highest peaks of R_H were recorded by AP in May (2.65 µmol CO_2 m^{-2} s^{-1}) and August (2.08 µmol CO_2 m^{-2} s^{-1}). WP showed its R_H peak in July (1.81 µmol CO_2 m^{-2} s^{-1}) as well as AH (1.98 µmol CO_2 m^{-2} s^{-1}) and WH (1.98 µmol CO_2 m^{-2} s^{-1}) (Figure 3.1 g-h).

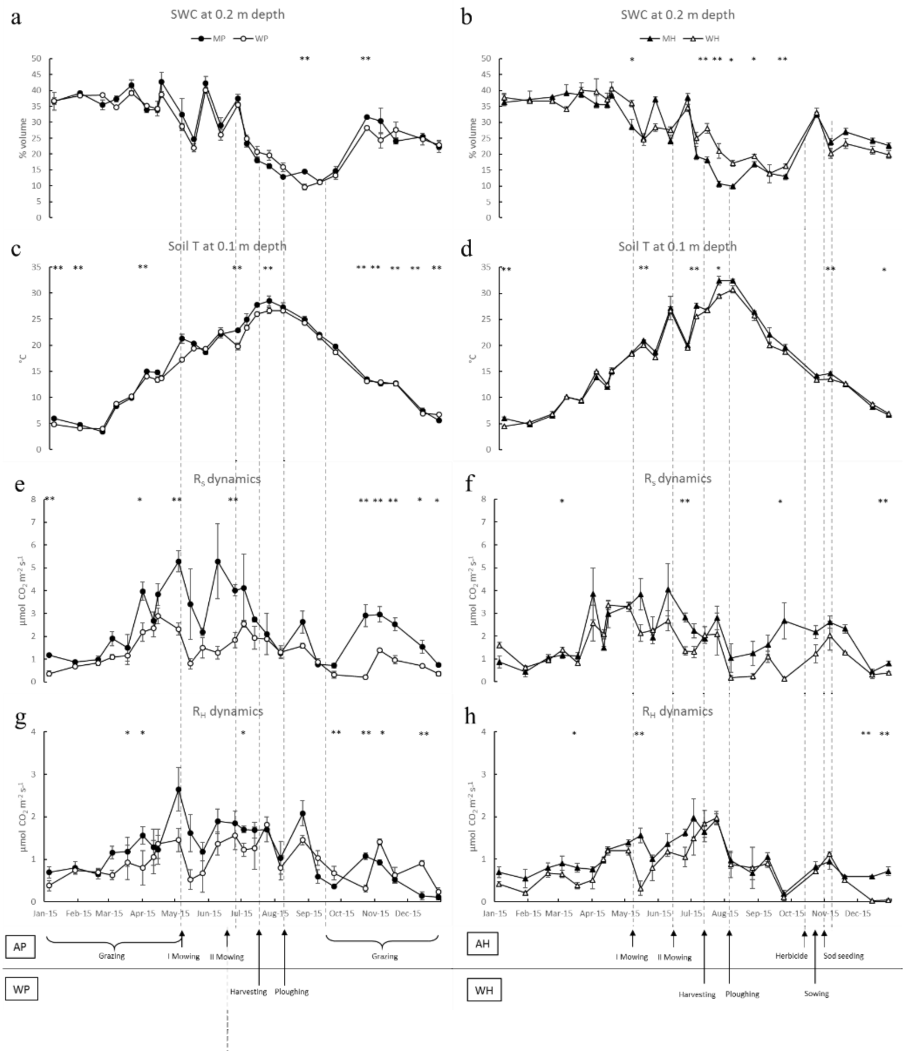

Figure 3.1. Seasonal dynamics of SWC at 0.2 m depth (a-b), soil T at 0.1 m depth (c-d), RS (e-f) and RH (g-h) during the study period (January-December 2015). Vertical dotted lines represent the different management practices adopted for the different crops in the study fields. Least significant differences (LSD) within date are expressed as "*" for p <0.05 and as "" for p<0.01. The vertical bars represent the standard errors. AP= alfalfa plain study site; WP= wheat plain study area; AH= alfalfa hilly study site; WH= wheat hilly study site.**

3.2. Relationship of soil respirations (R_S and R_H) with soil T and SWC

Significant exponential relationships between the seasonal variation of soil T with R_S and R_H in each field were recorded, with the exception of R_S of the two wheat fields (WP and WH). The R^2 values of exponential regressions explained 42-70% of the variation of the annual R_S rate and 63-71% of the annual R_H rate. When soil conditions were under the WP threshold value, no significant differences were observed between SWC and soil respirations (R_S and R_H) (Figure 3.2).

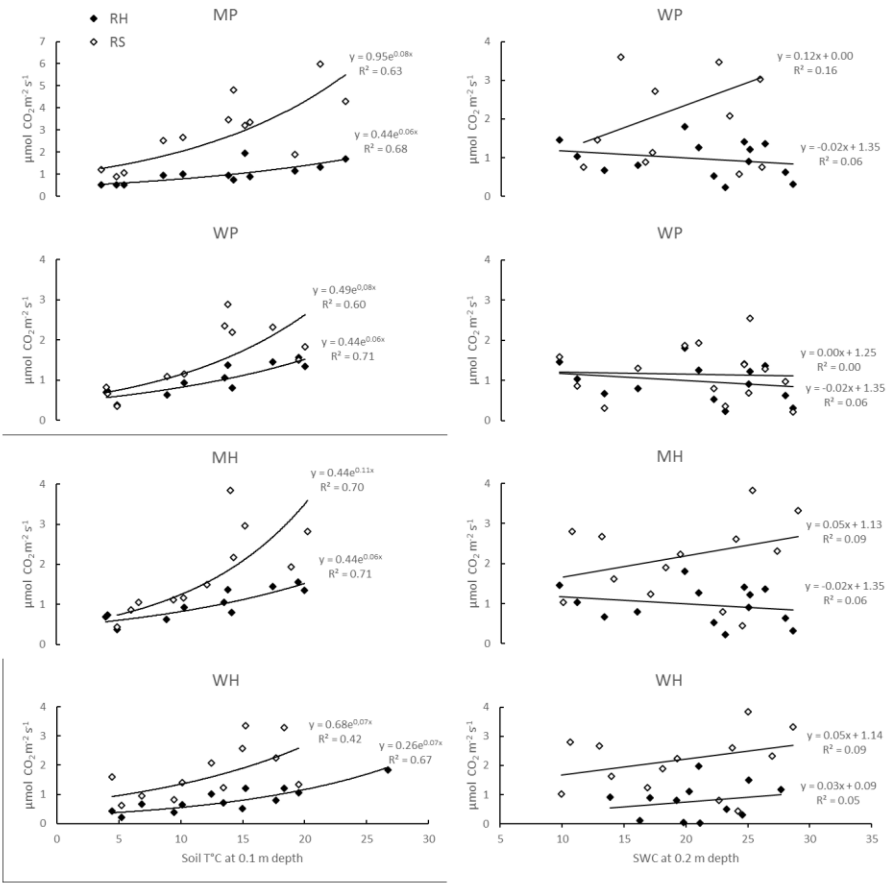

Figure 3.2. Relationships between the seasonal variation of RS (white dots) and RH (black dots) with soil T (0.1 m depth) and SWC (0.2 m depth). Significance is expressed as "*" for p < 0.05 and "" for p < 0.01.**

Q_{10} varied between and within each of the two study site ranging from 1.82 to 2.01 for R_H and from 2.01 to 3.00 for R_S (Table 3.1).

Table Errore. Per applicare 0 al testo da visualizzare in questo punto, utilizzare la scheda Home..**2. Reference respiration rate at 10 °C (R_{10}) and (Q_{10}) values for RH and RS in each field.**

Field	Type	a	b	R^2	p	DF	R_{10}	Q_{10}
AP	R_H	0.44	0.06	0.68	<0.01	11	0.80	1.82
	R_S	0.95	0.08	0.63	<0.01		2.11	2.23
WP	R_H	0.44	0.06	0.71	<0.01	9	0.80	1.82
	R_S	0.49	0.08	0.60	=0.01		1.09	2.23
AH	R_H	0.44	0.06	0.71	<0.01	10	0.80	1.82
	R_S	0.44	0.11	0.70	<0.01		1.32	3.00
WH	R_H	0.26	0.07	0.67	<0.01	12	0.52	2.01
	R_S	0.68	0.07	0.42	=0.06		1.37	2.01

Soil C Stock

The Soil C stock estimation at 0.3 m depth showed values of 64.10, 55.05, 39.97 and 29.73 t ha^{-1} in AP, WP, AH and WH respectively. The comparison of AP with WP (Plain fields) and AH with WH (Hill fields) showed significant differences ($p<0.05$ and 0.02, respectively) (Figure 3.3).

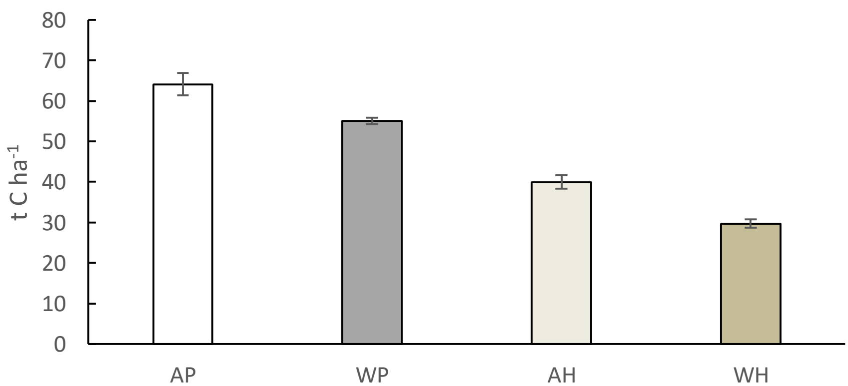

Figure 3.3. Soil C stock (t C ha^{-1}) of the different field in the two study sites. The bars represent the standard error.

4. Discussion

The strong seasonal variability of soil respirations (R_S and R_H), water content and temperature recorded in the study sites and between fields (Figure 3.1) confirmed the results obtained by Almagro et al. (2009) Lai et al (2012) and Rey et al. (2002) in Mediterranean climate areas. R_S in all the study sites and crops varied markedly over the period of observation showing a fast decreasing at the end of May corresponding to a fast SWC increase due to the high precipitation registered (Figure 3.1 e-f).

Many studies reported a single peak of R_S in sub-tropical (Deng et al., 2013; Fan et al., 2015), temperate (Fenn et al., 2010) or continental climate (Tüfekçioğlu et al., 2001). This study shows different dynamics that showed multiple peaks in line with what observed by Almagro et al. (2009), Casals et al. (2000) and Rey et al. (2002) in Mediterranean climate conditions. More specifically, In the plain study site, AP showed always higher R_S rates with the exception of one date of measurements in September. The drastic AP R_S rate reduction observed from May to June highlight the effect of the alfalfa mowing occurred on the 4th of May. A fast increasing of AP R_S rate occurred from the end of May to middle of July probably helped by the remarkable amount of rainfall occurred in this period (Figure 2.2). Subsequently, AP R_S rates started to decrease until the middle of September when sheep started the grazing activities. In this period significant ($P<0.05$) and very significant ($p<0.01$) differences were observed between AP and WP in each date of measurement until the end of the monitoring period suggesting that grazing influence the CO_2 effluxes from alfalfa field. The R_S rate of WP showed and increasing trend until the second week of April where it decreased to 0.8 µmol CO_2 m^{-2} s^{-1} on May 15th. Later the R_S rate of WP gradually increased again until July where the harvesting and, about one month later, the ploughing were performed. The heterotrophic component of R_S was almost always higher in AP with few exceptions (Figure 3.1-g). During the second grazing period (October to December 2015) the R_H of AP was always lower than RH of WP with the exception 24th September.

The crops in hilly study site showed similar R_S dynamics from January to the middle of June with the exception of 4th may where WH R_S was significantly higher than AH R_S Rate. From June to December, alfalfa showed higher respiration rates with the exception of the date of measurement immediately after the wheat harvesting (13th July). Despite significant differences where not always found, after the WH ploughing occurred on the first week august, the differences between AH and WH R_S rates tended to increase until the end of November. (Figure 3.1-f). From January to the 3rd of July, heterotrophic respiration rates of AH were always higher compared to WH rates while an inversion occurred in the middle of July immediately after the WH harvesting. Differences between R_H rates increased again after a sod seeding performed on AP ($P>0.01$).

The annual R_S means of the two alfalfa fields (2.46 and 2.03 μmol CO_2 m^{-2} s^{-1} in AP and MH, respectively) are higher compared to those reported by Paustian et al. (1990) (1.06 μmol CO_2 m^{-2} s^{-1}) in Sweden and Gong et al. (2015) (from 0.68 to 1.32 μmol CO_2 m^{-2} s^{-1}) in Inner Mongolia.

Concerning wheat, the annual R_S means of 1.37 and 2.03 μmol CO_2 m^{-2} s^{-1} recorded in WP and WH, respectively, are consistent with those observed in Missouri by Buyanovsky et al. (1987) (1.69 μmol CO_2 m^{-2} s^{-1}) and in Sweden by Paustian et al. (1990) (from 0.96 to 2.07 μmol CO_2 m^{-2} s^{-1}). Compared to R_S, the variation of R_H was less pronounced with the most marked differences registered on 15th of April for AP-WP and on 5th May for AH-WH.

The annual mean R_H rates (1.19, 0.96, 1.03 and 0.80 μmol CO_2 m^{-2} s^{-1} in AP, WP, AH and WH respectively) are consistent with what observed for alfalfa in different conditions (e.g. Alberti et al., 2010; Paustian et al., 1990) and in other agricultural ecosystems (Oyonarte et al., 2002). The higher R_H rates registered in alfalfa in the study sites (AP and AP) could be linked to the higher soil microbial activity (Bevivino et al., 2014) and TOC (Table 3.1) derived from the different field management of the last decade. Overall, AP and AH showed higher annual mean R_S and R_H rates compared to WP and WH (Figure 4.1).

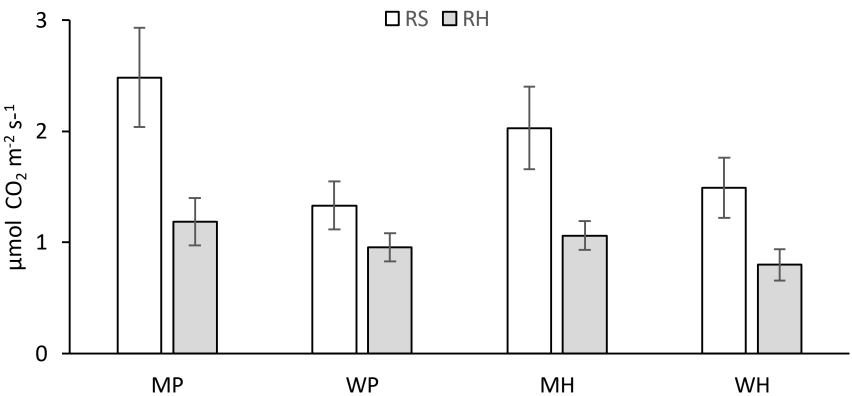

Figure 4.1. Mean annual soil respiration (RS) and soil heterotrophic respiration (RH) in each study field.

Scientific data on crops and grasslands tends to be fragmentary with results characterised by a strong variability. Hibbard et al. (2005) provided a review on soil respiration across northern hemisphere temperate ecosystems showing that strongest relationship between daily mean R_S and soil T occur in woodlands. Within Mediterranean climate areas, many authors reported high correlation values between soil respiration (both, R_H and R_S), and soil T (e.g. Almagro et al., 2009; Lai et al., 2012; Rey et al., 2002). In the present study, strong exponential relationships were observed between the R_S and R_H with soil T for each field in both the sites (R^2 from 0.42 to 0.71). These findings are in contrast with Conant et al. (2000) and Maestre and Cortina (2003) that observed low correlation of CO_2 effluxes with R_S in semi-arid climates.

Many authors reported very strong linear relationships between soil respiration (both, R_S and R_H) and SWC under Mediterranean climate, especially during the dry period when the limiting factor tends to be the SWC instead of soil temperature (Almagro et al., 2009; Lai et al., 2012; Tüfekçioğlu and Küçük, 2004). Despite this, in the study area the R^2 coefficient in all the analysed conditions was very weak and ranged from 0 to 0.16 and was never significant. However, it should be noted that soil respiration modelling under Mediterranean condition tends to be difficult especially if just SWC and soil T are taken into account (Cotrufo et al., 2011; Lai et al., 2012; Oyonarte et al., 2012). This because more factors like microbial activity and/or oxygen supply may play significant roles (Lai et al., 2014; Ryan and Law, 2005).

Q_{10} calculated from R_H exponential regressions were the same for AP and WP (1.82) suggesting no differences in the sensitivity of heterotrophic soil respiration to temperature of both, alfalfa and wheat in plain area. Conversely, in the hilly area, wheat showed a higher R_H Q_{10} (2.01) compared to alfalfa (1.82). The Q_{10} of R_S showed again identical values in AP and WP (2.23) but different results in the hilly area (3.00 and 2.01 in AH and WH, respectively). The results of R_S-Q_{10} of alfalfa are not very different from those obtained by Xu and Baldocchi (2004) in a Mediterranean grassland in California. Overall, the Q_{10} values of this study appear to be into the range reported by Raich and Schlesinger (1992) and by Xu and Qi (2001).

As expected, the soil C stock varied in each field and, in general, were higher in the plain area. The two wheat fields (WP and WH) which have been intensively tilled since at least 2006 (data obtained from interviews and not shown) showed a soil C stock of 55.05 and 29.73 t C ha^{-1}. Alfalfa fields showed a higher soil C stock (64.10 and 39.97 t C ha^{-1}) in AP and AH respectively if compared with the wheat fields.

5. Conclusion

In conclusion, in the study area, alfalfa proved to have different soil R_H and R_S dynamics compared to ordinary winter cereals cropping system. Although differences in Q_{10} were found only in the hilly study site highlighting a similar sensitivity of soil respiration rates to temperature increase in the plain study (Table 3.1), the alfalfa fields (AP and AH) showed higher soil C stock compared with the two wheat fields (Figure 3.3).

Alfalfa is widely used in Mediterranean areas but scientific literature concerning its GHG emission appears quite narrow. Typically, alfalfa last from three to five years before their replacement with other crops (e.g. wheat) by ploughing but just few data are available concerning SOC decay rates after ploughing. In addition, soil N_2O emission are related to different factors (e.g. fertilization) (Bouwman and Boumans 2002) and specific experimentations in this direction would help to fill some major knowledge gaps. The research results concerning driving factor od soil respiration dynamics tends to be uncertain or even contrasting especially under Mediterranean conditions. Therefore, further analyses are needed to fully understand the factors that regulate soil CO_2 effluxes of such contrasting cropping systems under Mediterranean conditions. In particular, incubation experiments may clarify the lack of evidence concerning the role of SWC as limiting factor during the dry season as observed by many authors (e.g. Davidson et al 1998; Rey et al., 2002). More broadly, field experiments both at plot and landscape scale would be recommended to investigate the spatial variation of GHG emissions especially under Mediterranean conditions.

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Chapter III: Soil respiration dynamics and Carbon Stock under different land cover changes (Monti Sibillini, Central Italy)

1. Introduction

Soil respiration (R_S) is defined as releasing of carbon dioxide (CO_2) from the soil including respiration of plant roots, the rhizosphere, microbes and fauna. R_S is the second carbon flux between ecosystems and atmosphere and due to climate change many scientific efforts focused on CO_2 emission in the last decades. Although the C stored in soils and living terrestrial biomass is estimated to be about three times bigger than the CO_2 present in the atmosphere (Falkowski, 2013), even small changes on R_S rates could change the concentration of CO_2 in the atmosphere (Schlesinger and Andrews, 2014). For these reasons it is crucial to study the interactions of humans and the C stored in terrestrial ecosystems, including soil respiration (Wu et al., 2014).

According to FAO (2014) grasslands[1] cover about 13% of the globe surface and they are considered pivotal for the biodiversity conservation (Tilman and Downing 1996). Grasslands play a significant role on greenhouse gas (GHG) emission since they have to potential to mitigate global warming (Oertel et al., 2015). Despite the importance of soil respiration is well recognized and studied for wide array of ecosystems still lacks of research are present for grasslands (Bahn et al., 2008).

Both Land use (LU) and Land cover changes (LCC) affect GHG emissions and thus affect global warming (e.g. Breuer *et al.*, 2006; Schoeneberger et al., 2012). Houghton et al. (2012) provided an excellent distinction between LU and LCC where the first refers to the

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[1] According to FAO (2014) Grasslands includes any geographical area dominated by natural herbaceous plants (grasslands, prairies, steppes and savannahs) with a cover of 10% or more, irrespective of different human and/or animal activities, such as: grazing, selective fire management etc. Woody plants (trees and/or shrubs) can be present assuming their cover is less than 10%.

management within a land-cover type (e.g. pastures or croplands) and the latter refers to the conversion of one cover type to another (forests to croplands and vice-versa). In agreement with these definitions, the main objective of this study was to investigate changes generated from a LCC from (grasslands to croplands) in a temperate mountain area. In particular, seasonal variation in soil respiration (R_S and R_H) was measured as well as soil quality indicators.

2. Materials and Methods

2.1. Site description

The study area was located in the territory of Castelluccio di Norcia (42°49'N, 13°13'E, central Italy) in the Monti Sibillini National Park. The site is characterised by a low input cropping system with winter cereals (e.g. spelt, oat) and legumes (mainly lentil) as main crops in a grassland-dominated landscape mostly used as summer grazing for cattle or transhumant flocks. The climate is temperate with an average annual precipitation of 908.6 mm and a mean annual temperature of 15.9 °C; the monthly mean air temperatures ranged from 7.3 °C in February to 28.0 °C in July. The rainiest month of 2015 was October (190.4 mm) and the most droughty was July (2.4 mm) (data provided by the Agrometeorological Extension Service of Umbria Region). The monthly mean air temperature and precipitation during the study period are shown in Figure 2.1.

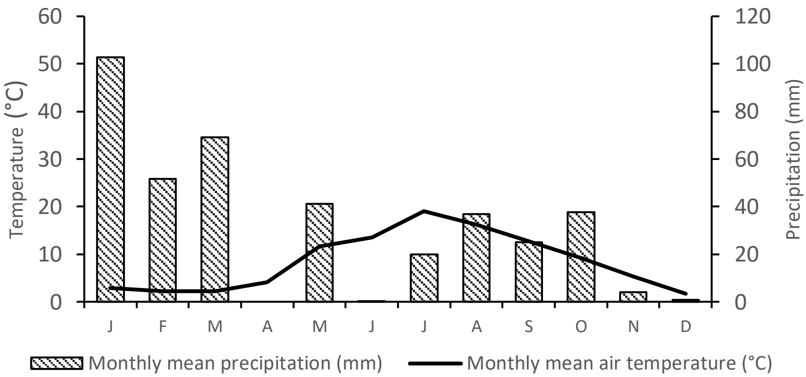

Figure 2.1. Seasonal mean precipitation and air temperature during the experimental period.

The soils were classified as Mollisol according to USDA (Soil Survey Staff 2014) and their basic physicochemical characteristics of the soil are shown in Table 2.1. A detailed description of Castelluccio di Norcia geomorphological evolution it is provided by Coltorti and Farabollini (1995).

Table Errore. Per applicare 0 al testo da visualizzare in questo punto, utilizzare la scheda Home..**3. Soil basic physicochemical characteristics obtained from 5 subsamples per field in the first 0-20 cm layer. Values are the mean ± standard error.**

Code	pH	Sand (%)	Silt (%)	Clay (%)	Gravel (%)	SDB (g cm^{-3})	WP (%)	FC (%)
GRS	7.6±0.13	60.83±1.36	35.70±2.65	3.47±1.31	65.33±5.03	0.92±0.04	45.30±4.23	56.16±6.53
LNT	8.0±0.04	43.37±1.00	45.53±0.46	11.10±0.62	72.33±2.52	1.10±0.66	24.64±1.78	37.21±1.29
SPL	7.9±0.01	48.03±2.08	44.87±1.42	7.10±2.72	67.67±4.04	0.80±0.06	33.39±1.96	44.64±3.61

The study area is composed by a homogenous portion of *Bromus erectus*-dominated grassland (GRS) and two adjacent arable fields that formerly were part of the same grassland: one cultivated with lentil (LNT) and one cultivated with spelt (SPL) in 2015.

2.2. Measurements description

In November 2015 six PVC collars (10 cm inner diameter and 10 cm high, with perforated walls in the first 5 cm) were inserted into the soil to a depth of 9cm. Three out of six PVC collars were placed on a root exclusion subplot where soil was isolated with a PVC cylinder (40 cm diameter, 40 cm high, opened at both ends), following the method described by (Alberti et al. 2010) to measure heterotrophic soil respiration (R_H). Soil CO_2 efflux was measured in situ using a portable, closed chamber, soil respiration system (EGM-4 with SRC-1, PP-Systems, Hitchin, UK). The measurement time was set to 120 s. Due to snow cover, measurements started in April 2015 and ended in November 2015 with a frequency from 2 to 3 times per month depending on the weather variability and agricultural practices (N=18 per field). Soil respiration was always measured between 8:30 and 12:00 am standard time in order to avoid effluxes fluctuation (Almagro et al., 2009; Fan et al., 2015; Xu and Qi, 2001). Soil temperature (T) was measured at each plot at the same time of CO_2 efflux measurement using the build-in temperature probe EMG-4 at 10 cm depth. Soil water content (SWC) was measured by taking soil samples in the top 20 cm and dried in oven to a constant weight at 105 °C.

2.3. Data analysis

Differences for R_S, R_H, soil T and SWC were tested with One-way ANOVA with repeated measures (Proc GLM, SAS) followed by a LSD test for the pairwise comparisons of the field plots. Exponential regression analysis was used to examine the relationship between soil respirations (R_S and R_H and soil T with the following formulae:

$$y=a\, e^{bx} \tag{1}$$

Where y was the measured R_S or R_H ($\mu mol\ CO_2\ m^{-2}\ s^{-1}$), b was the Soil T coefficient and x where the constant coefficient (Davidson et al., 1998). Q_{10} was calculate from the differences in the respiration rate of R_S and R_H at 10 °C interval using the exponential regression model with the following formulae:

$$R_{10} = a\, e^{10b} \tag{2}$$

$$Q_{10} = e^{10b} \tag{3}$$

Where R_{10} is the basal rate of respiration at the reference temperature of 10°C (Davidson et al., 2006). Soil C stock at 0.3 m depth was estimate with the following formulae:

$$TOC\ (g\ C/kg\ soil)\ x\ SBD\ (kg\ soil/ha)\ x\ depth\ (0{,}3m) \tag{4}$$

According to Putra et al. (2016), Humification rate (HR) was calculated with the following formula:

$$HR = (C\ HA+FA/TOC)*100. \tag{5}$$

All statistical analyses were performed with SAS/Studio® software, Version 3.5 of the SAS System for Windows.

3. Results

3.1. Soil temperature

Soil temperature showed fluctuating trends from April to August where the soil T started to decrease until December. Soil temperature reached the minimum on December 2015 and the maximum on July 2015 with variation from 1.72 to 22.65 °C in GRS; from 1.40 to 24.14 °C in LNT and from 1.37 to 24.10 °C in SPL. The average soil T during the course of the sampling period was 13.72, 13.84 and 13.90 in GRS, LNT and SPL respectively. Despite statistical differences ($p<0.05$) between the soil T recorded in in 8th May, 25th August, 1st October and 4th November (Figure 3.1), the seasonal average soil temperatures between the three fields were not different ($p>0.93$).

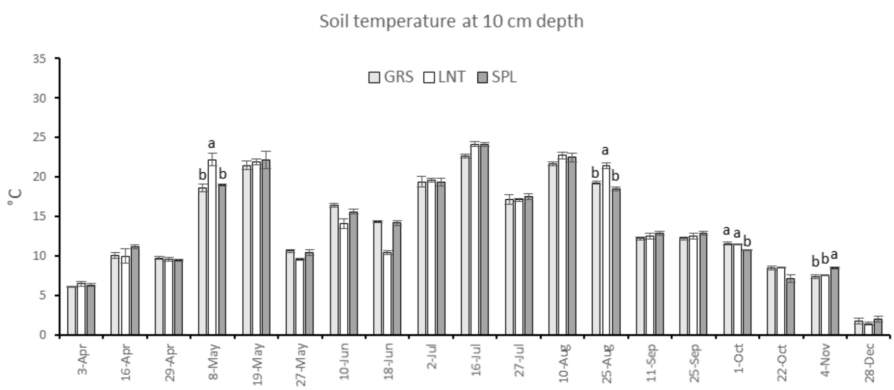

Figure 3.1. Seasonal dynamics of soil T at 10 cm depth. Bars labelled with the different lowercase letters at the same site denote significant difference at p<0.05. The vertical bars represent the standard errors.

3.2. Soil Water Content

Soil moisture (%Vol) varied markedly over the period of observation ranging from 6.92 to 35.34 in GRS, from 8.05 to 44.24 in LNT and from 3.92 to 20.46 in SPL. Significant differences ($p<0.05$) were present in several dates of measurement as well as in the annual mean SWC of the three fields ($p<0.01$) (Figure 3.2).

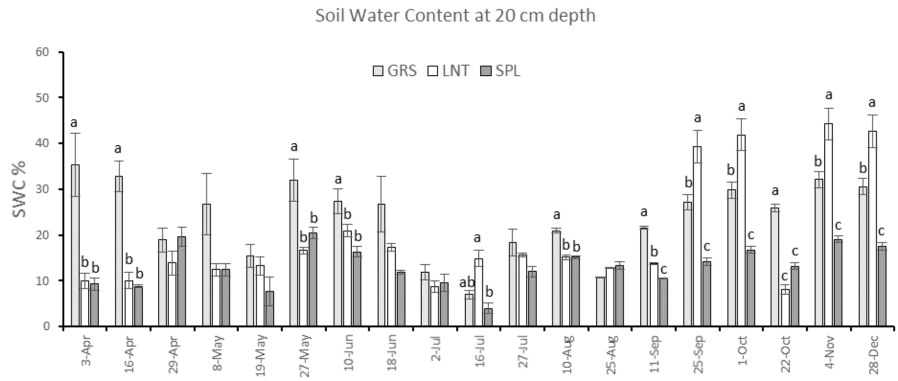

Figure 3.2. Seasonal dynamics of SWC at 20 cm depth. Bars labelled with the different lowercase letters at the same site denote significant difference at p<0.05. The vertical bars represent the standard errors.

3.3. Soil CO_2 emissions

Seasonal variations of R_S and R_H are summarized in Figure 3.3. GRS R_S fluxes increased gradually from April to the first half of May reaching the first peak on the 8th of May then decreased and increased again reaching the maximum in 18th June. A Third peak of R_S of GRS was registered in 10th august and then declined gradually until the end of the monitoring period. Soil respiration of LNT increased gradually from April reaching the first peak in 10th June then decreased rapidly in 18th June and increased again in 2nd July. Thereafter the R_S lightly increased until 1st October and rapidly decreased again until December. SLT soil respiration effluxes reached the maximum in 10th August and then rapidly decreased in 25th August, lightly increased in 11th and 25th September and then decreased until December. Significant differences ($p<0.05$) within date of measurements concerning R_S are showed in Figure 3.3-A where the most marked differences between fields were observed in 18th June and 25th August. Heterotrophic soil respiration showed spatial and temporal variability where the peaks of GRS (3.47 μmol CO_2 m^{-2} s^{-1}), LNT (1.70 μmol CO_2 m^{-2} s^{-1}) and SPL (3.22 μmol

CO_2 m^{-2} s^{-1}) where reached in 10th June, 2nd July and 27th July respectively. With the exception of 8th May, 2nd July and 27th July GRS showed always higher R_H rates compared with the ones of SPL and LNT.

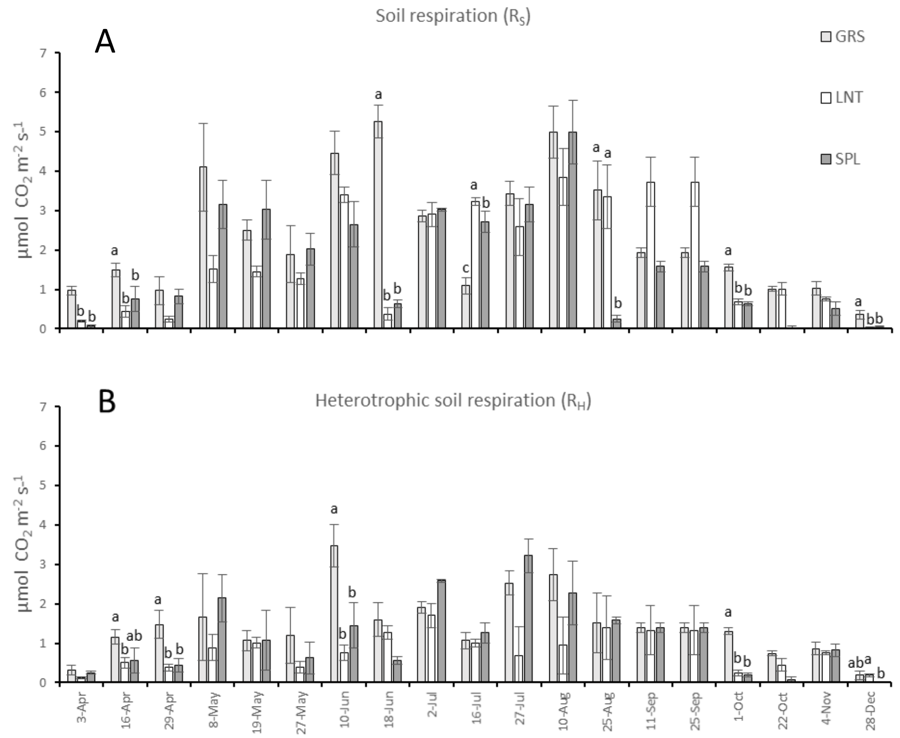

Figure 3.3. Seasonal variations of soil CO_2 emissions (A= Soil respiration; B= Heterotrophic soil respiration). Bars labelled with the different lowercase letters at the same site denote significant difference at p<0.05. The vertical bars represent the standard errors.

No significant differences ($p<0.28$) were observed in the annual mean R_S rates which amounted for 2.39, 1.83 and 1.66 µmol CO_2 m^{-2} s^{-1} at GRS, LNT and SPL respectively. Conversely, significant differences ($p<0.05$) were observed between the annual mean R_H rates which were 1.45, 0.81 and 1.15 µmol CO_2 m^{-2} s^{-1} at GRS, LNT and SPL respectively (Figure 3.4).

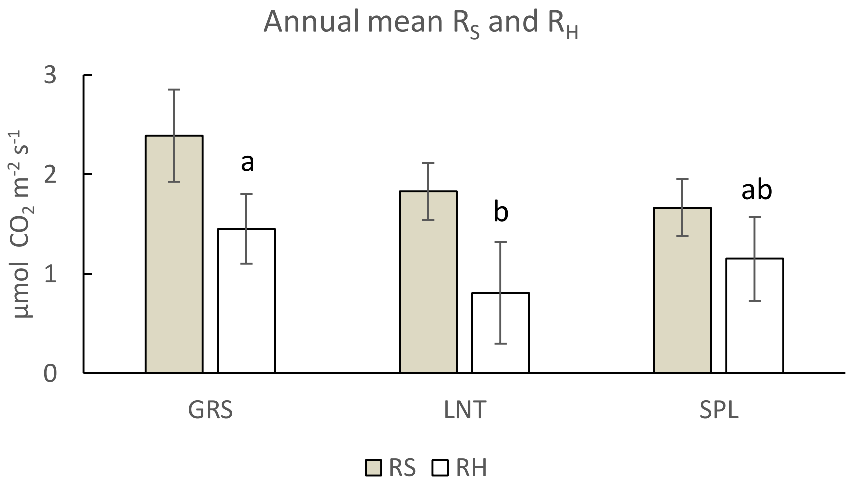

Figure 3.4. Annual mean R_S and R_H efflux in the three experimental fields. Bars labelled with the different lowercase letters at the same site denote significant difference at $p<0.05$. Vertical bars indicate standard error.

3.4. Relationship of soil CO_2 effluxes and soil temperature

There were significant relationships ($p>0.05$) between the seasonal variation of R_H and R_S with soil temperature at 10 cm depth in each field (Figure 3.5). The Q_{10} values calculated from R_H effluxes monitoring were 3.00, 3.67 and 2.01 in GRS, SPL and LNT respectively. The derived R_{10} and Q_{10} values are summarized in Table 3.1.

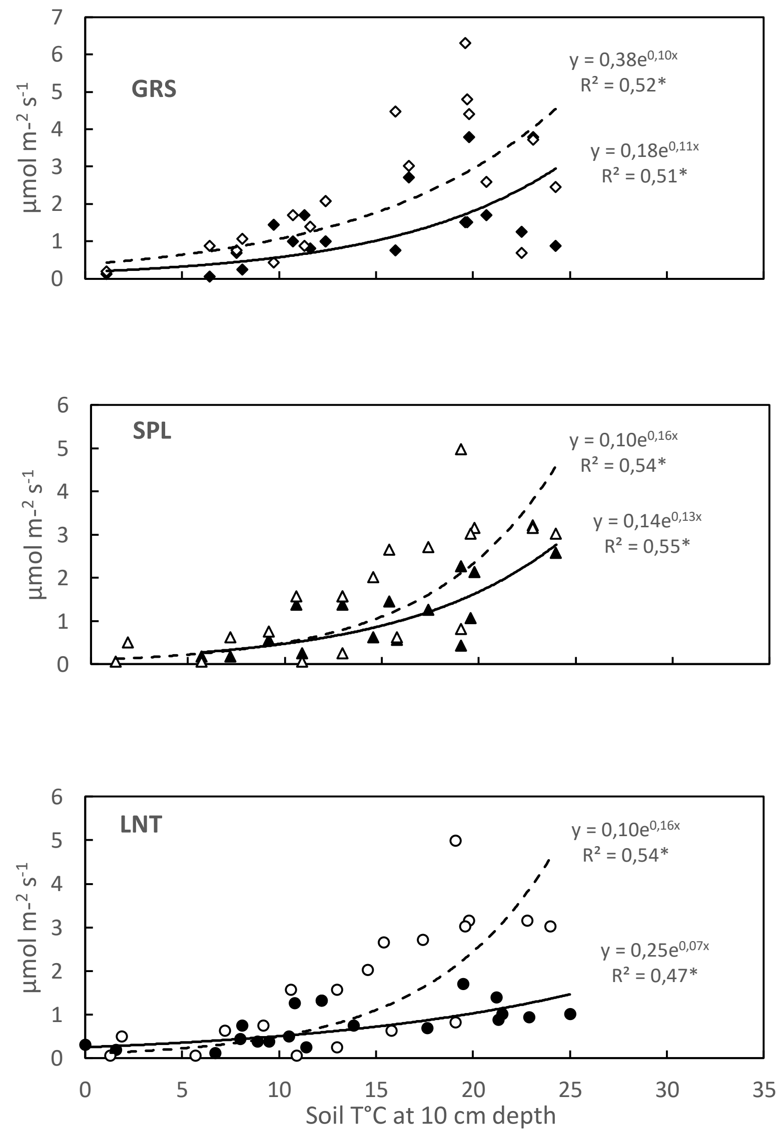

Figure 3.5. Relationship between the seasonal variation of R_S and R_H with soil T at 10cm depth. The data points are the mean of three plots per field. Significance is expressed as "*" for $p < 0.05$.

Table Errore. Per applicare 0 al testo da visualizzare in questo punto, utilizzare la scheda Home.**.4. R_{10} and Q_{10} values of the three fields.**

Field	Respiration	a	b	R_{10}	Q_{10}
GRS	R_H	0.18	0.11	0.54	3.00
	R_S	0.38	0.10	1.03	2.72
SPL	R_H	0.14	0.13	0.51	3.67
	R_S	0.1	0.16	0.50	4.95
LNT	R_H	0.25	0.07	0.50	2.01
	R_S	0.24	0.12	0.80	3.32

3.5. Soil quality indicators

The total organic carbon (TOC) values in 0-30 cm were significantly ($p>0.01$) higher in GRS (146.77±7.68 g Kg^{-1}) compared to SPL (104.00±13.08 g Kg^{-1}) and LNT (68.00±12.72 g Kg^{-1}). Soil total extractable carbon (TEC) was 80.20±2.13 g Kg^{-1}, 57.50±1.32 g Kg^{-1} and 41.53±7.26 g Kg^{-1} in GRS, SPL and LNT respectively with significant differences for $p<0,01$. Significant differences ($p<0.05$) were observed also on Humic and Fulvic acids (C HA+FA) values corresponding to 54.13±6.33 g Kg^{-1} in GRS, 44.83±2.85 g Kg^{-1} in SPL and 32.30±8.35 g Kg^{-1} in LNT (Figure 3.6). No significant differences were observed between the values of HR which correspond to 36.88 in GRS, 43.11 in SPL and 47.50 in LNT as well as for C:N ratio that showed values of 12.43±0.24, 11.17±0,22 and 10.03±0.23 in GRS, SPL and LNT respectively. Soil C stock was found to be higher in GRS compared to the other two fields ($p<0.01$) showing values of 265.25±1.40, 161.14±11.64 and 170.89±27.14 t C ha^{-1} in GRS, SPL and LNT respectively (Figure 3.7).

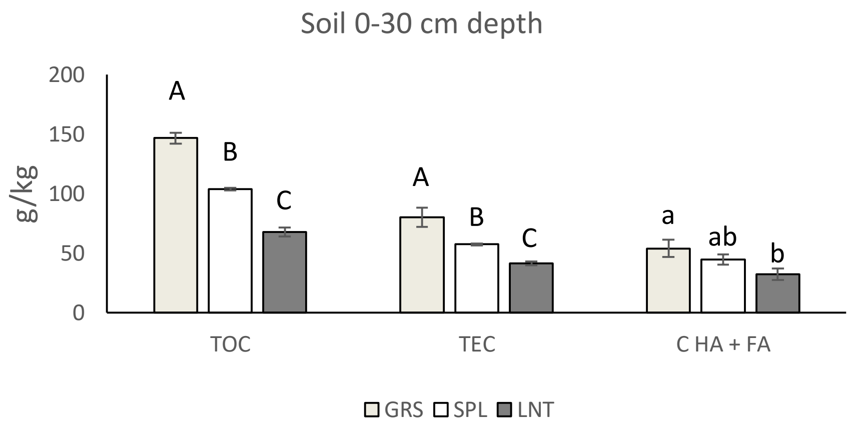

Figure 3.6. TOC, TEC, humic and fulvic acids (C HA+FA %) of the three fields. Quantification of TOC was performed with the Walkey-Black method. Extraction of TEC and HA-FA fractioning was performed using the method described by Ciavatta et al. (1990). Bars labelled with the different lowercase letters at the same site denote significant difference at p<0.05 while uppercase letters denote differences at p<0.01. Vertical bars indicate standard error.

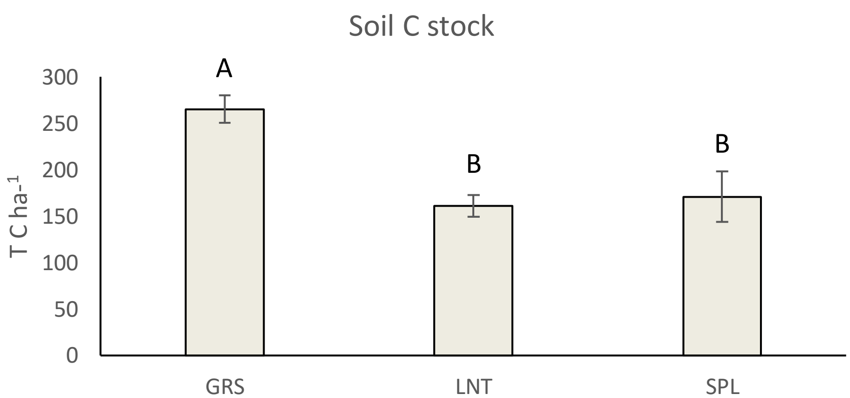

Figure 3.7. Soil C stock of the three fields. The bars represent the standard error. Bars labelled with the different uppercase letters at the same field denote significant difference at p<0.01. Vertical bars indicate standard error.

4. Discussion

In line with temperate climates, the soil T dynamic in each field shows a growing trend similar to the one of air temperature reaching the maximum around July then gradually decreasing (Figure 3.1). Soil moisture dynamics where similar in each field throughout the period of study with the exception of LNT in the month of October were appeared the most marked differences with GRS and SPL. This differences are probably ascribable to the LNT harvesting occurred in the beginning of September. Despite numerous studies in carried out in temperate climates have reported seasonal variation of the soil respiration with single peaks (e.g. Fenn et al. 2010; Kang et al. 2003; Luo et al. 2016) in this study seasonal variation of R_S and R_H showed multiple peaks similar to what observed by Almagro et al. (2009), Rey et al. (2002) and Zhu et al. (2016) in semi-arid climates.

A bibliography comparison on annual soil respiration between the fields of this study tends to be very difficult in light of the lack of researches on mountain crops like lentil or spelt. However, the annual mean R_S and R_H rates of LNT (1.83 and 0.81 μmol CO_2 m^{-2} s^{-1} respectively) and the ones of SPL (1.66 and 1.15 μmol CO_2 m^{-2} s^{-1}) are partially consistent with what reported by Raich and Tufekciogul (2000) which reported no significant differences between grasslands and nearby croplands.

GRS showed an annual mean R_S and R_H of 2.39 and 1.45 μmol CO_2 m^{-2} s^{-1} respectively. These results appear to be in line with what observed by Bahn et al. (2008) in different pastures over Europe but higher compared to than the annual respiration rates of world's major terrestrial biomes reported by Raich and Schlesinger (1992). The exponential regression models were able to explain 51% of R_H variation and 52% of R_S variation. The derived Q_{10} values (2.72 for R_S and 3.00 for R_H) calculated from the previous models are similar to the one reported by Cao et al. (2004) in the north-eastern Tibetan Plateau.

In agreement with Lal (2004) the results of soil indicators analysis suggest that TOC and TEC values present in GRS were higher compared with the ones of the two arable fields. Despite this, significant (p>0.05) differences of C HA+FA where detected only between GRS and SPL. The HR values at 0-30 cm soil depth ranged from 36.88 in 47.50 are higher than the one reported by Putra et al., (2016) but comparable to the ones reported by Moraes et al. (2011).

The estimation of Soil C Stock of GRS (265.25±1.40 t C ha^{-1}) showed higher values than many other researches performed on grasslands (e.g. Drewnik et al., 2016; Matsuura et al. 2012; Nakagami et al. 2009; Leifeld and Kögel-Knabner 2005). Consistently, also the soil C stock showed by SPL (161.14±11.64 t C ha^{-1}) and LNT (170.89±27.14 t C ha^{-1}) were much higher than other researches conducted on mollisols (e.g Puget and Lal, 2005; Novelli et al.,2017). Overall, soil C Stock values reported in this work seems to be comparable to the ones reported by Ausseil et al. (2015) in wetlands mineral soils of New Zealand or by Liu et al. (2016) in swamp grassland of Qinghai Plateau in China.

Guo and Gifford (2002) provided a meta-analysis on Soil C stock and land use changes that reported that a conversion from pasture to cropland reduced the amount of C of at least 50%. This study reports a slightly lower percentage which is about 44.93% in LNT and 37.85% in SPL.

5. Conclusion

In conclusion, the land cover changes from pasture to croplands determined different seasonal R_S and R_H spatial and temporal dynamics. Despite differences between GRS and LNT annual R_H means, no differences were found among annual R_S means between the three experimental fields. Overall, both temporal variations of R_S and R_H were controlled by soil temperature showing R^2 values ranging from 0.52 to 0.54 for R_S and from 0.47 to 0.55 for R_H. Q_{10} calculated from R_S exponential regression model showed the lower value in GRS (2.72) compared to the ones of SPL (4.95) and LNT (3.32). Q_{10} values obtained from R_H values showed different results corresponding to 3.00 in GRS. 3.67 in SPL and 2.01 in LNT. The LCC determined significant differences concerning TOC, TEC C HA+FA and C stock where GRS showed always the highest values except C HA+FA which were not different between GRS and SPL. No differences were also found concerning HR and C:N ratio of the three analysed fields.

This paper, in line with many other researches (e.g. Conant, et al., 2001; Diekow et al., 2005; Guo and Gifford, 2002; Houghton et al., 2012; Lal et al., 2004) , highlights that the LCC from a grazing to an arable land led to a depletion in terms of quantity and quality of the soil. For these reasons, a more general evaluation appears necessary to evaluate the LCC effects (Falkowski 2013) because trade-offs between ecosystem services (e.g. carbon stocking food provisioning or habitat for species) are likely to occur (Duraiappah et al. 2005).
If in one hand the lack of international references concerning soil C dynamics of mountain crops makes extremely difficult any comparisons, on the other hand this highlights the need to carry out further research on such systems. In this vision, the development of indicators that take into account the trade-off between ecosystem services could represent a valid contribute for many stakeholders such as tourists, policy makers and farmers.

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General Conclusions

The presence of Sheep transhumant systems in central Italy represent a source of biodiversity therefore an opportunity to preserve or enhance Ecosystem Services. By definition, transhumant systems envisage seasonal movements of flocks, that is the reason why this research is included in a wider project that include the analysis of many ecosystem service along a transect that follows the seasonal movements of the animals. From the analysis clearly emerges the need to preserve sheep transhumant system in order preserve the regulating ecosystem services of soil carbon sink. That is the presence of transhumant allows the presence of alfalfa fields in areas where otherwise conventional cropping systems (e.g. wheat) would take place. It is not a new issue that semi-permanent or permanent grasslands improve soil carbon stock, but from a point of view of Ecosystem service approach, the stocking of carbon in soils is only one, although crucial, service to humankind.

It would be unreasonable to conceive a future scenario where each Ecosystem Service is provided at a maximum level because tradeoffs are likely to occur: for example, transhumant system in the mountain area provide lamb meat (provisioning service) but at the same time the sheep grazing activity prevent shrub encroachments. This in turn lead to decrement of biodiversity but on the other hand, systems closer to forest act as better carbon sink. Moreover, the disappearance of transhumant sheep system from the mountain area would mean the disappearance of alfalfa fields of costal area of Central Italy. Within such a complex system, in accordance with the main results of Ecosystem Service review, the inclusion of cultural Ecosystem Services seems of paramount importance. Possible future research to developed from the results of this thesis could be the development of an indicator based not only on bio-physical parameters but that take in consideration also the aesthetic value of ecosystems.