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Analysis of ecosystem services provided by agro-pastoral systems to support co-design of agri-environment-climate measures

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Abstract

Grazing systems can provide a wide array of ecosystem services, defined by the Millennium Ecosystem Assessment as "the benefits that people obtain from ecosystems" and classified in four main groups: i) supporting (necessary for the production of all other ecosystem services; e.g. primary production), ii) provisioning (products obtained from ecosystems; e.g. food, fresh water), iii) regulating (benefits obtained from the regulation of ecosystem processes; e.g. climate regulation) and iv) cultural (non-material benefits people obtain from ecosystems; e.g. aesthetic experiences).

The thesis aims to investigate some relevant ecosystem services provided by extensive agro-pastoral systems in the territory of the Marche region (central Italy) to support the co-design process of agri-environmental climate measures. The thesis originates from a review paper that analyses the trends and approaches used in the analysis of relevant ecosystem services provided by grazing system, according to the framework principles of the Millennium Ecosystem Assessment. In a following step, the research focuses on the analysis of the greenhouse gases emissions under different cropping systems and management options: soil respiration from a *Bromus erectus*-dominated grassland under different mowing intensities in the uplands and N_2O emissions in a alfalfa-wheat system with biochar application in the lowlands. A final chapter analyses the design process of agri-environmental measures at landscape scale implemented in several case studies.

The literature review revealed a misunderstanding concerning the concept of ecosystem services among stakeholders. The biodiversity was considered an ecosystem services *per se* and the anthropocentric vision of the ecosystem services was not accepted or understood, moreover a lack of a multi-sectoral approach in the analysis of ecosystem services and the integration of different knowledge emerged. Furthermore, cultural ecosystem services were poorly studied despite being considered the most relevant for local and general stakeholders and with some other relevant services, could foster agri-environmental schemes.

From the analysis of the soil respiration from a *Bromus erectus*-dominated grassland emerged that more intensive use did not significantly impact soil respiration and primary production on the short term. The analysis of the N_2O emissions in the alfalfa-wheat system suggests that: i) postponed tillage in autumn may mitigate nitrogen losses as N_2O after alfalfa termination; ii) the effects of biochar application on N_2O emissions and crop productivity should be analysed in a long term perspective to verify the 'biochar aging'.

The results obtained from the research could feed the hybrid knowledge, that with shift of the stakeholder role in the system are key elements for the co-design process of site specific, shared and landscape agri-environmental measures.

Riassunto

I sistemi pastorali possono fornire una vasta gamma di servizi ecosistemici che vengono definiti dal Millennium Ecosystem Assessment come "i benefici che le persone ottengono dagli ecosistemi" e vengono classificati in quattro gruppi: i) supporto (necessari per la produzione di tutti gli altri servizi ecosistemici; es. produzione primaria), ii) approvvigionamento (prodotti forniti dagli ecosistemi; es. cibo, acqua), iii) regolazione (benefici ottenuti dalla regolazione dei processi ecosistemici; es. regolazione del clima) e iv) valori culturali (benefici non materiali che la popolazione ottiene dagli ecosistemi; es. esperienze estetiche).

L'obiettivo principale della tesi è stato quello di utilizzare le analisi di alcuni dei più importanti servizi ecosistemici forniti dai sistemi agro-pastorali estensivi della regione Marche (Italia centrale), a supporto dei processi di co-progettazione di misure agro-climatico ambientali. Nel capitolo iniziale, attraverso un lavoro di review della letteratura scientifica, la tesi esamina le tendenze e gli approcci utilizzati nell'analisi di alcuni servizi ecosistemici forniti dai sistemi pastorali alla luce dei principi del Millennium Ecosystem Assessment. Successivamente, la tesi si concentra principalmente sull'analisi delle emissioni di alcuni gas ad effetto serra generati da diversi sistemi colturali e diverse gestioni: (i) respirazione del suolo di una prateria montana a dominanza di *Bromus erectus* sottoposta a diverse intensità di utilizzazione; (ii) emissioni di N_2O in un sistema medica-frumento, con e senza l'applicazione di biochar, in un'area collinare. Nell'ultimo capitolo, la tesi analizza i processi di progettazione di misure agro-ambientali per la gestione di problematiche ambientali a scala territoriale in numerosi casi di studio.

L'analisi della letteratura ha rivelato una generale confusione da parte dei portatori d'interesse relativamente al concetto di servizio ecosistemico. La biodiversità viene considerata un servizio ecosistemico di per sé e la visione antropocentrica dei servizi ecosistemici non è accettata o compresa. Dalla review emerge inoltre lo scarso utilizzo di un approccio multi-settoriale nell'analisi dei servizi ecosistemici, nonché quello di un'integrazione delle diverse conoscenze. In aggiunta, i servizi ecosistemici culturali risultano scarsamente studiati, nonostante siano considerati di grande importanza per i portatori d'interesse sia locali che generali e che, insieme ad altri importanti servizi ecosistemici, potrebbero favorire l'adozione di politiche e misure agro-ambientali.

Dall'analisi della prateria a dominanza di *Bromus erectus*, non emerge nel breve periodo un impatto significativo dell'intensità di utilizzazione sulla respirazione del suolo né sulla produzione primaria. Nel sistema medica-frumento, l'analisi delle emissioni di N_2O suggerisce che: i) la lavorazione posticipata in autunno può mitigare la perdita di azoto sottoforma di N_2O; ii) gli effetti dell'applicazione del biochar dovrebbero essere analizzati nel lungo periodo per verificare i possibili effetti dell'invecchiamento del biochar, anche sulla produttività della coltura.

I risultati ottenuti da questa ricerca potrebbero contribuire alla condivisione di conoscenze e alla formazione di una conoscenza ibrida che, insieme al cambio di ruolo dei portatori d'interesse del sistema, sono risultati elementi chiave del processo di co-progettazione di misure agro-climatico ambientali sito specifiche, condivise e a scala territoriale.

Contents

General Introduction

1. Preface

Human needs are related to the services provided by the ecosystems (Haines-Young and Potschin, 2010). Ecosystems can provide a wide range of material goods (e.g., wool, food) and also non-material benefits contributing, for example, to the spiritual well-being creating opportunities for the enjoyment of nature (Haines-Young and Potschin, 2010). People transform the landscapes though the land use management affecting the status of ecosystems and consequently the ecosystem services (ES) supply (Quintas-Soriano *et al.*, 2016).
All economies are connected to the provisioning of ES (Alcamo *et al.*, 2003) although many ES have an intrinsic value that cannot be included in a market framework (Farley, 2012).
The widely accepted definition of ecosystem services is provided by the Millennium Ecosystem Assessment (MA) (Alcamo *et al.*, 2003) where they are defined 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, nutrient cycling), where the impact on people is either indirect or occurs over a very long time period; (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 control; and (iv) Cultural: non-material benefits people obtain from ecosystems through spiritual enrichment, cognitive development, reflection, recreation and aesthetic experiences.
Grazing systems is defined as complex structure that emerges from the interaction of human activities aimed at raising livestock with natural resources. In these systems a community of breeders shares productive direction, traditions and cultural values (Caballero *et al.*, 2009). In this production system, breeding is mainly based on grazing which is often the best way to use and maintained the marginal rural areas (i.e., European Less Favoured Areas) over particular biogeographical region in which this system are actives and can also be defined as Large Scale Grazing System (LSGS) (Caballero *et al.*, 2009). These pastoral or agro-pastoral systems can include a gradient of intensification and met two main threats: intensification and abandonment (that can be considered the most extreme form of extensification) (Caballero et al., 2007).
These production systems, based on grazing as one of the main management practices adopted across the grazing lands (Allen et al., 2011), cover just less than half of the world's

usable surface (De Haan *et al.*, 1997) and can provide a wide array of ES (e.g., food supply, C storage, water quality, aesthetic value) (Ford et al., 2012) included in all of the four categories (e.g., primary production, food, climate regulation, natural heritage) (D'Ottavio *et al.*, 2018).

These ES are, in turn, dependent on the system characteristics, such as the grazing animals used in the systems or the different management practices (Fischer et al., 2010; Steiner et al., 2014), such as different grazing regimes (Ford et al., 2012).

'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 (i.e., rangelands and artificial pastures, but also permanent and temporary meadows, ley crops and any aftermath-grazed meadows).

According to relevant literature (e.g., Caballero et al., 2009; Seré *et al.*, 1996), grazing systems can be classified as follows: 1) *pastoral systems* (i.e., pastoralist, outdoor system), in which natural and/or semi-natural permanent vegetation (i.e., rangelands, grasslands, shrublands) is exploited through grazing livestock mobility throughout the whole year; 2) *agro-pastoral systems*, distinguished in (a) *extensive* (outdoor, in-/out-door), in which natural and/or semi-natural permanent vegetation (i.e., rangelands, grasslands, shrublands) and other crops (i.e., temporary crops, temporary pastures/meadows, aftermath-grazed meadows, fallow) and woodlands are exploited through grazing livestock mobility during the whole year (outdoor) or a part of the year (in-/out-door); (b) *intensive* (outdoor or in-/out-door), in which temporary (artificial) forage crops (i.e, meadows, pastures, aftermath-grazed meadows, other crops) are exploited through grazing livestock over the whole year (outdoor) or over a part of the year (in-/out-door), with or without the adoption of livestock mobility; 3) other systems (i.e., agrosilvopastoral, silvopastoral systems), defined as land-use systems or practices in which trees/shrubs are deliberately integrated with crops and/or animals on the same land management unit.

The abovementioned classification, mostly based on the different *origin of the main forage resources* and the *feeding management* used in the system, includes different main criteria: (i) the *breeding method adopted in the system*: 1) outdoor: outdoor grazing throughout the whole year; 2) in-/out-door: outdoor grazing during the favourable season and indoor stalled animals during the unfavourable season; (ii) the *mobility method* adopted in the system: 1) nomadic, in which constant movement to provide feedstuff to the livestock is adopted, mainly by feet and involving a full time engagement of a shepherd; 2) transhumant, constant by feet movement of the livestock (great distance displacement of animals) in order to provide sufficient feedstuff, in case, accompanied by short-term vehicle transport of the animals between several large blocks which are not reachable by feet (e.g., urban areas, long distance between pastures). Nowadays the old transhumant system has been almost entirely replaced by small-vertical movement (100 - 200 km) of the animals and it's defined as trasterminance; 3) sedentary, in which grazing is performed in one or more different large grazing blocks that are close or have direct contact with the farmstead and in which livestock movements are performed by feet or by short-term vehicle transport; (iii) the *forage resources* used in the systems: 1) rangelands, lands on which the native vegetation (climax or natural potential plant community) are predominantly grasses, grass-like plants, forbs, or shrubs suitable for grazing or browsing use. They include natural and/or semi-natural grasslands, savannas, many wetlands, some deserts, tundras, alpine communities and marshes and certain forb and shrub communities, but also areas that have been seeded with introduced specie which are extensively managed like native range; 2) artificial (temporary) pastures, those lands that have been seeded, usually with introduced species and intensively managed using agronomy practices and control of livestock. They are temporary, short-term (two to five years) especially sown pastures comprising grasses and legumes and used for grazing in succession to other crops after a predetermined period.

2. Aims and structure of the thesis

In line with the framework principles of the MA (Alcamo *et al.*, 2003), the thesis aims to analyse some of the main ES provided by extensive and large-scale agro-pastoral systems active in the territory the Marche region (central Italy).

In central Apennine the livestock breeding has been one of the most important economic activities, in particular the pastoral activities influenced significantly the environment (Caballero *et al.*, 2009). Although after the Second World War, a severe crisis affected the livestock sector with reduction of the stocking rates or abandonment of pastoral practices (Caballero *et al.*, 2009), this system is still existent and active in all the regional territory. Both cattle and sheep in-/out-door system (mainly in the mountain and high hill areas) and cattle in-door system (mainly in low hill and plain areas) are present. Transhumance system (mainly sheep-based) is implemented through the trasterminance in all the regional territorial areas.

In particular, in the mountain area both sedentary and transhumant systems adopt vertical trasterminance, moving the livestock to upland permanent grasslands (e.g., spontaneous or secondary origin and natural or primary origin) in the summer period (e.g., from May-June to September-October, or a longer period with favourable weather conditions). While at the end of this period, sedentary farms move animals to pastures at lower altitudes and during the winter back to stables, the transhumant farms move the flocks to lowlands on temporary forages (mainly on long-term alfalfa meadows), but also on vineyards, orchards, olive groves and marginal lands (Caballero et al., 2009).

In view of the above, this PhD thesis aim to analyse some ES (i.e., greenhouse gases mitigation, primary production and crop production) provided by: (i) permanent grasslands, commonly used during the summer period as pasture or aftermath-grazed meadows in mountain areas, and (ii) long-term alfalfa, commonly used during the winter as pasture or aftermath-grazed meadows in low hill area included in ordinary winter cereal-based rotation.

These experimentations are part of a wider research that takes into account the analysis of other ES (i.e., primary production from the mountain permanent grassland, soil fertility, nutrient cycling through enzyme activities analysis) and of the plant biodiversity, that according to Alcamo *et al.* (2003) is the necessary condition for the delivery of all ES.

The thesis originates from a paper (D'Ottavio *et al.*, 2018) which reviews the trends and approaches used in the analysis of some relevant ES provided by grazing systems, in line with the framework principles of the MA.

The thesis mainly focused on the analysis of the greenhouse gases emissions, crop and primary production from grassland under different management. The results of these studies can be used as dialogical tools for the co-design of agri-environment-climate measures (AECMs) or to any other concerted stakeholders' action (e.g., short supply chains).

The agricultural intensification of the last decades in Europe, led to a sharp reduction of landscape diversity with a consequent decline in biodiversity (Stoate *et al.*, 2001) and increase of environmental problems. To respond to this problems of environmental quality demand by the society and to support the development of rural areas, in 1985 the European Union introduced agri-environmental measures (AEMs) and later these were included in the Rural Development Programmes (RDPs) (Toderi *et al.*, 2017). The AEMs are adopted by farmers on a voluntary basis and to be successfully and accepted by stakeholders the co-design process should be shared among them and include the knowledge bases (e.g., scientific, local, policy) (Toderi *et al.*, 2017).

As emerge from the last chapter of the thesis, the adoption of a virtuous and participatory process with the inclusion of the scientific knowledge, is necessary for the emergence of shared, site specific and landscape AEMs. In this view, the results from the two experimental field studies can therefore be used for supporting the definition of AECMs.

The thesis contains four chapters, based on the research activities performed during the PhD. Each chapter was written as a stand-alone manuscript and the first, second and last chapter were published on peer review journals.

The body of the thesis is the following:

1. A review paper entitled "**Trends and approaches in the analysis of ecosystem services provided by grazing systems: A review**". This chapter dealt with the ES framework that is one of the most accepted instruments for the evaluation of ecosystems and their functions. This article reviews the trends and approaches used in the analysis of some relevant ES provided by grazing systems, in line with the framework principles of the MA. This review includes 62 eligible papers and highlighted misunderstandings concerning the concept of ES: (i) biodiversity was considered as an ES *per se,* (ii) anthropocentric vision of the ES were not accepted or understood, (iii) lack of a multi-stakeholder's approach, (iv) lack of ES interaction analysis, (v) ES concept is not clear for many stakeholders.
2. A full research paper entitled "**Soil Respiration Dynamics in *Bromus erectus*-Dominated Grassland under Different Management Intensities**". In this chapter is investigated the influence of different management intensities, in terms of mowing regimes, applied for three consecutive years on soil respiration, soil temperature and soil moisture, over the short-term in a permanent mountain *Bromus erectus*-dominated grassland. From this study it emerged that the different mowing frequency had no effects on the soil water content over the three year. Occasionally mowing frequency changed the soil temperature. These changes did not have any impact on seasonal mean soil respiration with the exception for the first growing season when intensive utilization promoted higher seasonal mean soil respiration compared to the abandonment. Moreover, within the same mowing frequency, the soil temperature was the main driver of soil respiration only when the soil water content was above a

threshold; below this threshold the soil respiration was driven by soil water content. This suggests that a more intensive grassland use did not significantly impact soil respiration of *Bromus erectus*-dominated grassland on short term and that integrated analysis of multiple case studies, also using modelling application, would contribute to confirm the dynamics observed for the whole *Bromus erectus*-ecosystem. Future studies should clarify the role of the root, mycorrhizal and microbial respiration in the light of climate change considering also the projected scenarios of the seasonal redistribution of precipitation patterns.

3. A full research paper entitled: "**Nitrous oxide emissions as affected by perennial crop termination and biochar application in alfalfa-wheat rotation under Mediterranean environment**". In this chapter are investigated: (i) the impact of a legume perennial crop (six-year-old alfalfa) termination performed by spading and postponed in autumn (contrary to the traditional tillage system that require deep tillage performed in summer) on N_2O emissions in an alfalfa-wheat system under Mediterranean environment, (ii) the effectiveness of biochar incorporation on N_2O emissions mitigation. Cumulative, weekly and daily N_2O emissions are measured in three different treatments: (i) six-year-old alfalfa, (ii) durum wheat (*Triticum turgidum* L. ssp. *Durum* (Desf.) Husn.) after six-year-old alfalfa termination in autumn and (iii) durum wheat after six-year-old alfalfa termination in autumn amended with biochar. From this study emerged no differences among the treatments in terms of N_2O emissions mainly due to the: (i) low soil temperature and moisture during the autumn tillage that created unfavourable environmental conditions for denitrification processes and (ii) presence of the wheat (after alfalfa termination) that after a first increase of N_2O emissions (early wheat's stages) used the mineralized N reducing the N_2O emissions. Furthermore, no mitigation effects are visible in this first year after biochar application on N_2O emissions nor effect on crop productivity. This study provides new data concerning the effects of legume perennial crop termination, under Mediterranean environment, on N_2O emissions and the effects of biochar application after one year on N_2O emissions and wheat yield and quality. These results highlight the importance to: (i) postpone the legume perennial crop termination in autumn when the soil conditions are less favourable to denitrification process and (ii) a more evaluation to analyse the possible biochar aging effects on N_2O emissions and crop productivity.

4. A full research paper entitled: "**Bottom-up design process of agri-environmental measures at a landscape scale: Evidence from case studies on biodiversity conservation and water protection**". This study analysed in 9 case studies how different agri-environmental agreements at the landscape scale (AEAs) and the AEMs design process led to site-specific and landscape scale AEMs. From this analysis emerged: (i) that local policy makers should identify a set of targets in the RDPs without predefined measures, to create room for the bottom-up emergence of the AEMs, (ii) that the AEMs should emerge from participatory analysis of site specific condition by local stakeholders, (iii) the importance to involve stakeholders from the beginning and in all phases of the design process.

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Chapter I: Trends and approaches in the analysis of ecosystem services provided by grazing systems: A review

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Abstract

The ecosystem services (ES) approach is a framework for describing the benefits of nature to human well-being, and this has become a popular instrument for assessment and evaluation of ecosystems and their functions. Grazing lands can provide a wide array of ES that depend on their management practices and intensity. This article reviews the trends and approaches used in the analysis of some relevant ES provided by grazing systems, in line with the framework principles of the Millennium Ecosystem Assessment (MA). The scientific literature provides reports of many studies on ES in general, but the search here focused on grazing systems, which returned only 62 papers. This review of published papers highlights that: (i) in some papers, the concept of ES as defined by the MA is misunderstood (e.g., lack of anthropocentric vision); (ii) 34% of the papers dealt only with one ES, which neglects the need for the multisectoral approach suggested by the MA; (iii) few papers included stakeholder involvement to improve local decision-making processes; (iv) cultural ES have been poorly studied despite being considered the most relevant for local and general stakeholders; and (v) stakeholder awareness of well-being as provided by ES in grazing systems can foster both agri-environmental schemes and the willingness to pay for these services.

Keywords. climate regulation, food, habitat services, land degradation prevention, moderation of extreme events, natural (landscape) heritage, primary production, regulation of water flows, water quality regulation

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, Wilson, & Boumans, 2002). This is particularly the case since the publication of the Millennium Ecosystem Assessment (MA) (Fisher, Turner, & Morling, 2009). The MA (Alcamo et al., 2003; Millennium Ecosystem Assessment, 2005) represents one of the most extensive and widely accepted studies on the links between human well-being and the world's ecosystems. It 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.” According to Alcamo et al. (2003), the goal of MA is to establish the scientific basis for actions that are needed to enhance the contributions of ecosystems to human well-being without undermining their long-term productivity. The MA conceptual framework assumes that there is a dynamic interaction between people and ecosystems that requires a multi-scale approach, as this better reflects the multi-scale nature of decision making. Effective incorporation of different types of knowledge into ES assessment can both improve the findings and help to increase their adoption by stakeholders. The MA conceptual framework places human well-being as the central focus for assessment.

The MA identified four groups of ES: (i) Supporting: services necessary for the production of all other ES (e.g., soil formation, nutrient cycling), where the impact on people is either indirect or occurs over a very long time period; (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 control; and (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, The Economics of Ecosystems and Biodiversity (TEEB, 2010), 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), while considering supporting services merely as ecological processes, and not strictly as ES.

Although it is recognized that each ecosystem can produce a large number of ES (Alcamo et al., 2003; Millennium Ecosystem Assessment, 2005), ecosystems can 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 to group together a wide variety of anthropocentric arguments for the protection and sustainable use of ecosystems by humans. Schröter et al. (2014) also counter-argued six other main critiques to the ES concept that were derived from the 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 that also takes into account the local knowledge of stakeholders is a key requirement in assessing ES (Alcamo et al., 2003; Millennium Ecosystem Assessment, 2005; Reed, 2008).

According to Alcamo et al. (2003) and TEEB (2010), ecosystems and biodiversity are closely related concepts, although biodiversity is not strictly considered as an ES, but rather as a source or a regulator of the ecosystem (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., Harrison et al., 2014; Jax & Heink, 2015; Sircely & Naeem, 2012).

Livestock systems occupy about a third of the ice-free land surface of the planet, and they represent an important source of income; indeed, they can even be essential for the survival of vulnerable human communities. In these systems, grazing land can provide a large and differentiated number of ES (Porqueddu et al., 2016; Tarrasón, Ravera, Reed, Dougill, & Gonzalez, 2016). These ES are, in turn, dependent on the different management practices (Fischer et al., 2010; Steiner et al., 2014), such as different grazing regimes (Ford, Garbutt, Jones, & Jones, 2012).

This article reviews the trends and approaches used in the analysis of some relevant ES provided by grazing systems, in line with the framework principles of the MA. In the context of this review, grazing systems include production systems in which grazing is one of the main management practices adopted across the grazing lands (Allen et al., 2011). In this review we analyse: (i) if the papers follow the principles of the MA, and the main reasons behind their missed adoption; (ii) which are the most analysed ES, and which require further investigation within grazing systems; (iii) how different types of knowledge have been incorporated into ES assessment, as requested by the MA; and (iv) how ES concepts have fed the decision-making process. It is intended that the results of this review can be used to derive recommendations for research activities in the analysis of ES.

2. Links between biodiversity and ecosystem services

Biodiversity is the variability between living organisms, and it includes diversity within and among species and ecosystems. Biodiversity 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). 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 that is affected by the drivers of global change (e.g., climate, change in land use) and a factor that modifies ecosystem processes and ES, and indirectly, human well-being (e.g., health, freedom of choice and action). Changes in human well-being can lead to modifications of management practices, with direct effects on ecosystem processes and biodiversity (Figure 1). Although the MA describes a unilateral relationship between biodiversity and ES, some authors consider biodiversity as a service in its own right; e.g., as the basis of nature-based tourism (van Wilgen, Reyers, Le Maitre, Richardson, & Schonegevel, 2008). However, others consider that biodiversity can have different roles as a regulator of ecosystem processes, as a service in itself, or as a good (Mace, Norris, & Fitter, 2012).

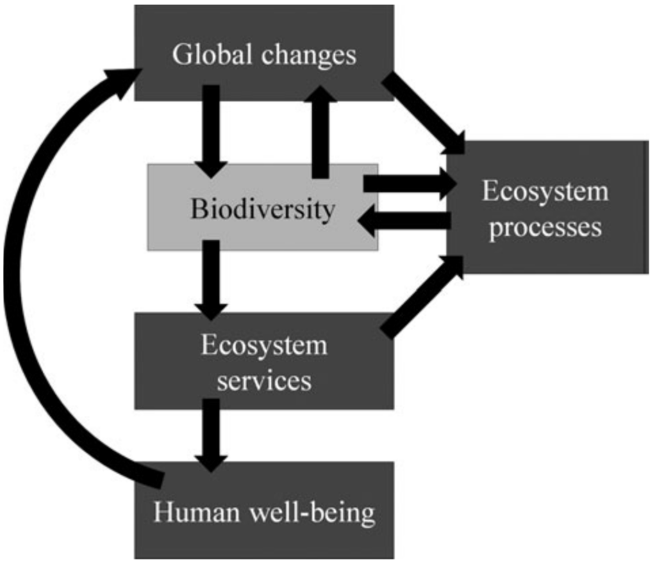

Fig. 1. Interrelationships between biodiversity, ecosystem functioning, and ES (modified from Millennium Ecosystem Assessment, 2005)

Habitat provisioning is one of the main ecosystem services that links the effects of livestock grazing to the biodiversity of the host ecosystem (Hoffman, From, & Boerma, 2014). Habitat services arise from the direct interactions of animals with their environments, and are hence related to land-management practices, especially in relation to grazing systems. Unlike the MA (Alcamo et al., 2003; Millennium Ecosystem Assessment, 2005), the TEEB (2010) considers habitat services as a separate category. In agreement with these documents, this review considers habitat services within supporting services, because of their interconnected nature and their shared roles in underpinning the delivery of other services.

3. Bibliographic search and analysis criteria

This review is based on the ES provided by grazing systems as categorized and defined as prominent by Hoffman et al. (2014; Table 1). Among these, the ES relevant to the expertise and background of the authors were analysed in detail: primary production (PP), habitat services (HS), food and other livestock-related products (FP), land degradation and soil erosion (LD), water quality regulation/ purification (WQ), regulation of water flows (WF), climate regulation (CR), moderation of extreme events (EE), and natural (landscape) heritage (NH) (Appendix 1).

Published papers dealing with ES were sampled in January 2016 using the Web of Science^{TM} (WoS). Within the search option of “topic”, 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”. To have a preliminary selection for each analysed ES, specific search terms were added to the basic string according to the keywords (Table 1) included in the Food and Agriculture Organisation report (Hoffman et al., 2014). The additional strings used for the preliminary selection are reported in detail in Table 2.

All of the papers extracted with the basic string (155 papers) were analysed to verify the adoption of the MA framework and the attribution of the papers to each ES, which was corrected as necessary. The analysis of the extracted papers allowed the identification of which ES were analysed for each paper in the light of the MA, and which did not take the MA into account (i.e., “ecosystem services” and/or “millennium ecosystem assessment” were cited merely in the Introduction or Conclusions sections).

After the analysis of the extracted papers the following manuscripts were excluded from this review: (i) papers dealing with an ES that was not analysed (ii) reviews, editorials and meta-analyses; and (iii) papers that did not adopt the MA framework.

Table 1. Papers dealing with ecosystem services provided by grazing systems returned by the basic string from the Web of Science™ and after selection according to the review criteria. Each paper can deal with more than one ecosystem service

Ecosystem services group	Ecosystem service	Description	Papers		
			Extracted[a] (n)	Satisfying analysis criteria[b] (n)	(%)
Supporting	Maintenance of soil structure and fertility	Nutrient cycling on farms and across landscapes; soil formation	12	n.a.	n.a.
	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	n.a.	n.a.
	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
	Fertiliser	Manure and urine for fertiliser	9	n.a.	n.a.
	Fuel	Manure and CH_4 for energy, manure biogas, etc.	11	n.a.	n.a.
	Power	Draught animal power	0	0	0
	Genetic resources	Basis for breed improvement and medicinal purposes	10	n.a.	n.a.
	Biotechnical/ medicinal resources	Laboratory 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	n.a.	n.a.
	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 run-off/ flooding	44	15	24
	Climate regulation	Soil carbon sequestration, greenhouse gas mitigation	60	31	50
	Moderation of extreme events	Avalanche and fire control	19	4	6
	Pollination	Yield/ seed quality of crops and natural vegetation; genetic diversity	17	n.a.	n.a.
	Biological control and animal/ human disease control	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	n.a.	n.a.
	Knowledge systems and educational values	Traditional and formal knowledge about breeds, grazing and socio-cultural systems of the area	23	n.a.	n.a.
	Cultural and historic heritage	Presence of the breed in the area helps to maintain elements of the local culture that are valued as part of the local heritage; cultural identity	21	n.a.	n.a.

	Inspiration for culture, art and design	Traditional art/ handicraft; fashion; cultural, intellectual and spiritual enrichment and inspiration; pet animals, advertising	12	n.a.	n.a.
	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 and the human life-cycle, such as religious ceremonies, funerals or weddings	0	0	0

n.a., not analysed.

[a]155 papers extracted from the Web of ScienceTM, for a total of 529 findings.

[b]62 papers, for 149 findings, satisfying the analysis criteria.

Table 2. Basic and additional strings used for the extraction of the papers, according to the keywords included in the Food and AgricultureOrganisation report (Hoffman et al., 2014)

Ecosystem service analysed	**Extraction string**
Ecosystem services (basic string)	"ecosystem service*" and ("grassland*" or "rangeland*" or "shrubland*" or "scrubland*") and "grazing"
Primary production	("primary production" or "vegetation growth" or "vegetation cover" or "vegetation" or "NPP" or "net primary production")
Habitat services	("species" or "habitat" or "life cycle")
Food and other livestock related products	("meat" or "milk" or "honey" or "wool" or "leather" or "hide" or "skin" or "wax")
Land degradation and soil erosion	("land degradation" or "erosion" or "cover crop*" or "vegetation cover")
Water quality regulation/ purification	("water quality" or "water regulation" or "water purification" or "water filtering in soil")
Regulation of water flows	("water" or "natural drainage" or "drought prevention" or "runoff" or "rainfall" or "flooding")
Climate regulation	("climate" or "soil carbon" or "greenhouse gas*" or "GHG" or "CO_2" or "CH_4" or "N_2O")
Moderation of extreme events	("avalanche*" or "fire" or "extreme event*")
Natural (landscape) heritage	("landscape" or "aesthetic" or "inspiration")

4. Trends and approaches in ecosystem services analysis

4.1. The extracted papers: numbers, exclusion, and reasons for exclusion

The basic string search returned a total of 155 papers (Table 1) with an increasing trend from 2010 (Figure 2). The multiple occurrence of different ES within single papers results in a total of 529 findings within the 155 papers. Most papers dealt in particular with supporting (mostly for PP and HS), regulating (in particular, CR and WF) and cultural (NH) ES. Only a few papers dealt with FP, and surprisingly, very few with food itself. The addition of some other terms to the basic string would have resulted in additional papers. For example, by adding or "good*" to the basic string, the total number of papers for FP would increase from 12 to 38. This highlights that many authors did not analyse food as an ES according to the MA framework. Similar considerations can be stated for the other ES analysed.

The total number of extracted papers is surprisingly low compared, for instance, with the far more numerous papers that have analysed grazing systems from an economic and/or biophysical perspective, but which did not adopt the MA framework. Indeed, by removing the keyword "ecosystem service*" from the basic string and maintaining the same time span, the number of papers reached 5,983.

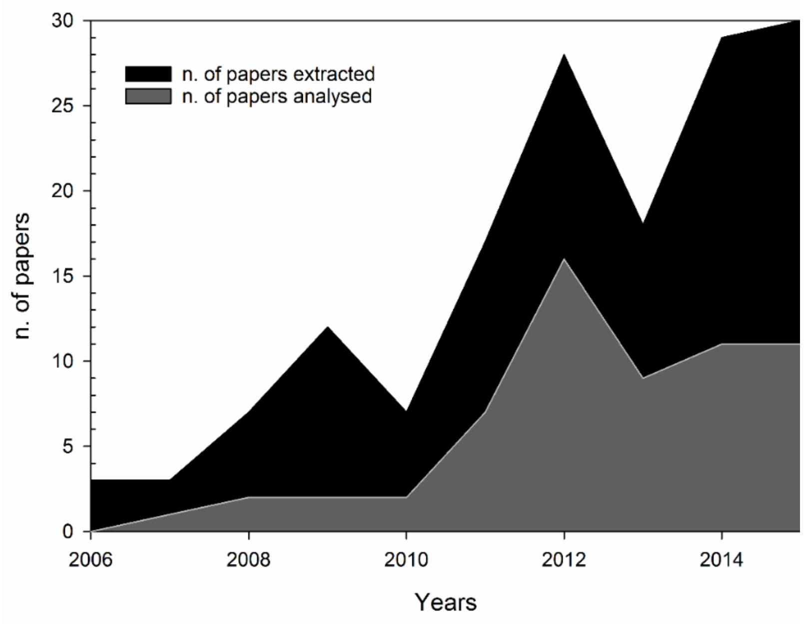

Fig. 2. Number of papers extracted from the Web of ScienceTM and analysed according to the review criteria

According to the review criteria, twenty-nine papers were excluded from this review, as reviews, editorials or meta-analyses, and sixty-four papers were excluded as they dealt only with ES that were not analysed in this review (e.g., fuel, power, pollination; nine papers) or because they did not adopt the MA framework (55 papers). In these papers, the term “ecosystem service” was present in the text (e.g., in the Introduction), and for this reason they were extracted.

Sixty-two papers (149 findings) were eligible for the present analysis. Natural heritage (NH) was apparently assessed in 25% of the papers, although it proved to be analysed as a cultural ES in only 6% of the papers (of 39 publications, four were eligible; Table 1). In the papers excluded from the NH category of ES, the landscape was considered: (i) for the effects that it can have on biodiversity (e.g., Cole, Brocklehurst, Robertson, Harrison, & McCracken, 2015; Kearns & Oliveras, 2009; Lindborg et al., 2009; Littlewood, Stewart, & Woodcock, 2012; Sanderson et al., 2007); (ii) as support for improving or maintaining other ES, but not as an ES *per se* (e.g., Lavorel et al., 2011, 2015; Schaldach et al., 2013); (iii) as an assessment scale for other ES (ee.g., Hussain & Tschirhart, 2013; Kimoto et al., 2012; Medina-Roldan, Paz-Ferreiro, & Bardgett, 2012; Peringer et al., 2013); and (iv) for the effects that different drivers had on it without directly analysing the consequences on its cultural value (e.g., Cousins, Auffret, Lindgren, & Tr€ank, 2015; Lamarque, Meyfroidt, Nettier, & Lavorel, 2014; Schaich, Kizos, Schneider, & Plieninger, 2015). The limited number of papers dealing with the landscape as a cultural ES might be explained by the difficulty for the measurement of this aspect, and to the few currently available indicators (Feld et al., 2009; TEEB, 2010). Rather than being considered as an EE, fire was analysed in some papers as a management tool for the enhancement of other ES (e.g., habitat provisioning, prevention of wildfires), and for this reason these papers were excluded from the EE analysis. For example, Joubert, Pryke, and Samways (2014) investigated the effects of annual burning on plant species richness, composition and turnover in three firebreak types, and under different cattle grazing levels. Boughton, Bohlen, and Steele (2013) conducted an 8-year split-plot experiment to study the effects of the season of burn on the plant composition of a semi-natural grassland in Florida (USA), where in addition to prescribed winter burns, natural historical wildfires occurred on abandoned ranchlands. The response of vegetation disturbance was studied (Hancock and Legg, 2012) for prescribed fire management in pine forests and ericaceous heathlands in the UK. These papers were excluded from the NH and EE analyses, but were included in the other ES analysed in this review; e.g., Lavorel et al.

(2011) was excluded from NH but was included in the HS, PP and CR analyses. “Landscape” and “fire” were considered as particular cases, as these can have different meanings (e.g., scale of investigation or management tools). The main reasons for the exclusion for the rest of the papers (e.g., Bai et al., 2012; Loucougaray et al., 2015; Zeng, Wu, & Zhang, 2015) were the lack of adoption of the MA approach or for only mentioning the term “ecosystem service” in the text (e.g., in the Introduction or Abstract). Table 1 summarizes these review categories according to the numbers of papers for each ES extracted by the strings, the numbers of papers eligible for the analysis, and the attribution of these papers to each ES.

4.2. The eligible papers: most and least analysed ecosystem services in combinations with each other

The predominance of papers dealing with PP (63% of the papers), HS (55%) and CR (50%) that emerged in the extracted papers was confirmed for the eligible papers. Although livestock production is clearly related to the forage characteristics of pastures (e.g., yield, quality, species diversity, plant active compounds) (Lieber et al., 2014), only five papers included PP and FP ES in the analyses (Figure 3). From the deep review of the papers, it clearly emerged that PP, CR and HS were often analysed together; that is, PP was assessed in 80% of the papers dealing with CR (e.g., Medina-Roldán et al., 2012; Oňatibia, Aguiar, & Semmartin, 2015) and in 60% of the papers dealing with HS (e.g., Duru, Jouany, Theau, Granger, & Cruz, 2015; Marriott, Fisher, Hood, & Pakeman, 2010), while HS was analysed in 40% of the papers dealing with PP or CR. At the same time, these three ES were assessed with at least one other ES (e.g., Lamarque et al., 2014; Miller, Belote, Bowker, & Garman, 2011); that is, PP was analysed in more than 70% of the papers dealing with FP (e.g., Koniak, Noy-Meir, & Perevolotsky, 2011) or LD (e.g., Giese et al., 2013), HS was analysed in 100% of the papers dealing with NH (e.g., Fontana et al., 2014), CR was analysed in about 70% of the papers dealing with FP (e.g., Ford et al., 2012) and in 60% of the papers dealing with WQ (e.g., Roche, O’Geen, Latimer, & Eastburn, 2014) or with WF (e.g., Fisher et al., 2011) (Figure 3). In the grazing systems, PP and HS were classified as supporting ES, and were thus placed at the base of all of the other ES. This explains the high number of papers that dealt with PP and HS. As a regulating ES, CR is a well-investigated topic because it is strongly linked to urgent climate-change issues. Indeed, even if CR was one of the most analysed ES, its analysis was mostly at a global scale, in terms of its role in net sequestration or net emissions of greenhouse gases, while none of the papers analysed how changes in land cover can affect both temperature and precipitation at local levels. There appears to have been little analysis of the relationships between the supporting ES, PP and HS and the other regulating ES; that is, WQ was assessed only in 3% and 4% of the papers dealing with PP and HS, respectively, while WF was analysed in about 20% of the papers dealing with PP or HS. Also, while 80% of the FP papers analysed the relation with PP and about 67% analysed the relation with HS, only 13% and 11% of the papers that assessed PP or HS included FP. A similar consideration can be derived for the cultural ES NH, where 100% of the papers analysed the NH relationship with HS, and 80% with PP. On the contrary, only 12% and 8% of the papers dealing with HS or PP included the effects of different management options on NH within their study (Figure 3).

This analysis highlights that the authors tended to concentrate their research on ES very close to each other in terms of their characteristics and relationships, and that they mostly focused on the supporting and regulating ES. Indeed, papers that dealt with ES that are distant from each other represented the minority; e.g., between HS and FP or NH. In the next section (4.3), the literature was analysed in terms of the advantages that derive from a multisectoral analysis which also includes the provisioning and cultural ES, and how this analysis allows inclusion of different stakeholders in the definition of shared management options or support policies (e.g., “Payments for ES” or “agri-environmental schemes”).

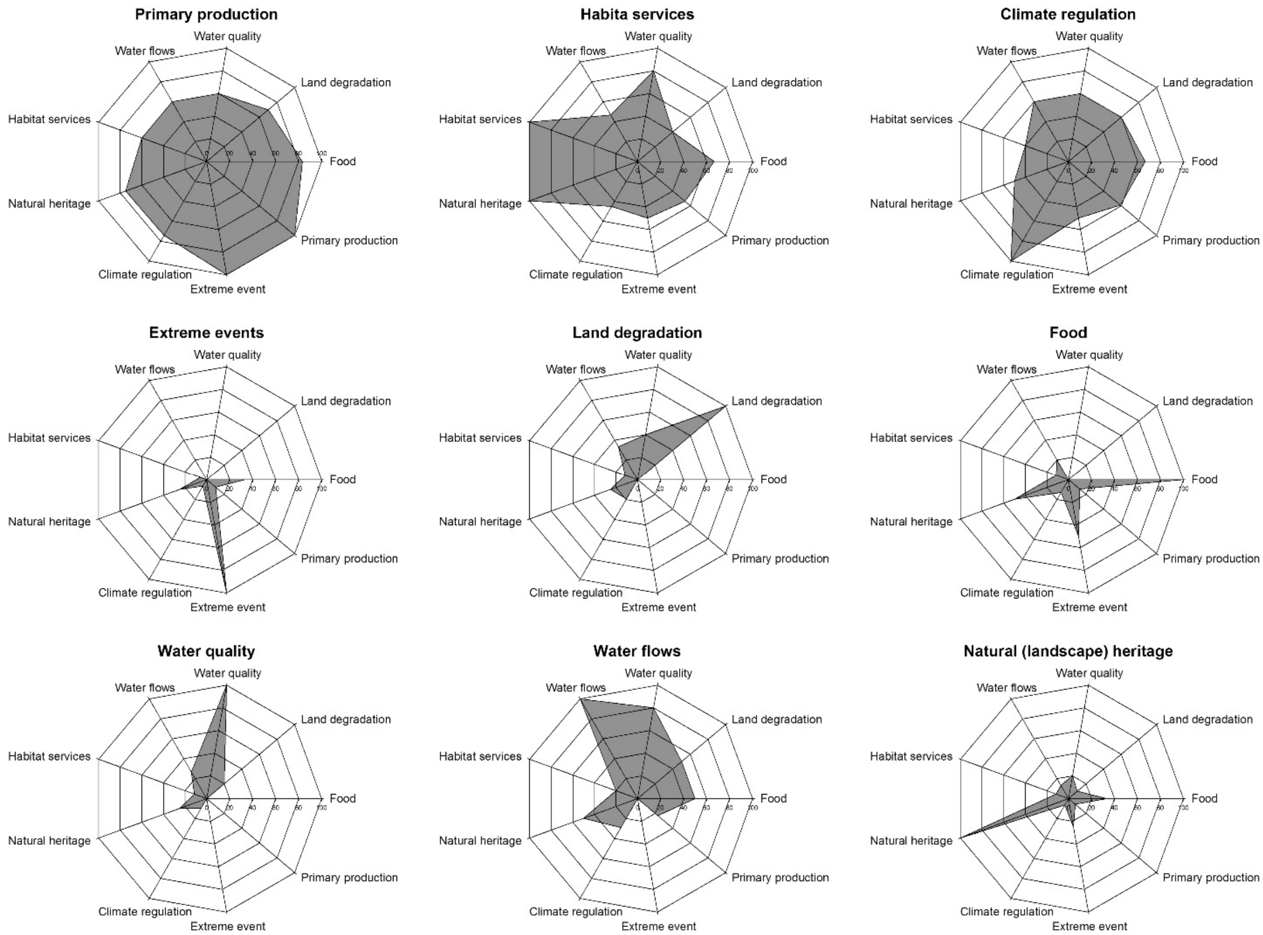

Fig. 3. Multisectoral approach in the 62 papers eligible for the review in which the Ecosystem Services are analysed in combination with each of the others. For example, Food is analysed only in 13% of the papers dealing with Primary production or Water flows is analysed in 45% of the papers dealing with Natural (landscape) heritage

4.3. Millennium Ecosystem Assessment principles in the eligible papers

Despite the MA (2005) recommending the implementation of a multisectoral approach to fully evaluate changes in ES, their interactions, and the trade-offs and impact on people, 34% of the 62 papers analysed just one ES (i.e., 10 out of 35 papers for HS; five of 31 for CR, and three of 39 for PP), and 23% analysed only two ES (Figure 4). Only 11% of the papers dealt simultaneously with more than five ES.

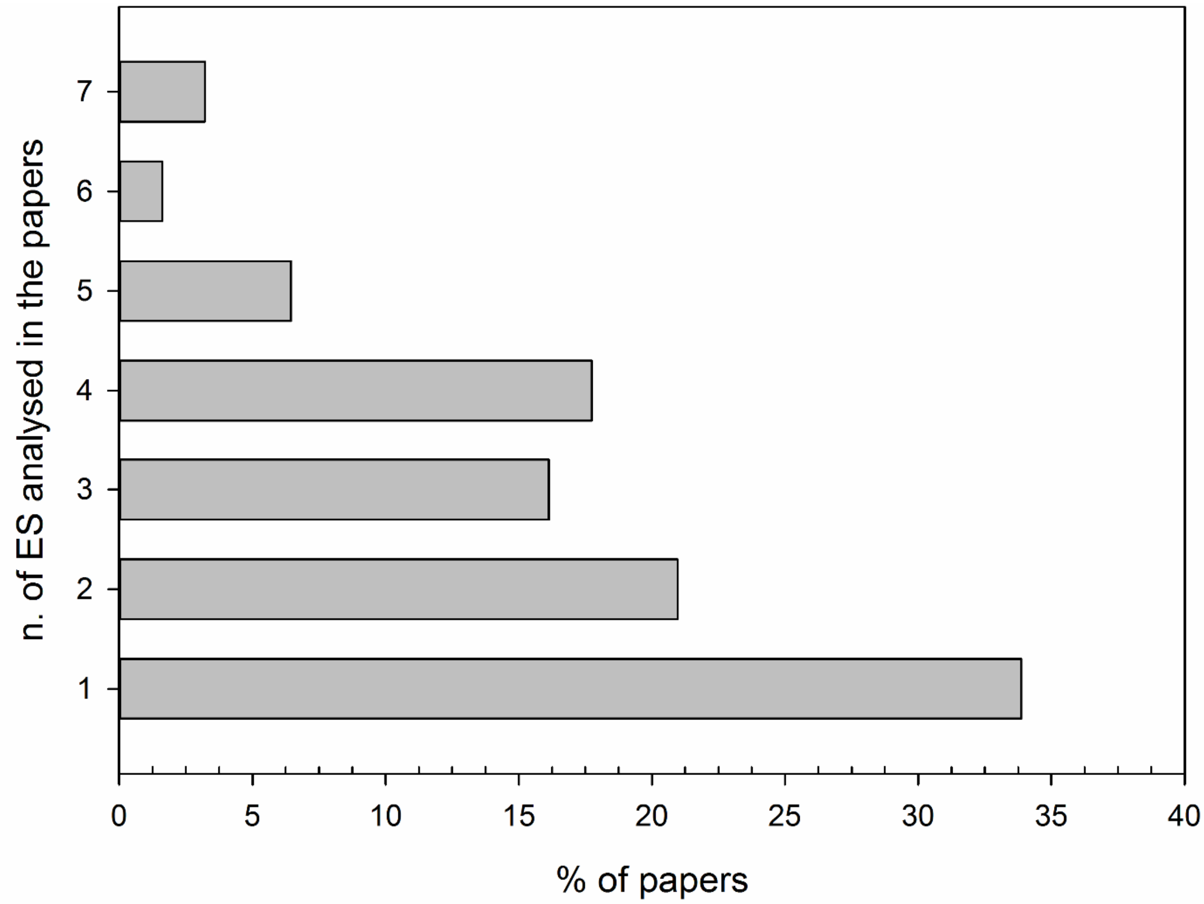

Fig. 4. Percentage of papers eligible for the analysis (n = 62) that dealt with one or more ecosystem services

The papers that dealt with one or a few ES turned out to be a very detailed analysis of the single ES, and at the same time, they lost the overview of the system and the potential other effects and trade-offs on the other ES. For example, Kimoto et al. (2012) analysed the effects of different intensities of livestock grazing on native bees, and they concluded that maintaining a heterogeneous landscape with some areas grazed and other not grazed, or with rotation of grazing, might be necessary to support native bee diversity. However, the consequences on FP and NH were not investigated by these authors.

In two interesting papers, Cole, Brocklehurst, Elston, and McCracken (2012), Cole et al. (2015) analysed the effects of the main physical and botanical attributes and of the different management options of riparian field margins on ground beetle and pollinator diversity, and they concluded that wide riparian margins, strategically placed within the landscape, can enhance taxonomic and functional diversity. Nevertheless, this study did not analyse the effects on the landscape as cultural ES (i.e., the aesthetic value) generated by the different management options, and so they missed the opportunity to highlight further positive effects or trade-offs.

Another example is provided by Peringer et al. (2013), who analysed silvopastoral systems as traditional components of the landscape in the Swiss Jura Mountains, for the prevention of the loss of species-rich open grasslands and forest-grassland ecotones. In this paper, the landscape was an assessment scale for the other ES (i.e., HS), and so it was not an ES.

Other authors enlarged their analyses to other ES, to highlight potential trade-offs or existing relationships; e.g., between different management options on FP or on the aesthetic value of the landscape to produce income from tourism. In this vision, Fontana et al. (2014) analysed the effects of management changes of larch grasslands in the Italian Alps (abandonment and intensification *vs*. traditional management) on PP, HS and pollination, and also on valuable cultural ES (i.e., scenic beauty, traditional healing plants). They conducted a phyto-sociological study on plots that were randomly selected using geographic information systems. For each plant species recorded, three 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 use in 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, and with specific financial support for a traditional agroforestry system.

Other authors analysed the effects of several scenarios (e.g., climate change, policies, management) on FP and on other ES for a more holistic analysis; e.g., Koniak et al. (2011) addressed issues related to honey production, and developed a mathematical model which predicted the dynamics of multiple services in response to management scenarios (grazing, fire, and their combination) mediated by vegetation changes. These authors combined the potential contribution to honey production with other ES from different groups into one "ES basket" (e.g., carbon retention for CR, forage production for PP, density of geophytes for HS), despite their different natures, which can help land managers to evaluate the effects and trade-offs of alternative management scenarios. Another example of a holistic approach is provided by Dong et al. (2012a), Dong, Yang, Ulgiati, Yan, and Zhang (2012b); Dong et al. (2014), who used the "emergy" approach to calculate the performance of several ES (i.e., CR, EE, FP, WR, PP) under different systems and scenarios, to support local resource management and larger-scale environmental resource decision making. "Emergy" was defined by Odum (1996) as the amount of available energy of one type, usually solar, directly or indirectly required to provide a given flow or storage of energy or matter.

Ford et al. (2012) used a wide range of ES for each of the MA category of ES to test the hypothesis that changes in grazing intensity of semi-natural grassland differentially affect individual services and alter the balance of supporting, provisioning, regulating and cultural ES provision. This holistic approach underlined that in addition to biodiversity measures of "success" in conservation, ES measures and trade-offs need to be taken into account when choosing an appropriate grassland management scheme. Reed et al. (2015) analysed a combination of many ES to produce tools and frameworks to support the stakeholder decision-making processes for land management. These authors identified new economic instruments (e.g., payments for ES) to enhance the flow of ES provided by grazing systems.

4.4. Ecosystem services, and different types of knowledge and decision making in the eligible papers

A further approach to the analysis of ES provided by grazing systems emerged from some papers that included the involvement of stakeholders in different phases of the evaluation process and with different aims. Some other authors applied a holistic approach which combined the ES analysis with stakeholder involvement to explore the relationships between land management and ES. This approach was intended to influence the decision-making processes, to increase the stakeholder ES knowledge and awareness of the consequences of their activity. Lamarque et al. (2014) applied a role game, in which farmers were faced with changes in ES (i.e., PP, HS, WQ, CR) 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 the farmers were not aware, for example, of the potential effects of their activities on nitrate leaching, and that feedback loops between

ES and land-management decisions can favour more sustainable ES management. A global-scale study was performed by Petz et al. (2014a) in South African rangelands that were affected by historical issues of land conservation and degradation due to overgrazing (e.g., vegetation cover, species diversity, soil erosion, carbon stock, water quality). These authors used the combined approach of a literature review, collected data, and models (i.e., "IMAGE-USLE") to study the interactions between input data, livestock density and ES to strengthen and optimize the choices of local stakeholders for the future management of the area in three different land-management scenarios. A further example of the effectiveness of the use of this approach to identify the best land-management options was provided by Fisher et al. (2011). These authors explored the variations in ES delivery that resulted 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, consultation with stakeholders and experts was carried out through workshops and meetings, to elaborate specific details of the management impact on CR, WQ and WR, linked to the range of management practices. These results are particularly relevant for the drafting of management plans which need to 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.

Other authors used different approaches for the analysis of ES, with the integration of scientific knowledge with local knowledge, to create "hybrid knowledge". In this vision, for a pastoral system of a semi-arid region of northern Nicaragua, Tarrasón et al. (2016) highlighted the importance of engaging relevant and interested stakeholders in dialogue with each other and with the researchers, and encouraging the participation of local stakeholders in the decision-making processes. They applied a participatory methodological framework to identify features of LD and links with other ES provisions. The study designed a four-step methodological framework to integrate local and scientific knowledge within a participatory assessment of land degradation. Field visits and in-depth interviews with key informants and farmers produced information that was integrated with the scientific knowledge that was validated by focus groups, and then used in a state-and-transition conceptual model. Field data on the cover vegetation and the plot life forms were used in thematic working groups with different stakeholders to discuss the results of the previous phases and to develop adaptive management options to maintain or improve ES.

The increase in awareness of local and general stakeholders (e.g., citizens, inhabitants, tourists) of the flow of ES provided by grazing systems was considered by some authors as a key element. The increased awareness of these stakeholders favours the acceptance of new economic instruments (e.g., Payments for ES), which increased their “willingness to pay” for ES. An example emerges from the analysis of Bernués, Rodríguez-Ortega, Ripoll-Bosch, and Alfnes (2014), who attempted to determine the socio-cultural and economic value of some ES delivered by mountain agroecosystems in northeast Spain (e.g., forest fires, habitats for species, aesthetic and recreational values of the landscape, product quality linked to the territory), by identifying stakeholder 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 that were derived from contrasting policies, and to test the willingness to pay for ES. Cultural ES were demonstrated to be a useful tool to engage with stakeholders to support grazing system policies. From this analysis, it emerged that the farmers were more interested in supporting ES, the local and general stakeholders were more interested in cultural ES, and the local stakeholders were more interested in the landscape than the general stakeholders. The willingness to pay for ES was higher compared with the current level of EU agri-environmental support.

5. Conclusions

The extraction criteria used for this bibliographic review resulted in a relatively small number of papers. The keyword “ecosystem service” was the dividing term between a vast literature that deals with biophysical and socio-economic features of the grazing systems and the relatively minimal number of results of papers in this analysis that used the ES concept.

Although the MA has been the most widely accepted ES assessment framework since 2003, the analysis of these extracted papers has highlighted misunderstandings concerning the concept of ES. One clear example is the confusion concerning biodiversity, which contrary to the MA, was considered in several papers as an ES *per se* (e.g., Lindborg et al., 2009; Mace et al., 2012). Furthermore, not all of the analysed papers understood or accepted the anthropocentric vision of the ES framework; e.g., some authors proposed biocentric solutions to reverse the inner dynamics of systems without taking into account stakeholder opinions or needs (e.g., 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), was applied in a number of these analysed papers (e.g., Koniak et al., 2011; Petz et al., 2014a). Management and development options should take into account the internal dynamics of systems and the biophysical components, and also the socio-economic, socio-cultural and institutional features (Caballero and Fernández-Santos, 2009). Despite this, only a few authors integrated a multi-stakeholder approach into their analysis of ES and the interactions between these (e.g., Bernués et al., 2014; Petz et al., 2014b; Tarrasón et al., 2016). The need for stakeholder involvement emerged in some papers that underlined how the ES concept was not familiar to stakeholders, and was often confused, for instance with the responsibility of humans to preserve nature (e.g., Bernués et al., 2014; Tarrasón et al., 2016). The use of ES as a basis for discussion might favour more sustainable practices, to increase the awareness of the effects of different management options on stakeholder well-being (e.g., Lamarque et al., 2014).

Other authors emphasized how the stakeholders and their knowledge inclusion is needed to improve the effectiveness of local decision-making processes (e.g., Lindborg et al., 2009; Tarrasón et al., 2016). The integration of local and scientific knowledge generates hybrid knowledge, thereby encouraging the participation of local stakeholders in the decision-making processes. This allowed the identification of adaptive strategies for key services to be maintained into the future (Francioni, Toderi, Catorci, Pancotto, & D'Ottavio, 2014; Lamarque et al., 2014), for example, through the implementation of *in-situ* experiments on native pasture management (Tarrasón et al., 2016). Many tools that are commonly used in scientific activities, such as mathematical models, future scenarios, indicators and biophysical data, were adopted by these authors to engage the stakeholders or to facilitate discussion with and between them.

In the analysed literature, cultural ES were poorly studied, despite these being considered the most relevant for local and general stakeholders (Bernués et al., 2014). This thus limited the ES framework to agriculture-related aspects. Better stakeholder awareness of the well-being provided by ES in grazing systems might foster agri-environmental schemes and the willingness to pay for these services. Many papers analysed and proposed different management options to improve the provision of ES (e.g., Cole et al., 2015), but did not analyse the effects on the natural heritage (e.g., the landscape aesthetic value), which can be relevant in policy-making processes (Bulte, Boone, Stringer, & Thornton, 2008) and, for instance, in the definition of Payments for ES. Compensation and market-related policies have gained prominence as mechanisms to encourage farmers, policy makers and land managers to change their behaviour, and these might represent a mechanism to align potentially opposing interests; e.g., in the areas of wildlife management and biodiversity conservation.

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APPENDIX 1

Ecosystem Services (ES) Provided by Grazing Systems Analysed in this Review

Primary production [PP] is a fundamental service that is defined in Alcamo et al. (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 (2005), primary production is considered as a provisioning ES when harvested and sold outside the commercial fields, or as a supporting ES if used as basic feed for wild or domestic animals.

Habitat services [HS] 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 in co-evolved landscapes. The most important clusters of habitat services provided by grazing systems are those that support the maintenance of species life cycles and those related to the connection of habitats.

Food and other livestock related products [FPs] in grazed ecosystem categories include provision of high-protein meat and dairy products, along with leather and other by-products of livestock production (Steiner et al., 2014).

Land degradation and soil erosion [LD] are regarded not just as loss of soil and fertility, but also as deterioration of balanced ecosystems and the loss of ES (Nachtergaele et al., 2011).

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 they can also help to filter and decompose organic waste introduced into inland waters (Alcamo et al., 2003).

Regulation of water flows [WF] deals with the timing and magnitude of run-off, flooding and aquifer recharge, which can be strongly influenced by changes in land cover, including alterations in the water storage potential of a system (Alcamo et al., 2003).

Climate regulation [CR] influences the climate, both locally and globally. For example, at local levels, changes in land cover can affect both temperature and precipitation. On a global scale, ecosystems have important roles in climate regulation, by either net sequestration or net emission of greenhouse gases. This ES is receiving increasing attention, as the effects of climate change over the next century are expected to affect (directly and indirectly) all types of ecosystems and ES (MA, 2005).

Moderation of extreme events [EE] is an ES that mainly refers to the prevention of avalanches and wildfires through livestock grazing.

Natural (landscape) heritage [NH] is mentioned in the MA among the cultural services, and this includes values as shaped by the animals themselves or as part of the landscape (e.g., aesthetic values).

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Chapter II: Soil Respiration Dynamics in *Bromus erectus*-Dominated Grasslands under Different Management Intensities

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Abstract

Reduction of soil greenhouse gas emissions is crucial to control increases in atmospheric CO2 concentrations. Permanent grasslands are of considerable importance in climate change mitigation strategies as they cover about 13% of the global agricultural area. However, uncertainties remain for the effects of management practices on soil respiration, especially over the short term. This study investigated the influence of different mowing intensities on soil respiration over the short term for Bromus erectus-dominated grasslands in the central Apennines. From 2016 to 2018, soil respiration, temperature, and moisture were measured under three different management systems: customary management, intensive use, and abandonment. Both soil water content and temperature changed over time, however mowing did not affect soil water content while occasionally altered soil temperature. The intensive use promoted higher seasonal mean soil respiration compared to the abandonment only during the 2016 growing season. Soil temperature was the main driver of soil respiration above a soil water content threshold that varied little among treatments (18.23%–22.71%). Below the thresholds, soil moisture was the main driver of soil respiration. These data suggest that different mowing regimes have little influence on soil respiration over the short term in Bromus erectus-dominated grasslands. Thus, more intensive use would not have significative impacts on soil respiration, at least over the short term. Future studies need to clarify the role of root mycorrhizal and microbial respiration in the light of climate change, considering the seasonal redistribution of the rainfall.

Keywords: carbon cycle; CO_2; greenhouse gases; mowing; Natura 2000; permanent grassland; semi-natural dry grasslands

1. Introduction

According to the fifth assessment report of the Intergovernmental Panel on Climate Change, annual greenhouse gas (GHG) emissions from agricultural production in 2000–2010 were estimated at 5.0–5.8 Gt CO_2 eq. yr^{-1} while annual GHG flux from land use and land-use change activities accounted for approximately 4.3–5.5 Gt CO_2 eq. yr^{-1} [1]. The same report estimated that over recent decades, the cumulative CO_2 emissions from agricultural land uses (i.e., croplands, forest lands, and grasslands) have increased by about 40%.

The CO_2 efflux released into the atmosphere by the biological activities of plant roots, soil microbes and animals is defined as soil respiration [2], and this represents the second-largest carbon flux between the terrestrial ecosystem and the atmosphere on a global scale [3]. As well as global warming, which might have positive feedback on soil CO_2 release from the soil carbon pool [4], human activities have crucial roles in soil carbon cycling, in terms of land-use changes and agricultural practices [5,6].

To face this issue, there is a need to adopt mitigation strategies to preserve the carbon pools of terrestrial ecosystems [5,7,8]. In particular, the soil carbon pool of the first soil layers is one of the main abiotic factors affecting soil respiration [9], thus mitigation strategies must include the best practices that address the soil (e.g., tillage, amendments application, land-use change), the crops (e.g., nutrients, residues, water management), and the livestock (e.g., feeding, manure management, stocking densities) to reduce GHG from agricultural systems [1].

Permanent grasslands might indeed be important for climate change mitigation strategies as they cover about 13% of the global agricultural area [10], which represents more than a third of the European agricultural area [11]. Projected scenarios have suggested that grasslands may face major issues due to temperature and atmospheric CO_2 concentration increase as well as changes in precipitation patterns [5]. The latter appears to be particularly relevant in areas with low summer rainfall (i.e., southern Europe) because precipitation after prolonged dry periods can enhance CO_2 efflux from the soil to the atmosphere [12,13].

Bromus erectus-dominated grasslands are semi-natural communities of secondary origin that are included in the list of habitats of European Community interest: Habitat code 6210(*) "Semi-natural dry grasslands and scrubland facies on calcareous substrates (*Festuco-Brometalia*) (*important orchid sites)" (EU Habitats Directive 92/42/EEC, Annex I). These are also among the most prevalent grasslands, with about 600,000 ha in the EU Natura 2000 network, with the largest areas reported for Italy (33%) and Romania (25%). Natura 2000 is a network of nature protection areas (both terrestrial and marine) in the territory of the European Union. It is made up of Special Protection Areas (SPA) and Special Areas of Conservation (SAC) designated under the Birds Directive (EU Birds Directive 79/409/EEC replaced by Directive 2009/147/EC) and Habitats Directive (EU Habitats Directive 92/42/EEC), respectively.

High levels of animal and plant biodiversity are linked to permanent grasslands [14,15], which in turn provide a large number of ecosystem services [10,16]. These include supporting services (e.g., primary production, nutrient cycling), cultural services (e.g., landscape aesthetic value, recreational experiences), provisioning services (e.g., food and forage production), and regulating services (e.g., carbon sequestration, erosion control) [17]. Secondary grasslands include numerous plant communities whose characteristics vary in relation to the environmental conditions and, in relation to their potential production, they are subject to different management intensities [18].

Many studies on permanent grasslands have focused on the effects of management practices on forage yield and herbage quality (e.g., [19]) and/or on plant diversity (e.g., [20,21]), but little is known about soil respiration compared to other ecosystems (e.g., forests) [22]. The need to further investigate their role in GHG mitigation strategies is highlighted by the few inconsistent data about the effect of management practices (e.g., grazing and/or mowing intensity) on soil respiration and its drivers [23]. Indeed, many papers reported contrasting results on the effect of management practices that involve herbage removal on soil temperature and/ or water content, which are among the main drivers of soil respiration. For example, herbage removal might increase soil temperature [24], but the tall-grass cover might cause a shading effect and thus decrease it [25]. In many case-studies mowing had no effect on soil moisture (e.g., [24,26]), while in some other cases the opposite effect is reported (e.g., [25,27]).

Results on the effect of management practices on soil respiration and its drivers appear even more uncertain when the time-scale is also considered. For example, it has been reported that over the medium to long term, heavy grazing on permanent grasslands might alter the soil respiration by alterations to the soil microbial communities, while light grazing has more limited effects [23]. A recent study on permanent grasslands reported that a land-use change from permanent grasslands to arable lands does not influence the microbial soil respiration in mountain areas, while the soil carbon stock can be halved over a few decades due to soil tillage and erosion [28]. Switching from the long-term to the short-term perspective (1–3 years), some studies have reported that mowing results in reduction of soil respiration [24,25], while other studies have reported that soil respiration of permanent grasslands might not be influenced by mowing [26]. A 2 year experiment in a Mediterranean grassland showed that defoliation reduced the carbohydrates translocated to the roots affecting/reducing both microbial and root-derived CO_2 effluxes [29]. In contrast, a short-term experiment (8–14 days) carried out in temperate mountain grasslands reported a soil respiration decrease due to a reduction in microbial activity (e.g., [30]). At the same time, root respiration was affected to a lesser extent since it was supported by carbohydrates present in the reserve organs. Thus, management practices appear to have significant effects on soil respiration even over the short-term, although on a basis that tends to be context-dependent.

To date, many studies have investigated the effect of management practices on soil respiration of dry or semi-dry grasslands (e.g., [24,26,31,32]), very few on *Bromus spp.*-dominated grasslands (e.g., [25]), and to the best of our knowledge, no one has reported data on the effect of management practices on soil respiration for *Bromus erectus*-dominated grasslands, despite covering large areas, being of the greatest conservational importance [10,16], and being largely used in large scale grazing systems [33].

In line with this perspective, the present study was designed to investigate the effects of different management intensities, in terms of mowing regimes applied for three consecutive years, on soil respiration, soil temperature, and soil moisture (i.e., dynamics and seasonal mean values) for a permanent mountain *Bromus erectus*-dominated grassland of the central Apennines.

2. Materials and Methods

2.1. Study area

The study area is located in the territory of Monte San Vicino (central Apennines, Marche Region, Italy) and within the Natura 2000 SPA IT5330025—“Monte San Vicino e Monte Canfaito”, which includes two SAC (IT5330015—“Monte San Vicino”; IT5320012—“Valle Vite-Valle dell’Acquarella”) and two main grassland habitats (i.e., habitat code 6210 and 6170; EU Habitats Directive 92/43/EEC). The bedrock of the study area is calcareous, and the climate is temperate-oceanic, which is characterized by an annual mean temperature of 13 °C and a mean precipitation of 865 mm, with higher values in autumn and spring, and the minima in summer [34]. Figure 1 shows the monthly mean air temperatures and precipitation during the study period (January 2016–December 2018), when the precipitation that occurred from June to September 2016 (374.2 mm) was almost double that of the same period in 2017 (191.8 mm) and 2018 (197.8 mm).

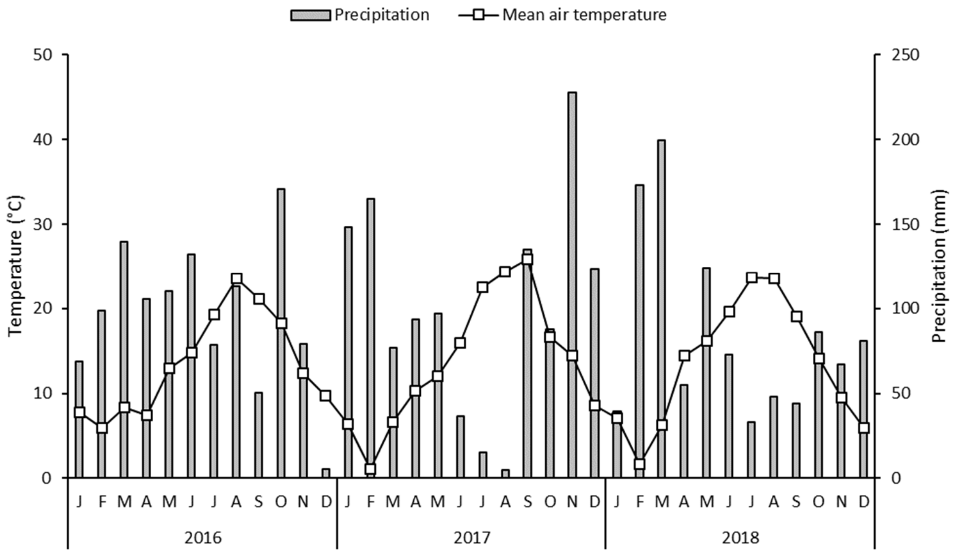

Figure 1. Monthly mean air temperatures and precipitation in the study area during the experimental period from 2016 to 2018 (source: Regione Marche, Servizio Protezione Civile).

2.2. Study site and experimental design

In November 2015, an area homogenous for soil, vegetation and topographic conditions (43°21′13.4″N 13°03′31.6″E; 900 m a.s.l.; NE exposure; 10% slope) was identified and fenced off to prevent any disturbance. The site was characterized by a semi-natural permanent grassland that was dominated by *Bromus erectus* and was considered as a priority habitat (i.e., 6210(*) Semi-natural dry grasslands and scrubland facies on calcareous substrates (*Festuco-Brometalia*) (*important orchid sites), Council Directive 92/43/EEC). It belonged to the association *Brizo mediae-Brometum erecti* Bruno in Bruno and Covarelli 1968 corr. Biondi and Ballelli 1982. The soil was classified as Mollisol, according to the United States Department of Agriculture Soil Taxonomy system [35]. The soil texture from 0 cm to 20 cm in depth was clay loam, with a rocky component of 7%, a pH of 6.79, a total C content of 5.92%, and a total N content of 5.25%.

A complete randomized block design with three replicates was applied to test the responses of the grassland to different use regimes from 2016 to 2018, as: (i) customary management (CST), with herbage mowing performed by the end of June and by the end of October each year; (ii) intensive use (INT), with herbage mowing performed every month from April until the end of October each year, when the herbage production was available; (iii) under abandonment (ABN), with no herbage mowing performed throughout the study period. Each experimental unit was 2.0 m × 2.0 m. The herbage mowing within the experimental units was carried out every year using a bar mower (cutting height, 5 cm) and a standard rake was used to collect and remove the cut herbage immediately after the mowing. For each experimental unit, the soil respiration, temperature, and water content were measured from March 2016 to October 2018, for a total of 52 surveys.

2.3. Measurement descriptions

For each experimental unit, one polyvinyl chloride collar was installed (inner diameter, 10 cm; height, 10 cm; with perforated walls), which was inserted ~9 cm into the soil to measure total soil respiration. The measurements were performed in situ using a portable CO_2 infrared gas analyzer with a soil respiration closed chamber (EGM-4 with SRC-1, PP-Systems, Hitchin, UK), equipped with a thermometer probe. For each measurement, soil temperature was measured at 10 cm soil depth. Soil water content was determined on soil samples collected from the top 10 cm layer, using oven drying at 105 °C to constant weight. Soil respiration, soil temperature, and soil water content were monitored between 9:00 am and 12:00 noon (standard time), to avoid efflux fluctuations [36].

2.4. Data analysis

Two-way mixed analysis of variance (General linear model procedure for repeated measure) was carried out to assess the effect of time, mowing frequency as well as their interactions on seasonal mean soil respiration, temperature, and water content. When a significative interaction emerged, a Tukey test was carried out on mowing frequency to detect differences in each date of measurements. Conversely (i.e., with no significant interaction between mowing and time) only the main effect of mowing frequencies was compared with Tukey honest significant difference (HSD) tests. Data had been previously tested for normality distribution by Shapiro-Wilk's test and sphericity by Mauchly's test.

Seasonal mean soil respiration, temperature, and water content were calculated by linear interpolation between close dates of measurement, assuming a linear flux change between sampling days [37].

Regression analysis was used to determine the relationships between soil respiration and soil temperature and water content [38]. Breakpoints within the functional relationship between respiration and soil water content were detected with a piecewise regression approach. The following equations were used:

For soil respiration and soil water content:

$$y = a + bx, \tag{1}$$

where y is the measured soil respiration (g CO_2 m^{-2} h^{-1}), a and b are the equation coefficients, and x is the measured soil water content.

For soil respiration and soil temperature:

$$y = a\, e^{bx}, \tag{2}$$

where y is the measured soil respiration (g CO2 m−2 h−1), a and b are the equation coefficients, and x is the measured soil temperature.

3. Results

3.1. Soil Water Content and Temperature Dynamics as Affected by Mowing Intensity

Both the soil water content and temperature varied markedly throughout the study period, with similar trends across all three of the management treatments, customary (CST), intensive (INT), and abandonment (ABN). In general, the peaks in the temperature coincided with the minimum water content in summer, while the minimum soil temperatures were recorded in winter along with the highest soil moisture (Figure 2a,b).

The soil water content at 10 cm in depth ranged from 11.12% to 35.45% for CST, from 10.13% to 36.18% for INT, and from 12.15% to 35.83% for ABN. During the 3 year study period, the mean soil water content at 10 cm depth was 24.89%, 24.24%, and 25.07% for CST, INT, and ABN, respectively.

The soil temperatures at 10 cm depth ranged from 0.63 °C to 26.72 °C for CST, from 0.98 °C to 26.90 °C for INT, and from 2.10 °C to 24.53 °C for ABN. During the 3 year study period, the mean soil temperatures at 10 cm depth were 15.39 °C, 15.71 °C, and 14.58 °C for CST, INT, and ABN, respectively.

Both soil water content and temperature changed over the growing or vegetative periods of the monitored years, however, within the dates of measurements, mowing did not affect soil water content, but occasionally altered soil temperature (Table 1).

Seasonal mean soil water content was never affected by the combination of time and mowing regimes. Conversely, the time was found to significantly affect the seasonal mean soil water content of each mowing regime with the exception of the 2017 stasis (p = 0.06). Mowing frequencies did not have significant effects in any of the monitored growing or vegetative periods (Table 1).

For seasonal mean soil temperature, a significant interaction between time and mowing regime was observed in all of the growing and vegetative stasis periods throughout the monitoring period (Table 1). Higher soil temperatures were occasionally observed in INT compared to ABN after the mowing performed in spring for INT in each year of the monitoring period. Conversely, during the vegetative stasis, occasional significantly higher soil temperatures were observed in ABN compared to the other treatments (Figure 2b).

3.2. Soil Respiration Dynamics as Affected by Mowing Intensity

The soil respiration varied markedly among and within the years of the study. This generally following the changes in the soil temperature over most of the year, except in summer, when the soil respiration followed changes in the soil water content (Figure 2c). Multiple peaks of soil respiration were observed for all three of the treatments throughout the 3 year monitoring period. In particular, in 2016, all of the treatments showed fluctuating dynamics that were characterized by three peaks for the soil respiration between March and October, with a less marked trend for ABN compared to CST and INT. In 2017, the first peak of the soil respiration was in the third week of May for all of the treatments (0.78, 0.72, 0.69 g CO_2 m^{-2} h^{-1} for CST, INT, ABN, respectively). This was followed by a marked drop in the soil respiration and a subsequent second peak (0.78, 0.50, 0.62 g CO_2 m^{-2} h^{-1} for CST, INT, ABN, respectively). In 2018, the first peak of the soil respiration occurred at the end of May for all of the treatments (0.67, 0.65, 0.78 g CO_2 m^{-2} h^{-1} for CST, INT, ABN, respectively), with a less pronounced subsequent drop in the soil respiration compared to 2016 and 2017. The second peak of the soil respiration in 2018 occurred at the end of July for CST (0.60 g CO_2 m^{-2} h^{-1}) and in the third week of September for ABN (0.63 g CO_2 m^{-2} h^{-1}), while there was no second peak here for INT (Figure 2c). During the whole study period, the soil respiration ranged from 0.02 to 0.87 g CO_2 m^{-2} h^{-1} for CST, from 0.03 to 0.94 g CO_2 m^{-2} h^{-1} for INT, and from 0.05 to 0.94 g CO_2 m^{-2} h^{-1} for ABN (Figure 2c). During the 3 year study period, the mean soil respiration was 0.36 g CO_2 m^{-2} h^{-1}, 0.33 g CO_2 m^{-2} h^{-1}, and 0.41 g CO_2 m^{-2} h^{-1} for CST, INT, and ABN, respectively.

Soil respiration changed over time and, within the dates of measurements, it was occasionally affected by the mowing regime (Table 1). In general, seasonal mean soil respiration was found to be much higher during the growing period compared to vegetational stasis for all the treatments (Table 1). A significant interaction between time and mowing regimes emerged in all the growing seasons while seasonal mean soil respiration was found to be lower in INT compared to ABN only during the 2016 growing season.

Within the 2016 growing period, the first cut performed in INT had no effect on the soil respiration rates, which were 0.83, 0.85, and 0.86 g CO_2 m^{-2} h^{-1} in CST, INT, and ABN, respectively (Figure 2c). Conversely, approximately 2 weeks after the second mowing performed for INT, a significant decrease in soil respiration was observed in INT (0.59 g CO_2 m^{-2} h^{-1}) compared to CST (0.79 g CO_2 m^{-2} h^{-1}) and ABN (0.81 g CO_2 m^{-2} h^{-1}). After the first mowing for CST (i.e., the third mowing for INT, Figure 2c) both CST and INT showed significantly lower soil CO_2 emission compared to the other treatment (0.39, 0.32, and 0.67 g CO_2 m^{-2} h^{-1} in CST, INT, and ABN, respectively).

Within the 2017 growing season, a significant decrease in soil respiration was observed only after the second mowing performed in INT when CST was higher (0.78 g CO_2 m^{-2} h^{-1}) than INT (0.50 g CO_2 m^{-2} h^{-1}). Despite showing always lower CO_2 emissions, the mowing performed in CST and/or INT did not result in significant emissions until the end of the 2017 growing season.

During the 2018 growing season, the first two mowings performed in INT did not result in any differences in terms of soil respiration among the treatments. However, approximately one month after the first mowing performed for CST (i.e., the third mowing for INT, Figure 2c), the soil CO_2 emission was higher for CST (0.19 g CO_2 m^{-2} h^{-1}) compared to INT (0.07 g CO_2 m^{-2} h^{-1}).

In general, the soil respiration for CST decreased after each mowing, with a sharper decrease after the first compared to the second. Similarly, the soil respiration for INT decreased after each mowing, except for those performed in the first week of June 2016 and in the second week of May 2018. In general, the soil respiration decreased more sharply for the mowing events performed for CST, compared to INT. For ABN, the soil respiration was generally higher than for CST and INT, especially in the periods after the mowing events, with more marked differences during 2016, compared to 2017 and 2018 (Figure 2c).

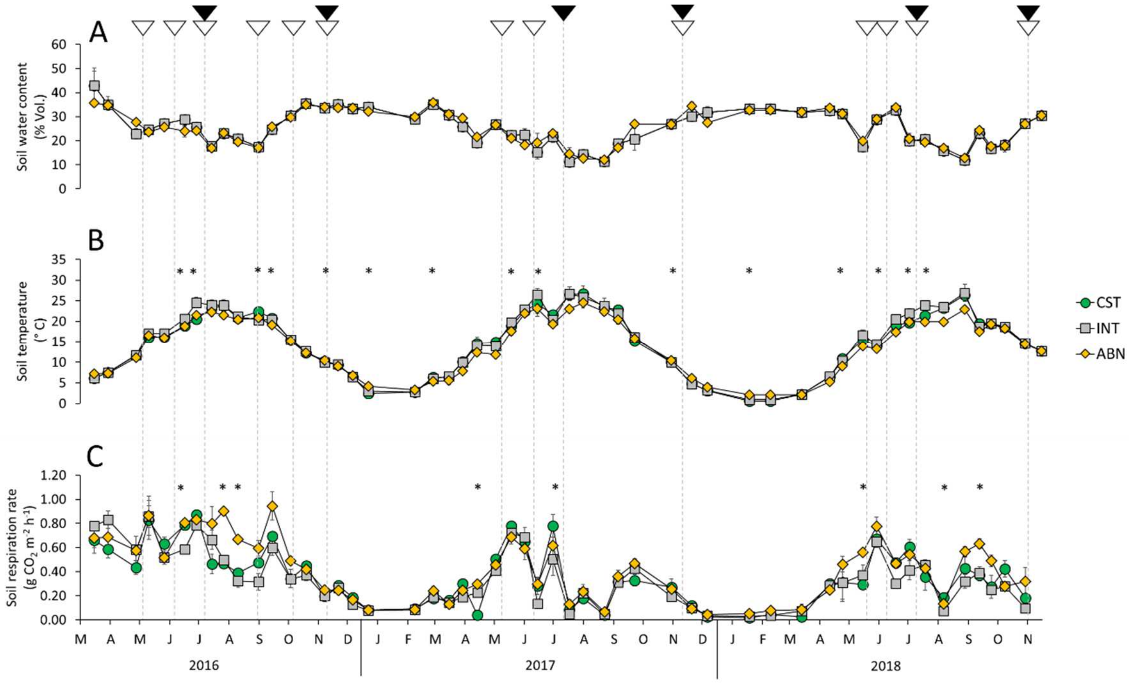

Figure 2. Seasonal variations of the soil water content (A, 10 cm depth), soil temperature (B, 10 cm depth), and soil respiration (C) during the study period from April 2016 to November 2018. Black triangles, dates of the mowing applied for customary management (CST); white triangles, dates of the mowing applied for intensive use (INT); no mowing was applied for abandonment (ABN). Vertical bars, standard errors. Data are means of three replicates per treatment. *, $p < 0.05$.

Table 1. Repeated measure ANOVA for soil water content, temperature, and respiration during the growing periods (G) and the vegetative stasis (S) for customary management (CST), intensive use (INT), and abandonment (ABN). Different letters indicate significant differences (Tukey HSD test, $p < 0.05$).

Variable	Source of Variation	*p* Values					Mowing Regime	Seasonal Mean ± Standard Error				
		2016		2017		2018		2016		2017		2018
		G	S	G	S	G		G	S	G	S	G
Soil water content (% Vol.)	Time	0.01	0.01	0.01	0.06	0.01	CST	26.91 ± 0.49	32.11 ± 0.22	19.28 ± 0.88	31.92 ± 0.44	22.62 ± 0.63
	Mowing	0.43	0.72	0.55	0.34	0.20	INT	26.13 ± 0.81	30.89 ± 0.70	19.29 ± 0.29	30.95 ± 0.41	22.21 ± 0.20
	Time × Mowing	0.49	0.30	0.29	0.08	0.95	ABN	25.94 ± 0.51	32.47 ± 0.50	19.98 ± 0.41	32.05 ± 0.61	23.16 ± 0.59
Soil temperature (°C)	Time	0.01	0.01	0.01	0.01	0.01	CST	16.85 ± 0.52	6.77 ± 0.31	20.10 ± 0.85 ab	3.45 ± 0.21	18.07 ± 0.60
	Mowing	0.13	0.73	0.01	0.16	0.10	INT	17.34 ± 0.53	6.76 ± 0.50	20.18 ± 0.95 a	3.54 ± 0.19	18.64 ± 0.83
	Time × Mowing	0.01	0.01	0.01	0.01	0.02	ABN	16.44 ± 0.28	6.58 ± 0.27	18.57 ± 0.73 b	3.92 ± 0.05	16.80 ± 0.25
Soil respiration (g CO_2 m^{-2} h^{-1})	Time	0.01	0.01	0.01	0.01	0.01	CST	0.56 ± 0.01 ab	0.19 ± 0.02	0.35 ± 0.02	0.05 ± 0.01	0.37 ± 0.05
	Mowing	0.04	0.40	0.28	0.23	0.23	INT	0.54 ± 0.01 b	0.16 ± 0.01	0.33 ± 0.02	0.06 ± 0.01	0.32 ± 0.04
	Time × Mowing	0.01	0.08	0.01	0.82	0.01	ABN	0.67 ± 0.05 a	0.18 ± 0.01	0.37 ± 0.03	0.07 ± 0.01	0.45 ± 0.01

3.3. Relationships Between Soil Respiration, Soil Water Content, and Temperature

The regression analysis showed that soil moisture was the main driver of soil respiration below a soil water content threshold that varied little among treatments (18.23%–22.71%). Above the thresholds, soil temperature was the main driver of soil respiration (Figure 3).

A positive correlation between soil water content and soil respiration was observed when the soil water content was below 22.51%, 22.71%, and 18.23% in CST, INT, and ABN, respectively. In that case, the linear model (Equation (1)) explained 42%, 44%, and 46% of the seasonal variation in the soil respiration for CST, INT, and ABN, respectively ($p < 0.05$). Above such threshold, a negative and weaker relationship was observed for soil respiration and soil water content only in CST and ABN, explaining 21% and 26% of the soil respiration seasonal variation, respectively. A not significant relationship was found in INT.

In contrast, a positive and significant relationship was observed with soil temperature where the exponential model (Equation (2)) explained 66%, 63%, and 77% of the seasonal variation in the soil respiration for CST, INT, and ABN, respectively ($p < 0.05$).

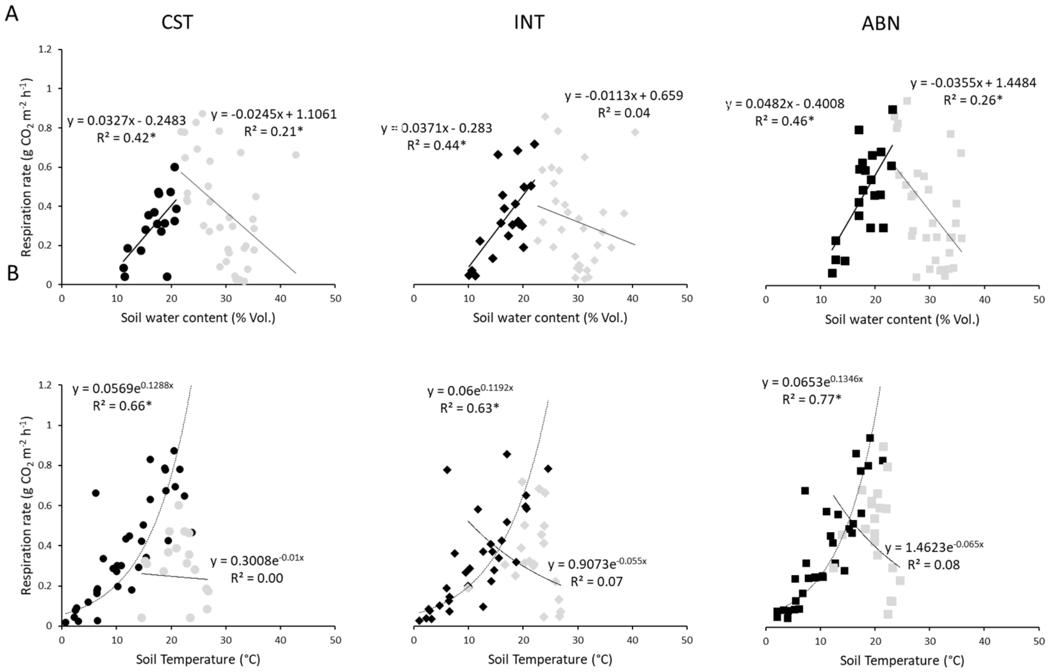

Figure 3. Relationships between the variations in the soil respiration and the soil water content at 10 cm depth (**A**) and the soil temperature at 10 cm depth (**B**) for customary management (CST), intensive use (INT), and abandonment (ABN). Black symbols refer to data below the threshold for soil water content modeling and above the threshold for soil temperature modeling. Gray symbols refer to data above the threshold for soil water content modeling and below the threshold for soil temperature modeling. Data are means of three replicates per treatment. * $p < 0.05$.

4. Discussion

4.1. Relationships Between Soil Respiration, Water Content and Temperature

A role for grasslands in climate change mitigation is widely recognized, and the numerous studies that have been carried out recently testify their importance at the global level [5,24,26,32,39-41]. Future climate scenarios in southern Europe suggest that changes in annual temperatures, precipitation patterns, and atmospheric CO_2 concentrations might negatively alter grassland biodiversity and the associated ecosystem services, including regulating services linked to carbon cycling, especially in areas with low summer rainfall [5,41]. It is well known that the soil respiration of grasslands is largely driven by the soil temperature and the soil moisture [13,26] although the data tend to be context-dependent and related to human disturbances, such as management practices (e.g., grazing, fertilization) and changes in land use [6].

In the present study, the soil temperature was the main driver of the soil respiration only when the soil water content was not a limiting factor. Indeed, the soil respiration dynamics generally followed the soil temperature trends from autumn to the end of spring, while in summer, its pattern was bound to the soil water content dynamics (Figure 2). This was confirmed by the regression analysis that highlighted the relationships between the soil respiration and the soil water content under certain thresholds (Figure 3). Similar data have been reported in other studies carried out in areas with dry summer seasons or where the main limitation is water shortage (e.g., [12,41]). In line with these studies, when water was a limiting factor here, the soil respiration increased sharply, together with the increase in the soil water content due to the rainfall events (Figure 2). Such soil respiration 'pulses' are typical of Mediterranean regions, and they are associated with rainfall events during the prolonged dry season and are controlled by both biotic (e.g., fine roots, mycorrhizal, and microbial activities) and abiotic (e.g., CO_2 released by soil carbonates) factors [12].

4.2. **The Effects of Management Practices on Soil Water Content and Temperature**

The soil water content was always influenced by time except for the 2017 stasis, but it was never influenced either by mowing or by the mowing × time interaction. In agreement with the present study, mowing did not affect the soil water content of tall grass prairies [25] or semi-arid grasslands [24,26]. In contrast, herbage mowing significantly altered soil water content in alpine meadows only when soil temperature was increased with infrared heaters [27]. As suggested by [26], the lack of a significant effect of herbage mowing on this study may be attributed to an offset of enhanced evaporation by decreased transpiration. Indeed, for both CST and INT the mowing caused a drastic reduction of the photosynthetic tissues that due to the aridity conditions in summer (i.e., high temperature and low rainfall) did not return to grow, except belatedly in autumn. At the same time, the low transpiration for ABN may be related to a low photosynthetic activity due to sward aging.

The mowing regime had a significant effect on soil temperatures measured at 10 cm. As observed by [24], herbage removal by mowing exposes soil to higher incident solar radiation and this might result in a stimulation of both microbial and plant root activities. At the same time, the shading effect of ABN is expected to result in lower soil temperature [25]. Indeed, in the present study a higher mean soil temperature was found in INT compared with ABN in the 2017 growing season (Table 1). Despite that, during the vegetative stasis, when significant differences emerged for soil respiration, ABN was found to be higher compared to CST and INT (Figure 2c). This may be attributed to a "shelter" effect of ABN which still had a relevant sward biomass.

Despite the mowing frequency affecting seasonal mean soil temperature in the 2017 growing season, this did not result in differences in terms of seasonal mean soil respiration rates within the same period.

4.3. **The Effects of Management Practices on Soil Respiration**

The results of the effects of use intensity on the soil respiration for grasslands tend to be highly context-dependent, and sometimes contrasting because many other biotic and abiotic factors can affect soil respiration (e.g., grazing, fertilization, substrate supply) and contribute to the soil CO_2 release [6]. For example, as reported by [26] for semi-dry grasslands in inner Mongolia, the soil respiration was not affected by mowing and, contrary to expectation this did not alter the soil water content nor the supply of C substrate to the soil microorganisms. On the other hand, as reported by [24] for similar semi-dry grasslands, mowing significantly decreased the soil respiration, because root production was significantly reduced after the mowing.

In the present case study, the effects of mowing on the soil respiration generally emerged 1 week or 2 weeks after its application, with significant effects only in the 2016 growing season. The effects of mowing led to soil respiration suppression that is explainable as a reduction in the photosynthetic assimilates supply from the above-ground plant parts to the roots [22], and in the labile C substrate for the rhizosphere microorganisms [25].

In the present case study, this mowing effect on the soil respiration was more visible in 2016, compared to 2017 and 2018, because of the limited amount of rainfall that occurred in these latter 2 years that might suppress both the microbial activities [42] and the root respiration [40]. Some studies have reported that mowing has an indirect effect on the soil respiration due to herbage removal and exposure of the soil to higher incident solar radiation, which might stimulate the soil microbial activity due to the soil temperature increase [24]. However, this was not the case for the present study as the soil temperature was not influenced by mowing frequency in the 2016 growing season.

These limited effects of mowing on the soil respiration of this *Bromus erectus*-dominated grassland regardless of use intensity suggest that future studies should also consider the effects on microbial respiration [6] and the potential synergies and trade-offs between the other ecosystem services [16,39].

5. Conclusions

The different mowing frequency had no effect on soil moisture over the 3 year monitoring period. Conversely, it occasionally changed the soil temperature both in the growing season and vegetative stasis. At any rate, such changes in soil temperature did not have any impact on seasonal mean soil respiration among the treatments, except for the 2016 growing season when the intensive use showed lower soil seasonal mean respiration rates compared to the abandonment. Such differences in soil CO_2 emissions were imputable only to the effect of mowing and not to the alteration of soil water content or temperature.

Within the same mowing frequency, the soil temperature was the main driver of the soil respiration only when the soil water content was above a threshold. Below this threshold, soil respiration was mainly controlled by the soil water content, as highlighted by soil respiration pulses that occurred after rainfall events during prolonged dry seasons.

These effects on soil respiration, that emerged only in the first growing season with high rainfall, suggest that in this study site, a more intensive use would not have significative impacts on soil respiration of *Bromus erectus*-dominated grasslands. The integrated analysis of multiple case studies, also using modeling applications, would contribute to confirming the dynamics observed for the whole *Bromus erectus*-dominated grassland ecosystem, regardless of the study site.

Future studies should include aspects such as the contributions of the root, mycorrhizal, and microbial respiration in the light of climate change, especially considering the project scenarios of the seasonal redistribution of precipitation patterns, with a focus on the effects of rainfall on prolonged dry periods.

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Chapter III: Nitrous oxide emissions as affected by perennial crop termination and biochar application in alfalfa-wheat rotation under Mediterranean environment

Abstract

Agricultural activities are potential sources of greenhouse gases (GHG) emissions and nitrous oxide (N_2O) is one of the most important non-carbon dioxide (CO_2) GHG. Perennial legumes, such as alfalfa (*Medicago sativa* L.), have a potential role for cropping systems to mitigate soil GHG emissions. With the aim to identify innovative practices to reduce N_2O emissions in an alfalfa-wheat system under Mediterranean environment, the study tested (i) alfalfa termination performed by spading and postponed in autumn, contrary to the traditional tillage system that requires deep tillage (i.e., ploughing) performed in summer with high soil temperature and N_2O emissions; (ii) the incorporation of biochar into the soil to reduce soil N_2O emissions, whose effectiveness is still under discussion. To test these hypotheses the following treatments were compared in terms of N_2O emissions: (i) six-year-old alfalfa (A); (ii) winter wheat after six-year-old alfalfa termination in autumn (W); (iii) winter wheat after six-year-old alfalfa termination in autumn amended with biochar (60 t ha^{-1}) (WB). In W and WB, wheat yield and quality were also analysed. The study was conducted at plot level (2.5 m x 13.0 m) by adopting a complete randomized block experimental design with three replicates, in a temperate oceanic sub-Mediterranean area (central Italy). The cumulative emissions were 0.72, 0.84 and 0.77 kg N-N_2O ha^{-1} for A, W and WB, respectively. Compared to summer, autumn tillage led to lower N_2O emissions due to the unfavourable conditions for denitrification processes. In wheat treatments higher N_2O emissions were recorded only in three dates soon after tillage probably due to the asynchrony between N released into the soil by alfalfa residues mineralization and wheat N-uptake. Despite these initial emissions no significant differences emerged in terms of cumulative N_2O emissions among the treatments. Moreover, at this rate of application, biochar amendment did not show N_2O emissions mitigation effects nor wheat production enhancement. The absence of biochar mitigation effects can be related to the specific characteristics of the soil, therefore further study on soil microbial activities could explain some of the biochar mitigation mechanisms. Moreover, the biochar mitigation potential could emerge in further years of experimentation due to the aging process.

Keywords: Greenhouse gas, Legume, Autumn tillage

1. Introduction

Many recent studies agree that the increase of greenhouse gases (GHG) emissions to the atmosphere is linked to human activities (Cayuela et al., 2017; Reay et al., 2012; Smith et al., 2014; Stehfest and Bouwman, 2006; Wang and Fang, 2009). Among the other GHG, N_2O is one of the most relevant non-CO_2 GHG (Forster and Ramaswamy, 2007) with a Global Warming Potential 265 times higher than CO_2 over 100 years' time horizon (Smith et al., 2014).

The soil is the largest natural source of N_2O (Stehfest and Bouwman, 2006; van Groenigen et al., 2010) and agriculture is responsible for around 60% of N_2O emissions, representing the largest anthropogenic source (Syakila and Kroeze, 2011) which originates through the alteration of the global nitrogen cycle (Reay et al., 2012).

Many factors play key roles on the nitrogen cycle and consequently on N_2O emissions from agriculture. The main factors that contribute to N_2O emissions from agriculture are the fertilizer type and application rates, crop type, climate conditions, soil physio-chemical properties including organic carbon content, pH and texture (Stehfest and Bouwman, 2006). The N_2O emissions are influenced by soil type, generally increasing in soil with higher clay content (Lesschen et al., 2011), compared with sandy soil due to the increase of anaerobic microsites (Signor and Cerri, 2013). Also soil pH can influence N_2O emissions (Šimek and Cooper, 2002); for example Van den Heuvel et al. (2011) in a forested riparian ecosystem in the Netherland, saw an increase in N_2O emissions in soil with lower pH values while Shaaban et al. (2018) through an incubation experiment with soil sampled in arable field of China found that the addition of dolomite in acidic soil enhanced N_2O reductase enzyme promoting the reduction of N_2O to N_2. The type of crop residues and the C:N ratio (low in alfalfa crop residues) can affect the N_2O emissions (Gomes et al., 2009); in a plot experiment carried out in Gray Lowland soil in Hokkaido (Japan), Toma and Hatano (2007) found that application of residues with a low C:N ratio lead to high N_2O emissions due to an easily mineralization of residues and a consequent greater probability of nitrification and denitrification processes (Huang et al., 2004). Also, the fertilizer type and application rate can affect the N_2O emissions as reported by Kim et al. (2013) in their meta-analysis, where many studies indicate an increase of N_2O emissions at higher N input rate than those required by crop. The adoption of best agronomic practices that take in consideration all these N_2O emissions' affecting factors and the use of some mitigation strategies such as the use of biochar can lead to reduce soil N_2O emissions.

The effects of N-fertilization on increasing N_2O emissions is reported by many authors (Malhi et al., 2010; Sanz-Cobena et al., 2017; Tenuta et al., 2019; Volpi et al., 2018). N-fertilization to soil results in high N_2O emissions since this gas is one of the by-products of the microbial process nitrification and denitrification. Legume crops are widely used as an alternative to chemical fertilization for their N-fixation ability and consequent potential role for perennial systems to mitigate soil N_2O emissions (Abalos et al., 2016) reducing direct soil N_2O emissions and indirect N_2O emissions arising for example from their production and transport (Aguilera et al., 2013). Alfalfa (*Medicago sativa* L.) is one of the most important forage crop all over the world (Tesfaye et al., 2006) and it is one of the most used perennial legume in organic farming system that is having an increase in terms of surface extension in all areas of the world (Willer and Lernoud, 2017). However, very few studies investigated the effects of perennial legume termination on soil N_2O emissions (e.g., Tenuta et al., 2019; Westphal et al., 2018), considering also the effects of crop residues (Jensen et al., 2012) able to increase these emissions (Autret et al., 2019; Basche et al., 2014).

The incorporation into the soil of biochar, that is a carbon-rich product obtained by pyrolysis in low-oxygen environment (Lehmann and Joseph, 2015), is a promising agronomic practice able to mitigate GHG emissions and increase the C-storage (Smith et al., 2014).

Biochar can affect N_2O soil emissions changing soil biological, physical and chemical properties (Lehmann and Joseph, 2015). For example, Cayuela et al. (2013) found a relation among biochar and the promotion of the reduction of N_2O to N_2 through the increase of soil pH around biochar particles (Borchard et al., 2019). Although biochar can mitigate N_2O soil emissions some authors found that its application does not reduce or even increase N_2O emissions. For example, Koga et al. (2017) found that biochar application at four different rates (0, 10, 20, and 40 Mg ha^{-1}) had no effects on N_2O emissions reduction in a Andosol field in northern Japan, while Verhoeven and Six (2014) observed an increase of N_2O soil emissions after the application of 10 Mg ha^{-1} of biochar in a Dierssen sandy clay loam soil of California. This suggests that biochar effectiveness on GHG mitigation is closely related to the context characteristics (e.g., soil properties, pH, biochar application rate, management practices).

Besides GHG mitigation, biochar may have beneficial effects on crop yield enhancement sebbene il suo effetto richiede a deeper understanding of the processes to reduce the risk of misinterpretations (Lehmann and Joseph, 2015). Indeed, as reported by Liu et al. (2013) in their meta-analysis, the increase of crop productivity is closely related to the type of soil, crop and biochar. Biochar addition were found to increase wheat yield in a Mediterranean climate with biochar rate of 30 and 60 t ha^{-1} (e.g., Vaccari et al., 2011), while the wheat yield increase found by Wang et al. (2012) after biochar application (0, 10, 25, 50 t ha^{-1}) were dependent on N fertilization (urea at 0 and 200 kg N ha^{-1}). On the contrary, Koga et al. (2017), that applied wood residue-derived biochar in a plot experiment, found no effects on wheat crop yield under none of the four biochar doses applied (e.g., from 0 to 40 Mg ha^{-1}). Also Martos et al. (2019) found no effects on barley crop yield at three biochar addition rates (0, 5, and 30 t ha^{-1}), in a Mediterranean environment.

The traditional tillage system usually accomplished in clay soils of the study area requires deep tillage (i.e., ploughing) performed in summer (i.e., July or August) even out of the optimal humidity, followed by subsequent soil refining (e.g., grubbing and harrowing) performed before the sowing. Under these conditions (i.e., dry soil and high temperature), soil moisture and temperature being drivers of N_2O emissions (Butterbach-Bahl et al., 2013; Rakotovololona et al., 2019), alfalfa termination performed in summer could result in high N_2O emissions (Krauss et al., 2017). For this reason and with the scope to identify N_2O emission mitigation practices within the traditional rotation system, in this study alfalfa termination is postponed to autumn by spading when the soil conditions are more cold and dry, as suggested by Krauss et al. (2017). Spading, that is a shallow tillage, is used instead of ploughing to preserve soil quality and soil organic matter pool (Laudicina et al., 2017).

Taking into account the paucity of information on N_2O losses after legume perennial crop termination (Jensen et al., 2012), this study investigate the impact of (i) alfalfa termination on the N_2O emissions, (ii) biochar application on soil N_2O emissions, and (iii) biochar application on crop yield, in an alfalfa-wheat system under Mediterranean climate conditions with main tillage postponed in autumn period.

2. Methods

2.1. Site description

The site (43° 33' N, 13° 25' E, 100 m a.s.l., SW exposure, 23% slope) is located in central Italy, in the experimental farm of the Polytechnic University of Marche. The bioclimate is temperate oceanic sub-Mediterranean variant (Agnelli et al., 2008), with a mean precipitation of 770 mm and a mean annual temperature of 14.1 °C (Kottek et al., 2006). Monthly precipitation and mean air temperature recorded by a weather station located 0.3 km away from the study site measured a cumulative rainfall over the study period (October 2017-July 2018) of 867 mm and mean air temperature of 12.3 °C as shown in Figure 1.

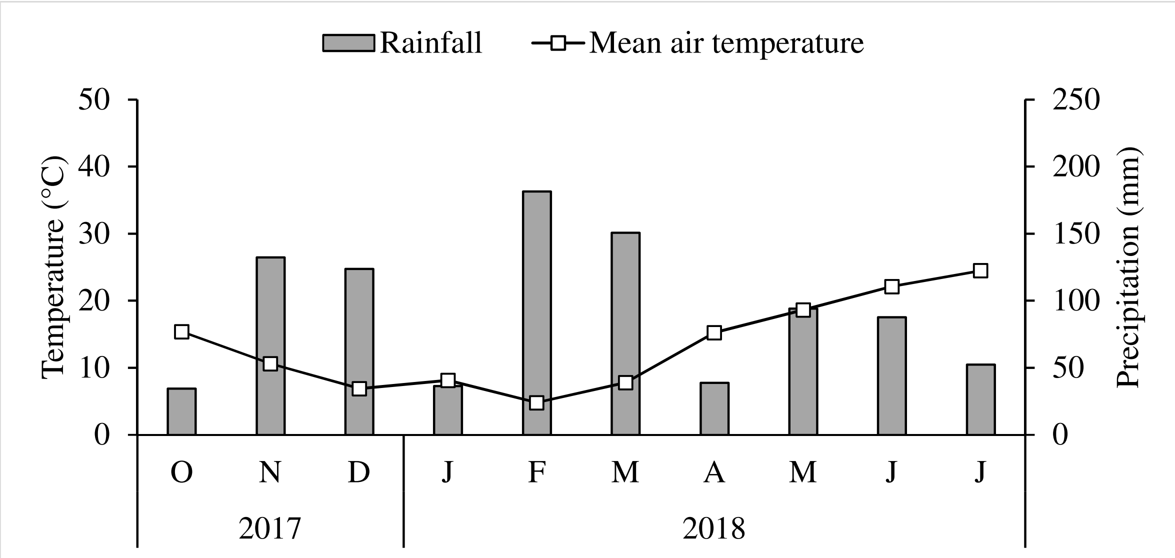

Figure 1. Montly mean air temperature and precipitation in the study area during the whole monitoring period (October 2017 – July 2018).

The soil at the study site was classified as Inceptisol according to USDA Soil Taxonomy system (Soil Survey Staff, 2014). Ancillary measurements were conducted at the beginning of the trial (i.e., October 2017) following the non-systematic "W" pattern described by Paetz and Wilke (2005). Soil samples were collected at a depth of 10 cm and 40 cm (Table 1).

Table 1. Basic properties of the soil at 0-10 cm and 10-40 cm depth: pH, carbon-nitrogen ratio (C:N), soil texture, soil organic matter (SOM), organic carbon (C_{org}), total organic carbon (C_{tot}), total nitrogen (N_{tot}), humic and fulvic acids (HAs-FAs), cation exchange capacity (CEC), field capacity (FC), wilting point (WP).

Depth	pH	C:N	Sand (g/kg)	Silt (g/kg)	Clay (g/kg)	SOM (g/kg)	C_{tot} (g/kg)	C_{org} (g/kg)	N_{tot} (g/kg)	HAs-FAs (g/kg)	CEC %	FP %	WP%
0-10	8.13	8.28	364.80	379.80	255.40	14.30	8.28	8.32	1.00	4.96	22.30	24.49	17.42
10-40	8.12	8.62	353.40	386.20	260.40	15.56	8.76	9.04	1.04	4.90	23.06	24.40	17.81

2.2. Experimental design

This study was conducted from October 2017 to July 2018. In October 2017, an area (20.0 m x 20.0 m) homogeneous for soil, crop vegetation and topographic conditions in a 6-year-old alfalfa field was identified and fenced to prevent any disturbance. A complete randomized block design with three replicates (individual plot size was 2.5 m x 13.0 m) was applied to test the response of the following treatments: (i) six-year-old alfalfa, (ii) durum wheat (*Triticum turgidum* L. ssp. *durum* (Desf.) Husn.) after alfalfa termination and (iii) durum wheat after alfalfa termination amended with a dose of 60 t ha^{-1} biochar, as commonly adopted in other similar field experiments (Castaldi et al., 2011; Rogovska et al., 2014). The biochar applied in WB plots was a commercial charcoal obtained from by-products from river beds (beech, pine and fir) produced at a pyrolysis temperature of 800-900 °C.

The management applied during the study period included the following practices (Table 2): (i) in the alfalfa plots, the herbage mowing was performed at the beginning of the crop flowering (end May and early July) and was carried out by using a bar mower (0.05 m cutting height) and a standard rake was used to collect and remove the cut herbage immediately after the mowing; (ii) in the wheat plots (W and WB), the alfalfa termination was performed at the beginning of October 2017 (i.e., 11th October) by using a spading machine (0.20 m depth) followed by two consecutive (i.e., 13th October and 16th October 2017) rotary harrows (0.15 m depth) and alfalfa residues (from 1.83 to 3.37 Dry Matter (DM) t ha^{-1} with a mean of 2.53 DM t ha^{-1}) were incorporated into the soil; in WB plots, biochar was manually applied before sowing (16th October 2017), and partially buried with a rotary harrow over 0.15 m depth following the method reported by Castaldi et al. (2011). Durum wheat plots were sowed at the end of November 2017 (i.e., 23th November) in rows with a sowing rate of 400 seeds per m^2 and manually harvested at the beginning of July 2018 (i.e., 4th July) in a central plot area of 2 m^2. Manual weeding was performed in W and WB plots twice in the second half of May 2018 (i.e., 17th and 24th May) and the main species were *Convolvulus arvensis* L. and *Papaver rhoeas* L..

Because the effects of N-fertilization on N_2O emissions are well known and demonstrated by many authors (Aguilera et al., 2013; Liu et al., 2015; Wei et al., 2010), N-fertilization was not applied to isolate only the effects of alfalfa termination and biochar application on N_2O emissions.

Table 2. Management practices applied to the different treatments during the study period (Day: Julian days). A: alfalfa, W: wheat, WB: wheat + biochar.

	Management practice		Treatment		
			A	W	WB
Year	Type	Depth (m)	Day	Day	Day
2017	Spading	0.20	-	284	284
	Harrowing 1	0.15	-	286	286
	Biochar application	0.15	-	-	289
	Harrowing 2[§]	0.15	-	289	289
	Sowing	0.03	-	327	327
2018	Harvesting 1	-	131	-	-
	Weeding 1	-	-	137	137
	Weeding 2	-	-	144	144
	Harvesting 2	-	185	185	185

[§] Second harrowing was performed after biochar application in WB plots.

In each experimental unit, soil N_2O emission, temperature and water content were measured from October 2017 to July 2018, for a total of 32 surveys per each variable.

2.3. N_2O monitoring

Nitrous oxide was measured by using closed static chambers as described by Parkin and Venterea (2010). Chambers were made of PVC (0.15 m high x 0.25 m in diameter) and were equipped with thermometer to measure the variation of internal temperature during the sampling period. Two PVC base rings (0.25 m diameter) per plot (n= 6 chambers per treatment) were permanently installed into the soil (0.1 m depth) and have been removed only for soil tillage and immediately installed afterwards (Ghimire et al., 2017).

Gas samples were collected between 9:00 and 12:00 o'clock (Krauss et al., 2017) at a frequency of three or four days from tillage (11th October 2017) to sowing day (23rd November 2017) and after rainy events, and mainly every about 15 days depending on management occurrence (Volpi et al., 2018). Chambers were placed for 45 minutes during which four gas samples (30 ml each, equidistantly spaced over time: t_0, t_{15}, t_{30}, t_{45}) were withdrawn from the headspace of each chamber. The gas samples were injected into 30 ml glass vials sealed with a butyl rubber septum (previously vacuumed) (Parkin and Venterea, 2010).

2.4. Laboratory analysis

The N_2O concentration were determined by using gas-chromatograph electron capture detector (ECD). According to Koga et al. (2017), fluxes emissions (Equation 1) were calculated starting from the slope (ppm/hr) and cumulative emission were calculated by interpolation between sampling dates (Gelfand et al., 2016).

$$F = \frac{M}{V0}\frac{P}{P0}\frac{(273+T0)}{273+T}\,h\frac{dC}{dt}$$

(1)

T_0, P_0, V_0 are respectively air absolute temperature, atmospheric pressure and molar volume under standard conditions. M is a molecular weight of a gas X, P is the pressure outside the camber, $\frac{dC}{dt}$ is the slope of the curve of gas concentration variation with time, h is the height of chamber from base ring to the top.

2.5. Crop sampling and analysis

For each A plot a square area (1.0 m x 1.0 m) were randomly chosen to measure the aboveground biomass that was cut using electric scissor at 5 cm above the ground. The plant material was harvested, oven-dried at 105 °C for constant weight to determine the DM. Aboveground biomass samples were taken three times during this seventh A growth cycle (i.e., 11th May, 4th July and 6th September 2018).

From each W and WB plots, a central area of 2 m^2 was selected and all plants were manually harvested on 4th July 2018. Following the method of Monaci et al. (2017) the harvested plants were threshing using a Wintersteiger Delta combine in order to determine the wheat grain production. Protein content and moisture were determined by using an Inframatic 9500 NIR Grain Analyzer while hectolitre weight was determined by using Dickey John GAC 2000 grain analysis meter (Dickey-John, Auburn, IL).

2.6. Statistical analysis

Statistical analysis was performed by using the IBM SPSS Statistics version 25 (IBM, 2017). One-way ANOVA followed by Tukey's HSD post hoc test was used to compare seasonal cumulative N_2O emissions, weekly cumulative N_2O emissions and daily N_2O emissions. Significance for ANOVA was assumed at a limit value of $P<0.05$. Student t-test was performed to compare the wheat grain production and quality for W and WB treatments.

3. Results and Discussion

3.1. Effects of alfalfa termination and biochar application on seasonal cumulative N_2O emissions

Seasonal cumulative N_2O emissions did not differ significantly between treatments for all the period monitored (Figure 2) and were 0.72, 0.84 and 0.77 kg $N\text{-}N_2O$ ha^{-1} for A, W and WB, respectively. The range of cumulative N_2O emissions are consistent with those reported by other studies under similar climatic conditions. For example, Volpi *et al.* (2018) found in a Mediterranean environment on durum wheat seeded after clover harvesting and a minimum tillage (i.e., disk harrow, 0.10 m depth) without N-fertilization, a cumulative N_2O emissions of 0.87 kg $N\text{-}N_2O$ ha^{-1}. While some authors reported high emissions after alfalfa termination this is mainly due to N-fertilization and soil spring-thaw period in glacio-lacustrine clay floodplain (Tenuta et al., 2019; Westphal et al., 2018b), some other authors found that timing (i.e., spring, summer and late summer) and termination (i.e., tillage, herbicide and both) method had no influence on N_2O emissions after 7-year old alfalfa stand (Malhi et al., 2010). Well known are the effects of N-fertilization on N_2O emission; for example Malhi et al. (2010) found in a field experiment conducted in a Gray Luvisol with loam texture, a significant influence on N_2O emission after 7-year-old alfalfa termination due to N-fertilization, but the effects of a perennial legume on the subsequent cropping season in the context of Mediterranean cropping system is still uncertain.

The application of biochar in our Inceptisol fields did not significantly affect N_2O emissions from soil, as showed in Figure 2 comparing W and WB treatments. Some authors (Cayuela et al., 2015; Liu et al., 2019; Wang et al., 2012) found a reduction of N2O soil emission after biochar application due to the biochar application rate and to the biochar and the soil properties (Cayuela et al., 2014). Our results are in line with (Koga et al., 2017) that found no effects on N_2O reduction for none of the four doses (0, 10, 20, and 40 Mg ha^{-1}) of biochar applied on a Typic Hapludand soil. In their Andosol, Koga et al. (2017) did not find significant effects of biochar application on soil pH, this resulted in absence of the promotion of the complete denitrification to N_2 (Borchard et al., 2019) and thus absence of mitigation of N_2O soil emissions. Also Martos et al. (2019) did not find significant changes in soil pH after the application of wood chip biochar on a loamy Typic Calcixerept soil under Mediterranean climate condition with a alkaline soil. According to these authors, the absence of biochar's effect on the reduction of N_2O emissions in our experiment can be partially due to the alkaline pH of the studied soil that promotes *per se* a more complete denitrification to N_2 (Borchard et al., 2019).

In this first year of monitoring, the absence of N_2O mitigation effects after the biochar application could be due to the biochar aging into the soil normally occurring in the long term (Borchard et al., 2019), as reported by Hagemann et al. (2017) who found significant suppression of N_2O emissions in the third year of experimentation. According to Borchard et al. (2019), aging of biochar and the formation of organo-mineral complexes with a consequence retention of nutrient (Mia et al., 2017), in particular on NO_3^- (Joseph et al., 2018) could reduce the N_2O emissions produced by denitrification. The long-term effect of biochar aging on N_2O mitigation potential may emerge in the coming years of experimentation (Borchard et al., 2019).

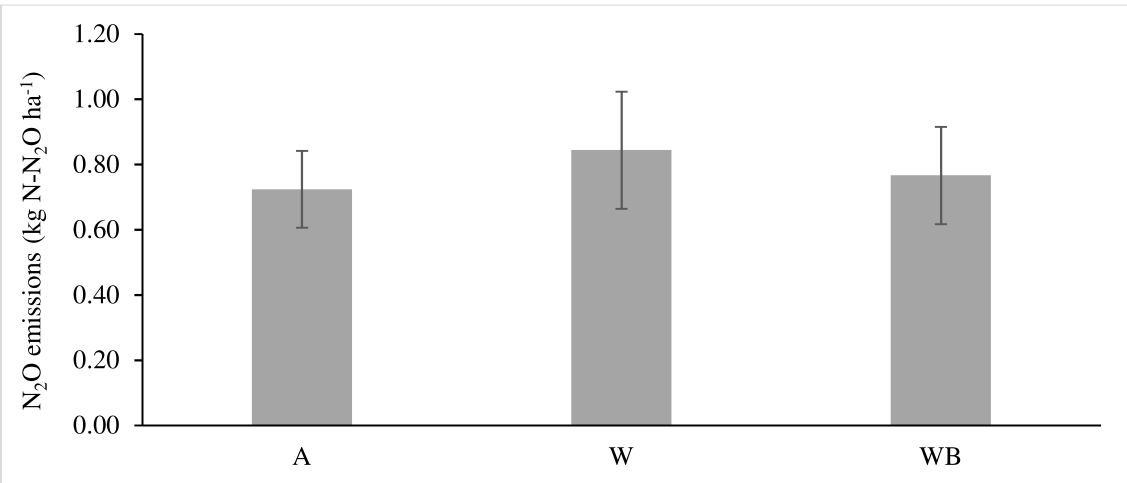

Figure 2. Seasonal cumulative N_2O emissions during the study period (from 18th October 2017 to 4th July 2018). A: alfalfa, W: wheat, WB: wheat + biochar. Vertical bars represent standard errors.

3.2. Effects of alfalfa termination and biochar application on cumulative weekly N_2O emissions

During the study period, the cumulative weekly N_2O emission ranged from 0.07 to 35.17 g N_2O ha^{-1} h^{-1} showing very similar trends in the W and WB that present higher emissions compared to A immediately after alfalfa termination (Figure 3; Table 3).

From the beginning of the study period till the half of February 2018, significant differences between A and wheat treatments (i.e., W and WB) emerged. These results can be related to the legume residues incorporated into the wheat plots and to their low C:N ratio (Basche et al., 2014) that can be easily mineralized (Toma and Hatano, 2007). With this regard, Huang et al. (2004) in an incubation experiment with conditions favourable to nitrification found negative correlation between N_2O emissions and C:N residues ratio. This was imputable to a microbial activity stimulation promoting the oxygen consumption creating temporary anaerobic microsites that enhanced N_2O production via denitrification processes. Similarly, in our experiment, soil tillage which promoted mineralization and nitrification processes, may have caused a temporary anoxic environment through increased microbial activity and respiration, which favoured denitrification processes as reported by Huang et al. (2004). Furthermore, soil tillage may have led to the breakup of soil aggregates (Álvaro-Fuentes et al., 2008), with consequent release of N_2O from their core with the main anoxic condition (Borer et al., 2018) that can promote the denitrification processes. The nitrogen released into the soil after alfalfa termination (11th October) and due to residues mineralization (Basche et al., 2014) was not used soon by wheat that was sowed about a month and a half later (23rd November). Also, during the initial wheat stage, the N uptake was low compared to the next stages (Delogu et al., 1998; Li et al., 2012) and this was probably reflected in high level of N into the soil derived by the legume residues mineralized after the previous tillage.

These initial higher N_2O emissions recorded in W and WB compared to A significantly affected cumulative weekly emissions from 25th October to 14th February 2018.

After 14th February until the first half of May, a visible difference between wheat treatments (i.e., W and WB) compared to A emerged, even if these differences were not significant. In this second phase, W and WB N_2O emission were lower compared to the first wheat growing phase and this could be attributable to the higher N uptake from wheat crop in this phase (Delogu et al., 1998), with a nitrate subtraction to a possible denitrification.

From the second half of May, the increases of the N_2O emissions in the wheat treatments (i.e., W and WB) could partly due to the lower N uptake from wheat after heading and in particular nearing maturity phases (Delogu et al., 1998; Li et al., 2012).

After alfalfa mowing at early May, N_2O emissions from A treatment increased immediately afterwards. This may be due to the removal of the photosynthetic tissues with a consequent change in N metabolism. In particular, herbage cutting could have led to a reduction of the mineral N forms uptake from the soil and nodule function (Erice et al., 2011).

The consequence of this trend is that at the end of the crop cycle the cumulative weekly N_2O emissions did not show significant differences among treatments.

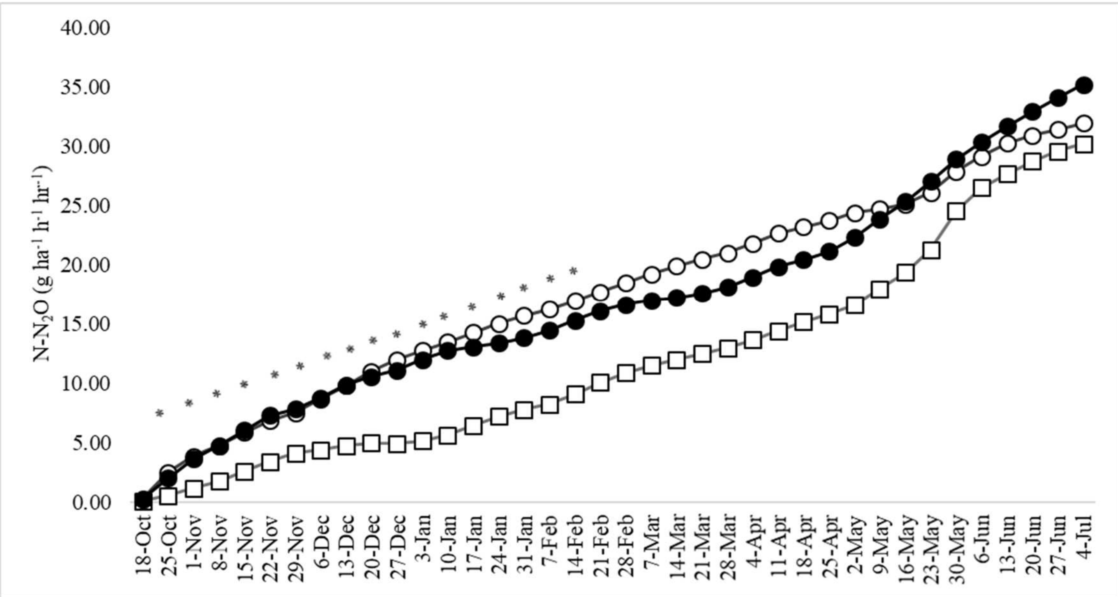

Figure 3. Cumulative weekly N_2O emissions over the study period. A: alfalfa (open squares), W: wheat (open circles), WB: wheat + biochar (closed circles). Significant difference is expressed as "*" for $P < 0.05$.

3.3. Effects of alfalfa termination and biochar application on daily N_2O emissions

The N_2O emissions rate over the monitoring period (Table 3) ranged from -0.02 ± 0.01 to 0.53 ± 0.14 g N-N_2O ha^{-1} h^{-1} for A, from 0.02 ± 0.07 to 0.37 ± 0.11 g N-N_2O ha^{-1} h^{-1} for W and from 0.03 ± 0.10 to 0.39 ± 0.10 g N-N_2O ha^{-1} h^{-1} for WB. Statistical analysis highlighted differences among the treatments ($P<0.05$) in three dates in the period immediately after tillage (Table 3); higher N_2O emissions from wheat plots (i.e., W and WB) compared to A treatments emerged in W on the 27th October, in WB on the 20th October and in both W and WB on the 6th November.

Emissions from October to end of December during seedling phase accounted for 16.49, 32.77 and 38.90% of the total emissions for A, W and WB, respectively. A reduction in N_2O emissions occurred during the wheat tillering/double ridge phase, between January and March, when N_2O emissions were 19.47, 14.47 e 18.85% of the total emissions for A, W and WB, respectively. In this second phase, a reduction in N_2O emissions in W and WB compared to the first period occurred and this was probably due to the higher use of N by wheat. These results are in line with Liu et al. (2015) who tested the effects of different N-fertilization levels on N_2O emissions under wheat crop cycle and found N_2O emissions mainly concentrated in the sowing-greening stages.

From March to the end of April (double ridge - jointing stages) the N_2O emissions were similar to the previous phase and in particular 18.00, 15.05 and 18.03% of the total emissions for A, W and WB, respectively.

The N_2O emission recorded during the last wheat stages, between the beginning of May and early July (harvest time) (booting – maturity), accounted for 46.03, 37.70 and 24.22% of the total emissions for A, W and WB, respectively. In the case of A this increase in N_2O emissions may be due to: (i) the mowing performed at early May that, as described above, may have reduced the N uptake from root system but also, (ii) the rainy events that at the end of May gave rise to three N_2O peaks (22nd, 24th and 29th of May). In the case of W and WB treatments, the increase of N_2O emissions compared to previous stages is probably due to the above mentioned reduction of the N uptake by wheat in the maturity phases (Delogu et al., 1998).

Table 3. N_2O emissions rate (g N-N_2O ha^{-1} h^{-1}) in A: alfalfa, W: wheat, WB wheat + biochar over the study period. SE = standard error. Significant difference (Sign.) is expressed as "*" for P < 0.05; different letters indicate significant differences (P<0.05) between treatments.

Date	A	W	WB	SE-A	SE-W	SE-WB	Sign.
18/10/2017	0.07	0.28	0.28	0.03	0.07	0.03	0.10
20/10/2017	0.00 b	0.12 ab	0.25 a	0.03	0.02	0.05	0.01*
24/10/2017	0.11	0.37	0.39	0.05	0.11	0.10	0.15
27/10/2017	0.12 b	0.29 a	0.23 ab	0.03	0.07	0.03	0.04*
31/10/2017	0.06	0.16	0.15	0.02	0.03	0.02	0.13
03/11/2017	0.09	0.14	0.09	0.02	0.03	0.02	0.31
06/11/2017	0.08 b	0.17 a	0.16 a	0.03	0.04	0.02	0.03*
09/11/2017	0.12	0.16	0.13	0.02	0.06	0.02	0.74
17/11/2017	0.12	0.24	0.21	0.01	0.05	0.08	0.42
22/11/2017	0.10	0.11	0.05	0.07	0.01	0.02	0.66
24/11/2017	0.11	0.06	0.06	0.03	0.03	0.05	0.71
28/11/2017	0.10	0.08	0.12	0.04	0.03	0.07	0.87
01/12/2017	0.04	0.11	0.19	0.04	0.01	0.07	0.21
05/12/2017	0.03	0.16	0.15	0.04	0.04	0.09	0.36
14/12/2017	0.07	0.14	0.17	0.03	0.05	0.14	0.54
21/12/2017	-0.02	0.05	0.16	0.01	0.01	0.06	0.07
03/01/2018	0.05	0.15	0.09	0.03	0.03	0.03	0.23
16/01/2018	0.13	0.02	0.12	0.07	0.07	0.06	0.46
05/02/2018	0.05	0.09	0.07	0.03	0.02	0.04	0.67
14/02/2018	0.16	0.13	0.11	0.02	0.02	0.01	0.48
08/03/2018	0.07	0.03	0.11	0.02	0.03	0.02	0.19
23/03/2018	0.07	0.06	0.07	0.02	0.03	0.01	1.00
06/04/2018	0.11	0.15	0.14	0.03	0.03	0.02	0.48
18/04/2018	0.10	0.06	0.05	0.05	0.05	0.04	0.82
27/04/2018	0.08	0.15	0.11	0.02	0.04	0.05	0.54
09/05/2018	0.24	0.25	0.03	0.05	0.13	0.10	0.43
17/05/2018	0.16	0.16	0.09	0.08	0.09	0.03	0.36
22/05/2018	0.34	0.30	0.14	0.11	0.10	0.01	0.26
24/05/2018	0.53	0.34	0.38	0.14	0.10	0.16	0.23
29/05/2018	0.42	0.22	0.16	0.17	0.10	0.09	0.09
07/06/2018	0.17	0.19	0.19	0.02	0.05	0.12	0.97
20/06/2018	0.15	0.17	0.07	0.14	0.12	0.10	0.81

3.4. Effect of biochar on wheat grain production

On average, wheat grain yield was very low and ranging from 1.39 to 2.34 t ha^{-1}, much lower than the average production recorded in the study area surroundings (4.40 t ha^{-1}) and probably due to the inefficacy manual weeding performed in our experiment to reduce competition with wheat. Although wheat yield in W was higher than in WB by 9.4%, the addition of high doses of biochar did not significantly affect wheat yield (Table 4), neither its quality (*t*-test, P < 0.05).

Addition of biochar as soil amendment can affect crop productivity influencing several mechanisms (i.e. nutrients dynamics, soils pH, reduced N losses by leaching and N_2O

emission) (Lehmann and Joseph, 2015). Many authors (e.g., Biederman and Harpole, 2013; Vaccari et al., 2011) reported significant increase in wheat yield with biochar application. Vaccari et al. (2011) in Mediterranean climate conditions found a positive effect of biochar application on winter wheat productivity at 30 and 60 t ha^{-1} of biochar rate, ascribing this positive effect to an increase in soil pH and a consequent rise in nutrient bioavailability. On the contrary, our results matches the results obtained by Koga et al. (2017) that did not find any significant increase in wheat yield under four rates of biochar application (0, 10, 20, and 40 Mg ha^{-1}). In that case, the authors did not find any differences under all the four doses of biochar on soil pH and available water capacity; they found minor changes only in soil porosity and dry bulk density with 40 Mg ha^{-1} but considering the specific physical properties of their soil the little changes observed did not affect the wheat production.

Weed biomass, sampled in subplots of 0.25 m^2 soon after durum wheat harvest (12th July 2018) and about two months later (29th August 2018) in W and WB plots and mainly represented by *Convolvulus arvensis* L., *Setaria viridis* (L.) Beauv. and *Polygonum aviculare* L.. Soon after wheat harvest weed biomass was lower in W than WB, even if the differences were not statistically significant (Figure 4). Unlike in our experiment, Vaccari et al. (2011) observed an increase of soil temperature in biochar treated plots which promoted an anticipation of wheat emergency limiting weed competition. In our experiment, biochar addiction in general did not affect soil temperature except on three dates (Figure 5) and no faster crop emergency was observed in WB. Thus, it is conceivable that biochar addiction did not influence weed competition in WB as no differences emerged soon after wheat harvest among the wheat treatments (i.e., W and WB). This corroborates what observed in our experiment in terms of wheat yield: the absence of a starter effect connected to initial higher temperatures in WB treatment contribute to explain the absence of differences between W and WB wheat production. Moreover, two months after wheat harvest, weed biomass of W was higher compared to WB weed biomass but also in this sampling the differences were not significant. In biochar treated plots in the post-harvest period, Vaccari et al. (2011) found an increase in weed biomass suggesting an improvement of the soil water status which promoted the weed growth. On the contrary, in our study the post-harvest weed biomass did not differ between W and WB treatments, as probably the applied biochar did not affect soil water status as observed by Vaccari et al. (2011) in their silty-loam sub-acid soil.

Table 4. Wheat yield, moisture, protein and hectolitre weight at harvest (4th July 2018). W: wheat, WB: wheat + biochar. Data are average of three measurements

Treatment	Wheat yield ± s.e. (t ha^{-1})	Moisture ± s.e. (%)	Protein ± s.e. (%)	Hectolitre weight ± s.e. (kg/hl)
W	1.98 ± 0.24	13.30 ± 0.10	16.37 ± 0.37	70.00 ± 2.08
WB	1.79 ± 0.25	13.33 ± 0.03	15.93 ± 0.58	71.83 ± 0.77

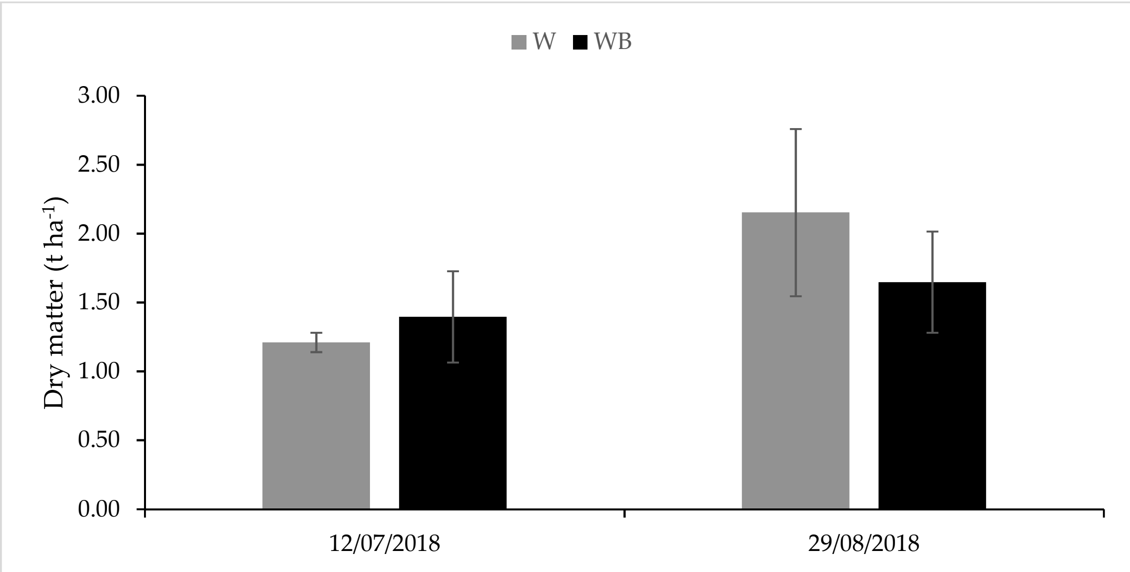

Figure 4. Weed biomass assessed soon after durum wheat harvest (12th July 2018) and after two months (29th August 2018). W: wheat, WB: wheat + biochar. Data are average of three measurements. Vertical bars represent standard errors.

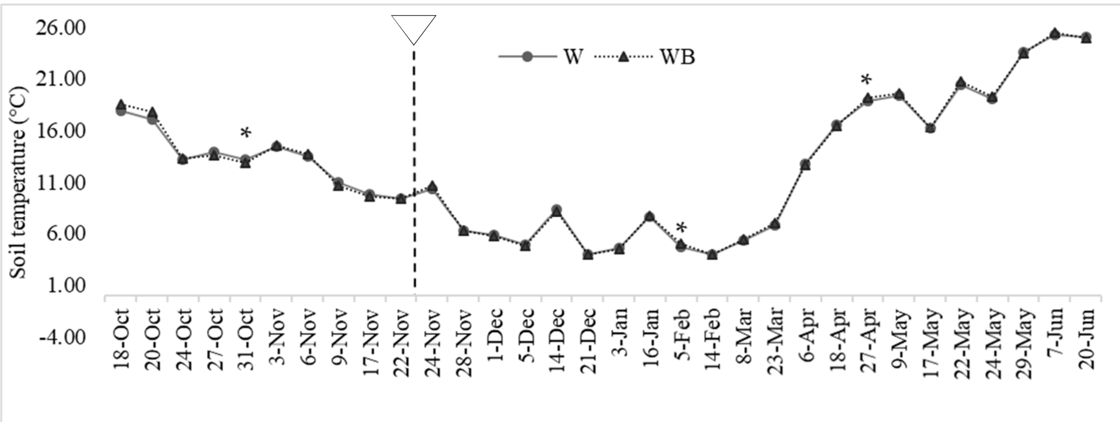

Figure 5. Seasonal variation of the soil temperature for the W: wheat and WB: wheat + biochar treatments. Soil temperature was defined at 10 cm in depth. Significant difference is expressed as "*" for P < 0.05. White triangle represents the sowing date.

4. Conclusions

The study filled important knowledge gap concerning the effects on N_2O emission determined by the change in type (from ploughing to spading) and in timing of the main tillage (from summer to autumn), compared to the traditional tillage system used for perennial crop termination in alfalfa-wheat rotation under Mediterranean environment.

Compared to the high summer temperatures leading to higher soil respiration and O_2 consumption with consequent promotion of N_2O emissions, the autumn lower temperatures reduce soil respiration, guarantee less anoxic environment and consequent lower N_2O emission. Postponing of the main tillage in early autumn, with still low soil humidity (that is one main N_2O driver factor), demonstrated to provide unfavourable conditions for the denitrification process and consequent lower level of N_2O emission compared to summer tillage as reported by relevant literature.

Moreover, our results highlight that alfalfa termination postponed in early autumn seems to contain N_2O emissions from wheat plots in which, compared to alfalfa plots, higher N_2O emissions were recorded only in three dates soon after tillage. This emission was probably due to mineralization process after tillage and the asynchrony between N released from alfalfa termination and the N uptake of the subsequent wheat. Anyway, this initial N_2O emission did not affect cumulative N_2O emission and no significant differences emerged among the treatments.

In conclusion, to reduce the impact of legume perennial crop termination is suggested (i) to postpone the main tillage in autumn when the soil temperature is lower and soil is still dry, and also (ii) to synchronize as much as possible the release of nitrogen into the soil by mineralization, occurring after soil tillage, with the N request from the next crop.

The specific characteristics of soil in the study area (i.e., the alkaline pH, loam soil texture) may have influenced the absence of effects of biochar application in terms of N_2O emissions and crop productivity. With this perspective, further studies could analyse the soil microbial activity as a function of the study soil and the biochar characteristics to better understand the mechanisms through which biochar can contribute to the N_2O emissions mitigation. Moreover, further studies could explore the effects of biochar aging in terms of N_2O emissions and crop productivity that could emerge in further years of experimentation.

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Chapter IV: Bottom-up design process of agri-environmental measures at a landscape scale: Evidence from case studies on biodiversity conservation and water protection

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Abstract

An Agri-environmental measure (AEM) is a payment to farmers to reduce environmental risks or to preserve cultivated landscapes. The single farm scale that is the basis for the AEM has often inhibited the achievement of the environmental goals since many biophysical processes (e.g. soil erosion, water pollution, biodiversity losses) occur at landscape scale. This creates a spatial scale mismatch between the implementation scale of the measures and the ecological processes controlling the target agri-environmental issues. In this paper, we propose how to address this spatial scale mismatch by analysing nine case studies of AEMs implementation at landscape scale concerning biodiversity conservation and water protection. The analysis highlights that the inclusion of the landscape scale in AEMs depends on the level of the involvement of the local stakeholders (SH) in the building process. When the authorities created the space for the SHs to participate in the defining process of AEMs, the inclusion of local knowledge led to the emergence of new landscape and site-specific AEMs which were not previously considered by the autorities. On the contrary, when the SHs were only allowed to choose among the AEMs predefined by the authorities, many site specificity and acceptance issues arose. The creation of space in Rural Development Programmes for collaborative, bottom-up and landscape scale AEMs and the overcoming of institutional constraints in the design of specific actions are the key ingredients for the successful adoption of measures and for enhancing their effectiveness. In this paper, we explore in depth what made these stories successful and provide a framework for the implementation of site-specific and landscape AEMs.

1. Introduction[1]

To support sustainable development of rural areas and to respond to increasing demands for environmental quality by society, the European Union (EU) introduced agri-environmental measures (AEMs) in 1985, with Council (EEC) Regulation 797/85. Later, the EU prescribed the mandatory implementation of agri-environmental programmes for all Member States (EEC Regulation 2078/92). The Agenda 2000 Common Agricultural Policy reform (EEC Regulation 1257/1999) then transferred AEMs into Rural Development Programmes (RDPs) (Defrancesco et al., 2008).

Agri-environmental measures can be defined at different levels (i.e., national, regional, local), and they are adopted by farmers on a voluntary basis. Most AEMs are management agreements that give compensation payments for the temporary adoption of specific practices, such as input-reduction, and landscape and habitat conservation measures (Uthes and Matzdorf, 2013). Several studies have highlighted the limitations of such AEMs. For example, some studies have stressed the "patchy success" of AEMs (Jones et al., 2016; Kleijn et al., 2006; Sutherland, 2004), with the objectives often too vague (Prager and Nagel, 2008). Others have stated that AEMs are not always suited for all kinds of farms (Evans and Morris, 1997; Hodge and Reader, 2010), and over/under compensation can be expected, in addition to several application problems (Klimek et al., 2008). On the other hand, there is evidence that the landscape spatial organisation can affect environmental processes like biodiversity conservation (Benton et al., 2003; Joannon et al., 2008; Kleijn and Sutherland, 2003) and water pollution (Beaujouan et al., 2001; Benoit et al., 1997; Toderi et al., 2007).

Existing incentive programmes typically neither require nor encourage landscape coordination, but instead favour a farm-level approach. However, many of the biophysical and ecological processes in agriculture do not occur at the farm level, but at the landscape scale (Kleijn et al., 2011; McKenzie et al., 2013; Prager et al., 2012). For these reasons, AEMs at the farm level can generate problems of spatial scale mismatch (Armitage et al., 2008; Cumming et al., 2006; Pelosi et al., 2010; Toderi et al., 2007).

The integration of knowledge from different stakeholders (e.g., farmers, scientists, experts) is considered a precondition for successful sustainable land management (Schwilch

[1] *Abbreviations:*AEA, agri-environmental agreement at landscape scale; AEM, agri-environmental measure; BIO AEA, biodiversity agri-environmental agreement at landscape scale; EU, European Union; NVZ, Nitrate-Vulnerable Zone; RDP, Rural Development Programme; WP AEA, water protection agri-environmental agreement at landscape scale

et al., 2012; Tarrasón et al., 2016). Participatory approaches and system perspectives for the identification and selection of options are becoming increasingly popular, and are required by the EU RDP (Prager and Freese, 2009). However, the unknown outcome for policy makers of a participatory process can limit its institutionalisation (Reed, 2008), and at all political levels, a big gap remains in the broad implementation of participatory processes (Rauschmayer et al., 2009). Stakeholder participation is increasingly seen as insufficient, and attention has shifted to social learning, co-management and empowerment goals as key issues (Armitage et al., 2008; Reed et al., 2008; Selin and Chavez, 1995).

Because the adoption of AEMs by farmers is voluntary, a high level of acceptance is required for their successful implementation. The perceived risk, effectiveness, scale of application (i.e., field, farm, landscape), and time and effort required for the implementation of measures are important factors that affect the willingness of farmers to join AEMs (McKenzie et al., 2013; Sattler and Nagel, 2010; Uthes and Matzdorf, 2013).

To involve stakeholders in the design of AEMs, and to overcome the spatial scale mismatch generated by the field/ farm level approach, the authority responsible for the control and coordination of RDPs in the Marche Region (central Italy) provided for agri-environmental agreements at the landscape scale (AEAs) in the RDP of 2007-2013 (Regione Marche, 2016). An AEA is defined as an agreement between public and/or private stakeholders to apply one or more shared AEMs in a specific territory of the region (e.g., a river basin, a protected area) above the level of farm, field or local-scale administration, with this designed to manage an environmental issue with a landscape dimension (e.g., water pollution, biodiversity conservation).

In the present study, we analysed how different AEAs and their AEM design process in nine case studies led to AEMs that are site-specific and/or that take into account biophysical phenomena on a larger scale with respect to the farm (a scale defined as "landscape AEMs" in this article). We also discuss how the differences between design processes: (i) affect local knowledge inclusion and stakeholder empowerment; (ii) have effects on the ability of stakeholders to generate innovative AEMs; and (iii) affect the degree of acceptance of the AEMs. From the analysis of these different case studies, we identified a design process of shared, site-specific and/or landscape AEMs with new roles for stakeholder involved.

2. Materials and methods

2.1. AEAs in the Marche Region RDP 2007-2013

According to the AEA procedure, stakeholders have to identify a lead partner who is responsible for: (i) administering an AEA; (ii) involving the stakeholders in a participatory process for AEM discussions; and (iii) planning the changes in the RDP with the regional authority (Regione Marche, 2010, 2011). In RDP 2007-2013, the Marche Region identified four major local environmental priorities on which to activate AEAs (Table 1). During the 2007-2013 planning period, the Marche Region activated AEAs exclusively on two of the priorities for which the stakeholders showed interest: one AEA on water pollution (WP AEA), and six AEAs on biodiversity (BIO AEAs) (Fig. 1). Two other attempts to create additional BIO AEAs were made, but these failed. Here, we also analyse the causes of these failures.

Table 1. Agri-environmental priorities and target areas identified by the Marche Region for AEA activation, and the case studies analysed.

Priority	Aims	Target areas	Agri-environmental agreements		
			Expression of interest	Successfully implemented	Analysed
Soil protection	Reduction of soil erosion and hydrogeological instability	Erosion hazard areas	0	-	-
Water conservation	Reduction of ground water pollution	Nitrate-Vulnerable Zones (Fig. 1)	1	1	1
Rural landscape conservation	Protection and recovery of hilly landscapes affected by agricultural mechanisation	High-value landscape zones	0	-	-
Biodiversity conservation	Conservation of biodiversity in protected areas	Natura 2000 sites (Fig. 1)	13	6	8

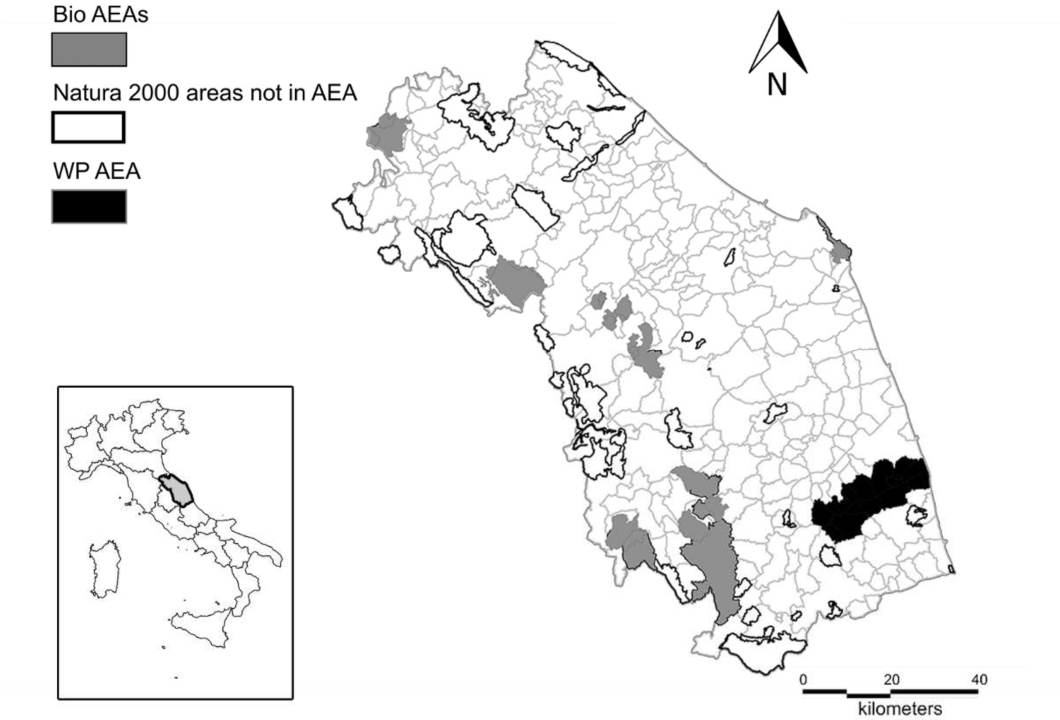

Fig. 1. Natura 2000 sites and AEAs activated in the Marche Region.

The WP AEA was activated in the Aso River valley, to reduce the high input of pesticides used in pest management by the dominant tree-fruit production-oriented farms. This included the territory of 15 municipalities, which were partially included in a Nitrate-Vulnerable Zone (NVZ) (EU Directive 91/676/CEE, and further modifications).

The BIO AEAs involved different Natura 2000 areas in terms of the pedo-climatic, environmental and socio-economic conditions. Five of the BIO AEAs were located in mountain areas, and one along the Adriatic coast. Natura 2000 sites in the Marche Region cover 136,900 ha, which corresponds to over 14% of the total area of the region. Specifically, the BIO AEAs require conservation of grassland habitats, as mainly the EU classifications of: 6210*, Semi-natural dry grasslands and scrubland *facies* on calcareous substrates (*Festuco-Brometalia*) (*important orchid sites); and 6510, Lowland hay meadows (*Alopecurus pratensis*, *Sanguisorba officinalis*), in the mountain areas where most of the grasslands are common pasturelands mainly subjected to customary grazing rights.

2.2. Theoretical framework adopted in the AEA analysis

The agri-environmental issues that occur at larger spatial dimensions than the farm/ field level are often resource dilemmas that are characterised by common pool resources, multiple stakeholders, interdependence, controversy, complexity and uncertainty (Blackmore, 2007; Ison et al., 2007). Inefficiencies occur and/or important components of the system are lost when there is a lack of alignment between the scale of the environmental variation and the scale of the social organisation, in which the responsibility for management resides. This can thus generate spatial scale mismatches. In these systems, long-term solutions will depend on social learning and the development of flexible institutions that can adjust and reorganise in response to changes in ecosystems (Cumming et al., 2006).

Reed et al. (2009) defined social learning as a change in understanding that goes beyond the individual, to become situated within wider social units or communities of practice through social interactions between actors within social networks. Collins and Ison (2010) considered social learning as an alternative governance mechanism and a process of systemic change and transformation undertaken by stakeholders in complex situations. Although more than one definition of social learning is available, the literature generally uses this term to refer to a "sustainability" type of transformative change that occurs at different levels, and in this, social learning is framed as a normative goal (Rodela, 2014). Armitage et al. (2008) analysed three potential loops of learning for co-management: fixing errors from routines (single loop); correcting errors by adjusting values and policies (double loop); and correcting errors by designing governance norms and protocols (triple loop).

Berkes (2009) identified the need for co-management for natural resources (i.e., the sharing of power and responsibility between government and local users), because of its complexity. Indeed, it is difficult for any one group or agency to have the full range of knowledge for environmental governance, and so the different partners have the potential to bring knowledge that is acquired at different scales to the discussion table, which will facilitate social learning. The important features of co-management include the sharing of authority, partnerships of government and local people, decentralised decision making, and vertical linkages for governance (Galappaththi and Berkes, 2015). Time-tested co-management with learning-by-doing turns into adaptive co-management. This can evolve spontaneously through feed-back learning over time from simple systems of management, and even if it does not appear to require legal arrangements to enable it, these might be required to sustain it (Galappaththi and Berkes, 2015). In this article, we highlight how legal arrangements that favour co-managment derive from a shift in the roles of policy makers in the system. When the shift in the roles of the policy makers does not occur, the co-management fails, or is at least delayed.

The integration of different types of knowledge into a “hybrid knowledge” for environmental management can foster collaborative approaches and social learning (e.g., Berkes, 2009; Prager et al., 2012; Raymond et al., 2010; Reed, 2008; Tarrasón et al., 2016). In this article, we argue that the integration of different knowledge is favoured by a shift in the roles of stakeholders, and that any interruption in this process will lead to interruption of the learning flux within the sytem. The shift in the roles of stakeholders is often unconscious, and it should be promoted in a stakeholder reflection process (Table 2, stakeholding). In the Social Learning for the Integrated Management and Sustainable Use of Water at Catchment Scale (SLIM) project (FP5-EVK1-2000-00695SLIM), which relates to NVZs, Natura 2000 and AEM issues, a heuristic tool was developed that can help stakeholders reflect on their own role in the management process (Blackmore et al., 2007; Ison et al., 2007; Steyaert and Jiggins, 2007). This diagnostic framework defines how a transformational change is positioned in a specific context (i.e., the history of the situation; Fig. 2, S1) that shapes current stakeholder practice and understanding (Fig. 2, S2). In addition, the diagnostic framework explains how changes in practice and understanding can be brought about by facilitation of the relationships among the stakeholders (i.e., the stakeholding), the ecological dynamics (i.e., the ecological constraints), and the whole complex of institutions and policies. These factors were identified as the four main variables that influence transformational changes (Table 2, Fig. 2), and also as variables in the sense that transformational changes can lead to transformation of each of the variables themselves. The diagnostic framework can be used to allow stakeholders to become aware of their role in transformational change (Steyaert and Jiggins, 2007), and for this reason, it was used to analyse the design and implementation processes of the AEAs

Table 2. The SLIM diagnostic framework variables.

Variable	**Description**
Stakeholding	Participatory process often leads to changes in the legitimacy of the stakeholder position or to the emergence of new stakeholders. The process by which stakeholders become aware of their role in the context is called “stakeholding”. Stakeholding takes over the concept of classical stakeholder analysis, and it monitors how the interests and social positions of the people involved can change over time, in relation to the issues at stake.
Ecological constraints	Stakeholders who live in and act on a specific territory deal with the components and processes that have to be taken into account. This variable analyses the stakeholder knowledge and awareness about these elements, called “eco-constraints”, because what is known about these processes tends to be fragmentary and based on expert sectorial knowledge.
Institutions and policies	This variable deals with the constitutive elements of the “institutional frameworks” (e.g., laws, social norms), constraints and deriving outcomes (e.g., new norms).
Facilitation	The facilitation in participatory process is a combination of the skills, activities and tools used to support the multi-stakeholder learning process. Moreover, the facilitation variable also analyses the stakeholder first-order learning (i.e., “what they are doing”) and second-order learning (i.e., “why they are doing what they do”), as described by Groot and Maarleveld (2000).

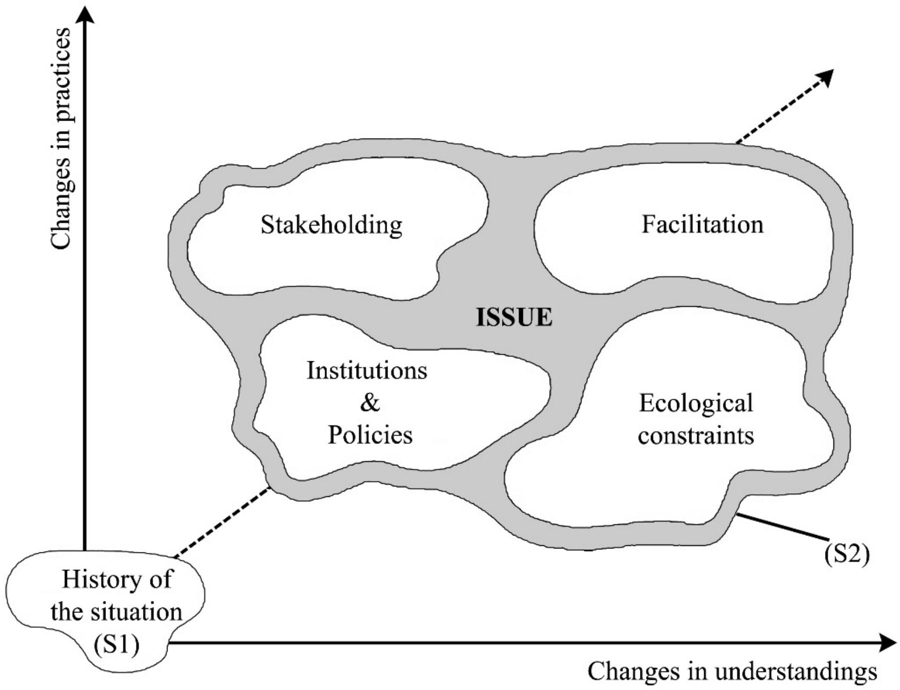

Fig. 2. The SLIM diagnostic framework. Heuristics for exploring the dynamics of transformational change, which are understood as changes in practices with changes in understanding, in complex and uncertain natural resources managing situations (S1–S3, situations one, two or three) (Steyaert and Jiggins, 2007).

2.3. AEA and AEM design process analysis

To assess the AEA design process, the stakeholders of the nine case studies were interviewed (i.e., one WP AEA; eight BIO AEAs, of which six were activated and two were aborted; Table 1). In the interviews, the stakeholders were asked to identify the issues that occurred in the design phase of the AEAs and in the later stages of their implementation, to identify potential cause–effect relationships.

The identification of the stakeholders was conducted as an interactive and iterative process (i.e., the snowball sampling technique). Therefore, the stakeholders interviewed were asked to identify other relevant stakeholders in the AEAs who can be interviewed. The survey started with two policy makers who were responsible for the Marche Region RDP.

Semi-structured interviews were performed after the AEAs were started, to discuss three main topics: issues that occurred in the AEA and/or AEM design and implementation processes; the stakeholder involvement; and the origin of the scientific knowledge used in the

AEM definition (Table 3). The interviews were recorded on a digital recorder and transcribed on a spreadsheet. The sentences obtained were clustered and analysed according to the SLIM diagnostic framework variables. Seventeen stakeholders were interviewed for the WP AEA, and 33 for the BIO AEAs (Table 4)

Table 3. Topics addressed in the semi-structured interviews to analyse the design and implementation processes of the AEAs/AEMs according to the related diagnostic framework variables.

Topic	Related diagnostic framework variables
Design and implementation of the AEA/AEMs	
How the RDP or the AEMs were modified according to your needs?	Institutions and policies, Facilitation, Ecological constraints
How the institutional and normative framework were included in the AEAs and AEMs?	Institutions and policies
Why the AEMs were/were not site-specific?	Ecological constraints
How would you like to improve the AEA/AEM design and/or implementation processes?	Institutions and policies
How and when was the stakeholder involved in the AEMs design and/or implementation processes?	Facilitation, Institutions and policies
The process of stakeholder involvement	
Who triggered the AEA activation, and how?	Facilitation, Stakeholder and stakeholding
Who contacted/ informed/ involved you, and how?	Facilitation, Stakeholder and stakeholding
Who was the facilitator, and how did they act?	Facilitation, Stakeholder and stakeholding
Who were the stakeholders involved?	Stakeholder and stakeholding
Were some relevant stakeholders excluded or not considered?	Facilitation, Stakeholder and stakeholding
Scientific knowledge supporting the design of AEA/AEMs	
Was the scientific knowledge discussed and in which phase of the design of AEAs/AEMs?	Facilitation, Ecological constraints
Who were the knowledge brokers and what were their roles in the design process?	Stakeholder and stakeholding

Table 4. Stakeholders active in the design and implementation processes of the AEAs/AEMs, their roles that emerged from the interviews, and the number of stakeholders interviewed in the WP AEA and the BIO AEAs. Grey shading, people who were not stakeholders in the AEAs/AEMs.

Stakeholder	Role in the AEA	WP AEA	BIO AEAs
Policy makers of the Marche Region Agriculture Service	Responsible for Marche Region RDP and AEMs	3	3
Policy makers of the Marche Region Environment Service	Responsible for Natura 2000 sites and their AEMs	No active role	3
Agents of the Regional Extension Service	Carrying out of local demonstration projects	2	No active role
Farmers	Implementation of AEMs	6	8
Policy maker of the local public administrative body	Lead partner of an AEA	1	No active role
Practitioners	Dialogue with farmers	1	3
Natura 2000 Site Managers	Lead partners of the BIO AEAs		8
Payment authority	AEM control of the eligibility for payments	0	0
Policy makers of the Agriculture Ministry	Responsible for Italian Natura 2000 sites management		1, but no active role
Farmers' Union staff members	Responsible for dialogue with farmers and policy makers	1	3
Inhabitants	Consumers of local products	3, but no active role	No active role
Managers of the bodies that regulate the customary grazing rights (e.g., municipalities, collective bodies)	Implementation of the AEMs in the common lands		3
EU Directorate-General for Agriculture and Rural Development	Responsible for the agriculture and rural development policy	0	0
University researcher	Research activities on grassland management and Natura 2000 site manager		1

2.4. Documents analysed to derive the historical context and AEA/AEM design process

To evaluate the process that led the Marche Region to activate AEAs, the following official reports were analysed (available online): (i) ex-post evaluation of EEC Regulation N° 2078/92 implemented in the Marche Region, and ex-ante evaluation of the Marche Region RDP 2000-2006 (*Ministero delle Politiche Agricole, Alimentari e Forestali,* 2016); (ii) ex-post evaluation of the Marche Region RDP 2000-2006 (EU Directorate-General for Agriculture and Rural Development, 2016); (iii) ex-post evaluation of the AEMs included in the Marche Region RDP 2000-2006 (*Rete Rurale Nazionale*, 2016); and (iv) ex-ante evaluation of the Marche Region RDP 2007-2013 (*Regione Marche*, 2016). The research team participated in the mid-term review and the ex-post evaluation of the effects of the AEMs included in EEC Regulation N° 2078/92 and in the Marche Region RDP 2000-2006.

3. Results

3.1. AEMs of the Marche Region: history of the situation

The historical background emerges from analysis of the official documents of the Marche Region. From 1998 to 2006, the research group performed an evaluation of the effects of the AEMs included in EEC Regulation N° 2078/92 and in RDP 2000-2006 of the Marche Region, on water soil erosion and nitrate leaching reduction. Ex-post evaluations showed that the application of AEMs at the field scale did not significantly reduce soil erosion and nitrate leaching, due to their landscape dimension (Toderi et al., 2007; Perugini et al., 2009). The results were discussed in several informal meetings with the policy makers, and it was recommended that AEMs at the landscape scale be designed also in cooperation with local stakeholders. As a consequence, in an ex-ante RDP 2007-2013 evaluation document, the Marche Region administration reported that: "*…the quantitative evaluation of AEMs in the RDP* [2000-2006] *highlighted the importance of adopting an integrated territorial approach that could be complementary to the farm payments and would foster greater awareness of the action by farmers...*". In the same document, the Marche Region considered the involvement and participation of stakeholders as indispensable for landscape-scale AEM definition. To address these issues, the Marche Region included the AEA approach in RDP 2007-2013.

3.2. The WP AEA design process

3.2.1. History of the situation

The interviews showed that in the Aso River valley, an agent of the Marche Region Extension Service was particularly active, and in the past, demonstration projects had been conducted in close collaboration with four local farmers to reduce the high input of pesticides in pest management for fruit production, and particularly for peaches. The trials focused on mating disruption, which is a situation where pheromones are released into a pest habitat in sufficient amounts to reduce the ability of the males to find females, or *vice versa* (Baker and Heath, 2005). The demonstration projects were successful only in some areas, because the farms were small and this technique is *"… effective only if implemented over wide areas, in order to avoid the entrance of mated females from non-treated areas…*" (an interviewed expert).

3.2.2. Stakeholder and stakeholding

The WP AEA was born as a result of the triggering of the regional Extension Service agent, who was well-informed about the AEAs. Based on the results of previous investigations performed in the area, the agent proposed the creation of a local AEA to four farmers. A first group of stakeholders was created, which included other local stakeholders (Table 4), and then they asked the policy makers of the Marche Region Agriculture Service to activate a new mating-disruption AEM under WP AEA. Considering the small area of the Aso Valley, the policy makers underlined the risk of the low participation compared to the complex procedures to renegotiate the RDP with EU officials. At the end of the AEA implementation, "*...the applications were so many that regional managers could not believe it…"* (the Extension Service agent), and this result "…*was related to the trust that the agent had with the stakeholders…*" (regional officer). Stakeholders designated the local public administrative body as the lead partner, but maintained control of the AEA. Almost 100 farms and about 1000 ha (80% arable land; 20% orchards) were included in this WP AEA.

3.2.3. Facilitation

Many stakeholders highlighted that the WP AEA measures were discussed in participatory meetings where the regional Extension Service agent demonstrated strong connections and mutual trust with the farmers, with whom she (probably unconsciously) had the role of a facilitator. The farmers were not passive in the design process of the AEA, but as the Extension Service agent said, *"…they discussed the AEMs with regional officers, they called me when necessary, they organised the meetings, and they went house to house to involve more farmers … we worked very well in synergy…"* and many other farmers joined the AEA design process after *"…they saw the results of the experiments…"* (a farmer).

3.2.4. **Institutions and policies**

Negotiations with the EU concerning some RDP modifications was required to include the new the mating-disruption technique AEM that had emerged in the participatory design process. The negotiations were carried out directly by the regional officers, who agreed to the requests of the local stakeholders step-by-step, and reported the objections of the EU to the stakeholders. Normative problems emerged concerning the farming areas; as the local RDP allowed the WP AEA only within the NVZs, the Marche Region and the EU negotiated the enlargement of the eligible area to make the application of the mating-disruption technique more effective.

Even if the EU showed interest in this AEA approach, the RDP renegotiation process was so laborious that a regional officer who was interviewed defined it as *"…a delirium…"* because *"…innovative bottom-up actions need to be translated into bureaucratic language, which is tricky, hostile and complex..."* (Agriculture Ministry officer).

The process to define and negotiate the new RDP AEMs lasted approximately 1 year, and led to the AEA measures that are listed in Table 5.

Table 5. The new WP AEA measures agreed between the stakeholders and included in the Marche Region RDP 2007-2013 after negotiations with EU.

Measure	Description
1.1.1.b	Training activities and information actions
2.1.4.a	Integrated farming with advanced integrated pest management (mating disruption)
2.1.4.b	Organic farming systems
2.1.4.c	Permanent swards

3.2.5. **Ecological constraints**

The only constraint that emerged in the WP AEA was related to the small farm areas, which would have constrained the application of the mating-disruption technique if this was applied by the individual farmers. The enlargement of the eligible area beyond the boundaries of the NVZ allowed the aggregation of sufficient orchard areas to effectively apply the techniques.

3.3. The BIO AEA design process

3.3.1. History of the situation

The design process of the BIO AEAs began in the final phases of the WP AEA design process. As for most of the other regional administrations in Italy, the Marche Region was late in the preparation of the AEMs for the management of the Natura 2000 sites, mainly due to strong conflicts with farmers.

In one of the Natura 2000 sites (called Torricchio), the manager was also a researcher from a local University who had previously carried out research into grassland conservation management in close collaboration with the local farmers, and who had a role similar to that of the Extension Service agent in the WP AEA. Some shared management practices emerged from these collaborations.

3.3.2. Stakeholder and stakeholding

The analysis of the BIO AEAs describes a different genesis path compared to the WP AEA. The same policy makers of the Agriculture Service involved in the WP AEA were the trigger for the BIO AEAs. Considering the unexpected success of the WP AEA, they hypothesised the activation of AEAs for the Natura 2000 sites. The policy makers proposed that the Marche Region Environment Service join in the drafting phase. Together, they identified the managing authorities of the Natura 2000 sites as AEA lead partners.

Each lead partner was asked to identify the AEMs to be implemented in their areas. Only the Torricchio manager proposed a set of AEMs shared with local stakeholders. These AEMs were then evaluated by the policy makers and, after a negotiation phase with the EU, they were included in the RDP with some modifications, and without any other consultations with the AEA stakeholders.

Each BIO AEA lead partner was then asked by the Marche Region to design their AEAs through participatory meetings with other local stakeholders, to select from among the proposed AEMs those that were most suitable and applicable in their area. To select the AEMs, the BIO AEA lead partners involved the municipalities that are included in the Natura

2000 sites, along with the farmers and the Farmers' Union staff members. Most of the BIO AEAs facilitators were practitioners with different backgrounds (e.g., agronomists, biologists) and/or local Farmers' Union staff members, while in the Torricchio BIO AEA, the facilitator was the researcher who was managing the area. Some other managers adopted all of the AEMs without any discussion with the stakeholders, and asked them to submit their applications to join the BIO AEAs.

In the BIO AEAs design process, no active role was taken by the Ministry of Agriculture. A Ministry officer who was responsible for the management of the Italian Natura 2000 sites stated when interviewed: *"…in Italy* [April 2013] *we have not spent enough on Natura 2000 yet…"* and highlighted how the AEA approach *"…is innovative but risky, in terms of payment, if the Payment Authority is not involved from the starting phase…"*. This lack of involvement in the BIO AEAs caused payment delays that discouraged other farmers from submitting applications.

3.3.3. **Facilitation**

Unlike what was observed for the WP AEA, in the BIO AEAs only the the Agriculture Service, the Environment Service, and the Natura 2000 site lead partners shared this process. The Torricchio BIO AEA was an exception here, where the measures were designed in close collaboration with the local stakeholders. In all of the other cases, the local stakeholders were involved only in the later stages, where they were only able to choose which AEM to be implemented in their AEA, and which to exclude.

The missed opportunity for modification of the AEMs restricted the number of farmer applications, and created conflicts and uncertainty. In these cases, the facilitators were perceived as, *"…people who did not understand the environmental context of the place…"* (Farmers' Union staff member) or even as *"...dictators…"* (farmer). Moreover, *"…the AEA was seen as a new restriction to the farmers' activities…"* (Farmers' Unions staff member) due to the impossibility of adapting the AEMs to local conditions. Other conflicts emerged between the managers of the Natura 2000 sites and the farmers concerning the constraints linked to grassland management. For example, a farmer stated: *"…cutting a shrub in a pastureland was impossible* [due to strong vegetation protection measures], *and the managing authority has to understand that pasturelands must be managed to be maintained…*".

Different outcomes emerged in the Torricchio BIO AEA. A Farmers' Union staff member stated that, "*…the initial number of the application forms were around 60 in all of* *the BIO AEAs, with about 40* [of these] *from the Torricchio AEA…*”*,* which was a consequence of the involvement of the stakeholders in the definition of the AEMs and of the past co-reasearch activities, and thus the AEMs were site-specific. Despite this, some of the AEMs modified by the Marche Region generated uncertainty among local stakeholders, due to their lack of knowledge and understanding of the modified AEMs "…*because we did not know how to apply the AEM prescription…*" (farmer).

For the Natura 2000 sites, where all of the predefined AEMs were adopted by an AEA lead partner without any discussion, the lack of involvement of the stakeholders created high levels of conflict with the farmers. As a result, some of the AEMs were refused and the lead partners were forced to withdraw from the AEA implementation.

3.3.4. **Institutions and policies**

Independent of the area and the site-specific conditions, each AEM was mostly the same in each of the BIO AEAs. For this reason, some farmers faced paradoxes, like "…*the request to control non-present invasive species…*" (farmer) in their area (e.g., *Brachypodium* sp.), or the request to increase wooded hedges in woodland-dominated areas. Some AEMs were refused in the EU negotiation phase because they were *"…not controllable by the Payment Authority…"* (regional officer). As mentioned in section 3.3.2. (Stakeholder and stakeholding), in the first 2 years of the BIO AEAs, the farmers experienced long delays in the payments due to property issues that were linked to the use of common pasturelands. These delays were overcome after long negotiations between the Marche Region, the EU, and the Payment Authority.

The AEMs proposed for the BIO AEAs are listed in Table 6, although not all of these AEMs were necessarily adopted in each of the BIO AEAs.

Table 6. BIO AEA measures included in the Marche Region RDP 2007-2013 after negotiations with the EU. The local stakeholders could choose those to be applied or not in their AEA, but no changes were allowed to their content.

Measure	Description
1.1.1.b	Training activities and information actions
1.2.5.a	Improvement of drinking troughs in pasturelands
2.1.1.a	Natural handicap payments for farmers in mountain areas
2.1.3.a	Natura 2000 compensation payments
2.1.4.b	Organic farming compensation payments
2.1.4.d	Conservation of native endangered germplasm resource compensation payments
2.1.6.a	Non-productive investments measures

3.3.5. **Ecological constraints**

The ecological constraints that emerged were closely connected to the different climatic and environmental conditions of the different AEA areas. As the Marche Region applied similar AEMs in each of the BIO AEAs, the farmers perceived some measures as not being site-specific, and therefore as inadequate for their conditions; e.g., postponed ploughing on clay soils for winter cereals in mountain areas. Similar issues emerged for the measures aimed at the conservation of 6210* grassland habitats without taking into account the behaviours of the different grazing animals, as was suggested by the local farmers.

As for the WP AEA, the interviews highlighted some constraints related to the BIO AEA eligibility areas. Farmers with smaller farm areas included in the Natura 2000 sites did not obtain any economic advantages from joining the AEA. Therefore, some of the lead partners did not reach any agreement with these farmers, and the AEA design process failed.

Six BIO AEAs are currently ongoing throughout the Marche Region, which cover around 52,000 ha, and are mainly for conservation management of 6210* grassland habitats.

4. Discussion

The analysis of the AEA design process applied in the different case studies (Table 7) allowed the identification of the key elements that led to AEMs that were well accepted, site-specific, and took into account the landscape dimension of the biophysical processes (landscape AEMs). In the following paragraphs, we analyse in more detail the consequences of the different design pathways that were used in the AEA case studies.

Table 7. Differences in the design processes of the AEMs adopted in the AEA case studies.

<table>
<tr><th>Phase</th><th>WP AEA</th><th>Torricchio BIO AEA</th><th>Other BIO AEAs</th></tr>
<tr><td>AEA trigger</td><td>Local stakeholders proposed the activation of an AEA to the Marche Region. They planned their own AEA, identified their lead partner, got other stakeholders involved, and managed the participatory meetings to plan their shared measures.</td><td colspan="2">The policy makers understood the potential of the WP AEA and tried to apply the same framework to the BIO AEAs. The policy makers involved the Natura 2000 site managing authorities, who were designated as the AEA lead partners and were asked to define the AEMs for the target areas in close cooperation with the stakeholders.</td></tr>
<tr><td>AEM design</td><td>In participatory meetings, local stakeholders planned their shared measures. The introduction of the mating-disruption technique was the result of several demonstration projects that had been carried out with farmers previously, and it was shaped according to their needs.</td><td>A University researcher and the Torricchio Natura 2000 site manager discussed the AEMs to be defined for their area with the local stakeholders. The AEMs were already well known by the local stakeholders from previous research activities in the area.</td><td>The lead partners of all of the other BIO AEAs had poor relationships with the local stakeholders. Furthermore, the farmers were never involved in the local research activities. For these reasons, they were not involved in the design processes of the AEMs.</td></tr>
<tr><td></td><td>The Marche Region discussed each single modification requested by the EU with the stakeholders.</td><td colspan="2">Due to the lack of further proposals and on the grounds of urgency, the Marche Region adopted the measures proposed by Torricchio for all of the BIO AEAs, with some changes that had not been agreed with the Torricchio stakeholders. In the following steps, the local stakeholders of all of the BIO AEAs were only allowed to choose between the AEMs that were proposed based on the Torricchio experience.</td></tr>
<tr><td></td><td>A new site-specific and landscape scale AEM emerged, to take into account the landscape dimension of the pest management of orchards. The new measures included were therefore highly accepted by the local stakeholders.</td><td>The AEMs were site-specific, and some of them were also AEMs at a landscape scale. Some other modified AEMs were considered of little use or improvable by the farmers, but probably not detrimental to their income. Despite the uncertainties generated from the modified measures, the new measures were highly accepted by the local stakeholders.</td><td>No site-specific or landscape scale AEMs emerged. Some of the AEMs were considered to be of little use, improvable, or even detrimental to their income by the farmers. Many conflicts emerged between the stakeholders.</td></tr>
<tr><td>EU negotiations</td><td colspan="3">A long phase of negotiations was needed to modify the Marche Region RDP and to justify the AEMs not presented in the RDP ex-ante evaluation.</td></tr>
<tr><td>AEA applications</td><td colspan="2">AEAs were successfully implemented with a high number of farmer applications.</td><td>Five AEAs were implemented with different results between the areas. The AEA implementation failed in two case studies.</td></tr>
</table>

4.1. Inclusion of local knowledge leads to site-specific AEMs

In the case studies analysed, the design process of the AEAs highlighted the different levels of inclusion of local knowledge and the different effects on the site specificity of the AEMs, and thus on their acceptance (Table 7, AEM design phase).

The WP and Torricchio BIO AEAs were the most successful in terms of stakeholder agreement. Probably unconsciously, some of these stakeholders will have acted as key stakeholders and carried out the role of knowledge brokers (Reed et al., 2009), thus shifting their institutional role in the system (Table 4). In these two case studies, the AEMs were defined in close cooperation with the farmers from the beginning, and arose from the combination of trust and local knowledge that had been generated in previous research activities and in the participatory meetings, and were therefore site-specific and well known, and thus also well accepted. Essential conditions for 'win–win' agri-environmental policy making are: interest in the issue, decision alternatives, trust among the parties, transparency of the process, and dedicated personnel (Prager and Freese, 2009); stakeholder participation right from concept development and planning (Reed, 2008); responsibility for developing management solutions remains with farmers (Burton and Paragahawewa, 2011); and flexible schemes that are adaptable to changing circumstances (Emery and Franks, 2012). Similar indications emerged also from the WP and Torricchio BIO AEAs, which also emphasises the need for a shift in the roles of all of the stakeholders (Table 4). The farmers shifted from the passive role of "implementors of AEMs" to the active role of "AEM designers". The Farmers' Unions staff members, who are usually in charge of the lobbying activities, shifted their role to "supporters of the participatory process".

The flow of local knowledge should not be interrupted in any phase of the AEM design process. Indeed, despite the similarities between the WP and Torricchio BIO AEAs, some differences can be seen. In WP AEA, the measures were planned as a result of cooperation between local stakeholders, and the policy makers shifted their "command and control" role (Table 4) to stakeholders working in collaboration with other stakeholders. Indeed, there was continual debate between the stakeholders at all of the steps of the AEA design process (e.g., negotiations of the AEMs with the EU), and local knowledge flow was fed into each step, facilitated by the shift in the stakeholder roles. In addition to this shift, in WP and Torricchio BIO AEAs, it was possibile to observe: (i) sharing of authority; (ii) partnerships of government and local people; (iii) decentralised decision-making; and (iv) vertical linkages for governance; these are features that were listed as very important for co-management by Galappaththi and Berkes (2015). On the other hand, in the Torricchio BIO AEA, policy makers returned to the original “command and control” role when they changed some measures without sharing these with the local stakeholders. Sharing of authority was replaced with a linear transfer of knowledge (Ison et al., 2011), and the farmers switched back to the role of “predefined AEM implementors”. The flow of local knowledge was interrupted, creating diffidence among the stakeholders, and problems for the site-specificity of some of the AEMs (Table 7, AEM design). However, for both of the case studies, co-management spontaneously emerged and evolved through feed-back learning (i.e., past research and demonstration projects).

In the other BIO AEAs, stakeholder involvement took place only at the later stages of the design processes and with limited decision alternatives (Table 7, AEM design), which in turn limited the trust among the parties and in the process (i.e., lack of co-management features). For these reasons, the design process was perceived by local stakeholders as not being transparent, and this created a lack of empowerment (i.e., limited or no sharing of authority), and the facilitators were perceived as mere executors of the decisions of the lead partners. Thus the shift of the stakeholder role in the system (Table 4) did not occur in these cases. This situation arose due to the lack of reflection by the policy makers on their new role in the system. In particular, they did not analyse the reasons behind the success of the WP and Torricchio BIO AEAs and they hypothesised the same AEMs for the other BIO AEAs.This decision led to the implementation of meaningless (from the stakeholders perspective) or inapplicable measures. Some studies have observed similar dynamics, where historical and contextual differences have led to policies that were successfully adopted in one area and were inappropriate or refused in other similar areas (Armitage et al., 2008; Steyaert and Jiggins, 2007).

As suggested by Burton and Paragahawewa (2011), to produce agri-environmental goods, farmers need to learn about the connections between their land management practices and environmental outcomes. In terms of this vision, The policy makers should have analysed the whole of the WP AEA process (e.g., using the SLIM diagnostic framework and its variables) to highlight the reasons behind its success.

According to Galappaththi and Berkes (2015), the other BIO AEAs could be classified as cases of “unsuccessful top-down co-management”, due to the lack of knowledge of the features needed for a co-management process.

4.2. AEAs legal arrangement allows inclusion of a landscape approach in AEMs

In the WP and Torricchio BIO AEAs, the aggregation of the farmers favoured the emergence of the landscape dimension of some environmental issues, and overcame the farm-level approach.

In the WP AEA, the stakeholder understanding of the landscape dimension emerged during the AEA design process. The request to expand the target areas to include farms located outside the NVZ came from the stakeholders, and was designed to improve the effectiveness of the measures based on the mating-disruption technique. McKenzie et al. (2013) observed high willingness of farmers to participate in collaborative AEMs as long as these were applied only to portions of their farm, and not to their whole farm. We believe that the learning that was generated from the beginning in the WP AEA design process strongly increased the “willingness” of the farmers. Indeed, the new site-specific and landscape AEM were applied to the entire farms, even for the most valuable products, like the orchards.

In the Torricchio BIO AEA, the stakeholders showed an understanding of the landscape dimensions by analysing their areas as a continuum to be managed collectively, and not as a collection of fields. Indeed, during the meetings, the stakeholders identified some areas where the grazing period for biodiversity conservation should be earlier, and others in which it should be postponed. This confirms what was observed by Prager and Freese (2009), that farmers or local stakeholders can identify fields of cooperation, find innovative solutions to problems identified, and generate win–win situations.

In the other BIO AEAs, despite the imposing of the AEMs by the policy makers, the AEA design process was sufficient to create some attempts for landscape planning, co-management, and a tentative shift in the stakeholder roles for farmers, similar to Torricchio BIO AEA. This highlights that the AEA formal arrangements allowed the emergence of a landscape dimension and favoured the shift in roles also under these less than ideal circumstances for knowledge inclusion and sharing. For example, collective management of the fragmented properties and of the pasturelands under customary rights was proposed by the stakeholders during the participatory meetings. However, these proposals were not accepted because the AEMs were already included in the Marche Region RDP after the negotiations with the EU Directorate-General for Agriculture and Rural Development, and they were not modifiable again within a reasonable time. In this case, the “rulebook” for land management, that was criticised by Burton and Paragahawewa (2011) because it was seen to constrain the abilities of the farmers to develop unique and innovative solutions to reach scheme targets, limited the possibility of generating shared choices in most of the BIO AEAs.

At a landscape scale, Prager et al. (2012) stressed the need for participatory and collaborative approaches that facilitate the processes of communication, negotiation and feedback, to allow joint monitoring, learning, and scheme adjustments. The AEAs did indeed favour the co-management of natural resources and scheme adjustments whenever the shift in stakeholder roles and local knowledge inclusion occurred.

Prager and Nagel (2008) observed that authorities tend to feel threatened by participatory approaches due to the risk that they might lose their power and legitimacy. The analysis of WP and Torricchio BIO AEAs highlights the empowerment of some stakeholders, which however was not a consequence of power loss or legitimacy from other stakeholders (i.e., policy makers), but derived from the sharing of power and responsibility. The policy makers could have stopped the process triggered by the AEA legal arrangement at any moment, as they actually did in other BIO AEAs. However the policy makers themselves were involved in the process, and they shifted their role and participated in the sharing of knowledge. For these reasons they were not delegitimised by the process and they were encouraged to propose the BIO AEA.

However, a missing part of the process that was not envisaged by the policy makers was the definition of the monitoring and feedback mechanisms that should generate, in turn, learning among stakeholders. Moreover, the "stakeholder involvement" within the AEA process remained vague, which according to many studies (e.g., Prager and Freese, 2009; Reed, 2008; Tarrasón et al., 2016) highlights the need to include skilled facilitators, and the institutionalisation of the participatory processes. The results of this evaluation carried out by our research group were discussed in several informal meetings with the policy makers responsible for coordination of the Marche Region RDP 2014-2020, who decided to include a "skilled facilitator" among the eligible costs for the new call for AEAs (AEA 2.0) that was published in September 2016.

Also, the reciprocal trust among the stakeholders generated in the AEA design process was seen to create an informal learning platform and favourable conditions for further concerted actions at a landscape scale. Starting from the WP AEA experience, the Aso Valley stakeholders have activated a supply production chain of fruit based on the mating-disruption technique, with the deeper involvement of the inhabitants and local environmentalists. In the

Torricchio BIO AEA, the stakeholders have started to reflect on the possibility of applying for a short food-supply chain based on lamb meat.

The emergence of social learning within a process of co-management turns into adaptive co-management (Galappaththi and Berkes, 2015). In a review, Rodela (2014) suggested that learning and social learning are not interchangeable. An attempt to distinguish between these was provided by Reed et al. (2009), who proposed that if learning is to be considered "social learning" then it must: (i) demonstrate that a change in understanding has taken place in the individuals involved; (ii) demonstrate that this change goes beyond the individual and becomes situated within the wider social units or communities of practice; and (iii) occur through social interactions and processes between actors within a social network. From what we observed for WP and Torricchio BIO AEAs, it is possible to highlight many examples of learning processes generated from the knowledge sharing. Indeed, we observed in the stakeholders (and not only for the farmers) signs of "changes in understanding" and learning that occurred "through social interactions". We cannot say the same with any certainty with regard to point two of Reed et al. (2009) (i.e., "goes beyond the individual"), which is probably still ongoing (i.e., supply production chain processes). However, it is possibile to identify a double loop of learning in the design process of both the AEAs, which is defined as correcting errors by adjusting values and policies (Armitage et al., 2008).

According to Galappaththi and Berkes (2015), from our case studies it emerged that adaptive co-management can evolve from simple systems of management spontaneously through feed-back learning over time (i.e., the past research and the demonstration project). It emerged also that a formal arrangement was necessary to sustain this, which was provided by the approach of the AEAs in the local RDP.

4.3. Framework for the design of AEMs at a landscape scale, as emerged from the case studies

To improve the definition of site-specific and/or landscape scale AEMs, an iterative process that was based on the experiences of the AEA case studies was identified (Fig. 3). In light of the EU environmental priorities, local policy makers should only identify a set of targets in the RDPs without predefined measures, to create room for the bottom-up emergence of the AEMs (i.e., sharing of authority, decentralised decision making). In this vision, the AEMs should emerge from a participatory analysis of the site-specific conditions by the local stakeholders (i.e., shift of the stakeholder role in the system, knowledge sharing and inclusion), who need to be involved from the very first phases of the design process. In a following step, co-analysis of the proposed AEMs is required that includes the local and EU policy makers (i.e., partnerships of government and local people, vertical linkages for governance), and also other relevant stakeholders that can be identified (e.g., researchers). Policy makers need to avoid making any changes to the AEMs without sharing the reasons with the other stakeholders.

In this framework, the stakeholders will improve their system knowledge, which will lead to: (i) identification of new stakeholders who were not included at the beginning (e.g., the Payment Authorities); (ii) analysis of the ecological constraints that affect the system (e.g., the need for a landscape approach for the mating-disruption technique); or (iii) taking into account the institutional and policy framework (e.g., time constraints for RDP fund expenditure, grazing rules on common land) that might limit or create new conditions for the stakeholder actions.

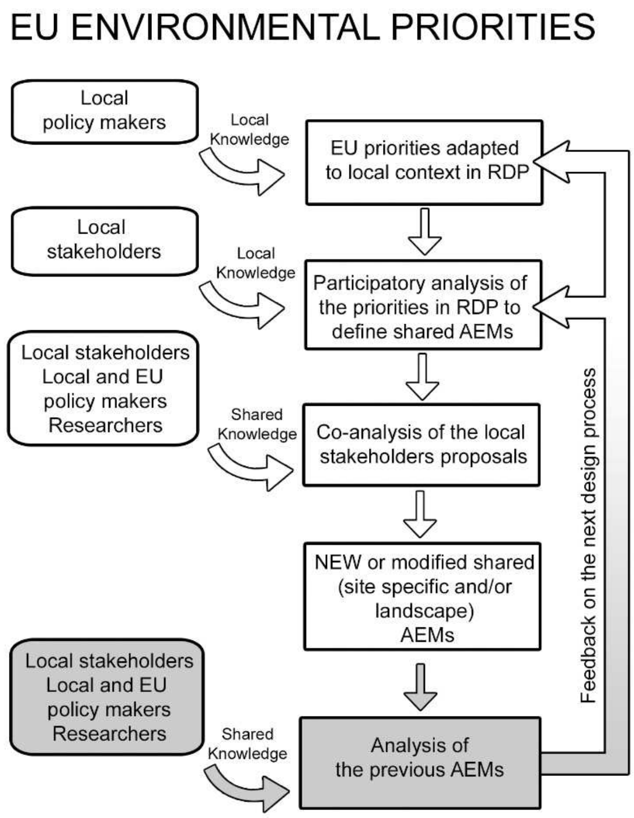

Fig. 3. The iterative process for the design and evaluation processes of the AEMs.

In agreement with the SLIM diagnostic framework and with other studies (e.g., Berkers, 2009; Prager and Freese, 2009), an analysis of the previous measures by the stakeholders should be performed (Fig. 3). This is because the socio-ecological conditions of the ecosystems are constantly changing, and the successive loops of learning and problem solving in learning networks can incorporate new knowledge that can be used to deal with problems at increasingly larger scales (Berkers, 2009).

The design process described was applied in the WP AEA, mostly applied in the Torricchio BIO AEA, and not applied at all in the other BIO AEAs (Fig. 4). As the WP AEA had all of the characteristics mentioned above, new shared site-specific and landscape measures emerged. In the Torricchio BIO AEA, the process was altered by the intervention of the regional authorities that imposed some AEMs, which thus created problems with farmer acceptance. In the other BIO AEAs, many conflicts arose because of the lack of local stakeholder involvement. There was no evaluation process in any of these case studies because it was not required by the local RDP.

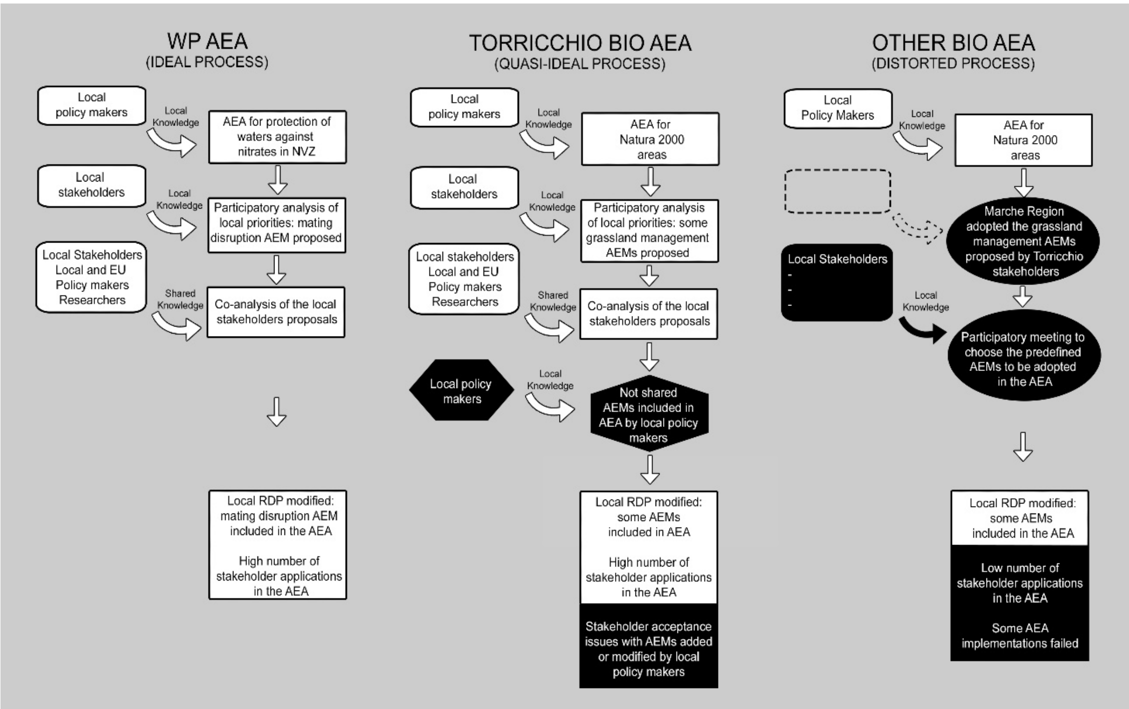

Fig. 4. The ideal design process that was applied in the WP AEA, and the differences seen for the other case studies.

In a review on AEM, Uthes and Matzdorf (2013) underlined that "…*there is not yet extensive literature on collaborative AEM, but the existing studies suggest that collaboration* *among farmers on a larger or even landscape-wide scale may be promising, if properly designed and implemented*". In a viewpoint article, from other case studies, Prager et al. (2012) derived the factors and overlapping phases needed in the design and implementation process, which started from the various types of information collection needed to feed the process. Prager (2015) underlined many different aspects that lead to collaborative agri-environmental management: awareness of a problem; good horizontal and vertical communication; access to high quality advice and support; support of existing groups and networks; trust; flexibility in scheme design; funding for feedback; monitoring and evaluation of the results. Except for the last two (i.e., funding for feedback; monitoring and evaluation of the results), these features were present in the successful examples of the AEM and AEA design processes that we have discussed here. Our case studies add to these features the need for a shift in the stakeholder roles in the system, and the framework we propose creates a legal arrangement that targets the shift in the roles of all of the stakeholders, including the policy makers. Indeed, delegation of stakeholders to produce AEMs facilitates the shift in their roles and implies the need to involve other different stakeholders with different knowledge in the analysis and design processes (e.g., researchers, but also policy makers and other land managers).

Again, the main driver of the successful AEA and AEM design process was sharing and inclusion in the participatory process of the different knowledge bases (e.g., local, scientific, policy). In the cases where this occurred, it was driven by shifts in the stakeholder roles that were allowed by the formal AEA arrangement.

This role shifting might have occurred in other contexts too; e.g., for the "Bordeproject Lower Saxony" case study (Prager and Freese, 2009), the "dialogical tools" case study (Toderi et al., 2007), the "collaborative management in Sri Lankan shrimp aquaculture" case study (Galappaththi and Berkes, 2015), and the "pastoral systems in northern Nicaragua" case study (Tarrasón et al., 2016). However, in all of these examples, the authors never directly referred to any shift in the stakeholder roles in the system.

This shift in the stakeholder roles might be seen as a signal of empowerment of some stakeholders (e.g., farmers) in a co-management process. However, with a more general vision, and also including other stakeholders in the analysis (e.g., the policy makers), this can be seen as a feature of a co-management process.

The design process will be faster in systems where there is already spontaneous co-management, and slower in situations where it is necessary to initiate co-management from the start (i.e., in the cases here of the BIO AEAs, where we observed tentative shifts in the roles). However, this is likely to run into the time constrains of the RDP (as emerged in the other BIO AEAs here), as these processes are time consuming.

5. Conclusions

The analysis of these case studies has highlighted the key elements that are needed to create site-specific and/or landscape AEMs within an RDP that will have high levels of acceptance among the stakeholders: (i) stakeholder involvement must take place from the very beginning of the design process of the AEAs and/or the AEMs; (ii) stakeholder involvement must take place at each phase of the process, to avoid changes in the AEMs without sharing of the reasons for these changes with the stakeholders; (iii) a predefined "rulebook" must not be imposed, as the stakeholders must be allowed to design their own measures to create site-specific AEMs with a landscape dimension; (iv) the AEA legal arrangement must allow the analysis to be focussed on specific local conditions, and lead to the emergence of the "landscape dimension" of the environmental issues addressed. In this process, a shift in the stakeholder roles in the system is required from the very begining, because this favours the flow of local knowledge in the design of the AEMs. This shift in the stakeholder roles in the system is a feature of co-management.

However, the creation of room within RDPs for bottom-up and stakeholder actions might be lost by the long and complex EU bureaucratic procedures for the implementation of what emerges from the stakeholder involvement. If this problem is of minor importance when spontaneous co-management processes are already present, this can create time constraints for the expenditure of funds within the RDP period, which is a relevant issue that concerns policy makers.

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General conclusions

The agro-pastoral farming system in central Italy are strongly influenced by the climate and landscape characteristics. Dry summer, wet winter and high inter-annual rainfall variability characterize Mediterranean climate. The high variability of the climatic conditions of the Mediterranean area lead to a great intra-annual variation of grassland production (Carmona et al., 2013). One of the best uses of the available resource following the seasonal change of grassland production is pursued through the transhumance system or the sedentary systems, both in highlands (summer grazing on permanent grasslands) but also in lowlands (when the common forage crops are grazed throughout the year). Despite the crisis in the livestock system since the middle of the last century, such breeding system is still active in Marche region.

In line with the framework of the MA (Alcamo *et al.*, 2003), this research highlighted the importance to analyse the ES trade-off and interactions, involving the stakeholders to integrate different knowledge in the design process. From this analysis emerged that among the ES, the cultural ES were poorly studied despite their important role for local and general stakeholders. Also, from this analysis emerged the need to adopt a multisectoral approach on the analysis of the ES provided by grazing system. This multisectoral analysis of the ES allows the inclusion of different stakeholders and their knowledge, that can improve the effectiveness of local decision-making process. The creation of tools (e.g., mathematical models, indicators, future scenarios) and frameworks to support the decision-making process can be adopted in order to facilitate discussion between stakeholders and lead to the choice of the most appropriate management methods.

The analysis of some relevant ES provided by one of the most wider mountain permanent grasslands of the central Apennine shows that intensification of management does not affect the soil respiration on short-term perspective. If these results will be confirmed in long-term and with the aim of grasslands conservation is suggested to avoid applying too restrictive measures for grassland management (e.g., utilisation calendars), rather introducing innovations grassland management with the stakeholders' involvement.

The research focused on the analysis of ES provided by a long-term alfalfa used by transhumant farms and included in the winter cereal-based rotation provided interesting and novel insights to mitigate GHG emissions. Despite the relevance of the legume perennial crop, their termination in terms of N_2O emissions is little studied in the Mediterranean environment. Postponing of the main tillage from summer to early autumn, with low soil temperature and humidity, provides unfavourable denitrification conditions with potential lower N_2O emissions after tillage. The alfalfa termination resulted in an initial high residues mineralization with consequent N_2O emissions due to the asynchrony between N realised and N demand by the subsequent crop. The typical alfalfa-wheat rotation however did not result in different cumulative N_2O emissions between alfalfa and wheat. The biochar mitigation strategies did not show effects in this first year of application nor in terms of N_2O emissions mitigation nor on wheat yield, probably partly due to the specific soil characteristics. It is conceivable that mitigation and yield promotion effects may emerge in the second or third year of experimentation.

Results can be used in the participatory process for the definition of site-specific and/or landscape AECMs with a high level of acceptance among the stakeholders.

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