Efectos del uso y del manejo del suelo sobre la erosión y la generación de escorrentía en pastos permanentes en Europa

Land use and management effects on erosion and runoff generation in permanent grassland in Europe.

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Córdoba, Julio 2023

UNIVERSIDAD DE CÓRDOBA

PROGRAMA:

DINÁMICA DE FLUJOS BIOGEOQUÍMICOS Y SU APLICACIÓN

TITULO: *Land use and management effects on erosion and runoff generation in permanent grassland in Europe*

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DOCTORANDA/O

Filippo Milazzo

TÍTULO DE LA TESIS:

Efectos del uso y del manejo del suelo sobre la erosión y la generación de escorrentía en pastos

INFORME RAZONADO DE LAS/LOS DIRECTORAS/ES DE LA TESIS
(se hará mención a la evolución y desarrollo de la tesis, así como a trabajos y publicaciones derivados de la misma)

El doctorando ha completado la tesis dentro del plazo establecido y ha ido compliendo los hitos en cada evaluación anual. La
calidad final de la tesis es excelente, como demuestran las siguientes tres publicaciones derivadas de ellas, todos con factor de
impacto:
1) Milazzo, F., Francksen, R. M., Zavattaro, L., Abdalla, M., Hejduk, S., Enri, S. R., Pitarrello, M., Newell-Price, P., Schils, R., Smith,
P., Vanwalleghem, T. (2023). The role of grassland for erosion and flood mitigation in Europe: A meta-analysis. Agriculture.
Ecosystems & Environment, 348, 108443QQq.
Q1 “Agriculture”. IF 6.57.
2) Milazzo, F., Francksen, R. M., Abdalla, M., Ravetto Enri, S., Zavattaro, L., Pittarello, M., & Vanwalleghem, T. (2023). An
Overview of Permanent Grassland Grazing Management Practices and the Impacts on Principal Soil Quality Indicators.
Agronomy, 13(5), 1366.
Q1 “Agriculture”. IF 3.95.
3) Milazzo, F., Fernández, P., Peña, A., & Vanwalleghem, T. (2022). The resilience of soil erosion rates under historical land use
change in agroecosystems of Southern Spain. Science of The Total Environment, 822, 153672.
Q1 “ Environmental, Multidisciplinary”. IF 10.75.
Adicionalmente, el doctorando es primer autor de una cuarta publicación, que actualmente está en revisión y que forma parte
de esta tesis y de una quinta publicación como co-autor, que no aparece dentro de esta tesis.
Estas publicaciones son fruto de un trabajo de investigación de alta calidad y el candidato ha demostrado su alta capacidad de
análisis y capacidad investigadora. El doctorando también ha completado esta tesis con menión internacional, ya que ha
realizado una estancia de 3 meses en el grupo del Dr. Luca Broccaen el CNR-IRPI, Italia. Ha realizado presentaciones en varios
congresos internacionales y ha asistido a reuniones del proyecto europeo Super-G en el marco del cual se ha desarrollado esta
tesis.
Por todo ello, ha adquirido las competencias necesarias para la obtención del grado de Doctor

Por todo ello, se autoriza la presentación de la tesis doctoral.

Córdoba, a ____ de ____ de ____

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Nombre y Apellidos

Lista de publicaciones originales

Esta tesis se presenta en forma de compendio de publicaciones, de acuerdo con el artículo 53 Reglamento 57/2020 de Consejo de Gobierno, con fecha de publicación el 1 de diciembre de 2020 en BOUCO, y regulado a través del Real Decreto 99/2011, de 28 de enero (modificado por el RD 534/2013; por el RD 43/2015 y por el RD 195/2016), que recogen las enseñanzas oficiales de los estudios de Doctorado. Dichas publicaciones recogen los resultados que han sido obtenidos en los diferentes trabajos de investigación desarrollados con el fin de alcanzar el objetivo fijado para la realización de la tesis. A continuación, se listan las referidas publicaciones (3) que constituyen los capitulos de la presente tesis.

This thesis is presented in the form of a compilation of publications, in accordance with Article 53 of Regulation 57/2020 of the Government Council, published on December 1, 2020, in BOUCO (Official Gazette of the University of Córdoba), and regulated through Royal Decree 99/2011, dated January 28 (amended by RD 534/2013, RD 43/2015, and RD 195/2016), which establish the official requirements for doctoral studies. These publications contain the results obtained from different research works carried out in order to achieve the set objective for the completion of the thesis. The following (3) publications are listed as chapters in this thesis.

1) Milazzo, F., Francksen, R. M., Zavattaro, L., Abdalla, M., Hejduk, S., Enri, S. R., ... & Vanwalleghem, T. (2023). **The role of grassland for erosion and flood mitigation in Europe: A meta-analysis. Agriculture**. Ecosystems & Environment, 348, 108443QQq.
Agriculture, Ecosystems & Environment, Q1 "Agriculture". IF 6.57.

2) Milazzo, F., Francksen, R. M., Abdalla, M., Ravetto Enri, S., Zavattaro, L., Pittarello, M., ... & Vanwalleghem, T. (2023). **An Overview of Permanent Grassland Grazing Management Practices and the Impacts on Principal Soil Quality Indicators**. Agronomy, 13(5), 1366.
Agronomy, Q1 "Agriculture". IF 3.95.

3) Milazzo, F., Fernández, P., Peña, A., & Vanwalleghem, T. (2022). **The resilience of soil erosion rates under historical land use change in agroecosystems of Southern Spain**. Science of The Total Environment, 822, 153672.
Science of The Total Environment, Q1 " Environmental, Multidisciplinary". IF 10.75.

Actualmente, el cuarto capítulo titulado "**NDVI Prediction of Mediterranean Permanent Grassland Using Soil Moisture Products**" se encuentra en proceso de revisión en IEEE Transactions on Geoscience and Remote Sensing. Esta revista es altamente reconocida en el campo de la ciencia ambiental y la teledetección, con un factor de impacto de 8.12.

Currently, the fourth chapter titled "**NDVI Prediction of Mediterranean Permanent Grassland Using Soil Moisture Products**" is undergoing the review process in IEEE Transactions on Geoscience and Remote Sensing. This journal is highly regarded in the field of environmental science and remote sensing, with an impact factor of 8.12.

Abstract

Grassland ecosystems are widely recognized for their ability to reduce runoff and mitigate erosion and flooding compared to arable land and forest ecosystems. However, significant knowledge gaps exist regarding the quantitative assessment of grassland's role in erosion and flooding mitigation. This research activity addresses these gaps by evaluating the flood and erosion mitigation potential of grassland at a European scale, and comparing it with arable land and forest land.

We begin by highlighting the main erosion processes in grassland and identifying key knowledge gaps in terms of modeling, quantification, and conservation strategies. Moreover, we emphasize the importance of understanding the factors contributing to grassland degradation, including climate change, land use change, and inappropriate management practices that are currently threatening permanent grasslands worldwide.

Next, we provide a comprehensive overview of sustainable grazing management practices across Europe that aim to protect grassland soils. We discuss the potential benefits of these management strategies in maintaining the integrity of grassland ecosystems and their role in erosion prevention.

Furthermore, we investigate the impact of land use change on erosion generation in grassland and assess the potential risks of mitigation at a regional scale. Finally, by developing a novel model based on machine learning techniques and remote sensing data, we forecast the degradation of grassland vegetation status and assess their implications for sheet and rill erosion.

Overall, this research contributes to a better understanding of the quantitative role of grassland in erosion and flooding mitigation at a European scale. The findings highlight the urgent need for sustainable management practices to protect and conserve grassland ecosystems and emphasize the potential of innovative modeling approaches for erosion prevention and management.

Resumen

La investigación aborda las brechas existentes en cuanto a la evaluación cuantitativa del papel de los pastos permanente en la mitigación de la erosión y las inundaciones, centrándose en los ecosistemas de pasto permanente en comparación con zonas de cultivo y zonas forestales. El estudio comienza identificando los principales procesos de erosión en los pastos y destacando las lagunas de conocimiento en términos de modelado, cuantificación y estrategias de conservación y prevención.

Además, se subraya la importancia de comprender los factores que contribuyen a la degradación de los pastos permanentes, como el cambio climático, el cambio en el uso de suelo y las prácticas de manejo inadecuadas. Estos factores representan una amenaza para los pastizales permanentes en todo el mundo.

La investigación también proporciona una visión general exhaustiva de las prácticas de manejo sostenible del pastoreo en toda Europa, con el objetivo de proteger los suelos de los pastos permanentes. Se discuten los beneficios potenciales de estas estrategias de manejo en la preservación de la integridad de los ecosistemas de pastizales y su papel en la prevención de la erosión.

Además, se investiga el impacto del cambio en el uso de suelo en la generación de erosión en los pastizales y se evalúan los riesgos potenciales de mitigación a escala regional. Esto proporciona una comprensión más completa de cómo los cambios en el uso de suelo pueden afectar la erosión en la zona de pasto y permite identificar medidas preventivas.

Por último, se desarrolla un modelo novedoso basado en técnicas de aprendizaje automático y datos de teledetección para pronosticar la degradación de las capas de vegetación de los pastizales y evaluar sus implicaciones en la erosión superficial. Este enfoque innovador en el modelado permite una mejor comprensión de los procesos de erosión y ofrece herramientas para su prevención y manejo.

En general, esta investigación contribuye a una mejor comprensión del papel cuantitativo de los pastos permanentes en la mitigación de la erosión y las inundaciones a escala europea. Los hallazgos resaltan la necesidad urgente de prácticas de manejo sostenible para proteger y conservar los ecosistemas de pastos, y enfatizan el potencial de enfoques de modelado innovadores para la prevención y el manejo de la erosión.

Keyword: Grassland, erosion, flooding, Dehesa

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Chapter 1: introduction

Permanent grasslands are one of the most important ecosystems on our planet, providing a range of ecosystem services, such as carbon storage, nutrient cycling, and habitat for a wide variety of plant and animal species (Schils et al., 2022). At a global scale, grasslands cover approximately 60% of the Earth's land surface, making them one of the largest biomes on our planet. They are found in a variety of climatic zones, from the tropical savannas of Africa to the temperate permanent grassland of North America and Eurasia (Zhao et al., 2020). In addition, permanent grasslands play a crucial role in mitigating climate change by storing carbon in their soils and vegetation, they also help to prevent soil erosion and promote soil fertility, making them essential for sustainable land management (Abdalla et al., 2018; Zhu et al., 2015). For all these reasons permanent grassland ecosystems are vital for supporting biodiversity, providing ecosystem services, and sustaining human livelihoods and cultural heritage. The European Union classifies grassland types based on fodder age and rotation. Permanent grassland is defined as land used to grow herbaceous forage, either self-seeded or sown, that has not been included in the crop rotation for at least five years, while temporary grassland is land used to grow herbaceous forage as part of a crop rotation (European Commission, 2007). In Europe, permanent grassland covers around 30% of the agricultural area, making them one of the most widespread habitats on the continent (EUROSTAT, 2020), supporting animal production, carbon and nitrogen cycling (Hejcman et al., 2013). In the Mediterranean region, permanent grasslands are mainly present in the Iberian Peninsula, covering an area of about 5.5 million hectares. The Iberian permanent pastures are Savanna-like systems, classified as agro-silvopastoral systems (Maranon, 1988). In fact, at least 50% of the farm surface is occupied by permanent grassland with dispersed oak trees (Quercus ilex) covering between 5% and 60% of rangeland surface (Campos et al., 2010). Regionally and globally, permanent grasslands are widely recognised for they ability to improve the physical condition of the soil by promoting water infiltration, enhancing an important ecosystem service such as mitigating erosion and flooding. Indeed, in comparison with croplands, the grassland low intensity soil management promote the structural improvement of the top soil layer, which is reflected in the decrease of bulk density and the increase of hydraulic conductivity, organic matter, porosity, and thus water infiltration followed by a reduction of sheet and rill erosion by runoff (Marshall et al., 2009; Panagos et al., 2015). Anthropogenic impacts, in terms of land use and management, is the main cause of soil degradation and erosion. Permanent grasslands, like forest lands, are globally recognized for their low erosion ratio due to high vegetation cover and low management intensity (Borrelli et al., 2020). However, it would be mistaken to think that European permanent grasslands are exempt from the risk of soil degradation. Indeed, it is essential to acknowledge their ecosystem service of erosion and flood reduction and their significant role at a large scale. However, it is also necessary to quantitatively measure their degradation processes, their soil quality, and potentially reward farmers for their service delivered to the community. Currently, there are three major causes that have led, and continue to lead, to the erosion of permanent grasslands in Europe and globally: climate change, management intensification and land-use change. Climate change is affecting the health and existence of all terrestrial ecosystems, including permeant grasslands. Several studies have estimated the negative effect of the climate change on grasslands and their provision of ecosystem services i.e. biomass production, loss of biodiversity, greenhouse gas emissions, pollination. Long periods of drought followed by intense rainfall and extreme climatic events are causing exceptional erosion phenomena, which permanently degrade permanent grasslands soil. In fact, globally 49.25% of grassland ecosystems are affected by degradation and nearly 5% of these grassland areas ranged from severely degraded to extremely degraded (Gang et al., 2014). Climate change is the main cause, leading to 45% of the deterioration compared to 32% due to human activities (Gang et al., 2014). On the other hand, anthropogenic impacts can derive from a land use change due to economical and political reasons, or from unsuitable management. In Europe after the World War II, the increment of food demand drove an intensification of management promoting a land use conversion to cropland or to annual grassland cultivation. Most remote areas such as wetlands and hilly zones were abandoned, facilitating the transition to bushland and forest areas (Habel et al., 2013). However, at the end of the last century, an inverse movement of land use change occurred in Europe (especially in the East and Mediterranean countries), from cropland to permanent grassland due to land abandonment, leading to an increase of meadows and pastures by 6.5% (Kuemmerle et al., 2016). Conversely, intensive management decreases grassland multifunctionality and ecosystem services (Schils et al., 2022). Grassland management involves different practises and aspects of the grassland cultivation, for example cutting management, nutrient control, weed control and grazing management. In this thesis, only the grazing grassland management will be explored. Grazing intensity, which is characterized by high spatial heterogeneity, is an important parameter for accurately representing human disturbance and its impact on grassland soil (Akiyama and Kawamura, 2007). Indeed, intensive grazing management can irreversibly erode soil endorsing gully, piping, landslide, or other soil erosion promoting processes such as soil compaction by trampling. The mentioned soil erosion processes are the result of a loss of soil quality. Soil quality is a broad concept, which is still under discussion, and is defined by a balance of chemical, biological and physical soil properties to guarantee the environmental sustainability without overlooking yield (Doran and Parkin, 1994; Harris et al., 2022). The monitoring of soil quality, through the periodic assessment of soil quality indicators (chemical, biological and physical soil properties) is a widespread practise to evaluate the sustainability of soil management. Above ground management, such as livestock grazing, influence directly soil quality, in terms of chemical (i.e soil organic matter, pollutants and nutrients), biological (i.e microbes biomass, microbes activity and soil biodiversity) and physical properties (i.e bulk density, aggregates, structure, and hydraulic conductivity) (Abdalla et al., 2018; Aubault et al., 2015; Blanco-Canqui et al., 2016). While unsuitable grazing management, such as overgrazing, degrades the grass-layer dynamic and soil quality, viceversa, a sustainable grazing management promotes above and below-ground quality (Bilotta et al., 2007; Byrnes et al., 2018). The grazing management effect on soil quality can be evaluated with different methods, indeed it can be assessed by laboratory analyses, sampling soil on the field, by visual soil quality framework (VSA) and by remote sense application. Laboratory analysis are usually complicated and time and cost consuming (Yu et al., 2018), VSA is a rapid and cheap method that requires an expert evaluation and it is often related to soil structure and related properties (Cui et al., 2014), while remote sense applications may be expensive but it allows to have continuous monitoring of management and grass layer status (Xu and Guo, 2015). For the reasons mentioned above, the integrity of the grass-layer is essential to control soil erosion and flooding. Thus, a not degraded and uniform grass layer implies a sustainable grassland management and a good hydraulic soil quality. Therefore, remotely sensed vegetation indexes are used as a proxy of grassland degradation. One of the most used vegetation index is the Normalized Difference Vegetation Index (NDVI), that is a indicator of the greenness. NDVI has been used for several purposes in grassland erosion and flooding assessment, for example, Ma et al. (2019) and Zhao et al. (2022) used the historical NDVI data to assess the degradation effect of grazing intensity at large scale, while Sternberg et al. (2011) and Chen et al. (2021) tracked the grassland erosion and sustainability in China. NDVI is also used to monitor grass layer development and several scholars attempted to forecast grass biomass development using NDVI (Jianlong et al., 1998; Wang et al., 2005). The forecasting of NDVI can be a useful tool for farmers to predict grassland degradation in advance and apply mitigation measures in time. The grassland development it is often related to climate conditions, such as rainfall events and temperature, which can be assessed by analysing the NDVI evolution. In Ireland, grassland development is well predicted using weather forecasts relating grass growth to rainfall (McDonnell et al., 2019). In Mediterranean climate, where rainfall is not homogeneously spread along the year, the relationship between rainfall and grass development is not very clear. Indeed, the dynamics of vegetation are closely linked to soil moisture, as permanent grasslands effectively utilize subsurface water resources during the dry season (García-Gamero et al., 2021). Thus, the possibility of relating soil moisture to the phenological dynamics of permanent grassland, is a clear indication of how soil moisture can be used as an environmental predictor in the qualitative assessment of vegetation status, serving as a tool for sustainable grassland management.

In the light of the previous discussion, the general objective of this thesis is to improve the knowledge on the erosion and flooding mitigation potential of permanent grasslands, and to quantify the importance of land use and management changes. This has implications for the quantification of ecosystem services in permanent grasslands, as a part of the H2020 project “Developing SUstainable PERmanent Grassland systems and policies” (Super-G), in which this research was developed. The main objective of this research activity is to gain a comprehensive understanding of the role of permanent grassland in mitigating the risk of soil erosion and floods, while considering their future role based on climate change, from regional to European scale. To achieve this objective, the following specific goals investigated are:

I. Quantification of the importance of permanent pastures in mitigating soil erosion and floods at a European scale through a meta-analysis of published European studies, comparing them with agricultural and forest land. Identification of knowledge gaps in quantifying soil degradation in permanent grasslands.
II. Identification of grazing management practices to mitigate erosion and flooding phenomena in Europe.
III. Quantification of the historical importance of permanent grassland in soil erosion control at the regional level in Andalusia.
IV. Development of a management support model for farmers to mitigate erosion and flooding issues using new technologies such as remote sensing and machine learning.

To reach these specific objectives, the thesis is structured in the following four chapters, as it is shown in Fig.1:

The first chapter quantifies the role of permanent grassland on erosion and flood mitigation at the European level, in comparison with arable land and forestry. Furthermore, the major erosion and flooding phenomena and their promoting processes will be qualitatively analysed, discussing the current knowledge gaps inherent to the study of erosion processes in grasslands. This is carried out through a systematic review and the application of meta-analysis methods on European scientific publications (from 1980 to 2018) The quantification of the erosion and flooding rate on permanent grassland, is based on the meta-analysis of 4 indicators, such as hydraulic conductivity, bulk density, runoff generation and sheet and rill erosion rate. Additionally, a literature review assessed qualitatively the current state of research on erosion and flooding processes in permanent grassland in Europe.

In the second chapter, the focus was on the assessment of Soil Quality Indicators as a monitoring measure for the soil quality status of permanent grassland, aiming to provide valuable information to preserve permanent grassland in Europe. The importance of Sustainable Grazing Management was also examined, specifically regarding its effect on soil quality. Additionally, the significance of new plant species capable of countering the effects of climate change was reviewed.

The third chapter studies the effect of land use change on erosion. This study is centred in Andalusia, encompassing its full extent (8 million hectares), modelling the rate of erosion produced based on land use change from 1956 to 2018. The model approach is based on modelling the Cover management factor of RUSLE model, which is representative of the land use impact on soil erosion. Furthermore, the erosion rate, based on forecasting land use scenarios provides an effecting assessment of the soil erosion and flooding mitigation role of permanent grassland in Andalusia.

Lastly, a useful model has been developed for farmers to predict potential degradation of the grass layer due to drought, thereby preventing erosive phenomena caused by the exposure of bare soil to weather agents. This model is based on the observation of soil moisture. Through machine learning methods, two prediction models have been developed: one for 7 days and another for 30 days, using the Normalized Difference Vegetation Index (NDVI) as a proxy for plant development. The soil moisture data used in the models were collected in the field through TDR sensors and via satellite-based soil moisture observations from the Copernicus Metop ASCAT project. This study was conducted at the farm level, taking the Santa Clotilde Dehesa farm located in the province of Cordoba as a case study, demonstrating a possible application of this methodology and addressing the study of implementing these models on a larger scale.

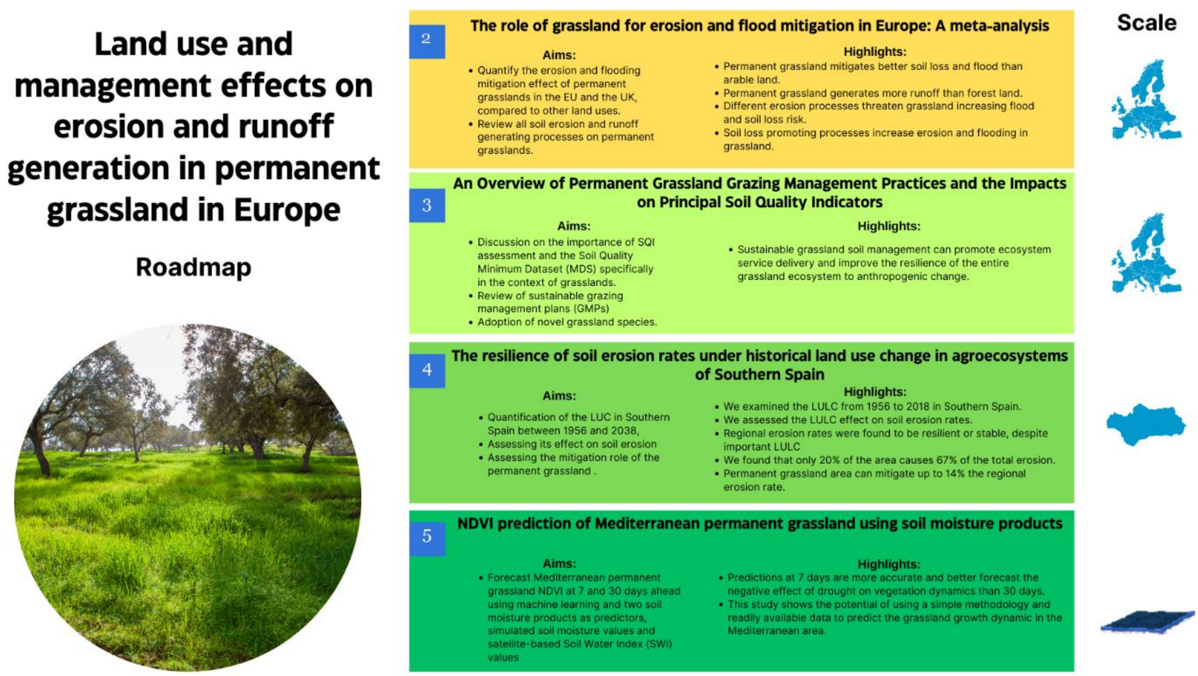

Fig. 1: Structure of the thesis, with a summary of aims and highlights per chapter. The considered scale ranges from EU (Chapter 2 & 3), regional (Chapter 4) to farm-field (Chapter 5)

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Chapter 2: The role of grassland for erosion and flood mitigation in Europe: a meta-analysis

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Abstract

Permanent grasslands are widely recognized for their role in protecting the landscape against soil erosion and flooding. However, this role has not yet been comprehensively quantified. Also, the degradation of grasslands is accelerating at an alarming pace, leading to erosion and runoff generation. This study aims to (i) quantify the erosion and flooding mitigation effect of permanent grasslands in the EU and the UK, compared to other land uses; (ii) review all soil erosion and runoff generating processes on permanent grasslands. First, a meta-analysis compared four erosion and flooding-related indicators: bulk density, hydraulic conductivity, runoff and soil loss between permanent grasslands, arable land and forests. The results show that permanent grassland soils had generally lower bulk density and higher hydraulic conductivity than arable soils, and generated less runoff and soil loss. Differences are less clear-cut in comparison with forests, although permanent grasslands had higher bulk density and runoff values. Secondly, a qualitative, in-depth review was performed to identify knowledge gaps related to the characteristics, importance and driving factors behind relevant soil erosion processes affecting grasslands in the EU. This identified six processes with appreciable knowledge gaps: trampling-induced erosion, gullying, piping, landsliding, snowmelt erosion, and avalanche erosion. Additionally, three processes were identified that promote runoff generation and soil erosion: compaction, hydrophobicity and wildfires.

Key words: Grassland, Erosion, Ecosystem services, Flooding

1. Introduction

Climate change, land use change and management intensification all increase the vulnerability of European soils to increased runoff, flooding and soil erosion. Both flooding and soil erosion are projected to increase under future climate change in the EU. Alfieri et al. (2015) project a 220% increase in flood risk in Europe by 2080. Panagos et al. (2021) predicted an increase of 13-22.5% in soil loss for the EU and UK by 2050 due to water erosion. Both processes are closely linked, and mitigation measures require policy measures that promote soil conservation, and land use planning policies promoting land uses with high soil water holding capacity, low runoff generation potential, high vegetation cover and erosion resistance. Grasslands have an enormous potential to make our landscapes more resilient to floods and erosion (Bengtsson et al., 2019; Hussain et al., 2021; Yang et al., 2021), while contributing to the production of forage and other ecosystem services (Schils et al., 2022). Grasslands cover more than 30% of the earth's terrestrial surface, more than double the surface of cropland (Lemaire et al., 2011), and 35% of the European (EU-28) agricultural area (EUROSTAT, 2020). The European Union differentiates grassland type based on the age of the fodder and rotation. Permanent grassland is defined as land used to produce herbaceous forage, self-seeded or sown, not included in the crop rotation for at least five years. Whereas, temporary grassland is land used to grow herbaceous forage included in the crop rotation (European Commission, 2007). In uplands, both permanent and temporary grassland reduce soil erosion, surface runoff and downstream flooding (Macleod et al., 2013). In lowlands, grasslands are capable of withstanding flooding better than other land uses and promote water infiltration (Strock et al., 2022). However, recent studies have shown that significant soil erosion can occur (Hancock et al., 2015) and that while erosion on well-conserved permanent grassland is generally low, these are increasingly under threat of intensification. Globally, 49% of grasslands has been degraded to some extent, and this process is accelerating in many parts of the world (Bardgett et al., 2021). Degraded grasslands are subject to severe erosion and runoff generation. In mountainous areas, such as the Swiss alpine uplands, water erosion can be severe and varies from 0.14 to 1.25 t $ha^{-1}month^{-1}$ depending on the phenological stage of grasses (Schmidt et al., 2019). Other processes, such as landslides or trail erosion contribute to sediment production (Zweifel et al., 2019) and have received little attention. In dryland regions, degradation and abandonment leads to increased woody plant encroachment and fire risk, which in turn exposes bare soil, increasing soil loss by 60% (Johansen et al., 2001) and creating feedback loops that accelerate degradation.

Panagos et al.(2020) reported that 25% of European soils have erosion rates higher than the sustainable threshold (2 t ha^{-1} yr^{-1}) and 6% of agricultural land exceeds 11 t ha^{-1} yr^{-1}. These areas are mostly under cropland and permanent crops, while grassland and forests have a lower impact on erosion generation (Cerdan et al., 2010; Panagos et al., 2015, 2021). However, widespread agricultural intensification, either by grassland conversion or management intensification, inevitably leads to increases in soil erosion. Therefore, it is important to quantify the erosion and flooding mitigation potential of permanent grassland compared to other land uses. This will aid evaluation of the impact of policies designed to influence land use and maintenance of permanent grassland, such as the Eco-schemes proposed under the new Common Agricultural Policy (CAP) 2023-2027 (European Commission, 2021a). It is also necessary to better understand the main soil erosion and runoff processes under intensified permanent grassland. Much of our knowledge is limited to sheet and rill erosion, but it is necessary to look beyond the processes that can be modelled using RUSLE (Quine and Van Oost, 2020), such as gullies, landslides or, in the case of grasslands, trampling or trail erosion due to overgrazing. In this study we aim to present a comprehensive overview of soil erosion and flooding issues that affect European permanent grassland by performing: (i) a quantitative meta-analysis of the soil erosion and flooding mitigation role of permanent grassland (ii) and a qualitative evaluation of additional erosion and flooding-related processes that threat permanent grassland in Europe.

2. Materials and methods

We quantified the role of permanent grassland in erosion and flood mitigation in contrast with arable land and forest land by performing a meta-analysis. that focusses on four indicators: bulk density, hydraulic conductivity, runoff and soil loss. These four indicators were selected for two reasons: (i) because they are widely acknowledged to be well related to runoff and erosion generating processes, and (ii) because they are widely used in literature and enough studies are available that report on the land use contrasts studied here. Bulk density is widely considered an important soil quality indicator that reflects the soil structure and soil compaction, and is directly related to other soil quality parameters such as soil porosity (Hernanz et al., 2000; Topa et al., 2021). Hydraulic conductivity is an important property in natural flood management for the understanding of the surface permeability of soil with the view of increasing rainfall infiltration and runoff reduction (Bens et al., 2007; Marshall et al., 2009; Talsma, 1987). Runoff and soil loss considered in this study are direct measures of the amount of water and soil loss and are assessed at field scale by using runoff plots. Although the relation between plot and catchment scale is complex, both indicators are well suited for comparing the response of land use or management types (Maetens et al., 2012). To evaluate the additional erosion and flooding risk in permanent grassland, we review the main soil degradation processes and the related promoting processes that foster erosion and flooding.

2.1. Search strategy

In the end of 2019, a systematic literature search was performed to identify studies reporting on the effect of grasslands on soil erosion and flooding. The literature was screened based on the criterion that a selected set of indicators were reported in a land use contrast: either permanent grassland-arable land, or permanent grassland-

forest. The selected indicators are: (1) hydraulic conductivity (mm h^{-1}); (2) bulk density (g cm^{-3}), (3) runoff (mm); and (4) soil loss (t ha^{-1}), (Tables S1, Appendix 1).
The search was limited to articles published from 1980 to 2018, and within the Europe-27, including also the EU-27 neighbourhood countries such as United Kingdom, Albania, Belarus, Bosnia Herzegovina, Kosovo, Macedonia, Moldova, Montenegro, Norway, Serbia, Switzerland and Ukraine. The research was conducted in Scopus and CAB abstract, using a keyword string aiming to collect the wider radius of scientific papers regarding soil degradation issues in permanent grassland land, as described in Schils et al.(2022) (Table S2, Appendix 1).

2.2. Data extraction and inclusion criteria

The screening process was implemented using "EPPI reviewer 4 tool" (http://eppi.ioe.ac.uk/cms/). Valid data sampled by the full text screening were extracted and transcribed in MS Excel form, creating a database of the number of field assessments, mean value, and standard deviation. In this first step of the systematic search, 14203 articles were collected, of which 3150 articles were removed due to duplicates, leaving a net total of 11053. A second screening process was then carried out: by title, by abstract and by full text. Exclusion criteria were set retaining only papers in English language that report on results of field experiments or measurements, rejecting model studies and reviews. At the end of the screening process, only 24 scientific papers were included in the meta-analysis. The full selection process is shown in Table1.

2.3. Reviewer bias

The processes of screening and data extraction were carried out by experts, consisting of a head-reviewer and two co-reviewers. The assessment of the head-reviewer was used as a benchmark against which the co-reviewers' decisions were compared. At least 5% of papers were double-screened to assess the rate of discrepancy between the head-reviewer and the co-reviewer's decision, identifying the "false exclusion rate". If the false exclusion rate was higher than 10%, the processes were discussed, and the issue was adjusted.

2.4. Weighted Meta-analysis

The extracted data were analysed using the logarithm response ratio weighted meta-analysis approach (Hedges et al., 1999). For every single entry, the effect of land use on the selected contrast was assessed as the natural logarithm response ratio (LnRR) of the mean of the contrasting land uses.

$$\mathrm{LnRR} = \ln \frac{\overline{\mathrm{X_x}}}{\overline{\mathrm{X_{PG}}}}$$

Where the LnRR is the natural log of the mean of forest or arable groups ($\overline{\mathrm{X_x}}$) against the mean of the permanent grassland group ($\overline{\mathrm{X_{PG}}}$)

The variance for each group was calculated as:

$$\mathrm{Var} = \frac{\mathrm{SD_x^2}}{\mathrm{N_x}\overline{\mathrm{X_x}}} + \frac{\mathrm{SD_{PG}^2}}{\mathrm{N_{PG}}\overline{\mathrm{X_{PG}}}}$$

Where SDx is the standard deviation of forest or arable groups, SDPG is the standard deviation of permanent grassland group; Nx is the sample size of forest or arable group, and NPG is the sample size of permanent grassland groups.

A random-effects model (RE) was fitted to the data. The amount of heterogeneity was estimated using the restricted maximum-likelihood estimator (Viechtbauer, 2005). The studentized residuals and Cook's distances are used to examine whether studies may be outliers and/or influential in the context of the model (Viechtbauer and Cheung, 2010). The analysis was carried out using R (version 4.1.2) (R Core Team, 2020) and the metafor package (version 3.0.2) (Viechtbauer, 2022).

3. Results and discussion

3.1. Weighted Meta-analysis

Meta-analysis results are shown in Figures 1 and 2 for the comparison of permanent grassland with arable land and forestry land respectively. In contrast to arable land, there were no significant differences in bulk density (RE [95%CI] = 1.17[-0.07; 0.40], n = 9 studies yielding). An examination of the studentized residuals revealed that the study of Pardini et al. (2017) had a value larger than 2.77 and may be an outlier in the context of this model. Therefore, 44% of the entries reported positive response rates that were significantly above 0, while 33% of entries reported a ratio of mean higher than 0but do not show significant differences. According to the Cook's distances, two studies (Nunes et al., 2011; Pardini et al., 2017) could be considered overly influential. This surprising result is probably related to the evolution of bulk density after tillage in relation to when the measurements were taken. Soil bulk density decreases with every tillage operation, but then changes very fast. Osunbitan et al. (2005) reported an increase in bulk density of up to 61% in only 8 weeks after tillage. Alletto and Coquet (2009) reported a similar increase in bulk density in a study in France. Since none of the included studies evaluated the temporal evolution of soil properties, nor details of the time of sampling and the time passed since the last tillage operation, this could easily explain some of the non-significant and negative entries.

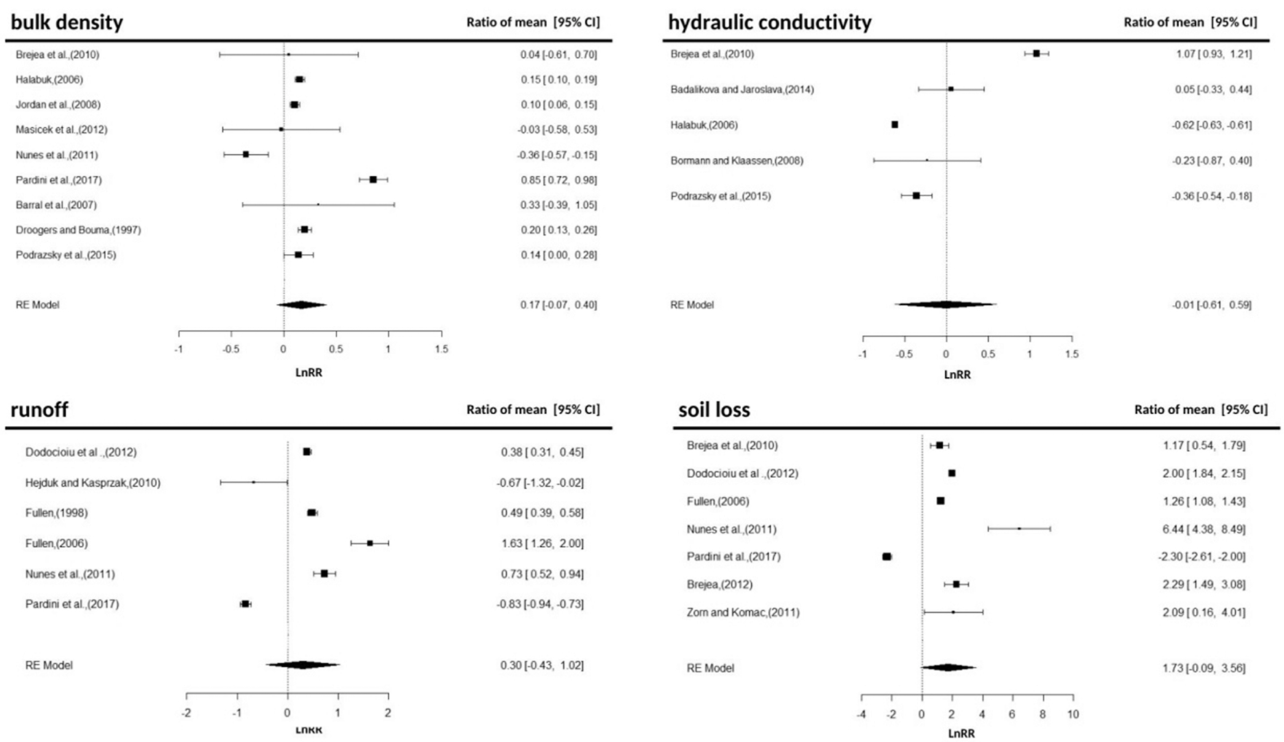

Figure. 1. Weighted mean effect (log response ratio, LnRR) and 95% confidence interval of permanent grassland vs. arable land on bulk density, hydraulic conductivity, runoff, and soil loss. A LnRR> 0 indicates a higher value of the indicator under arable land, while LnRR< 0 indicates a lower value under arable land, compared to permanent grassland. Effects are significant (P ≤ 0.05) where confidence intervals do not intercept 0.

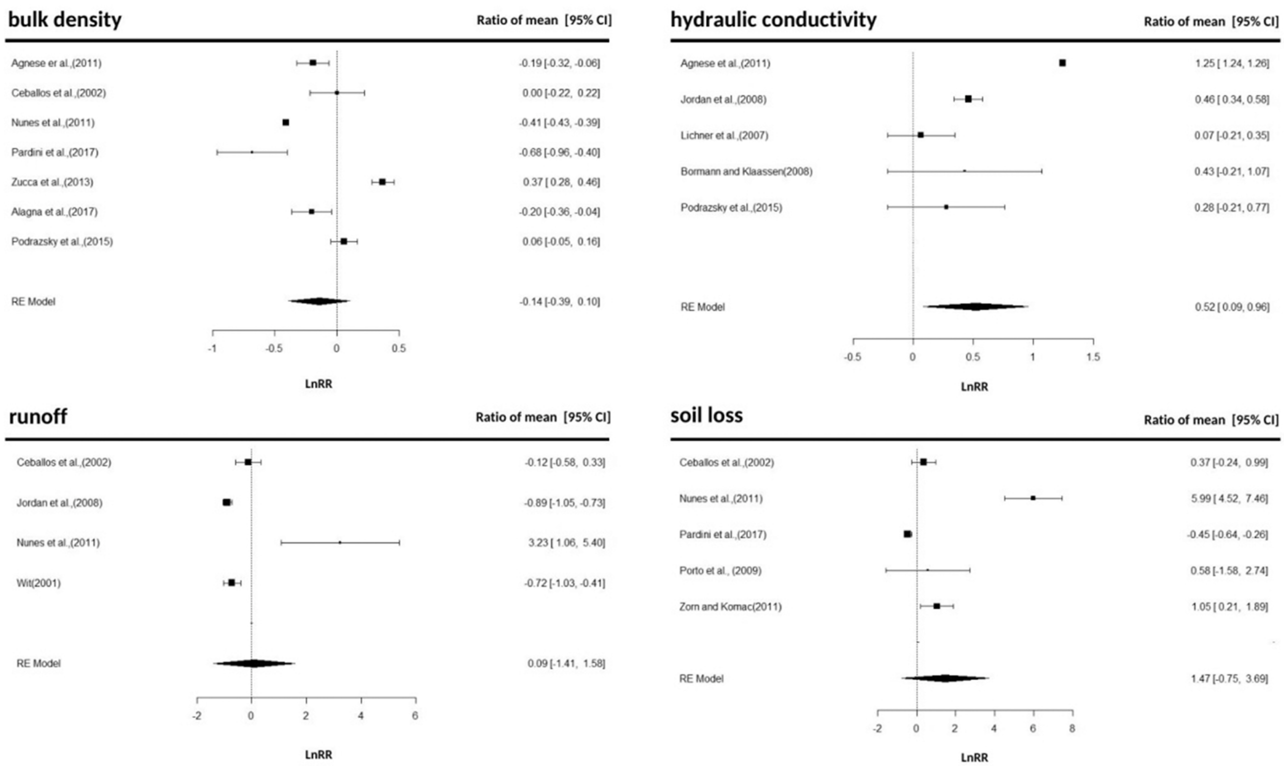

Figure. 2. Weighted mean effect (log response ratio, LnRR) and 95% confidence interval of permanent grassland vs. forest land on bulk density, hydraulic conductivity, runoff and soil loss response. A LnRR> 0 indicates a higher value of the indicator under forest land, while LnRR< 0 indicates a lower value under forest land, compared to permanent grassland. Effects are significant (P ≤ 0.05) where confidence intervals do not intercept 0.

Also, no significant differences have been reported in hydraulic conductivity (RE [95%CI=-0.01[-0.61;0.59] n=5 studies yielding), although the majority of estimates are negative (60%). An examination of the studentized residuals revealed that the study of Brejea et al. (2010) has a value larger than 2.57 and may be a potential outlier in the context of this model, influencing indeed the RE outcome. Again, tillage operations temporarily modify the physical status of topsoil increasing the hydraulic conductivity, although this effect quickly disappears after a couple of weeks (Kool et al., 2019).

The estimated average response rate based on RE of runoff is positive, although it is not significantly different from zero (RE [95%CI=0.30 [-0.43; 1.02] n=6 studies yielding). Most of the studies display a higher runoff generation in arable land (67%). Pardini et al. (2017) have a large sample size (n=20) influencing negatively the weight of RE. According to our assessment, soil loss generation is higher in arable land than in permanent grassland, although it is also not significant (RE [95%CI=1.73 [-0.09; 3.56] n=7 studies yielding). In fact, the estimated RE outcome is significantly higher in arable land in 86% of the analysed studies. Pardini et al. (2017) have a studentized residual value larger than 2.69 and it is considered an outlier in the context of the model. Local environmental conditions can overturn the erosion and runoff mitigation effect of permanent grassland. For example, Pardini et al. (2017) observed a higher runoff and soil loss under permanent grassland. Nonetheless, this can be understood because this study measured the erosion generated in a permanent grassland area regrown after a fire event in Catalonia. Fire severely compromises some soil properties, increasing the bulk density, as organic matter is lost, and the soil structure can collapse completely. This enhances soil loss and runoff. The role of fire on soil degradation and the effect of permanent grassland will be discussed in more detail in the next section. Also, Hejduk and Kasprzak (2010) observed a higher runoff in permanent grassland compared to arable land in the Czech Republic, and attributed this to quicker snow melting process in permanent grassland.

In terms of the contrast between permanent grassland and forest, the differences are not as clear-cut as compared to arable land. In contrast to forest land, there were no significant differences in bulk density (RE [95%CI] = -0.14[-0.39; 0.10], n = 7 studies yielding). Overall, 57% of entries were significantly negative, while only the study of Zucca et al. (2010) was significantly positive [LnRR [95% CI] = 0.37[0.28;0.46]. Hence, the average outcome is estimated to be negative. Bulk density is the only indicator that is generally lower in forestry land, except for the study by Zucca et al. (2010), which underlines the role of permanent grassland management on this indicator. In terms of hydraulic conductivity, similar results have been assessed, the observed response ration ranged from -0.21 and 1.26, with the totality of estimates being positive. The estimated average response ratio differed significantly from zero (RE [95%CI] =0.52 [0.09; 0.96]. An examination of the studentized residuals revealed that one study (Agnese et al., 2011) had a value larger than 2.57 and may be a potential outlier in the context of this model. The observed response ratios of runoff ranged from -0.89 to 3.23, with the majority of estimates being negative (75%). The estimated average response ratio did not vary significantly from zero, (RE [95%CI] =0.09 [-1.41; 1.58]. Only one study exceeded the studentized residuals values of 2.49 (Nunes et al., 2011) and may be a potential outlier in the context of this model. Also for the soil loss indicator, the estimated average response ratio based on the RE did not differ significantly from zero (RE [95%CI] =1.47 [-0.75; 3.69]. Moreover, the observed response ratios ranged from −0.45 to 5.99. Also, in this case, the study of Nunes et al. (2011) is considered a potential outlier within the RE.

In conclusion, while it is generally assumed that converting permanent grassland to arable land leads to more runoff, soil loss and flooding, and, overall, the results of our systematic analysis do indeed confirm this, the results are not always clear and significant. Numerous exceptions were found where the effect was found to be negative or non-significant. A deeper analysis of local conditions helps explain some of these differences, for example the effect of fire or snow-melt erosion led to a negative effect under permanent grassland. The effect of tillage on arable land is also important. Vegetation conditions, bulk density and hydraulic conductivity are highly dynamic and the time of the measurement with respect to tillage operations was not always well detailed in the analyzed studies. The comparison between permanent grassland and forest showed that the difference was even less clear, with no significant differences, except for hydraulic conductivity. This study indicates that permanent grassland is similar to forest in terms of erosion and flooding mitigation.

3.2. Additional erosion and degradation processes.

The four simple indicators analysed in the meta-analysis give a first diagnosis of erosion and flooding problems and are well related to runoff generation and sheet and rill erosion. However, grasslands are threatened by additional important erosion processes. We identified six additional erosion processes: trampling-induced erosion, gullying, piping, landsliding, snowmelt erosion, and avalanche (Figure 3), that are poorly studied and will be discussed in detail below. Also, we identified three erosion promoting processes, hydrophobicity, fires and compaction, that are related to grassland soil management, which exacerbated these erosion processes.

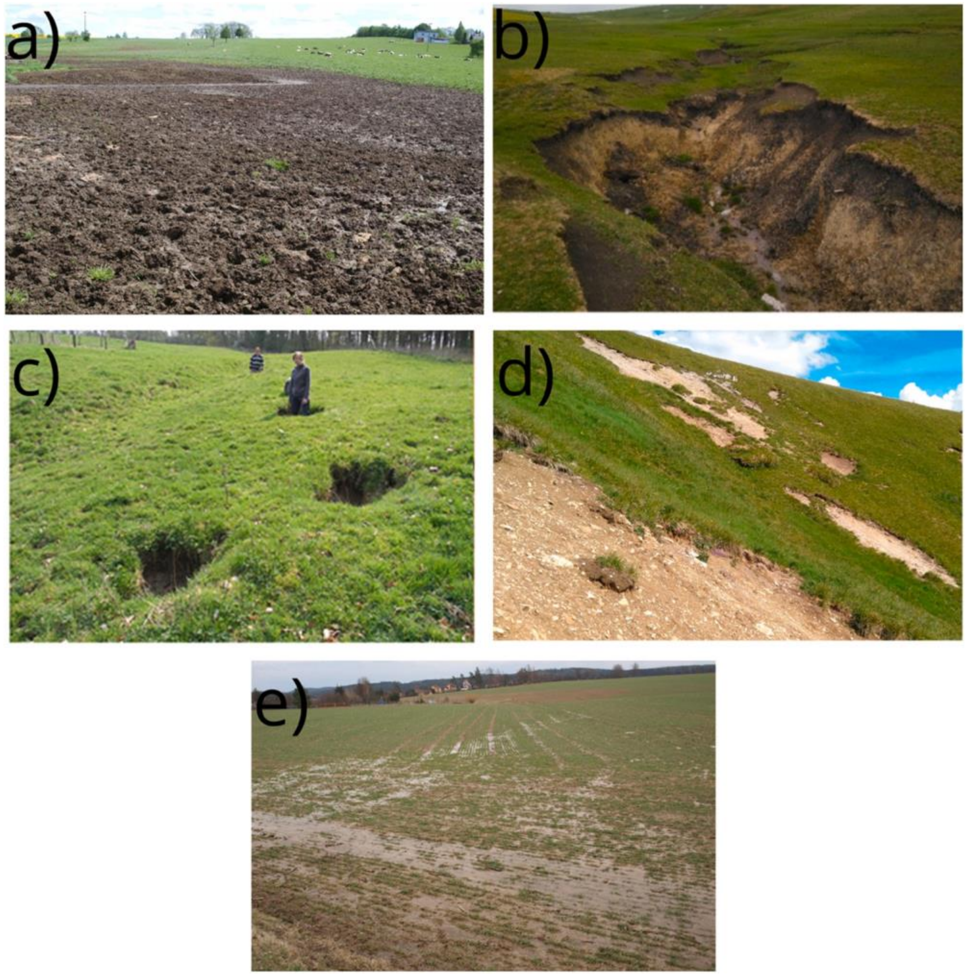

Figure 3. Main soil erosion processes: a) trampling-induced erosion in the Czech Republic; b) Gully erosion in Romania (Nicu, 2018); c) Pipe erosion in Belgium (Verachtert et al., 2011); d) Landsliding in UK (DEFRA, 2010); e) Snowmelt erosion and flooding in Czech Republic.

3.2.1. Trampling-induced erosion

Animal trampling increases soil erosion by degrading local vegetation cover, disturbing soil and unconsolidated materials (Apollo et al., 2018; Torresani et al., 2019; Marzen et al., 2019). Trampling also decreases water infiltration, which in combination with high runoff, reduces both soil health and permanent grassland productivity (Dubeux Jr. et al., 2009; Yang et al., 2013). The vegetation plays an important role in restricting the damage due to trampling by reducing soil moisture during the warm weather and increasing the potential of soils to absorb water during periods of rain (Pande and Yamamoto, 2006; Liu et al., 2016). Permanent grassland degradation by livestock trampling depends on different local factors, such as soil structures, soil wetness, grass and livestock types, and the period when livestock roam (Bilotta et al., 2007). Cole et al. (1995) observed the trampling effect on 18 grassland sites that were trampled between 0 and 500 times, concluding that there is no linearity between trampling intensity and vegetation cover disturbance. Indeed, the degradation was better described by a second-order polynomial function underlining a multi-fold relationship. Manthey and Peper (2010) studied the trampling effect in semi-arid rangeland, finding no linear relationship between grassland degradation and livestock intensity, but with a better relationship with the temporal distribution of the animal roaming. Trampling processes are particularly important in areas of high livestock concentration, such as livestock trails or around drinking or feeding areas, although specific studies on the extension and associated erosion rates are rare. Samarin et al. (2020) mapped a threefold increase of livestock trail erosion in a 26 km^2 alpine valley in Switzerland over the last 20 years, from 1 to 3 ha.

3.2.2. Gully erosion

Gully erosion is the formation and subsequent expansion of channels in the soil as a result of concentrated overland flow. In grazing areas in Australia, it has been documented that gully erosion is one the largest sediment contributors (Wilkinson et al., 2018). This study found that gully sediment yields were reduced by 77% if cattle was excluded from grazing within and around the gullies, therefore concluding that reducing livestock grazing pressure is crucial for gully erosion control. In Europe there has been less research on gullies in grazing areas, especially in permanent grassland. Torri and Poesen (2014) reviewed 39 publications on topographic thresholds for gullying. They identified 19 out of 49 sites where gullies had formed under permanent grassland and concluded that soils under permanent vegetation were almost four times more resistant to gully erosion than cropland. Only four of these studies were done in Europe, most of them in the Mediterranean region. Vandekerkhove et al. (2000) measured and compared gullies under rangeland in SE Spain and Lesvos Island, Greece, but noted that in the first case gullies actually formed when the area was still cultivated, and in the latter case that the vegetation cover was highly degraded due to overgrazing and frequent fires. Zucca et al. (2006) pointed to overgrazing as the main cause for the formation of gullies in their study area in Sardinia, Italy. Gutiérrez et al. (2009) studied gullies in the dehesa landscape of Southern Spain, a type of permanent grassland consisting of grass layer with dispersed tree cover. They found that gullying was significantly related to grazing intensity. Strunk (2003) reported gullying in mountain pastures of N Italy, and also linked this to overgrazing. Menéndez-Duarte et al. (2007) studied severe gully erosion in the north of Spain, an area with agroclimatic conditions comparable to UK, Ireland and Northern Europe (Ceglar et al., 2019). In a recent study, Nicu (2018) explored the relation between overgrazing and gullying in Romania, and mapped 677 gullies in a 550 km^2 area, using a combination of aerial photos and field mappings. The lack of more detailed studies indicates that there is an important research gap here.

3.2.3. Piping erosion

Piping erosion is an underground process, which consists of the displacement of soil through empty spaces (macropores, roots or biological channels) by concentrated water flows, that can collapse and become a discontinuous gully (Hagerty, 1991). This phenomenon is more widespread than often assumed, and it occurs in almost all the bioregions and it is prompted by different factors such as climate, soil properties, topography, land use and management (Carey and Woo, 2000; Zhu, 2012; Faulkner, 2013). Pipe erosion is often followed by other soil erosion process such as landslides (Jones, 2004; Hencher, 2010; Verachtert et al., 2013) and gullies (Jones, 1981; Gutiérrez et al., 1997; Faulkner, 2013). Due to the underground nature of the process, it is challenging to detect, control and measure, and is usually only discovered when the roof of the pipe collapses (Verachtert et al., 2013; Bernatek-Jakiel et al., 2016). Relatively few studies have specifically assessed pipe erosion rates, the majority focussing on cropland by measuring sediment yield (Farres et al., 1990; Øygarden et al., 1997; Sogon et al., 1999) or tracers as Pb^{210} and Cs^{137}. Verachtert et al, (2011) found the pipe soil loss rate of a Belgium permanent grassland to be between 2.3 and 4.6 t ha^{-1} y^{-1}, which is considerably above the superficial European mean soil loss rate in permanent grassland, excluding the Mediterranean region (Cerdan et al., 2010).

3.2.4. Landslides

Landslides are defined as the movement of a mass of rock, debris, or earth down a slope by the force of gravity and thereby, the loss of one or more soil functions. It is one of the major local soil threats in Europe's mountainous regions and slopes (European Commission, 2008). This phenomenon is widespread in the European Alps and has enduring degradation effects on permanent grassland (Wiegand and Geitner, 2013). In the Austrian Alps, a landslide is locally called "Blaike" which is a German word that refers to an extremely eroded spot surrounded by undisturbed grassland (Stiny, 1910). Indeed, Landslide is promoted by intense rainfall events, snowmelt abrasion (i.e avalanche and snow gliding) or a combination of both (Geitner et al., 2021; Wiegand and Geitner, 2013). However, many other factors such as topography, soil and bedrock, vegetation and human activities are interacting with slope stability (Bíl and Müller, 2008; Stolte et al., 2015). An increase in animal stocking rates also has significant impacts on landslide incidence (Meusburger and Alewell, 2008). In the European Alps and other mountains regions, for example the Spanish Pyrenees, shallow landslides, where superficial erosion removes a layer of soil in a small area, between 2 and 200 m^2, exposing the mother rock (Geitner et al., 2021; Wiegand and Geitner, 2013), are a widely spread phenomena in grasslands and happen when prolonged precipitation or snowmelt displaces the topsoil layer (García-Ruiz et al., 2010; Zweifel et al., 2019). Recently, landslide events in Europe have increased regionally with different intensities (Kundzewicz, 2019; Van Beek and Van Asch, 2004). Crozier (2010) expected more future landslides due to global warming and extreme precipitation events.

3.2.5. Snowmelt erosion and flooding

Snowmelt runoff is an important factor in flooding and soil erosion in higher and cold regions of the world. In Nordic countries of Europe, snowmelt processes significantly affect water resource recharge but also the occurrence of natural hazards (overland flow, flooding and shallow erosion) (Øygarden, 2003; Kremsa et al., 2015). The mechanism of surface runoff formation from frozen soil is completely different compared to surface runoff caused by torrential rains on unfrozen soil (Hejduk and Kasprzak, 2004), which means that permanent grasslands are more prone to generate snowmelt erosion and runoff compared to other land uses. When snow melts, the magnitude of the runoff event depends on the soil frozen layer structure, which is increased from the discontinuity and the heterogeneity of the icy layer. Tillage and fertilization practices increase the volume, the surface roughness and the formation of a heterogeneous soil frozen layer that increases the runoff formation, which explains its importance for permanent grassland (Nyberg et al., 2001; Miller et al., 2017). Kremsa et al. (2015) studied the snow layer in forest and in grasslands, observing that in forest areas, the snow-depth was 26% higher. This was explained particularly by wind effects and higher snow erosion in the open landscape. The final snowpack depletion in the forest occurred over 44 days, compared to 25 days in grassland areas, with a mean melt intensity 7 versus 10 mm day ^{1}. Chanasyk et al. (2003) found that surface runoff from grasslands in Alberta (Canada, c. 1,300 m a.s.l.) were much higher during snow melt in early spring compared to summer runoff after heavy storm rains. The early spring runoff accounted for 78 and 96% of total annual runoff from mown and grazed grasslands, respectively. Snow thaw was much faster on ungrazed grassland (2 days) compared to grazed (10 days), probably due to residual biomass on ungrazed stand and deeper frost penetration on the grazed stand due to soil compaction (higher heat conductivity). Hejduk and Kasprzak (2004) compared surface runoff from arable land and grasslands in winter seasons. They reported that mown grasslands had a higher susceptibility to formation of surface runoff than winter wheat (sown into tilled soil). In grasslands, a higher surface runoff was caused by quicker thawing of snow cover 'hanging' of the grass stubble and slower melting of soil (icy layer insulated by grass biomass). Soil erosion in winter and early spring can be particularly severe in connection with rain on partially thawed soil (Øygarden, 2003), when infiltration is restricted and fast water flow can detach particles from the thawed soil surface. However, Ulen at al. (2012) stated, that in contrast to rain, snowmelt is a gentler process.

3.2.6. Avalanche erosion

The problem of soil erosion by snow is becoming increasingly relevant. Besides that occurring in spring by snowmelt discussed in the previous paragraph, winter avalanches might contribute strongly to soil erosion in alpine grasslands (Ceaglio et al., 2011; Jomelli and Bertran, 2001). Identifying and classifying avalanche formation is complex; its multifactorial nature means that local conditions influence its pathway and dynamics (Schweizer et al., 2003). Snow avalanches are both an erosional and flooding process, and may modify or produce other erosion processes, such as gullies and landslides (Luckman, 1977). Meusburger et al. (2014) assessed the importance of snow gliding or avalanches for soil erosion in grasslands of the Swiss alps. They compared modelled erosion rates using the RUSLE model with measured erosion rates, using Cesium-137 radioisotopes, and found a large difference that could be attributed to the effect of avalanche erosion. They also measured soil deposition by avalanches directly during one year, obtaining soil erosion rates between 0.03 to 22.9 t ha^{-1} yr^{-1}. Stanchi et al. (2014), developed the winter factor (W-factor) to adapt the RUSLE model. W-factor is the ratio between the ^{137}Cs derived erosion rates, including all erosion processes, and erosion rates modelled by RUSLE, that only include sheet and rill erosion. However, avalanche parameterization and the soil erosion assessment derived from it, are still relatively new and much more research is needed.

3.3.Processes promoting erosion and flooding

3.3.1. Compaction due to trampling and wheeling, poaching and pugging

Soil compaction is the process of densification and distortion of soil leading to lower soil pore volume, resulting in loss of one or more of the soil's functions (Akker et al., 2004). Soil compaction is a major soil threat in Europe where about 32% of soils are highly susceptible and 18% are moderately susceptible to it (European Commission, 2021b). In permanent grasslands, soil compaction occurs due to animal trampling, machinery wheeling and poaching or pugging (i.e. penetration of soil surface by the animal hooves). It represents one of the main factors that leads to degradation of soil physical quality (Imhoff et al., 2000). It negatively affects soil structure, water retention, water uptake, soil porosity, soil nutrients and grass production (Freddi et al., 2009; Hargreaves et al., 2019; Silva et al., 2015). Heavy animal grazing and introduction of larger machinery in European grasslands has led to compaction becoming a more common phenomenon. In addition, poaching or pugging can stimulate water runoff, expose soil surface to water erosion and cause damage to swards (Evans et al., 1999). Johnson et al. (1993) found that pasture with a reduction in growth due to pugging/poaching can be effectively renovated by undersowing. The structural damage of soil due to compaction in Europe can be very serious, especially when the soil conditions become wet. In Ireland, Bondi et al. (2021) noticed that poorly drained fields were highly vulnerable to wheeling intensity. One of the effective indicators for soil compaction in grazed pastures is penetration resistance (resistance of soil matrix to penetration by growing roots) (Benevenute et al., 2020), which is very sensitive to compaction by animal trampling (Scholz and Hennings, 1995). Mapfumo et al. (1999) and Ludvíková et al. (2014) reported that heavy grazing, even for a short period, significantly increased the penetration resistance, and reduced vascular plant richness, overall plant species composition, plant cover and sward height.

3.3.2. Hydrophobicity and water repellence

A specific phenomenon that occurs especially on grasslands with sandy and organic soils is called hydrophobicity or soil water repellence (SWR). It can decrease the infiltration rate of soil and increase the potential for surface runoff in response to rainfall (Bauters et al., 2000; Dekker et al., 2001). The likelihood of SWR increases as the soil surface dries out in warmer months (McDowell et al., 2020). One of the factors that creates SWR can be manure applications, or the presence of certain plant waxes on soil particles (Miller et al., 2017). Infiltration in hydrophobic soils is limited to preferential pathways that increase leaching of pesticides and nitrates from the soil (Aamlid et al., 2009). The cause is usually the coating of soil particles with hydrophobic compounds, which are produced by the plants themselves (leaf waxes, root exudates) or microorganisms (especially fungi). Hydrophobicity is also often caused by irreversible changes in organic colloids as the soil dries out. To reduce the negative phenomena caused by the hydrophobicity of soils, it is possible to use soil wetting agents, the application of which, however, is justified in view of the high price only in intensively treated turfgrasses and in fruit orchards (Moore et al., 2010). Lichner et al. (2011) measured the differences between topsoil (sand with roots and organic matter) and subsoil (pure sand) of grassland on sandy soil in SW Slovakia. They found that grassland soil had an index of water repellence about 10 times that of pure sand and the persistence of water repellence almost 350 times that of pure sand. Hydraulic conductivity and saturated hydraulic conductivities in the grassland soil were 5% and 16% of those of the pure sand. The grassland soil was substantially more water repellent and had three times the degree of preferential flow compared to pure sand. Runoff is likely to be exacerbated by water repellence, as it decreases infiltration rates, enhances overland flow and increases the risk of soil erosion (Doerr et al., 2000). Water repellence is a transient soil property, which tends to be both spatially and temporally highly variable. It often disappears after periods of prolonged soil wetting, but will usually re-emerge during drier periods when soil moisture falls below a critical threshold (Dekker et al., 2001). Water flow paths, once created, persist over time during summer, but over annual cycles their spatial arrangements can change completely (Wessolek et al., 2009). Grass cover can induce water repellence in all soil types ranging from sands (Dekker et al., 2001) to clays (Dekker and Ritsema, 1996) by both root exudates and thatch (the layer of organic matter between the mineral soil and the green grass).

3.3.3 Fires in Mediterranean pastures

Fire is an important natural landscape shaping agent, and the Mediterranean area is the most fire-prone zone of Europe due to land use and climate (Pausas, 2004). Depending on the fire characteristics, as the intensity or the severity, it can cause shorter or longer-term impacts (Vieira et al., 2015). The most important short-term impact, in terms of soil erosion risk, is the reduction of the vegetation cover which increases runoff and erosion (Soler and Sala, 1992; Zavala et al., 2014). Moreover, fire events can cause deterioration, partially or completely, the soil structure, the porosity and increase the bulk density (Mataix-Solera et al., 2011). Consequently, fire produces negative impact on the soil hydraulic properties (Imeson et al., 1992). Studies reported that it can also reduce soil aggregate stability that can contribute to an intensification of soil detachment (Llovet et al., 2009; Ubeda and Bernia, 2005), and, raise soil repellence (Doerr et al., 2000). Pardini et al. (2017) assessed the effect of fire on runoff and soil loss in grassland and olive orchard, observing a remarkable runoff and soil loss mitigation of permanent grassland. Despite these negative effects summarized above, prescribed fire is a common management practice in the Mediterranean, aimed at burning bushland in favour of pasture. It is considered an efficient and cost-effective land management practice for livestock feed production. In arid and semi-arid climates, prescribed fire executed in late spring exposes erosion-prone sites to elevated summer runoff and soil loss events (O'Dea and Guertin, 2003). The effect of fire on erosion is widely studied. Shakesby (2011), reviewed the post-wildfire soil erosion in the Mediterranean basin and found only 6% of the reviewed studies focussed on the permanent grassland land use. Vieira et al. (2015) reviewed 109 studies globally about the effect of post-fire on erosion and runoff generation, claiming that 63% of studies are located in the USA , 25% in Spain, and only 10% of those are focussed on permanent grassland land uses. According to these global studies, it is clear that fire risk affects mainly forest land, and fire risk in permanent grassland is lower, but if it occurs, it causes significant damage.

4 Conclusions

Our study provides a deeper overview of the importance of permanent grassland for erosion and flood mitigation in Europe and the UK. Firstly, a quantitative meta-analysis evaluated four erosion and flooding-related indicators, bulk density, hydraulic conductivity, runoff and soil loss, between three land uses: permanent grassland, arable land and forests. In total 24 articles were analysed, after screening over 14203 articles. The results showed that on the one hand, in comparison with arable land, results are often in contrast to the widespread opinion of topsoil structural amelioration of grassland. In fact, no significant differences have been reported comparing bulk density and hydraulic conductivity and soil loss, highlighting the temporary effect of tillage and of the local environmental conditions that can promote soil degradation (i.e. fire). On the other hand, permanent grassland mitigates better runoff than arable land. In contrast with forest land, differences are not clear cut, suggesting that soil erosion and runoff mitigation condition are similar between the two land uses, except for the hydraulic conductivity which is higher in forest land.

However, these general indicators are limited in scope. A second, broader review showed how European permanent grasslands suffer from additional land degradation hazards. This additional review identified six processes important for soil erosion in European grasslands: trampling-induced erosion, gullying, piping, landsliding, snowmelt erosion and avalanche erosion. All these processes were documented in European grassland to have cause significant erosion problems locally. At present, their extent and regional impact is mostly unknown. These are boosted by several promoting processes related to soil management and environmental conditions: compaction, hydrophobicity and wildfires. In summary, although permanent grasslands are considered crucial for the reduction of soil loss and flood, they are under degradation risk. Due to the complex nature and the interconnection between erosion and flooding processes, and the lack of knowledge on many of the processes involved, their assessment, understanding and modelling are still often challenging. Therefore, these processes must be studied more in detail in order to get a good view of the status of European permanent grasslands. This will help with designing a site-specific soil management strategy for European grasslands, aiming at the zero net land degradation goals promoted by the Green Deal.

Acknowledgements

This research is funded by the European Union, under the Horizon 2020 project “Developing SUstainable PERmanent Grassland systems and policies (Super-G)”, grant no. 774124. Views and opinions expressed are those of the author(s) only and do not necessarily reflect those of the European UnionEU or the European Research Council. Neither the European Union nor the granting authority can be held responsible for them. Vanwalleghem and Milazzo also acknowledge additional financial support from the Spanish Ministry of Science and Innovation, the Spanish State Research Agency, and through the Severo Ochoa and María de Maeztu Program for Centers and Units of Excellence in R&D (Ref. CEX2019-000968-M).

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Chapter 3: An overview of permanent grassland grazing management practices and the impacts on principal Soil Quality Indicators

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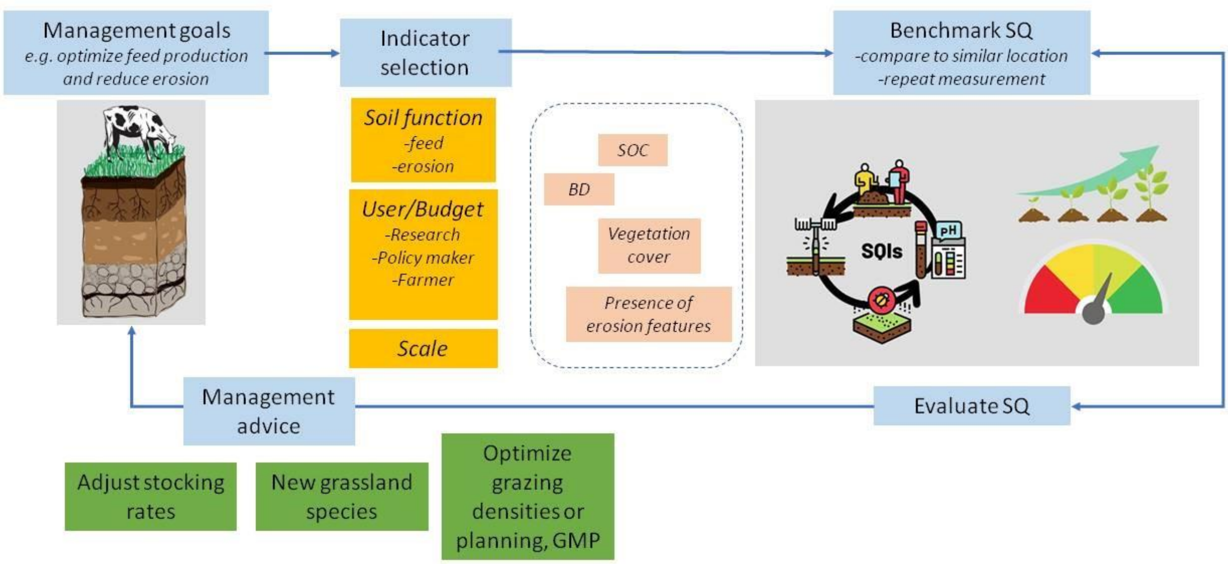
Abstract:

Grasslands are at risk of degradation due to unsustainable management practices and climate change. Here, we review the principal soil quality indicators (SQIs), to evaluate the sustainability of different grassland management practices globally. We discuss the importance of SQI assessment and the Soil quality Minimum dataset (MDS), specifically for the context of grasslands. We then review two potential solutions. One is adapting the grazing management, where sustainable Grazing Management Plans (GMPs) offer great potential. The other solution is the development and adoption of novel grassland species which may improve either drought-resistance or infiltration rates, erosion and flooding. Sustainable grassland soil management can promote ecosystem service delivery and improve the resilience of the entire grassland ecosystem to anthropogenic change.



Keywords: Soil quality indicators, Grazing management, Ecosystem services, Permanent grasslands, Management practices

1. Introduction

Grasslands cover more than 30% of total cultivated land in Europe and 69% globally (EUROSTAT 2021, Suttie et al., 2005), and are generally recognized for their role in soil erosion control and ecological multifunctionality (Milazzo et al., 2022; Schils et al., 2022). Grasslands are experiencing degradation due to desertification and intensive grazing (Zhang et al., 2019). Grazing plays an essential role in grassland preservation and well-managed grazing can promote soil quality, biodiversity and other related ecosystem services (Metera et al., 2010). Grazing affects the nitrogen cycle (Silveira et al., 2013), soil organic carbon (SOC) (Steffens et al., 2009), soil water content (Thomas et al., 2008), bulk density (Zhou et al., 2010) and soil biodiversity (Esch et al., 2013). However, overgrazing can also promote several soil degradation processes affecting entire grassland ecosystems (Zhan et al., 2020). Overgrazing compacts soil and triggers a series of subsequent issues related to the increase in bulk density, such as soil loss, runoff and flooding (Centeri, 2022; Milazzo et al., 2023). Moreover, soil compaction leads to depletion of SOC and total nitrogen affecting the soil microbiota (Bagchi et al., 2017). For these reasons it is important to consider appropriate livestock densities that avoid these negative effects of overgrazing. However, the definition of heavy or light grazing, at a European level, may be too broad to assist farmers in their grazing decision-making. For instance, Klipple and Bement (1961), define grazing density based on the ability of grass species to maintain themselves as forage for grazing animals. Optimum grazing density is then usually defined in terms of grass biomass production, and aims for a balance between carrying capacity and animal requirements. Research has shown that grass growing capacity varies with climate, grass species, animal type and soil type. Milazzo et al. (Milazzo et al., 2023) highlighted the importance of protecting permanent grassland from the various erosive phenomena that threaten these ecosystems Europe-wide. In particular, they described that unsustainable grazing management, which depletes soil quality, promotes erosion and flooding phenomena. Therefore, it is challenging to establish grazing limits in practice and it might be necessary to consider other soil quality indicators (SQIs) that can alert of soil degradation. In other words, to make adequate management decisions on grazing densities or practices that promote soil health, it is necessary to include SQIs that can assist farmers in establishing objective limits to grazing densities, or in other corrective measures. Several studies discuss grazing advantages and disadvantages, synthesizing a large volume of scientific evidence, often providing a qualitative assessment of different grazing practices (di Virgilio et al., 2019). They generally focus on the comparison between different types of grazing management, i.e. short duration grazing (Lawrence et al., 2019), continuous vs rotational grazing (Ma et al., 2019), holistic versus continuous grazing (Oliva et al., 2021), and usually, they evaluate the impacts of practices on grass productivity (di Virgilio et al., 2019), but not on soil properties.

Possibly the most important threat to grasslands is climate change. This is threatening grasslands globally by exposing soils to prolonged droughts, making them prone to water erosion (Dong et al., 2020; Wang et al., 2015; Zheng et al., 2019). Heatwaves are also endangering global grassland productivity (Ciais et al., 2005), particularly in semi-arid and arid climates where irregular and high-intensity precipitation is enhancing flooding and erosion (Wang et al., 2015). Liu et al. (2019) assessed grassland degradation worldwide, asserting that more than 45% of grassland areas have experienced degradation processes by human activities and climate change. Moreover, they stated that anthropogenic activities are more dominant in North America and Europe, while the Asian region is more affected by climate change. In the Chinese Loess Plateau, human activities and climate change contributed to 42% and 58%, respectively, of the total grassland degradation (Zheng et al., 2019). Currently, several studies explore the need to breed and select new drought-resistant grassland species to preserve the grassland provisional service (Fernández-Habas et al., 2023; Foster et al., 2012). However, in the global context of climate change and less water availability for grassland, breeding of new drought-resistant grassland species can reduce yield gaps, bare soil conditions, and control soil degradation processes.

Soil quality is defined as the ability of soil to perform ecosystem functions (Karlen et al., 2003). It is a broad concept that is not limited to the biological, physical and chemical soil properties, but it also involves productivity and animal and human health (Doran and Parkin, 1994). The soil quality concept was introduced in 1977 by Warketin and Flacher (Warkentin and Fletcher, 1977), to respond to increasing stakeholder's concerns about soil resources and to evaluate land use decisions made in the institutional context. The interest in soil quality increased after the publication by Council et al. (Council et al., 1993), when academia focused on critical soil function identification and a common soil quality assessment framework (Doran and Parkin, 1994). Nowadays, soil quality has obtained more attention for monitoring land management, sustainable development and ecosystem restoration through the evaluation of the Soil Quality Indicators (SQIs) (Gholamhosseinian et al., 2022; Muñoz-Rojas, 2018a). However, due to the wide variety of soils, climate, land uses and management systems, it is challenging to standardize the SQIs benchmark for a universal assessment. Indeed, there are two problems that can be identified: (1) there is no universally accepted set of optimum SQIs that should be considered, and (2) there is no ideal or exact index value that can universally standardize soil quality assessment. However, using a framework that prioritizes soil quality goals and evaluate the management operation to achieve those specific soil functions can help (Karlen et al., 2003). Indeed, the periodic estimation of SQIs can guide farmers in management decisions, and even on inherently "poor" soils, positive effects can be seen if compared to an initial measurement or to an appropriate local benchmark. In this sense, it is important to select the appropriate SQI, weighing cost and benefits, and considering local conditions and objectives. (Karlen et al., 2003).

In this study, we aim to give a global perspective of grassland soil quality assessment and management, before applying the lessons with relevance to mitigate soil grassland degradation in European and UK. Firstly, we review the importance of SQIs for sustainable grazing management methods to avoid land degradation risk. Secondly, we present an overview of the new drought-resistant grass species that improve soil quality and reduce soil loss.

2. Soil Quality Indicators for grassland

SQIs are defined as measurable physical, chemical, and biological attributes which relate to functional soil processes and can be used to evaluate SQ status, and that are sensitive to changes in management (Lal, 2011), Table 1. These attributes are commonly soil properties, although in a wider sense also non-soil properties can indirectly inform on soil quality, for example yield, surface vegetation cover, or presence of erosion features. The latter are often easier and faster to assess by farmers and land owners. Commonly used chemical indicators include organic/total carbon and nitrogen, extractable phosphorus and potassium, pH, electrical conductivity, and cation exchange capacity. Biological indicators include microbial respiration rates, microbial biomass, nitrogen mineralisation rates, macrofauna (often earthworms), nematodes, microbial community composition and enzymatic activity. Physical indicators include soil bulk density, structure, texture, aggregate stability, porosity, water storage, hydraulic conductivity and infiltration (Muñoz-Rojas, 2018b). In relation to soil erodibility and flood risk reduction, the physical indicators are the most directly relevant, because they influence rainfall-runoff dynamics and water storage capacity, which sustains and regulates river flows, and thus contributes to stream flow buffering (Buytaert et al., 2002). Nevertheless, many of the chemical and biological SQIs play important indirect roles through their influence on the soil physical properties. For instance, soil structure and aggregate stability are both related to SOC, which in turn depends on a range of biological soil properties (Meurer et al., 2020; Sullivan et al., 2022). As such, many of these SQIs are inter-related, and while physical properties are likely to have the biggest direct impact on soil erosion and flood risk, chemical and biological SQIs could serve as useful proxies to assess management. Bünemann et al.(2018) showed that the most commonly used physical SQIs are water-holding capacity, water content, bulk density and texture. Several studies have assessed the soil hydraulic properties of grasslands and compared these to those of cropland soils. Abdalla et al. (2020) reviewed the overall soil loss and SOC loss under different land uses, and found a remarkable protection capacity of grassland when compared to orchards, croplands, and forests. While total rainfall and slope were the key drivers of soil erosion, high soil surface cover, SOC and clay content all limited soil loss. Several studies accompanied SQI observations with measurements of SOC and quality, due to its strong link with soil physical properties. Ghimire et al. (2019) in the USA, among others, showed that SOC, microbial biomass and total nitrogen the most commonly used SQIs (Bünemann et al., 2018) were all higher under permanent grasslands compared to croplands, owing in a large part to the lower degree of soil disturbance in permanent grasslands. Lehtinen et al. (2015) analysed the distribution of soil aggregates and assessed quality, quantity, and distribution of soil organic matter (SOM) in two unimproved and four improved (two organic and two conventional) grasslands in subarctic Iceland. They found a higher macroaggregate stability in organic farming practice compared with conventional farming, due to higher organic inputs. However, few attempts have been made to relate the grassland species composition to soil erodibility and SOC content and stock. Enri et al. (2021) highlighted the importance of grassland species composition in affecting SOC stock in Alpine pastures, while topographic attributes had negligible effects. Root characteristics are also important for increasing SOC stock, as well as determining the capacity of grasslands to resist erosion. Horrocks et al. (2019a) demonstrated a strong effect of forage species and variety on the aggregate stability, friability and SOC, in tropical environment grasslands in Colombia. These studies demonstrate the importance of vegetation type influencing SQIs. Whilst physical indicators provide a direct link to a grassland ability to reduce erosion, a large and increasing number of studies now emphasise the vital role of biological indicators on soil health and quality (Muñoz-Rojas, 2018b).

Table 1. Published studies on the biological, chemical and physical SQIs assessment in grassland and the related ecosystem services such as provisional of animal feed (p), water purification(w), biodiversity(b), climate regulation(c), erosion and soil degradation processes (e)

Reference	*Biological*	*Chemical*	*Physical*	*Country*	*Study period*	*Ecosystem service*
(Horrocks et al., 2019b)	microbial community	SOC	aggregate stability, friability,	Colombia	1	p, e
(Yu et al., 2018)	Root growth, Microbial biomass carbon, Alkaline phosphatase, catalase	SOC, C/N, C/P, N/P	Water content	China	4	p
(Rezaei et al., 2006)		SOC, Nutrient Cycling index	Soil Stability index, Infiltration index	Iran	1	p
(Askari and Holden, 2014a)		SOC,C/N	Bulk density, aggregate size distributio	Ireland	1	p

			n			
(Mueller et al., 2013)			Structure, porosity, compaction, penetration resistance	Germany	1	p, e
(Kavdır and Smucker, 2005)		N cycle, SOC,	aggregate	Michigan	2	p ,e
(Askari et al., 2015)		SOC, magnesium, C/N	penetration resistance, aggregate size distribution	Ireland	1	p
(Newell-Price et al., 2013)			Structure, compaction,	England	1	e
(Cui and Holden, 2015)	Microbial activity, enzyme activity,	SOC, N,	Porosity, bulk density, texture	Ireland	1	b
(AbdelRahman et al., 2019)		pH, Electrical conductivity, cation exchange capacity, P, N, nutrient availability	Bulk density, water stable aggregates,	Egypt	1	e

(Valle and Carrasco, 2018)		pH, extractable Al, P, SOC	bulk density, porosity,	Chile	2	p, e
(Paruelo et al., 2010)		SOC, C flux		Argentina, Uruguay, Brazil	4	c
(Dong et al., 2012a)		SOC,N,K,P	Bulk density, soil water holding capacity	China	5	e
(Devi et al., 2014a)	Microbial biomass,	SOC. N,P		India	1	b, w
(Silva et al., 2014a)		SOC		Brazil	1	p
(Franzluebbers et al., 2000a)		N,SOC,	Water-stable aggregate, bulk density, texture	Georgia ,USA	24	e
(Zhang et al., 2017a)		pH, electrical conductivity, SOC	Bulk density	China	1	p
(Jiao et al., 2016)		pH, electrical conductivity, SOM ,N,K,P	Bulk density	China	7	p
(Pauler et al., 2019a)		P,K Mg, pH		Germany	5	p
(Larreguy et	belowgroun	SOC,		Argentin	1	p

al., 2017)	d biomass			a		
(Barnard et al., 2006)	Microbial biomass,	N, C/N		France	1	b
(Gardi et al., 2002)	Microbial diversity	pH, SOC, carbonate	Texture, bulk density	Italy	1 punctual analysis over long period observatio n	b
(Li et al., 2023)	microbial biomass	pH, SOC, total N, C\N, electrical conductivity		China	3	b
(Han et al., 2020)	microbial biomass	Total C, total N, total P, pH,	Bulk density,	China	1 punctual analysis over long period observatio n	b

3. Soil quality Minimum dataset (MDS) for grazing management assessment

Grassland soil quality assessment cannot be defined by estimating single soil properties, and it would be impossible to use all soil properties for evaluating soil quality. Previous studies have attempted to create a minimum dataset (MDS) including a core set of soil characteristics which help to monitor soil quality taking into account multiple physical, chemical and biological SQIs (Maurya et al., 2020; Yu et al., 2018). The selection of the soil properties to be analysed is an important process, which can affect the quality and ease of monitoring. Indeed, the analysis of some physical and biochemical soil proprieties (i.e hydraulic conductivity, soil water capacity, microbes biomass) can make the soil status assessment cumbersome and complicated as they require complicated and/or expensive laboratory procedures. Many SQIs are interrelated, so the analysis of one may be sufficient for the determination of others. For example, bulk density and hydraulic conductivity are inversely related, so measuring the former can give us an indication of whether the latter is increasing or decreasing. Rezaei et al. (Rezaei et al., 2006), in semi-arid grassland, studied the importance of the use of a soil quality MDS taking into account time and economic costs. In fact, they compared two MDSs concerning the prediction of management goals of soil productivity and stability: the first, which did not take budget constraints into account, measured the physical properties of the soil and the landscape function analysis method that considers rangelands as landscape systems; the second considered only the measurement of soil physical properties. They found that the latter MDS optimally predicted pasture production underlying the high relationships between soil physical properties and grassland growth. Askary and Holden (2014a) analysed soil quality in temperate grassland by measuring twenty-one indicators for the assessment of grassland management (including grazing), stating that only SOC, C/N ratio and bulk density were decisive for assessing the management effect on soil quality. Complementary to laboratory analysis, several farmer tools-kits have been developed to assess SQIs giving an overall evaluation of the main grassland functioning related to the soil ecosystem services delivery. Ditzler and Tugel (2002), developed the "*Soil Quality Test Kit Guide"* providing a simple field assessment for 11 SQIs. This tool is potentially applicable for all agriculture and agro-forestry systems and permits a 3-level description of the main chemical, physical and biological SQIs. Nevertheless, Visual soil assessment (VSA) is widely used and is known to be cost-effective, practical and to provide rapid results (Ball et al., 2013). VSA gives reliable information about soil structure, presence of telluric fauna, soil porosity, root development and soil colour. This information can be related to pH, bulk density, soil organic matter (Sonneveld et al., 2014). For example the Visual Evaluation of Soil Structure (VESS), are functional and reliable methods for assessing soil structural quality (Askari et al., 2013; Mueller et al., 2013). VESS mainly focuses on soil physical quality indicators that influence several soil functions such as fertility, biological activity, root development, and nutrient cycling (Kavdır and Smucker, 2005). But it must be considered as a support assessment methods, indeed it can be useful for the grazing management assessment but not for the biochemical proposes (Askari et al., 2015). Despite the mentioned limitation, several authors proved the reliability of VESS in grassland. Newell-Price et al. (2013) showed the applicability of the Peerlkamp method for the bulk density assessment, while Cui et al. (2015, 2014) used VESS to score bulk density, total carbon, nitrogen and microbial activity. To meet the needs of farmers to assess SQIs in the field, and avoid the time-consuming and expensive laboratory analysis, different High-Tech solutions are available on the market, such as mobile apps and remote sensing. The SLAKES smart phone application, developed by the University of Sydney, assess the wet aggregate stability based on the slaking index soil aggregates (inversely correlated to aggregate stability) in less than ten minutes (Bagnall and Morgan, 2021). Aggregate stability is related to microbial activity, OM, soil structure, and it is susceptible to management operations (Blankinship et al., 2016). SLAKES app is an easy method, scientifically reliable for quantifying soil quality become available to non-scientists or groups with limited funding for soil analysis (Flynn et al., 2020). In addition, grassland SQIs can be monitored continuously using remote optical sensors, that give useful information for assessing management and soil status (Marsett et al., 2006). The use of satellite information, for grassland health and degradation assessment, is becoming popular due to their extensible scalability. Xu et al. (2015) reviewed the grassland health remote monitoring methods globally, collecting 1057 studies from Web of Science, published between 1984 and 2015, observing that 70% were about vegetation status of which 29% were about livestock management, 30% were about soil status and 25% were about the environmental system. As a matter of fact, with the newest remote sensing approaches it is possible to retrieve, at field resolution, several SQIs, such as SOC, soil erosion, heavy grazing degradation, soil salinity and water logging (AbdelRahman et al., 2019; Zhou et al., 2020).

4. Grazing management for improving soil quality

Sustainable grazing management practices aim to maintain or improve soil quality to prevent land degradation and increase biomass yield over time (Askari and Holden, 2014b; Kemp and Michalk, 2007). As such, grazing timing, grazing density, time between grazing events, and livestock species are crucial considerations for the sustainable management of grasslands.

Grazing effects are species specific (animal/plant) and vary with management types, bioclimatic regions and soil properties (Barber-Cross et al., 2022; Hickman et al., 2004). The interaction between climate and unsuitable farm management strategies, can compromise the soil status and thereby, promote flooding and erosion events (Bartley et al., 2014; McIvor et al., 1995). Due to the main grassland purpose of providing livestock feed, grassland soils are subject to grazing pressure that promote soil quality degradation in the base of the grazing intensity (Bilotta et al., 2007). Nevertheless, grazing intensity definition, in terms of heavy or light grazing is maybe too broad to assist farmers in the grazing decision-making and it varies in the base of grassland productivity and climate, Table 2. In fact, Klipple and Bement (1961), define heavy density grazing as the degree of grazing that does not allow pasture species to maintain themselves; moderate grazing as the degree of grazing that allows grass species to maintain themselves but decreases their mix diversity; light density grazing as the degree of herbage utilization that permits palatable species to maximize their herbage capability. However, this definition takes into account as a reference the fodder production and does not consider the effect on the soil quality. For instance, an increase in grazing intensity is generally related to a decrease in SOC, and conversely light grazing intensities ameliorates increases SOM and reducing soil erosion events (Lu et al., 2017; Zhou et al., 2017). Abdalla et al. (2018) performed a meta-analysis on the effect of grazing intensity on SOC stock globally, highlighting a clear climate-dependent effect. They stated that in a dry warm climate, the grazing effect negatively influence the SOC stock at all levels except for light grazing which increases SOC by almost 6%, instead, in the moist climates, SOC declined in all grazing intensity management. Indeed, animal trampling compacts soil, destroying soil aggregates and altering the soil microbial community, boosting nitrogenous losses by denitrification, and therefore, contributing to grassland degradation (Dong et al., 2012b), see Figure 1. Devi et al.(Devi et al., 2014b) in sub-tropical grassland, showed that moderate grazing intensity promotes the nutrient cycle increasing, in this climate, grassland sustainability. Franzluebbers et al. (2000b), in the Southern Piedmont USA, stated that long-term light grazing increases SOC, biological activity and soil quality. Many studies, across all the bioclimatic regions globally, stated that grazing intensity increases the bulk density, pH leading to higher denitrification processes raising soil erosion risk (Enriquez et al., 2015; Zhang et al., 2017b). Jiao et al. (2016) instead analysed the effect of different grazing management types, asserting that heavy-grazing and no grazing management, significantly increase the bulk density compared to light and moderate grazing, underlining even more the positive effect of a well-controlled grazing management. Generally, heavy grazing is commonly recognised as the dominant factor that increases soil erosion and runoff generation in grassland (Bilotta et al., 2007; Donovan and Monaghan, 2021). For instance, heavy grazing can promote runoff generation by up to 117% compared with rotational light grazing, while the latter has a positive impact in reducing flood risk (Döbert et al., 2021; Park et al., 2017). The choice of livestock breed is also important for farm productivity. New high-productive cattle breeds have different grazing behaviour and anatomic characteristics that impact grass composition and soil quality. Pauler et al.,(2020a, 2020b, 2019b) observed the grazing behaviour of low-productive cattle (Original Brauvieh) and high-productive breed Angus × Holstein in the Swiss Alps, highlighting some significant differences in grassland impact. The Original Brauvieh, on average, is 100 kg lighter than the high-productive breed and prefers to graze in flat areas close to the water point. Instead, the highly productive grazer roams long distances selecting higher-quality forage influencing the grassland species composition. Thus, grazing density and breed behaviour must be taken in consideration for the sustainable soil grazing strategies. However, the wide variability of grazing densities found in literature show that grazing density alone is not a good indicator of sustainability and must be completed by assessment with SQIs.

Table 2. Overview of the classification of grazing intensity (LSU ha^{-1}) in different studies.

Country	***Light grazing (LSU ha^{-1})***	***Moderate grazing (LSU ha^{-1})***	***Heavy grazing (LSU ha^{-1})***	***Reference***
China, Nanzhang county	0.16	1.75	2.58	(Wei et al., 2023)
China, Gansu Province	2.7	5.3	8.7	(Wang et al., 2023)
Ethiopian	0.48	1.44	2.4	(Pauler et al., 2019a)
Nebraska	2		4	(Blanco-Canqui et al., 2016)
North Dakota	1.04	2.16	3.52	(Patton et al., 2007)
Canada		1.92	3.84	(Zhang et al., 2020)

Colorado	0.8	1.2	2	(Derner and Hart, 2005)

[1] The livestock unit (LSU) is the European stocking rate reference unit. 1 LSU is equal to one adult dairy cow producing 3 000 kg of milk annually, without additional concentrated foodstuffs.

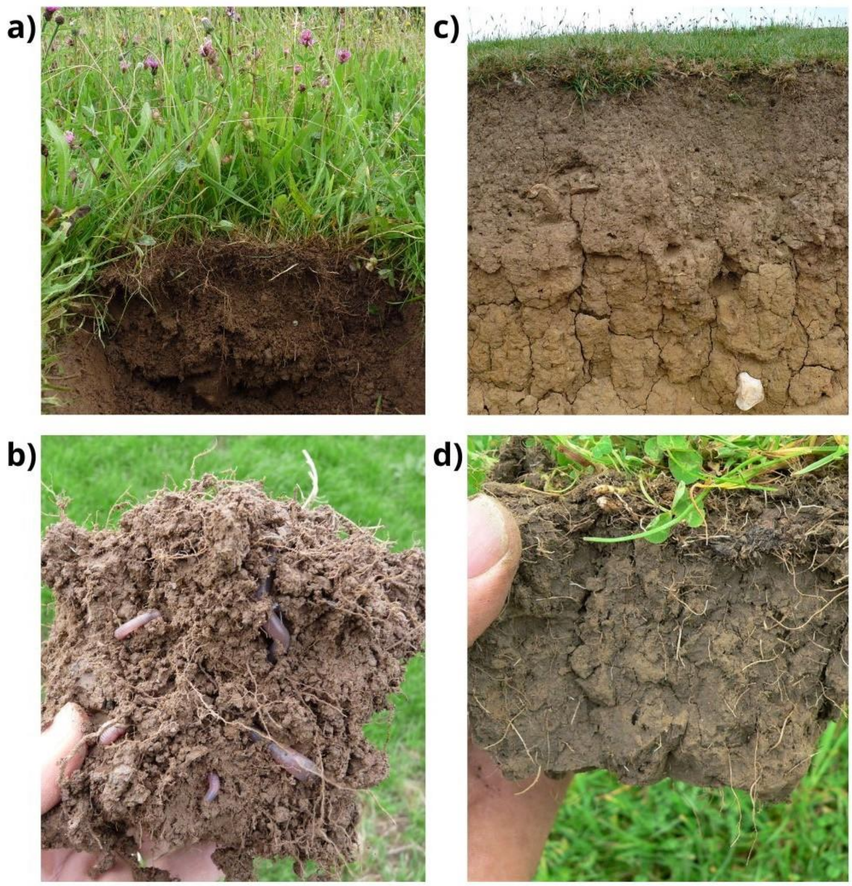


Figure 1. No-compacted grassland soil vs compacted soil (Northern England). a) zoom on no-compacted grassland soil; b) zoom on the non-compacted soil layer; c) zoom on compacted grassland soil; d) zoom on the compacted grassland soil layer (R. Smith 2023).

5. Grazing strategies for grassland soil conservation

In mountain regions of Europe, eco-climatic, topographic and vegetation characteristics of pastures can widely vary even in small spatial ranges then affecting overall stocking rates and fine-scale livestock site use intensity (Pittarello et al., 2021). In turn, animal excreta are heterogeneously distributed over the pastures, consequently influencing soil features, nutrient availability and biocycling and, thus, plant species composition. Defining a numerical threshold of each grazing management intensity is becoming an important need to prevent grassland degradation and mitigate the future soil loss and flooding hazard due to climate change. Therefore, the objective for sustainable grazing management should be to address the enhancement of grazing spatial distribution for a more homogeneous exploitation of the pastures by livestock. When livestock is allowed to roam freely, they show a selective and spatially aggregated grazing pattern (Probo et al., 2014), which leads to the overgrazing of most favourable areas (e.g. flat areas, near to water sources, etc.), Figure 2. A Grazing Management Plan (GMP) is a tool that has been successfully adopted in North-West Italian Alps (Perotti et al., 2018; Pittarello et al., 2019) and funded by the 2007-2014 EU Rural Development Program with the purpose of enhancing farm productivity and, at the same time, preserving plant and animal biodiversity, soil, and landscape. To obtain a more even selection of available resources and hence reducing local overgrazing, GMP defines grazing management practices aimed at balancing the animal stocking rate with the grassland carrying capacity (Allen et al., 2011). This means that, when considering the forage productivity and quality, grazing will occur over an area for a defined time period without causing degradation of the grazing land. To accomplish this, pastures are subdivided in paddocks grazed in rotation so that livestock is induced to homogeneously exploit the available resources while limiting overgrazing as much as possible (Probo et al., 2014). However, different studies comparing continuous and rotational grazing found small differences between the two management regimes in terms of grass production, underlining the importance of stocking rate and climate condition as distinctive degradation drivers (Briske et al., 2008; Zhou et al., 2019). Virgilio et al.(2019) performed a meta-analysis on the effect of grazing strategies on different indicators of rangeland sustainability, such as vegetation dynamics and soil quality. They found that multiple species grazing before complete destocking can ameliorate the vegetation composition of the grass layer. Rotational grazing has a minor impact on the vegetation status compared to continuous grazing, even if the impact of the latter is strictly related to livestock density. Indeed, according to them, livestock density is the main factor of grass and soil degradation. Regardless of the grazing strategy, some measures can be applied to avoid grassland degradation, for example, attractive points such as drinking and feeding troughs and salt supplementations can be placed in underused areas (e.g. steep and shrub-encroached sites) to enhance livestock spatial distribution and reduce overgrazing in the most accessible sites (Pittarello et al., 2016). Moreover, it is necessary to herd livestock into barns when the pasture soil is wet or saturated or, when possible, to reduce the length of the grazing period and to avoid rainy seasons. This minimizes the soil disturbance and can represent other valuable solutions to avoid overgrazing (Bilotta et al., 2007).

a)

b)

Figure 2: Grassland degradation due to overgrazing and trampling, in a) near to water sources of Northern France (F. Milazzo 2022), in b) in a flat clay soil area of United Kingdom (R. Smith 2023).

6. New grass species for grassland soil resilience

In addition to overgrazing, warmer and drier weather due to climate change is threatening grasslands by reducing grass diversity and productivity. Therefore, future new experiments need to consider new management practices such as grass species resilience (Li et al., 2018) not only to ensure productivity but also to preserve grassland soil. Grassland soil quality is strongly related to vegetation health, indeed the reduction of some species may decrease the soil carbon stock (Larreguy et al., 2017). Moreover, in degraded grassland prolonged drought situations with high CO2 emissions, can deplete the soil microbial community and promote a shift of the telluric biodiversity, decreasing SOC stock and modifying biochemicals cycles (Barnard et al., 2006; Pinay et al., 2007). Furthermore, vegetation cover is a principal factor that influences soil erosion rates in grasslands. The capacity to resist erosion greatly depends on the traits of the specific grassland plant community (Garnier et al., 2007; Macleod et al., 2013; Volaire et al., 2014). Grassland species and varieties differ in their capacity to store water, stabilise soil with their root systems and increase SOM content, all of which are important factors in determining soil erosion rates (Gyssels et al., 2005; Jones et al., 2012). As such, the establishment of new species and varieties into grassland communities can be an important technique for mitigating soil erosion. This can be achieved through increasing the functional diversity and species richness of grasslands, or through the development of novel breeds or cultivars with desirable traits, which can then be incorporated into the grassland community. In areas experiencing severe soil erosion or where soil erosion rates are predicted to increase due to climate change and land-use change, for example semi-arid areas of southern Europe (Kairis et al., 2015), establishment of grassland communities that ensure ecological stability, is a key adaptation measure (Volaire et al., 2014). One way of increasing ecological stability is through promoting or establishing greater plant functional diversity in the grassland community (Quijas et al., 2010). In many parts of the world, efforts to reduce soil erosion through establishment of new grassland species have not met expectations. Partly to blame for this has been the use of mono-cultures with a simple root structure, which are therefore inefficient at reducing soil erosion compared to areas with greater community functional diversity (Zhu et al., 2015). It is common practice for species mixtures to be sown or encouraged on permanent grasslands to promote multifunctionality and encourage resilience to environmental stresses including soil erosion (Humphreys et al., 2014). Individual grassland plant traits are an important consideration when choosing species and mixtures that will deliver desired services such as reducing soil erosion. For example, belowground biomass, organic matter contribution by roots and productivity are all important plant traits that can greatly affect the capacity of a grassland system to resist soil erosion due to trampling (Garnier et al., 2007). A meta-analysis of studies in which plant species diversity was manipulated, found an overall positive effect of increasing plant diversity on belowground biomass, which was considered a key indicator of erosion control (Quijas et al., 2010). In their investigation of grassland restoration efforts aimed at reducing soil erosion, Zhu et al. (2015) showed that communities with a smaller root diameter and greater root tensile strength exerted the greatest control over soil erosion. Medicago sativa is a perennial legume that, as well as being a protein rich forage species, is planted for its ability to protect the soil from wind and water erosion through its deep roots that stabilise soil structure (Yuan et al., 2015). The incorporation of M. sativa into species-rich grassland mixtures can simultaneously increase forage quality and reduce soil erosion, and as such is an example where multifunctionality can be increased through establishing new species into the grassland community. Novel grassland varieties may extend the depth of sub-soils and range of soil biota by rooting deeper than traditionally used species, which can enhance protection against erosion (Humphreys et al., 2014). Ahmed et al. (2014) demonstrated a high genetic diversity of Lolium perenne, the major grass forage species in temperate regions, and stated that this diversity could be exploited to breed new varieties that are adapted to, and can mitigate against, erosion risk. Furthermore, Marshall et al. (2014) showed that hybridisation between Trifolium repens and T. ambiguum affected the root structure and density of offspring plants and this could affect soil porosity and consequently impact on erosion rates. Macleod et al., (2013) hybridised perennial ryegrass (Lolium perenne) with a more stress-resistant meadow fescue (Festuca pratensis), developing a new cultivar called xFestulolium loliaceum. Over a two-year experiment, they found that L. perenne 3 F. pratensis reduced surface run-off by 51% compared to the leading English nationally recommended L. perenne species. There have also been promising results from the breeding of grass species with deeper or more extensive root systems e.g. Festulolium (ryegrass x fescue hybrid) which has a greater resource use efficiency (e.g. water), high biomass productivity and high contribution to SOC (Humphreys et al., 2003; Kell, 2011). Grassland drought resistance is associated with deep-root water uptake (Lynch, 2007). For this reason, Chicory (Cichorium intybus L.), which is a deep-rooted species (>2m), is becoming widespread in temperate and continental climates. In Denmark, Rasmussen et al. (Rasmussen et al., 2020) compared the subsoil uptaking ability of Cichorium intybus L. with Lolium perenne L. and Medicago lupulina L. , assessing that Chicory benefits most from deep soil moisture (up to 2.3m depth). in Pennsylvania Skinner (2008), introduced the Cichorium intybus L. as a deep-rooted forb, to a pasture mixture composed of orchardgrass (Dactylis glomerata L.), white clover (Trifolium repens L.) and perennial ryegrass (Lolium perenne L.), observing an increment of drought tolerance when chicory constituted more than 24% of pasture composition. Another promising grass species for the semi-arid and Mediterranean Tedera (Bituminaria bituminosa (L.) C.H. Stirton var. albomarginata). Tedera is an evergreen perennial legume that due to its physiological properties endures high water deficit also in a warm and windy areas (DaCosta and Huang, 2006; Foster et al., 2013; Peña and Peña, 2004). Moreover, it regrows faster than lucerne after harvesting/grazing, reducing the bare soil condition and yield gap, representing a near-future alternative for the Mediterranean farmer to mitigate climate change effects (Foster et al., 2015). Since soil erosion by water is one of the most widespread forms of soil degradation worldwide, the ability of these new varieties to reduce bare soil condition, store greater amounts of soil water and reduce runoff could have significant effects on soil erosion rates.

7. Conclusions

In the context of climate change and increasing grassland degradation, it is essential to understand soil quality development for the resilience of the grassland ecosystems. We show the importance here of using a variety of SQIs, including physical, chemical, and biological indicators, is crucial for achieving different sustainable international goals. Soil quality preservation and maintenance, should be considered essential for environmental quality in general (Döring et al., 2015). The application of sustainable management cannot be separated from careful monitoring of soil quality development. Indeed, the assessment of the reviewed

SQIs is a reliable strategy for undertaking sustainable and good management practices. However, the efforts to assess soil quality qualitatively and quantitatively are not new, and the standardization of indicators remains an ambitious task. Therefore, due to the site-specific soil quality, the SQIs threshold should be selected according to the base of the soil function of interest. Thus, the development of a SQIs assessment framework, also for limited data availability, can support grassland managers to preserve soil quality. Despite the current limitation of standardization, there are several initiatives aiming to harmonize soil quality information (e.i the Global Soil partnership, the Global Soil Biodiversity Atlas) at a different scale, that can support the management decisions. Sustainable grazing strategies can be implemented and adapted to promote soil quality and the related ecosystem services delivered, with the aim to overcome climate change effects. Several Grazing Management Plan programmes have been designed, and promoted by the local authorities aiming to improve the quality of the sward layer, aspiring to promote biomass production. However, the reference indicators used framework for farmers, are not generally based on soil quality. Studies, both at the European and regional levels, should open a new pathway for sustainable grazing management that promotes soil quality and thus contributes to the achievement of SDG 15.3. The test of new grassland species, drought-resistant and with desirable traits for soil protection, must be explored for the different bioregions aiming to improve grassland resilience in terms of soil protection, production, and ecosystem services delivery.

Acknowledgments

This research is funded by the European Union, under the Horizon 2020 project "Developing SUstainable PERmanent Grassland systems and policies (Super-G)", grant no. 774124. Views and opinions expressed are those of the author(s) only and do not necessarily reflect those of the European UnionEU or the European Research Council. Neither the European Union nor the granting authority can be held responsible for them. Vanwalleghem and Milazzo also acknowledge additional financial support from the Spanish Ministry of Science and Innovation, the Spanish State Research Agency, and through the Severo Ochoa and María de Maeztu Program for Centers and Units of Excellence in R&D (Ref. CEX2019-000968-M). Finally we would like to knowledge Richard Smith from the English Environmental Agency for providing us some important pictures.

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Chapter 4: The resilience of soil erosion rates under historical land use change in agroecosystems of Southern Spain

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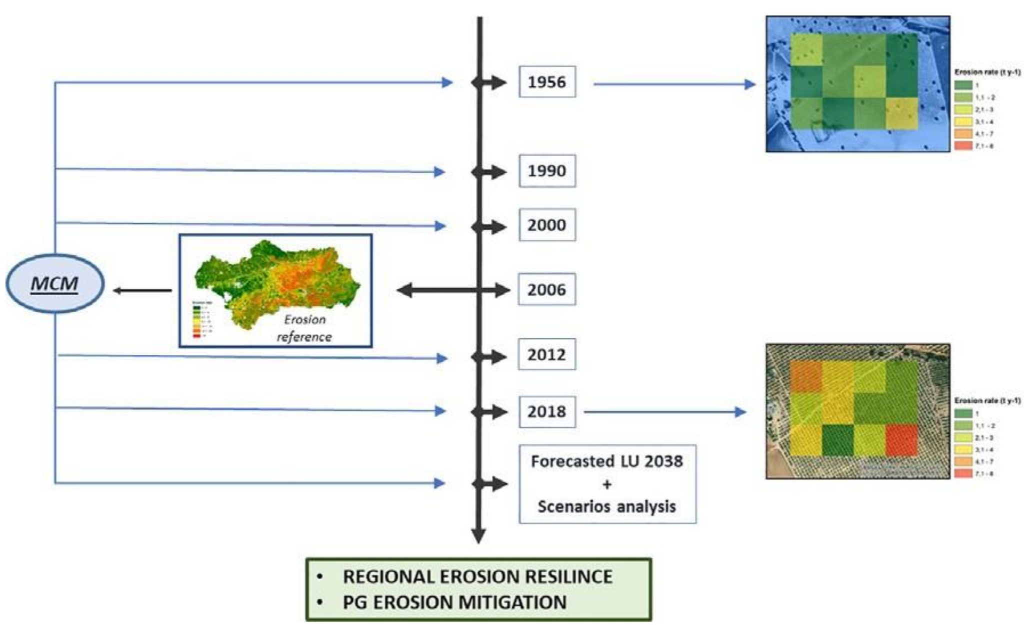
Abstract

Land use change (LUC) is identified as one of the main drivers of soil erosion in the Mediterranean. However, very little information exists regarding the relationship between land use and erosion over longer time periods and on regional scales. We quantified the LUC in Southern Spain between 1956 and 2018, examining its effect on soil erosion and assessing the mitigation role of the permanent grassland (PG). The land use influence on erosion is represented by the RUSLE's C-factor, which was modelled using the Monte Carlo Method (MCM) based on historical LUC. Moreover, future LUC scenarios by 2038 were developed by binary logistic model (scFS) and by a complete conversion of PG to cropland (scPC), permanent crop (scPP) and forest and natural (scFP). Historically, Southern Spain has experienced an impressive intensification of its agricultural system. While soil loss variation is noted within the classes, no big varia- tion is observed in cumulative erosion on a regional scale. The underlying reasons for this resilience are multifold, but mainly attributed to the fact that a small fraction of the total surface (20%), dominates total erosion (67%). The C- factor decrease in this area displays a LUC towards forest and natural area, suggesting an agriculture abandonment. On the other hand, the agricultural intensification that has taken place in the remainder of the area, contributes much less to overall soil erosion. Future LUC scenarios illustrate the importance of PG for erosion mitigation. scFS sce- nario does not project major changes. However, scCP and scPP, show an abrupt increase in regional erosion by 13% and 14%, while scFP shows a negligible reduction of erosion close to 0%. This allows to quantify the erosion mitigation offered by maintaining the PG and should be taken into account for future agricultural policy.



Keywords: Erosion, Land use, Permanent grassland, Remote sensing, Historical erosion

1 Introduction

Human-induced land use change (LUC) is often identified as one of the main drivers of accelerated soil erosion (Borrelli et al., 2020; Luetzenburg et al., 2020; Vanwalleghem et al., 2017). Erosion depends on different environmental factors, but land use and land management is definitely the main variable that can rapidly change over time and that is directly controlled by human action (Bakker et al., 2007; Nunes et al., 2011; Yang et al., 2003). Rapid land use change and intensification have led to strongly increased erosion rates after the second half of the 19th century (Bakker et al., 2007). Recently, soil erosion by water was identified as the major soil threat in the European Union (EU) by Panagos et al. (2015c) with a soil loss rate of $9{,}7 \times 10^8$ t ha^{-1} y^{-1}. Within the EU, the Mediterranean countries are the most susceptible to erosion and comprise 49% of the EU's total annual soil erosion (Panagos et al., 2015c). Soil erosion represents a serious concern for the EU, as reflected in the new EU soil strategy that contributes to the commitments set out in the EU biodiversity strategy 2030 (European Commission, 2019). The EU Commission, with the intention to mitigate the soil loss issues and achieve the Zero Net Land Degradation target by 2030, presented a pack of policy actions aimed at preserving soil, focusing on land management and land use (Montanarella and Panagos, 2021). Efficient soil management is one of the nine key policy objectives of the new Common Agriculture Policy (2023-2027), and soil conservation is central in the Eighth Environment Action Programme, and the EU Green Deal (European Commission, 2019). Field measurements of soil erosion rates have shown important differences between land uses. According to Cerdan et al. (2010) who reviewed soil erosion plot data across the EU, forest and grassland are characterized by the lowest loss soil rates, 0,2 to 0,4 t ha^{-1} y^{-1}, whereas permanent crops and arable land have the highest rates; in particular orchards and vineyards at, 3,1 and 17,4 t ha^{-1} y^{-1}, and arable crops at 3,6 t ha^{-1} y^{-1}. Any land use changes involving forests and grasslands can therefore have a major impact on soil erosion dynamics and sustainability. Different studies have addressed land use change dynamics at regional scales in the EU (EAA, 2017; Kuemmerle et al., 2016), and its effects on different ecosystem services, carbon sequestration for example (Cruickshank et al., 2000 and Gemitzi et al., 2021). Historical soil erosion modelling and its correlation with LUC is often carried out at a local or watershed scale, dealing with the reconstruction of land management and rainfall intensity (Kijowska-Strugała et al., 2018; Vanwalleghem et al., 2017). While studies on the effect of land use change on soil erosion dynamics at the catchment or regional scale (aprox. 10^3 - 10^4 km^2) are common however, studies on larger scales, and especially over longer time periods, are much less frequent. Borrelli et al. (2021) reviewed a total of 1697 articles on soil erosion modelling; of these, only 20,4% addressed land use change, and only 67 studies (2%) addressed the national scale. The studies that span time scales longer than a decade are even more limited. For example, Borrelli et al. (2017), evaluated the effect of global land use change dynamics on soil erosion, but the period evaluated was limited to 2001 – 2012. Panagos et al. (2021) forecasted the increase of soil erosion rates in the EU and the UK by 2050 due to land use dynamics and climate change, asserting that the conversion of cropland to pasture may reduce the continental erosion rate by 3%. The impact of land use and land cover management on erosion is well represented by the cover-crop factor (C-factor) of the Revised Universal Soil Loss Equation (RUSLE). This C-factor is a combination of different determinants such as tillage, soil cover, crop type and vegetation density. It reflects the effect of land cover management on soil erosion and ranges between 0 and 1, corresponding respectively to high-density forest without erosion and bare soil (Kinnell, 2010; Panagos et al., 2015b). Panagos et al. (2015b) mapped C-factors at a regional scale across EU. They reported the highest values for arable land (0,20-0,50), followed by permanent crops, such as olive orchards and tree cultivation (0,1-0,45). Natural grassland, forestry and agroforestry land uses had lower values than 0,1. Over the last few decades in the Mediterranean, particularly in Southern Spain, notable land use changes can be observed. Gómez et al. (2014) documented the expansion of olive cultivation since the late 18th century from 0,411 Mha to the present-day figure of 1,496 Mha. Local case-studies have shown that increased olive cultivation led to an increase in soil erosion rates (Gómez et al., 2014; Vanwalleghem et al., 2011), although no regional analysis has been carried out taking into account the complete land use dynamics. This is important, as not only the surface area, but also the spatial allocation of the land use is important for erosion dynamics. Adding to this complexity, in the highly variable environment of southern Spain, natural constraints such as soil type, rainfall or temperature regimes, limit the land's suitability for agricultural crops (Zabel, 2020). While it is well-known that permanent crops in the Mediterranean, especially olive orchards, have expanded greatly over the last 150 years, they have mostly replaced rainfed cereal cropping systems, which, on sloping lands, are also characterized by high erosion rates, as the soil is unprotected for the majority of the year (García-Ruiz et al., 2013). Therefore, at present, there is a need to assess the overall effect of the land use changes that have taken place over the last decades in the Mediterranean. In southern EU, Mediterranean forests, shrubland and areas of permanent grassland (PG) are crucial for soil protection (Liu et al., 2020; Torralba et al., 2016). The EU defines PG as land permanently used to grow grasses or other herbaceous forage for five years or longer (European Commission, 2007). In the Mediterranean, PG includes natural grasslands and agroforestry land uses (European Environmental Agency, 2019; Moreno and Pulido, 2012). The CORINE land use classification defines natural grassland as constituted by a PG with low human pressure and productivity, and agroforestry as the typical oak-woodland savanna named Dehesa or Montado, made up of 10-30% tree species (*Quercus suber, Q. rotundifolia),* as well as PG (>5 years old) where the soil surface is almost completely covered, or under the tree canopy (European Environmental Agency, 2019; Moreno and Pulido, 2012). In the Mediterranean, PG is dominated by this latter category of Dehesa or Montado, in terms of surface area, due to its ecological fitness that perfectly endures the typical semi-arid climate. Covering 1,5 Mha in EU and 1,0 Mha in North Africa (Bugalho et al., 2011; Pulido et al., 2001), PG is the most widespread land use within the Mediterranean zone (Porqueddu et al., 2016). Despite its importance, the area of PG in the Mediterranean is under threat, due to either land use conversion, intensification or abandonment. From the mid 1900s, grassland in the Mediterranean bioregion has shrunk considerably in favour of more intensive and profitable land uses (Auffret et al., 2018;

Kuemmerle et al., 2016; Oñate and Peco, 2005). Different studies have demonstrated this. For instance, over the last few decades, Italy has lost 29% of the Sardinian rangeland (Sedda et al., 2011), Turkish PG has decreased by 70% in the last 70 years (Bozkurt and Kaya, 2010; Celik, 2005) and furthermore, Tunisia and Morocco have lost a remarkable area of their agroforestry system, 40.000 ha in Tunisia and 190 ha in Morocco, which equates to21% of the rural land use (Chebli et al., 2018; Touhami et al., 2020; Wolpert et al., 2020). Since 1956, Southern Spain has also experienced the same impressive land use change, converting its agroforestry systems to arable land causing an increase the soil erosion problem (Anaya-Romero et al., 2011; Ledesma García, 2017). Land use change dynamics related to PG could also be affected by complex feedback loops. On one hand, where forests are replaced by PG, high stocking rates can lead to soil compaction and increased runoff and erosion, but on the other hand, the clearing of understorey vegetation in grassland areas reduces fire activity and erosion risk (Urbieta et al., 2019).

The main novelty of this work resides in the large temporal and spatial scale analysed, spanning several decades and an area of more than $8x10^6$ ha. A combination of erosion modelling and scenario analysis at this scale is rare. Only a few similar studies exist, but these are geared towards future scenarios only (Borrelli et al., 2020; Panagos et al., 2021), and this study is probably the first historical erosion study at regional scale. It is also the first to quantify the role of PG in erosion mitigation specifically. This study aims to quantify the long-term effect of land use change on soil erosion on a regional scale in Southern Spain, with a particular attention to the role of PG. For this, the specific objectives are:

(i) To quantify and analyze the historical Land Use (LU) changes between 1956 and 2018 in Southern Spain.

(ii) To predict forecasted LUC by 2038, using a binary logistic regression model.

(iii) To quantify the current importance of PG for soil conservation, by analyzing extreme LUC change scenarios based on expert opinion.

(iv) To calculate soil erosion rates for past, future and potential land use scenarios, by varying the C-factor of the RUSLE model through a Monte Carlo approach.

2 Materials and Methods

2.1 Study area

This study focuses on the southern region of Spain, Andalusia, located between latitudes 36° 00′ and 38° 44′ N and longitudes 1° 30′ and 7° 45′ W. The area covers 8.370.150 ha and is shown in figure 1. The regional climate is Mediterranean with an annual rainfall of 586mm, a mean annual temperature of 14,7°C, and a mean reference evapotranspiration of 830mm (Abd-Elmabod et al., 2020). The principal soil types are Cambisols 33% (sandy-loam), Regosols 20% (clay-loam), Luvisols 13% (loam) and Leptosols 11% (sandy), Vertisol 23% (clay) (Mudarra et al., 1988). Forestry and agriculture are the dominant land uses. Forest is typically composed of Mediterranean evergreen species such as oak, pine, firs and machis. Agricultural surface area is dominated by permanent crops, in particular olive trees, and to a lesser extent rainfed cereals and sunflowers (Junta de Andalucía, 2020). About 14% of the land is currently occupied by PG for livestock feed (Muñoz-Rojas et al., 2011).

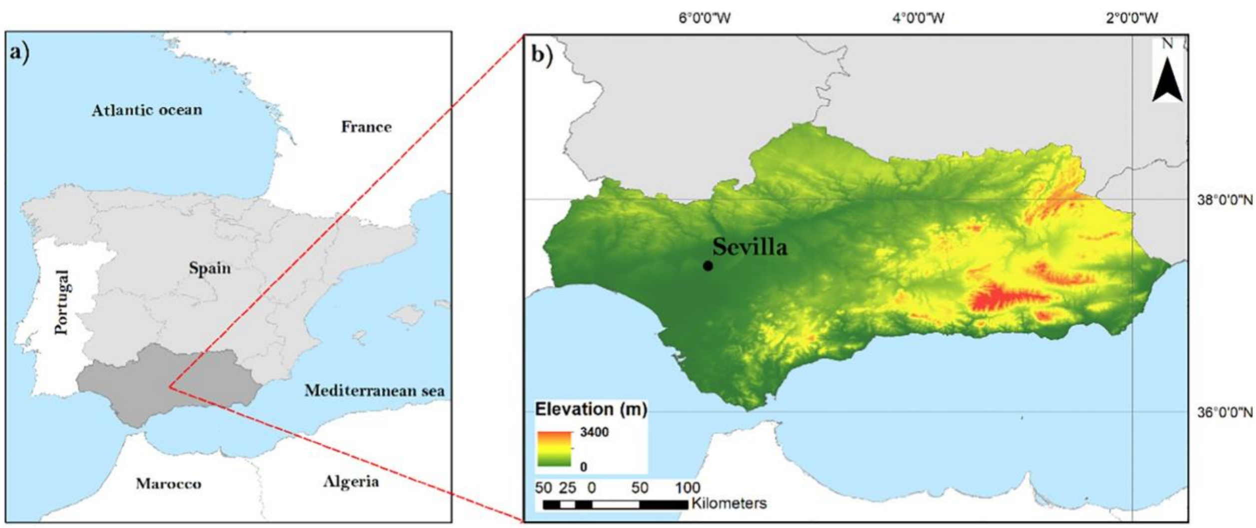


Fig. 1. Location and topography of Southern Spain

2.2 Classification of land use and land cover

The land use classification used for this study is derived from the CORINE land use map of Southern Spain, dating from 1990 to 2018. The CLC classification consists of 3 levels and is classified by 5 major categories at level 1: artificial surfaces, agricultural areas, forests and semi-natural areas, wetlands and water bodies (European Environmental Agency, 2019), and 44 different classes at level 3. In this study, we reclassified the original CLC classes into four main land use categories, as shown in Table 1. To ease the computational process of future land use modelling, "artificial surface" (CLC code 1.1.1 - 1.4.2) and "wetland and water bodies" (CLC code 4.1.1 - 5.2.3) classes have been excluded, because they do not generate soil erosion. The reclassification was needed for two main reasons. The first is that permanent crops, especially olive groves (CLC code 2.2.3), are very important in Southern Spain, and it is therefore important to separate the agricultural area into two separate classes in order to understand land use dynamics. On the one hand, permanent crops, and on the other hand, other agricultural crops, which include mostly non-irrigated arable land (CLC code 2.1.1). The second, and principal reason for this reclassification, is to allow for mapping the dynamics of PG. PG in Southern Spain chiefly consists of two CLC land use classes, natural grasslands (CLC code 3.2.1) and agro-forestry areas (CLC code 2.4.4). The latter are the typical oak-woodland savanna or 'Dehesa' agroecosystems ('Montado' in Portugal). These are woodland areas of Mediterranean oak (Quercus ilex, Q. suber) transformed by humans to provide pasture for livestock. In the CLC they are classified under forest and semi-natural areas, but in terms of erosion behaviour, they deserve to be treated separately, as their intense management has important implications for soil cover and fire incidence. The reclassified class “forest and natural area” includes all the level 3 classes of the original CLC level 1 class of “forest and

Category (used in this study)	***CLC classes***

seminatural areas”, except the natural grasslands (CLC code 3.2.1) that were included under the reclassified PG class. This includes all the vegetated forest land uses, all with a highly protective vegetative soil cover, but also scarcely vegetated area such as: dunes and sand (CLC code 3.3.1), bare rocks (CLC code 3.3.2), sparsely vegetated areas (CLC code 3.3.3), burnt areas (CLC code 3.3.4) and glaciers and perpetual snow (CLC code 3.3.5). The reason to include burnt areas, with a much higher erosion susceptibility, together with undisturbed forest areas in the same land use class is twofold. On the one hand, the original CLC also pools them together at level 1, and as mentioned before, for computational reasons only 4-5 classes can be handled in the land use modelling. Secondly, forest fires are an inherent part of Mediterranean forests. In that sense, since the objective is to calculate average soil erosion rates for periods of 6-10 years or longer, burnt patches can appear and disappear again. Data shows that within 2-3 years after burning, vegetation largely recovers (e.g Fernandez-Manso et al., 2016). To extend the analysis of historical land use to 1956, the land use and plant cover map of Southern Spain in 1956 was used

at a scale of 1:25,000 (MUCV25_56). The map is based on photo interpretation and consists of 162 classes (Junta de Andalucía, 1999). To have more homogeneous information, MUCV25_56 was reclassified into the same four main land use classes following the CLC nomenclature in Table 1.

Cropland	2.1.1 Non-irrigated arable land 2.1.2 Permanently irrigated 2.1.3 Rice fields 2.3.1 Pasture 2.4.1 Annual crops associated with permanent crops 2.4.2 Complex cultivation patterns 2.4.3 Land principally occupied by agriculture, with significant areas of natural vegetation
Permanent crop	2.2.1 Vineyards 2.2.2 Fruit trees and berry plantations 2.2.3 Olive groves
Permanent Grassland	2.4.4 Agro-forestry areas 3.2.1 Natural grasslands
Forest and natural area	3.1.1 Broad-leaved forest 3.1.2 Coniferous forest 3.1.3 Mixed forest 3.2.2 Moors and heathland 3.2.3 Sclerophyllous vegetation 3.2.4 Transitional woodland-shrub 3.3.1 Beaches, dunes, sands 3.3.2 Bare rocks 3.3.3 Sparsely vegetated areas 3.3.4 Burnt areas 3.3.5 Glaciers and perpetual snow

Table 1: Correspondence between land use classes used in this study (left) and CLC (right)

2.3 Analysis of future land use and land cover changes

A future scenario (scFS) was stochastically forecasted by the R package lulcc (Moulds, 2019) developed in R (R Core Team, 2020). Lulcc provides a flexible framework for land use modelling that allows the user to modify the source code and processes the input raster data in order to fit, evaluate and validate the spatially allocated changes (Moulds et al., 2015). This extensible R environment package provides two different land use and land cover changes (LUC) methods: first, the Ordered Model, which takes into consideration the perceived socio-economic value of each LU, based on the algorithm reported by Moulds (2015); and second, the CLUE-S model that performs the allocation of the spatial LUC upon statistical analysis (Verburg et al., 2013). In this study, the CLUE-S model with binary logistic regression was used, developing a possible LUC scenario based on the land use change class suitability for 2038. This model assumes that the LU input data is identically distributed, supposing moreover, their statistical independence (Overmars et al., 2003). To prevent autocorrelation of data, which reduces information by the prediction of its neighbours (Beale et al., 2010), the model was fitted by a random subset involving the 10% of data as described by Moulds (2015). The Southern Spain LUC was forecasted using only environmental data as input, specifically: the reclassified CORINE maps from 2000 to 2018 (note that the land cover maps of 1990 and 1956 were considered due to memory and computational limitations); the annual mean rainfall (Junta de Andalucía, 2013); the soil texture (Ballabio et al., 2016); the elevation (Junta de Andalucía, 2007); the slope length and steepness factor (Panagos et al., 2015a). The output evaluation was carried out by comparison between the CORINE 2000, CORINE 2018 and the forecasted land use map. Moreover, a statistical evaluation was supplied by the *ROCR*-package (Sing et al., 2005). The receiver operator characteristic (ROC) is usually applied to assess the performance of the forecasting model, evaluating if the change occurs (Pontius and Parmentier, 2014). The ROC curve is a global assessment ability test to discriminate whether a specific condition is present or not. It is drawn on a graph where the x axis represents the specificity, and the y axis represents the sensitivity of the ability test. A ROC curve that matches the reference line y=x produces false-positive results at the same rate as true positive outcomes (i.e. no better than a random chance). Consequently, a diagnostic analysis with fair accuracy must have a ROC curve in the upper left part above the reference line. An AUC of 0,5 represents a test with no discriminating ability, while an AUC of 1,0 describes a test with perfect discrimination (Hoo et al., 2017). In addition to this scFS, which could be considered the most likely evolution of land use in the future, we applied three extreme LUC scenarios, where PG was substituted completely. These were developed to evaluate the importance of PG areas in erosion mitigation. Although these scenarios are not realistic, they offer a good way to quantify the ecosystem service in terms of erosion protection of PG:

- PG total change to cropland (scCP)
- PG total change to permanent crop (scPP)
- PG total change to forest and natural area (scFP)

2.4 Calculation of potential erosion

Soil erosion was calculated by the Revised Universal Soil Loss Equation (RUSLE), following the methodology applied at EU scale by (Panagos et al., 2015c):

- $A = R \times K \times LS \times C \times P$

Where:

- A is the annual soil erosion rate (t ha^{-1} y-^{1}); R is the rainfall erosivity (t h ha^{-1} MJ^{-1} mm-^{1});
- K soil erodibility (MJ ha^{-1}h^{-1}y^{-1}); LS is the combined slope length and slope angle;
- C is the cover management (adimensional); P is the support practices.

For 2006, Panagos et al. (2015c) calculated the erosion rate, and each of these factors, on a EU scale at a resolution of 100m. In order to calculate the effect of past and future LUC on soil erosion rates between 1956 and 2018, it was assumed that all factors remain constant, except the C-factor. In other words, LUC changes are reflected by changing the C-factor when studying different years.

C-factor reproduces the effects of several aspects of the land cover and land use management on the erosion rate assessed by the RUSLE equation (Borrelli and Panagos, 2020). According to the original USLE method, it corresponds to the ratio of the erosion rate in a precise period of the phonological stage, weighted by the corresponding fraction of rainfall erosivity (Renard and Kenneth, 1997). Thus, this value varies continuously over time, particularly when there is a change to farming management or policy (Panagos et al., 2015a, 2015b). Generally, C-factor is no longer assessed in the field, as described in the original method, but instead remote sensing approaches are now widely used (Vrieling, 2006; Zhang et al., 2011). Different studies have related C-factors to vegetation indices, Normalized Difference Vegetation Index (NDVI) which is converted by statistical regression or correlated to field observation (Karaburun, 2010; Vatandaşlar and Yavuz, 2017). Panagos et al. (2015b) developed a new model, LANDUM, to estimate C-factors across EU. This model distinguishes between arable land and natural lands. For arable land, the mean C-value for different land use types is integrated with specific information on management practices, as derived from the European Farm Structure Survey 2010 (Panagos et al., 2015b). For natural lands, C-factor for each land use class is modulated using the fraction of vegetation cover, as derived from remote sensing. LANDUM is available as a supplement raster of 100 m resolution for EU-28.

In this study, in order to obtain C-value maps for different years between 1956 and 2018, the dataset of C-factors from 2006 by Panagos et al. (2015b) was used, and new C-values were assigned to pixels with changing land use by means of a Monte Carlo Method (MCM). The procedure is illustrated in figure 2. First, the C-factor distribution for 2006 was analysed for each LU class (step 1, figure 2). The probability density functions obtained were then used to feed a Monte Carlo-based bootstrapping procedure (steps 2 and 3) and assigned to pixels that changed land use (step 4). In figure 3, an example pixel changes from cropland to forest and natural area and is assigned a new C-value of 0,001. This was repeated over the entire study area, so modelled values were spatially distributed over the reclassified LU maps obtaining the C-factor maps of the different studied periods (step 5). This allowed new erosion maps to be calculated for the different years (step 6). The results of the obtained empirical distributions in step 1 are shown in figure 3 based on the four land use classes used in this study. It can be seen that cropland and permanent crops are characterised by the highest C-factor. C-factors for cropland have a narrow maximum around 0,24 and a second maximum around 0,16. This is because this land use category integrates different individual CLC classes (European Environmental Agency, 2019). Permanent crop has a maximum of around 0,23. C-factors for PG and forest and natural areas are much lower, between 0 and 0,08 respectively. Consequently, after reconstructing the land use map of each year, a new C-factor was generated for each pixel. This new C-factor was derived by applying the Monte Carlo Method (MCM) to the C-factors from the reference year 2006. MCM is essentially the generation of stochastic aleatory values that could represent "naturally", as part of modelling a real system, generally efficiently applied in large simulation (Kroese et al., 2014). In this study the MCM was applied using the Bootstrapping method, avoiding the possible problem of fitting non-parametric C-factor distribution, generating new value directly from the empirical distribution shown in figure 2. Successively, the simulated values were spatially allocated to the reclassified land use maps for the different years within the study period. Finally, in the last step, the erosion rate of different study years was calculated by applying the RUSLE equation. All calculations were performed at a resolution of 100m.

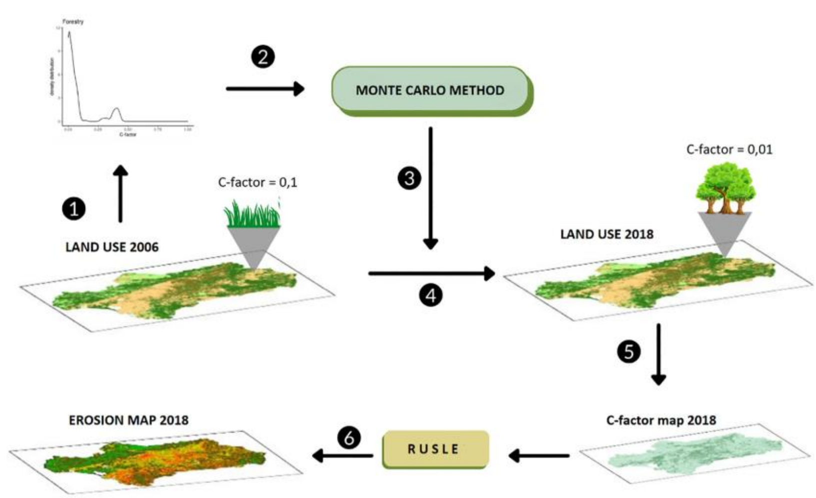


Fig. 2. Historical erosion rate model: 1) C-factor values extraction for each LU, data reference year 2006; 2) Monte Carlo method; 3) spatial allocation of the C-factor simulated values over Land cover map of different year; 5) Simulated C-factor map; 6) RUSLE model application.

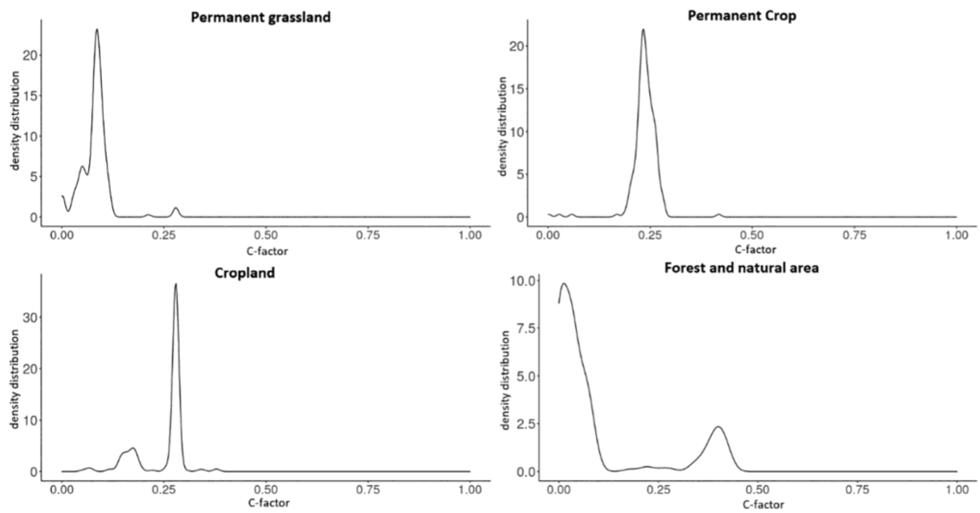


Fig. 3. Density distribution of C-factor values for the four land use classes.

3 Results and Discussion

3.1 Evolution of historical LUC

The evolution of LUC in Southern Spain over the last 62 years is shown in figure 4. In 1956, the most important land use category was forest and natural area, followed by cropland. Together, they occupied more than 59%. At that time, permanent crops, mainly olive orchards, and PG occupied only 41% of the total surface area. Since then, important changes have occurred in the different land use classes. Between 1956 and 1990, abrupt changes occurred for cropland and PG land uses: PG lost 53% of its area (9,69 x10^5 ha); permanent crop decreased by 15% (2,35 x10^5 ha); cropland, forest and natural area increased by 23% (5,18 x10^5 ha) and 33% (8,82 x10^5 ha) respectively, summing a total of 74%. This was a period of intense change in agricultural practices in Southern Spain. The end of the 1940s is characterised by a rural exodus and the introduction of mechanisation, which allowed for larger areas to be ploughed and cultivated (González de Molina et al., 2017; Simpson, 1996). Simpson (1996) reported that in 1960, approximately 40% of rainfed cereal was cultivated under fallow, while in 1990 this practice disappeared almost completely. González de Molina et al. (2017) document a sharp surge in mechanisation between 1960 to 1980, as fuel consumption increased sevenfold, from 275 to 1967 Gg y^{-1} during this 20-year period. Between 1990 and 2006, changes were relatively minor. PG remained stable while permanent crops increased slightly and the other two classes, forest and natural areas and cropland, decreased very slightly. A second phase of LUC, which affected the entire region, occurred between 2006 and 2012. PG and permanent crop area increased respectively by 43% and 20% (3,8x10^5 ha and 3,1x10^5 ha), while forest and cropland decreased abruptly by 11% and

16% (3,7 x10^5 ha and 4,1 x10^5 ha) respectively. Finally, between 2012 and 2018, land use distribution remained practically unchanged. Permanent crop and forest and natural area increased both by 2% (4,2x10^4 ha and 5,6x10^4 ha), while cropland remained stable. Only PG decreased somewhat more, by 8,5% (1,1x10^5 ha). The observed changes are similar to those in the rest of EU. Since the 1950s, the whole Mediterranean region has experienced expansion and intensification of the agricultural system. Karamesouti et al. (2015) named the period between 1950 and 1970 as the "cereal modernisation time", defining the cropland land use as the main pattern of land use in Greece. Alternatively, Roy et al. (2015) assessed a permanent crop reduction in Southern France between 1950 to 1982. Since 1990, the Mediterranean agro-system has become increasingly more intense due to agricultural policies (Feranec et al., 2010; Karamesouti et al., 2015).

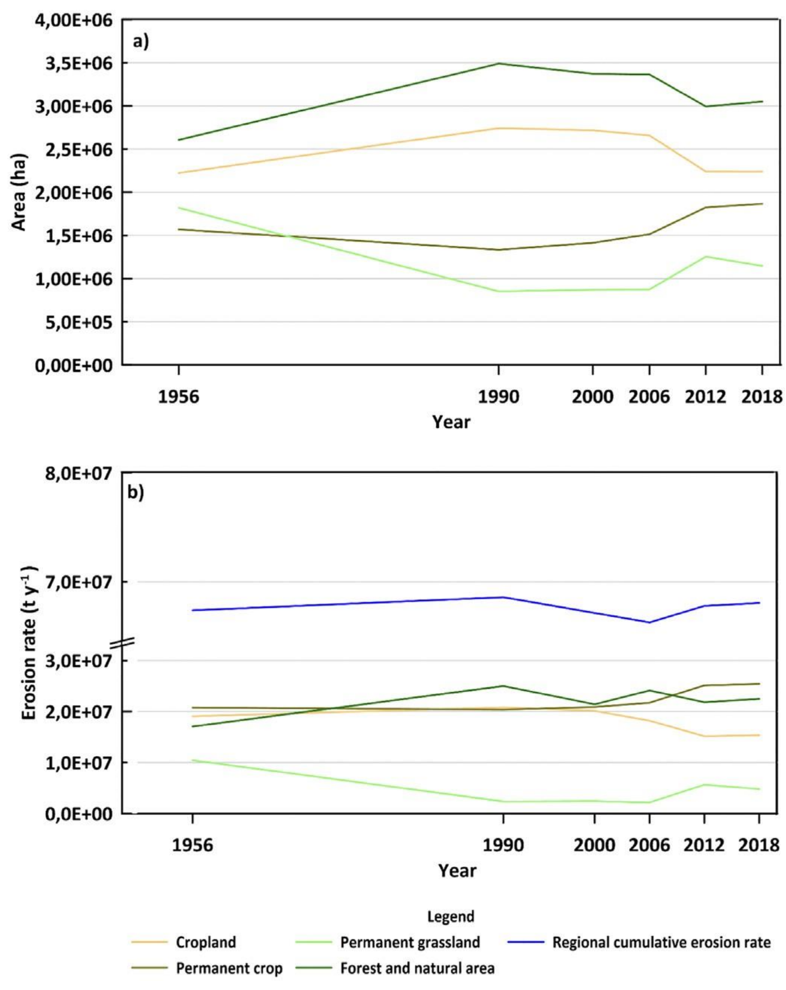


Fig. 4. a) Evolution of historical land use of Southern Spain between 1956 and 2018, summarized in four main classes. b) Evolution of the regional cumulative soil erosion rate between 1956 and 2018 in Southern Spain. Soil erosion is modelled by RUSLE and shown per land use class and cumulative.

The underlying reason for the change to PG between 2012 and 2018 is due to their nature in the South of Spain. As explained previously, PG there is mainly composed of savanna-like rangeland, so-called "Dehesa", which is a mixture of grassland and scattered trees obtained from forest cleaning and thinning (Pulido et al., 2001). If this PG area is left unmanaged without cleaning and grazing, it permits the recovery of understory shrubs and bushes. According to the European Environmental Agency (2019), an overgrown pasture with shrubs is classified as Transitional woodland-shrub (CLC 324) or Sclerophyllous vegetation (CLC 323), which corresponds to the forest and natural area LU of our classification, rather than Agro-forestry (CLC 244) which we classified as PG. As a result of the absence of human intervention, jointly with favourable climate conditions, Dehesa systems can easily shift back to forest over the course of a few years. Therefore, while no structural changes in the Dehesa system have occurred after 2006, LUC can be observed in figure 4, which corresponds to real changes in the vegetation and consequently in erosion rates. Overall, the most impressive LUC observed in this study is the reduction of PG and increase of cropland between 1956 and 1990. PG has decreased by less than half of its extension since 1956, although its area has recovered somewhat in recent years. Cropland, characterised mainly by arable crop production, has increased considerably since 1956, reaching its highest peak in 1990. However, it decreased by 21,9% ($5{,}0 \times 10^5$ ha) between 1990 and 2018, predominantly due to the expansion of the permanent crop area. This expansion of permanent crops after 1990 is the second focal change. This land use class, which consists mainly of olive orchards, has helped the South of Spain become the first olive producer in the country and one of the major exporters worldwide (Millán et al., 2014). These changes are of course driven by socio-economic changes and political actions, with direct effects on soil erosion. In Andalusia, the beginning of the study period in 1956 can be considered the highlight of the reverse migration to rural areas that started after the Spanish Civil War (Cabrera et al., 2015). After the access of Spain to the EU Economic Community in 1986, the agricultural sector experienced a profound transformation accelerated by the implementation of the EU Common Agricultural Policy (CAP) and related reforestation and afforestation programs. The first CAP in the eighties was characterized by price guarantee, emphasizing production, while the 1992 reform implied a paradigm shift towards a more sustainable and competitive agriculture, that was confirmed in successive reforms (European Commission, 2022). The influence of the CAP policy on the olive tree agriculture provided an additional income not subjected to the weather variability, thereby making it more attractive. It also allowed maintaining olives in mountainous areas where crop yields are marginal, but where agriculture is important for the vitality of rural communities. On the other hand, the increase of irrigation, through EU funds dedicated to modernizing agriculture, also contributed to the olive expansion observed in the early 2000s (Cabrera et al., 2015). Drip irrigation is frequent in permanent olive crops, allowing cultivation also in sloping areas that are affected by erosion. An in depth analysis of the underlying reasons goes beyond the scope of this study, but more details can be found in Amores and Contreras (2009), Manos et al. (2013). In figure 4, the spatial distribution of LUC between 1956 and 2018 is shown. In 1956, PG and forest and natural area were spread across the whole region, whilst cropland occupied the flatter zones with the most fertile soil. Around the Guadalquivir river, the central part of the study area, the presence of permanent crop was more limited. In 2018, the northern part of Andalusia experienced the lowest LUC, mostly transitioning from forest and natural area to PG, and vice versa. In contrast, in the south, PG was nearly entirely converted to forest and natural area. The LUC from PG to forest and natural area can occur easily due to the fact that the Dehesa system is created and maintained by bushland and forestry cleaning operations (Pulido et al., 2001; Sanjuán et al., 2018). Moreover, in the central part of the study area, important areas of cropland were converted to permanent crop, from the northeast to the central part of the region. Indeed, after 2014 in Spain, the revenue of crop land decreased notably (Ministerio de Agricultura, Alimentación y Medio Ambiente, 2020), so this, jointly with the rural development policy of 2014-2020 that supported both conversion and maintenance of organic orchard farming, boosted the conversion of permanent crop at the expense of cropland.

3.2 Evolution of historical erosion rates

Figure 4 shows the evolution of the erosion rates in Southern Spain between 1956 and 2018 per land use class and total. Surprisingly, the annual cumulative erosion rate does not vary widely during the studied period, only differing from 6,74 $x10^7$ t y^{-1} in 1956 to 6,80 $x10^7$ t y^{-1} in 2018. The highest peak was reached in 1990 (6,86 $x10^7$ t y^{-1}), and the lowest in 2000 (6,49 $x10^7$ t y^{-1}). This resilient behaviour with respect to erosion can be attributed to two main reasons. On one hand, despite important land use changes, the erosion behaviour between categories that replace each other is similar, so land use changes often do not result in changes in soil erosion rates. On the other hand, the spatial allocation of LUC is important, as will become clear from a deeper analysis of the variation of other RUSLE factors, in particular LS, over time. When analysing the underlying changes in individual erosion rates per LU class, indeed, figure 4 shows notable variation reflecting the previously documented land use changes. Between 1956 and 1990, there is a slight increase of the cumulative erosion rate produced by croplands and a steep increase of that produced in forest and natural areas. This is associated to an important increase in cropland and forest surface area, respectively. However, it should be noted that the mean soil erosion rate of the first class is higher than that of the latter, as can be seen in figure 5. Together with the slight increase in cumulative cropland erosion, it produced 4,5 $x10^7$ t y^{-1} (27%) more than in 1956. However, this is almost completely offset by the decline in cumulative erosion in PG and, to a lesser degree, in permanent crop. Only between 1990 and 2000 was there a reduction in cumulative erosion rates in forest and natural areas, not offset by the other classes and therefore leading to a small reduction in the overall cumulative erosion rate. Overall, cumulative erosion rates climbed back to their original level in 2012, and then remained stable up until 2018. The forest and natural area class has a high mean erosion rate value, which can be partly attributed to their location, as forests are often located on steep slopes unsuitable for agriculture or pasture, and partly to the natural pattern of this LU being characterised by degraded zones as bare rocks area, sparsely vegetated zones, and burnt area, respectively the Corine subclasses CL332, 333 and 334, which have C-values range between 0,26 and 0,48.

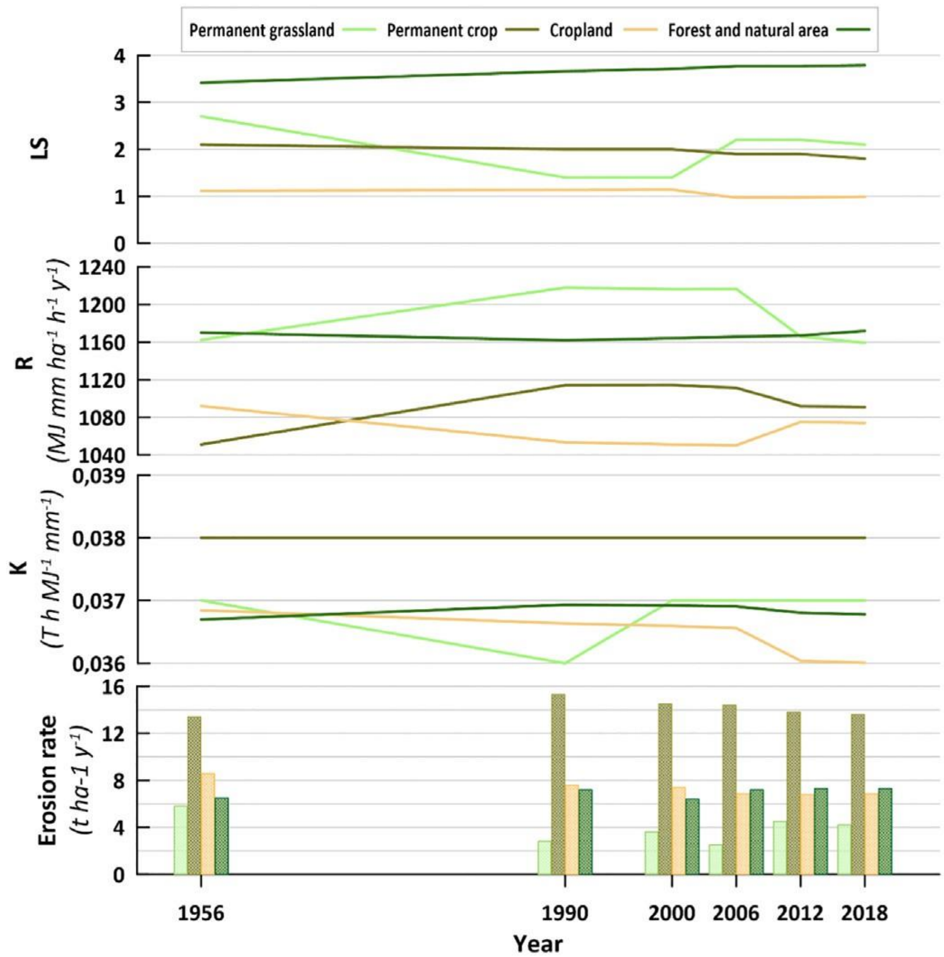


Fig. 5. Variation in the mean of the erosion factors LS, R, K and mean erosion rate between 1956 and 2018, per land use class.

Mediterranean forests compared with temperate forests, are more prone to soil erosion, especially after the drought season (García-Ruiz et al., 2013). The resulting distribution of C-factors shown in figure 2, illustrates this clearly, and while most of the forest areas are characterised by the lowest C-factor, several higher peaks in the distribution can be observed, for example around C=0,4. Cerdan et al. (2010) summarised field measurements at the EU level and under Mediterranean forest he reported an average erosion rate of 0,18 t ha^{-1} y^{-1}, two orders of magnitude higher compared to forest plots in the rest of EU. While Ricci et al. (2020) modelled the soil erosion rate for different LU in sloped areas, stating that forests mitigate the soil loss production up to 38%, Mediterranean forest vegetation is frequently disturbed by forest fires provoking short-term soil degradation phases, associated with bare soil conditions, hydrophobicity and lower aggregate stability. Therefore, erosion rates can be even higher, between 45 and 56 t ha^{-1} y^{-1} (Shakesby, 2011), until the vegetation recovers. Depending on the severity and fire frequency, that can vary between 6 and 20 years depending on site conditions (Chen et al., 2014), this can lead to long-term soil degradation. However, there is some uncertainty related to RUSLE model predictions under forest, as overestimations are possible, especially in areas with slopes greater than 5%, where Karamesouti et al. (2016) showed that RUSLE predicted up to double the erosion rates compared with the PESERA model. Due to the wide expanse of this forest and natural area class, it holds the highest cumulative annual erosion rate up until 2012, where permanent crop then overtakes, even though its expanse was 60% smaller. Permanent crops, in particular olive groves, but also vineyards, are well-known to be one of the most susceptible land uses to soil erosion within the EU, although there is a large variability. Cerdan et al. (2010) reports average soil loss rates between 1,67 t ha^{-1} y^{-1} and 8,62 t ha^{-1} y^{-1} while Gómez et al. (2009) found an average erosion rate of 23,2 t ha^{-1} y^{-1} for olive orchards. Cropland erosion rates are about the same as those in permanent crops until 2000. These two land use classes have similar C-factors, and, despite the wider extension of cropland, permanent crops are located in steeper zones having a mean LS-factor close to 2. The low LS-factor value of cropland mitigates the soil erosion rate despite the higher extension. After 2000, due to the decline of cropland area and the increase of permanent crops, respectively by -17,60% and +36,65%, the erosion rate of both classes start to diverge, and cropland erosion contributes less to the total. The expansion of intensive olive cultivation into marginal areas, which are characterized by frequent tillage, led to severe erosion problems (Borrelli et al., 2017; Kidane et al., 2019). Agriculture practices such as frequent tillage and weed management, when applied to areas of high slopes, result in serious erosion (Koulouri and Giourga, 2007; Napoli et al., 2016). The PG land use class is one that is characterised by the lowest average erosion rates. With respect to the observed resilience in soil erosion rates, we analysed the change in the different factors of the RUSLE in more detail since the variation of the erosion rate from LUC is not only linked to the C-factor change, but also to the spatial allocation of LU. This spatial allocation implies a change in the other RUSLE's factors related to topographic, soil and climatic conditions. Figure 5 shows the variation of K, R, LS factors per land use class, in addition to the mean soil erosion rate. The results show that climate and soil properties only have a limited influence of about 16% and 5% respectively, as they vary between 1139 and 1217 MJ $ha^{-1}h^{-1}y^{-1}$, and 0,036 and 0,037 t h Mh^{-1} mm^{-1}. The LS factor on the other hand varies between 1,1 and 3,8, which corresponds to a difference of almost 340%. This relative variation illustrates how the LS-factor has the most dominant influence over the variation in erosion rate. In the forest and natural area, the increase of area generally occurred in marginal and steep areas. This led to an increase of the mean LS-factor, which rose by 11% from 1956 onwards, boosting the mean erosion rate by 10%. The PG area, after the marked surface area reduction between 1956 and 1990, reduced its mean LS-factor by 48%. This implies that the remaining PG areas were in flatter zones and that it was the steeper parts that were converted to other land uses, in particular forest and natural areas, as is shown in figure 5. Consequently, the mean erosion rate of PG was reduced by 51%. After 2000, PG expanded its surface into steeper areas, leading to an increase of the mean LS-factor by 50%, and an increase of the mean soil erosion rate by 16%. In contrast, the historical LS of both permanent crop and cropland did not widely change. Cropland occupies the flattest and more fertile areas of southern Spain, with LS values between 1,11 and 0,98, while permanent crop has LS values between 1,8 and 2,1. These results suggest that the spatial allocation of the land use to areas with higher or lower LS factors in Southern Spain might be an important reason for the resilience in soil erosion rates despite land use changes. In fact, the steep areas only occupy a small fraction of the total study area and the regional erosion rates are very sensitive to what happens there. Important regional land use changes might not therefore be as influential as smaller land use changes that occur in these steep areas (Kijowska-Strugała et al., 2018; Meliho et al., 2020). This is explored in more detail in figure 6, where the relative contribution of the surface area to the cumulative erosion rate in the study area (red line, bottom panel), together with its LS factor (blue line, bottom panel) and the change in C factor of that area between 1956 and 2018 (green line, top panel) is shown. One can see how approximately 20% of the surface area (between 80% and 100% in figure 6) contributes to the 67% of the total erosion at regional scale. This area corresponds to the steepest zone in Southern Spain, as it has the highest LS-factor. However, part of the steep terrain also occupies the lower end of figure 6. At the lower end of the spectrum, 20% of the area (between 0% and the 20% in figure 6) generates only 0,59% of the total erosion, but actually corresponds to relatively high LS-factors as well. Both areas, the lowest and highest 20%, are in fact characterised by the most extreme changes in C value over the studied period. The remaining 60% of the study area is characterised by relatively stable C-values over the study period. The reason being is that this area either did not experience any land use change, or, if it did, that the land use conversion was between land use types with a similar C-value. Indeed, a conversion between permanent crop and cropland does not always imply an important change, as was discussed earlier, and as can be seen in figure 3, where both distributions peak around C = 0,25. The C-factor change, between 1956 and 2018, better explains the role of LUC and the combined influence with the LS-factor in the erosion generation.

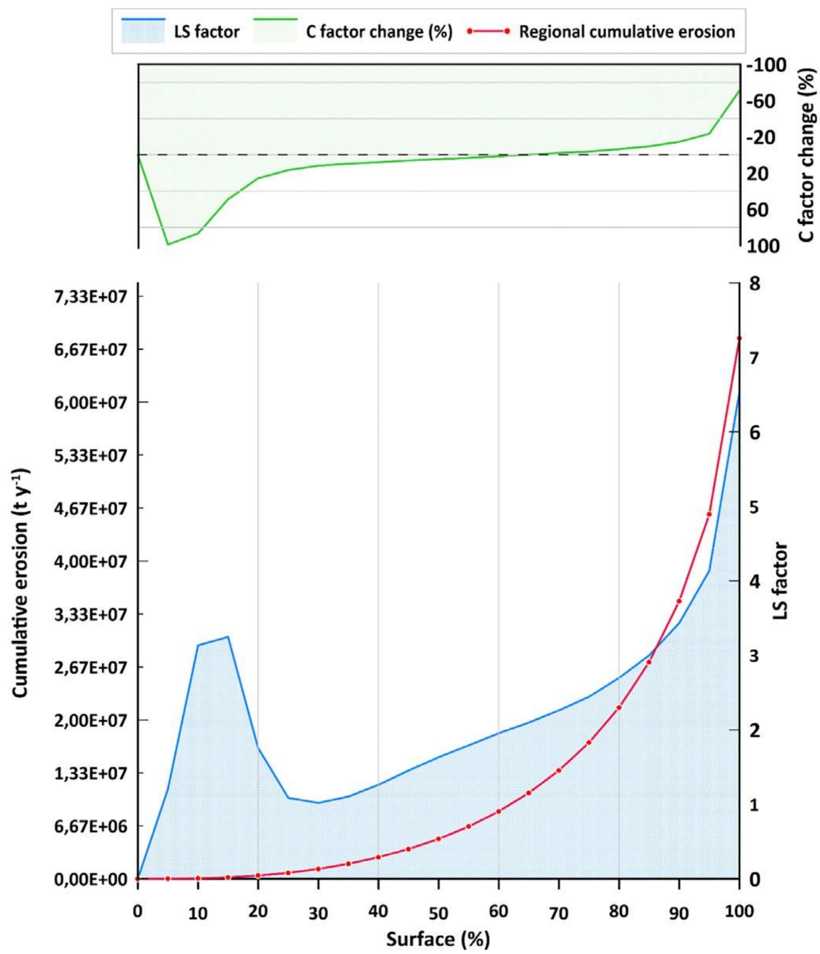


Fig. 6. Contribution of surface area in percentage to the regional cumulative erosion rate and relation with the associated LS factor, compared to the C factor percentage change between 1956 and 2018.

For instance, from 1956 and 2018, the C-factor increased by up to 100% in the flattest areas, barely influencing the total cumulative erosion; it then decreased by up to 70% in the steepest areas representing the highest cumulative erosion rate zones. These changes were mainly led by the change of rural policies that boosted the intensification of the agriculture system. As a matter of the fact, olive orchards were traditionally and commonly cultivated in marginal areas characterised by moderate and high slopes, but were then abandoned or converted into PG, with their cultivation being moved to flatter areas (Areal and Riesgo, 2014; Duarte et al., 2008; Loumou and Giourga, 2003). The results indicate that at regional level the system is quite resilient to changes in land use. This is in sharp contrast with many studies done at the local or catchment level (Amate et al., 2013; Gómez et al., 2014; Vanwalleghem et al., 2011, 2010). The rise in the soil loss rate in the Mediterranean is widely recognised to be due to the permanent crop land management intensification after the 1950s (Karamesouti et al., 2015; Roy et al., 2018). Historically in Andalusia, permanent crops have been pointed out to be the main driver of soil loss, especially after its massive expansion in the second half of the 19th century. Local studies of hill slopes and small catchments, observed that soil loss rates in olive orchards can reach 100 t ha^{-1} y^{-1} (Amate et al., 2013; Vanwalleghem et al., 2010) and reported a sharp increase in erosion rates in recent decades. Our results also confirm this, and average soil erosion rates for permanent crops are amongst the highest within the four land use classes. Its overall contribution to the soil erosion generated in Andalusia; however, is very similar to that of forests and natural areas. It was shown how the reason for this resilience is twofold: firstly, because land use changes can involve similar land use types in terms of erosion response, and secondly, because the generation of soil erosion in Southern Spain is strongly influenced by a small fraction of the total land area. The overall erosion response was found to be dominated by only 20% of the total surface. These findings show that in a Mediterranean setting with steep slopes, 20% is characterised by the highest erosion contribution. It follows that soil erosion control policies should pinpoint these erosion hotspots that are particularly sensitive to change. Similar conclusions were reached by González-Hilgado et al. (2007) with respect to the contribution of climate to the erosion response. By analysing daily soil erosion data from erosion plots, they found that in Mediterranean climates, the three most important rainfall events caused more than 50% of the erosion response. This implies that erosion can be controlled by a few single, extreme events, and it is important to assure a good ground cover to mitigate these events.

3.3 Forecasted land use change and erosion

The future 2038 LUC model forecasting ability is shown by the ROC curves in figure 7. The plot shows how well the model forecasts the cells in which change occurs. AUC for cropland and permanent grass is 0,86, while for forest it is 0,85, and for permanent crop it is 0,78. All values are significantly above 0,5, which would correspond to a completely random prediction (black line), so it can be concluded that the LUC model performs well across all land use classes. According to the output LUC forecasting model, shown in table 2, future changes will be relatively light in comparison to the historical changes documented between 1956-2018. PG and permanent crop area will lose 3% ($2,8x10^5$ ha) and 5,6 % ($4,6x10^5$ ha) of their total area compared to 2018 respectively. On the other hand, forest and natural area and cropland will grow 3,8% ($3,2x10^5$ ha) and 5,3% ($4,4x10^5$ ha) respectively. Despite the considerable LUC forecasted, the cumulative erosion rate does not importantly change, amounting to only 1% ($8,8x10^5$ t y^{-1}). The predictions in this study only take into account LUC, not climate change. Other studies have carried out projections of soil loss under climate change in the Mediterranean, and compared those to the effect of LUC. The results are quite distinct. Märker et al. (2008) simulated the future erosion rate in Mediterranean region, modelling the C-factor and the R-factor, basing the forecast on the A2 and the B1 climate change scenarios developed by IPCC (IPCC, 2000). They attributed a major role to LUC in future erosion dynamics. However, Panagos et al. (2021) ascribe to climate change the role of the main future soil erosion driver. Yang et al. (2003) forecasted the global erosion rate by 2090 and reported that the effect of climate change is larger than that of LUC.

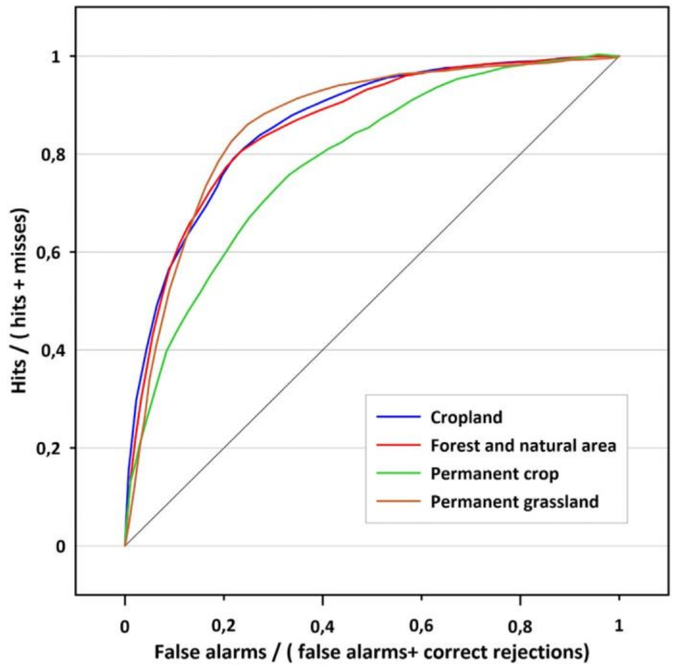


Fig. 7. Receiver-operating characteristic curves showing the ability of the binary logistic regression prediction model to predict each land use class in the data partition left out of the fitting procedure. The black line shows the expected performance of the null model.

3.4 The role of PG in erosion mitigation

PG mitigates more erosion events than other land uses (Souchère et al., 2003; Wu et al., 2020). By implementing the CLUE-S model, no important changes have been observed, making it harder to assess the soil erosion mitigation role of PG at the regional scale. To underline the importance of the PG area in the erosion risk mitigation, the soil erosion was therefore calculated under three hypothetical scenarios where the total PG area was substituted. Table 2 shows the erosion changes in these extreme scenarios, compared to 2018 for reference. In 2018, PG occupies 13,8% of the regional surface area and contributes less than 1% to the regional cumulative erosion. In scPP and scCP scenarios, the mean erosion rate of the converted area raises respectively to 11,6 and 12,3 t ha^{-1} y^{-1}, increasing the total erosion rate to $1{,}3 \times 10^{7}$ and $1{,}4 \times 10^{7}$ t y^{-1}. These changes also influence the soil loss on a regional scale, increasing the cumulative erosion by 13%, $8,8x10^6$ and 14%, $9,3x10^6$ t y^{-1}, incrementing the mean erosion rate of 1,1 t ha^{-1} y^{-1}. The total conversion of PG to forest and natural area (scFP) does not imply significant changes. This result underlines the impact of the spatial distribution of the land use, and in particular of the LS factors on the obtained erosion rates. While during the historically observed period from 1956 to 2018, forest and natural area always corresponded to higher mean erosion rates than PG (figure 5), this is not the case in the hypothetical scenario. The reason is that the MCM model simulation uses the original C-factor distribution shown in figure 3, but the spatial distribution changes. As shown in figure 5, the area in which PG is located is characterised by a lower LS-factor than that of the area covered by forest and natural area in 2018, and this greatly influences the obtained erosion rates. Figure 8, shows the spatial change of erosion rates of the extreme LUC scenarios, focusing on Cordoba province as a representative area with a high proportion. It is possible to visually appreciate the consequences of the LUC. Visual changes can be noticed between the scPP and scCP scenarios where it is possible to appreciate a raise of the erosion rate. Panagos et al. (2021) claim that the main driver of the C-factor change in EU and UK is the enlargement of the pasture area at the expense of cropland. This can mitigate the continental soil erosion by 3% by 2050. Field studies observed a lower generation of erosion in PG than permanent crop and cropland. Nunes et al. (2011) claim that in Mediterranean areas, PG should be encouraged to prevent erosion problems related to the abandonment of agricultural areas. Ceballos et al. (2002) compared the hydrology of the Dehesa system with cropland, confirming the reduction effect of PG on soil erosion.

Table 2: Erosion rate of future scenarios compared to 2018: scFS stochastically forecasted future; scCP total conversion of PG to cropland; scPP total conversion of PG to permanent crop; scFP total conversion of PG to forest and natural area.

	Permanent grassland		*Overall results*		
Scenario	*Cumulative erosion rate ($t\,y^{-1}$)*	*Mean ($t\ ha^{-1}\ y^{-1}$)*	*Cumulative change ($t\,y^{-1}$)*	*Cumulative change (%)*	*Mean ($t\ ha^{-1}\ y^{-1}$)*
2018	*$4,8x10^6$*	*4,2*	*0 [a]*	*0%*	*8,2*
scFS	*$5,5x10^6$*	*4,8*	*$8,8x10^5$ [a]*	*1%*	*8,2*
scPP	*$1,3x10^7$*	*11,6*	*$8,8\ x10^6$ [b]*	*13 %*	*9,3*
scCP	*$1,4\ x10^7$*	*12,3*	*$9,3\ x10^6$ [b]*	*14%*	*9,3*
scFP	*$4,7\ x10^6$*	*4,1*	*$-8,1x10^4$ [a]*	*0 %*	*8,2*

3.5 Model limitations

The RUSLE model used in this study is not calibrated specifically to this study area, and it is an assumption of this study that the modelled erosion rates correspond well with reality. Model validation is one of the major problems in erosion modelling, especially at regional scales (Jazouli et al., 2019). The problem with RUSLE is that it yields gross erosion rates, and not necessarily net erosion rates or even sediment delivery at the catchment scale (Alewell et al., 2019). Most model evaluation studies however focus strongly on comparing observed and predicted catchment response in terms of sediment and water output. Far fewer studies are available where a deliberate attempt is made to compare spatial distributions of predictions with spatially distributed observations (Van Oost et al., 2005). At present, no European-wide validation of RUSLE is available. Nevertheless, after a detailed review of different validation studies at local scale, Alewell et al. (2019) concluded that the RUSLE model can be considered a reliable method to examine the impact of land use change on erosion (Alewell et al., 2019; Kinnell, 2010; Terranova et al., 2009). Govers (2011) discusses misconceptions and misapplications of erosion models and also conclude that the (R)USLE model is among the best models available, and found that it cannot be outperformed by more sophisticated models due to the inherent variability of soil erosion processes (Panagos et al., 2014). Nevertheless, some care should be taken when interpreting average annual erosion rates, as also indicated by Alewell et al. (2019). Our conclusions complicate this even further as it becomes apparent that average, regional erosion rates can be insensitive to important land use changes, despite many studies at farm or catchment level that indicate a significant, but localized, increase in erosion rates due to land use changes (García-Ruiz, 2010).

Other limitations are related to the LUC forecasting applied in this study, as it does not include climate and policy projections, but is only based on past climatic and land use dynamics. Moreover, the LUC-forecasting model applied is time consuming and requires a high computational effort, for this we recommend to limit its use to catchment scale applications. Finally, the extreme LUC scenarios are not realistic but purely for to the purpose of quantifying the importance of PG area on erosion risk mitigation, with the final goal of raising awareness on the importance of this ecosystem amongst stakeholders.

3.6 Implications for management

The outcomes of the extreme LUC scenarios stress the importance of the PG area in the regional erosion risk mitigation. In Mediterranean areas, well-managed PG can provide different ecosystem services including its important role for soil erosion mitigation (Ceballos et al., 2002; Hao and Yu, 2018). PG can be used for soil conservation practices as an alternative to bushland or to mitigate the erosion enhancement of agricultural practices in crop and permanent cropland (Nunes et al., 2011). These results support the importance given to permanent pastures and their conservation in the potential eco-schemes that are being developed by EU member states. In this new policy perspective, grassland and PG habitat enhancement is widely considered within the eco-schemes of the two pillars (Bieroza et al., 2021), by implementing sustainable practices for: climate change mitigation, including reduction of GHG emissions; climate change adaptation, including actions to improve the resilience of food production systems; protection or improvement of water quality; prevention of soil degradation; and finally, protection of biodiversity (European Commission, 2021). Despite our result, PG in the Mediterranean climate can face different land degradation processes that are not considered in this study. For example, PG are more exposed to fire risk than cropland and permanent crop, and fires can create "windows" of higher soil erosion susceptibility (Pardini et al., 2017). Moreover, PG land can be exposed to bare conditions if the dry season lasts too long. This exposes the soil to high erosion events provoked by summer storms and hydrophobic soil conditions (Ceballos et al., 2002; Schnabel et al., 2010). It should be taken into account that the results presented here are based on modelling with RUSLE, and therefore any limitations related to this model also apply to the results. For example, only sheet and rill erosion are modelled. Under PG it is well known that erosion processes such as gullying can occur, or compaction related to overgrazing under inadequate management (Gutiérrez et al., 2009; Shakesby et al., 2002). We observed that only 20% of the regional area produces more than 60% of the total cumulative erosion, this is mainly influenced by the LS-factor. Future studies should be conducted to analyze this problem, proposing soil protection measures focusing on these regional erosion hotspots. Although our study has focussed strongly on the effectiveness of PG for reducing erosion in our study area, our results also show that forest yields statistically very similar results. Also from other studies it is well known that changing the land use from agricultural to forest will decrease soil erosion (Bakker et al., 2008, Panagos et al., 2015). In the light of this discussion, we can conclude that EU and national policy measures towards preserving or promoting PG, such as those reflected in the green ambitions of the CAP, but also those related to reforestation, such as the new EU forest strategy for 2030 (European Commission, 2021), are highly beneficial for mitigation of soil erosion in these erosion hotspots.

4 Conclusions

Land use change was quantified on a regional scale in Southern Spain and over a period spanning more than six decades, from 1956 to 2018. Abrupt land use dynamics were observed in this period of important socio-economic changes. Conspicuous loss of PG was observed, mostly for the benefit of forest and natural areas. Permanent crops, in particular olive orchards, also grew rapidly, while cropland first increased and then decreased towards the end of the studied period.

The variation of historical erosion rates was then calculated, using the observed frequency distribution of C-factor and a Monte Carlo approach for their spatial allocation. Despite the observed land use changes, the cumulative soil erosion rate on the regional scale remained constant, between 6,49 and 6,86 x10^7 t y^{-1}. The underlying reasons for this resilient behaviour was further explored, and, was attributed to different reasons. Firstly, land use changes occurred frequently between land use classes that had reasonably similar C-factor; cropland and permanent crops on the one hand for instance and PG and forest and natural areas on the other. This occurred within 60% of the terrain. Secondly, and possibly the most important reason, is that a small fraction of the regional area (20%) is responsible for most of the total cumulative erosion rate (67%). This area has an average LS-factor ranging 2,5 and 7,5, belonging mainly to the permanent crop and forest and natural area on higher slope gradient.

Finally, the importance of PG was quantified by applying four land use scenarios: one scenario of most probable change, based on a stochastic land use prediction model and three hypothetical scenarios of complete conversion. Future changes under the most likely scenario are predicted to be minor. However, this analysis shows how PG plays an important role on the regional cumulative erosion mitigation, as the PG total conversion to permanent crop and cropland can raise the regional cumulative erosion of 13% and 14%.

Acknowledgements

This research is funded by the European Union, under the Horizon 2020 project "Developing SUstainable PERmanent Grassland systems and policies (Super-G)", grant no. 774124. Views and opinions expressed are those of the author(s) only and do not necessarily reflect those of the European UnionEU or the European Research Council. Neither the European Union nor the granting authority can be held responsible for them. Vanwalleghem and Milazzo also acknowledge additional financial support from the Spanish Ministry of Science and Innovation, the Spanish State Research Agency, and through the Severo Ochoa and María de Maeztu Program for Centers and Units of Excellence in R&D (Ref. CEX2019-000968-M).

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Chapter 5: NDVI prediction of Mediterranean permanent grassland using soil moisture products.

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Abstract

Vegetation indexes are widely used as a proxy of vegetation status, they are often used to monitor and assess qualitatively and quantitatively the growing season. The Normalized Vegetation Index (NDVI) is the most widely used in agriculture, frequently as a proxy for different physiological and agronomical aspects, such as drought stress and crop yield losses evaluation. NDVI forecast is usually correlated to precipitation however, in Mediterranean and arid climates, it is not well correlated due to prolonged dry periods and sparse precipitation events. In this study, we forecast Mediterranean permanent grassland NDVI at 7 and 30 days ahead using machine learning and two soil moisture products as predictors, simulated soil moisture values and satellite-based Soil Water Index (SWI) values. Results show that both products can be used as reliable predictors of permanent grassland in Mediterranean areas. Predictions at 7 days are more accurate and better forecast the negative effect of drought on vegetation dynamics than 30 days. This study shows the potential of using a simple methodology and readily available data to predict the grassland growth dynamic in the Mediterranean area.

Key words: Soil moisture, grassland, SWI, NDVI

1.Introduction

Savanna-like agroforestry systems cover about 3.5 million hectares in Mediterranean Europe and about 1 million hectares in North Africa (Bugalho et al., 2011; Pulido et al., 2001). This land use is composed of scattered oak trees (Quercus rotundifolia), and permanent grassland (unrenewed herbage layer of 5 or more years) for livestock grazing (Pulido et al., 2001). In Iberian peninsula, savanna-like systems are called Dehesa in Spain and Montado in Portugal, these names refer to the fact that the land is divided into large plots bordered by stone walls, where rotational grazing (rangeland) is practised (Pulido et al., 2001). Grassland ecosystem plays an important role in sustaining the economy of marginal lands in the Mediterranean through livestock production and in preserving their endemic biodiversity and cultural heritage (Hadjigeorgiou et al., 2005; Moreno et al., 2016; Schils et al., 2022). In the Mediterranean climate, grassland production is limited due to frequent long dry summers, which cause a severe crop yield drops and hence, important economic losses. This susceptibility of Mediterranean grassland to drought is increasingly exacerbated by climate change (Brown et al., 2016; Iglesias et al., 2016). Therefore, accurate forecasts of grassland yield, in particular during dry periods, are crucial for both farmers and policymakers to apply mitigation measures and hence, ensure food security (He et al., 2019).

Different methods have been developed for predicting grassland dynamics and yield using weather forecasting. McDonnel et al. (2019), attempted 1 and 6 days forecasts of the grassland dynamics, using management inputs, such as fertiliser application, and weather inputs, such as temperature and precipitation. Trnka et al.(2006), developed an accurate grassland growth model, including as inputs, not only the weather and the fertiliser application, but also the soil moisture balance. The benefit of using soil moisture information is that it integrates weather, evapotranspiration, plant available water and hence, vegetation state information. Indeed, soil moisture is an essential driver of grassland dynamics, it can explain up to 60% of the grassland yield variability and when considered in productivity models greatly improve model performance (Krueger et al., 2021). In the literature, vegetation dynamics are commonly monitored by remote sensing techniques, and in particular by using the Normalized Difference Vegetation Index (NDVI). NDVI is a simple indicator of the vegetation greenness and it is widely applied to estimate vegetation density and crop yields (Wang et al., 2005; Xue and Su, 2017). Notably, in Spain NDVI is used for agricultural insurance purposes to quantify grassland yield losses during the growing season (Oct-Jun) and compensations due to drought or extreme weather conditions (BOE, 2022). Previous studies have shown that NDVI can be estimated from soil conditions, in particular, they show a good correlation in dry climates (Han et al., 2010). For instance, Chen et al. (2014), found a good correlation between soil moisture and NDVI in Australia's mainland. In climates characterized by prolonged dry seasons, such as South Spain, NDVI has shown to be better correlated to soil moisture than precipitation (García-Gamero et al., 2021). In the particular case of grassland areas, Wang et al. (2007) found a better correlation in semi-arid than humid regions of the USA.

Previous studies have shown the potential of using NDVI to support grazing and harvesting planning and in particular, NDVI predictions to anticipate water deficiencies and hence, yield losses (Escribano et al., 2014; Insua et al., 2019; Rodríguez and Ambrona, 2013). One approach to predict the NDVI is using autoregressive models, i.e. forecast future NDVI values using a linear combination of past NDVI values. This approach has shown high reliability in forestry land uses mainly thanks to the plant growth seasonality (Bounouh et al., 2022). Another approach not based on the use of past data, is the use of seasonal weather forecasts. Iwasaki (2009) in an arid climate, tried to predict NDVI distribution for 1-3 months using a seasonal weather forecast. They showed a weak prediction efficiency especially and advised against the use of precipitation forecasts for NDVI prediction in dry regions. Considering the NDVI as a proxy of vegetation growth, some studies have also used parametric crop growth models to forecast NDVI values. However, parametric crop growth models, have shown a low NDVI prediction accuracy (Ahmad et al., 2020) and a worse performance than an increasingly popular approach, machine-learning based methods (Berger et al., 2019).

With the advance of remote sensing methods and the informatization of the agriculture operation, machine-learning algorithms provide the possibility to develop forecasting or decision tools for land managers, farmers and other agro-forestry stakeholders (Casanova et al., 2014; Hadria et al., 2019; Htitiou et al., 2022). Machine-learning approaches provide powerful tools that are applied in different fields (Nieuwenhuizen et al., 2007; Rehman et al., 2019) such as weed detection (Pereira et al., 2012), soil analysis (Haghverdi et al., 2015), management zone clustering (Boydell and McBratney, 2002), irrigation and yield prediction and stress prediction (Liu et al., 2017; Park et al., 2017). However, predicting vegetation development remains a current challenge because several ecosystem processes affect vegetation dynamics (Anav et al., 2015; Xia et al., 2019). Currently, process-based models are not able to predict accurately the vegetation dynamic interrelating the multiple ecosystem processes that impact vegetation growth (Xia et al., 2019). For this reason, the use of machine-learning, due to its high performance, and multifold applicability quickly increases worldwide (Dokic et al., 2020). Different approaches have been widely applied to predict the vegetation dynamic, such as artificial neural networks, support vector regression, random forest and regression trees (Xie et al., 2018). These methods are characterized of the independence of the relationship between the predictors and predictive variable, particularly if compared to the traditional models as linear regression, which imply a Gaussian distribution for the input variables (Li et al., 2021). Roy (2021) compared the performance of some of the most used machine learning algorithms to forecast large-area average of NDVI in Bangladesh and showed that the Random Forest algorithm had the best performance.

In this study, we present an innovative NDVI forecasting model based on the application of the Random Forest machine learning algorithm and the use of past and present temperature and soil moisture information as predictors. Soil moisture information consists of two products: modelled daily soil moisture values and satellite-derived values of Soil Water Index (SWI) at a point and single-pixel scale respectively. Using each soil moisture product, we create two versions of the NDVI forecasting model that we tested and compared for 7-day and 30-day lead times in a Mediterranean permanent grassland.

2. Material and Methods

2.1 Study area

The study was carried out in Santa Clotilde commercial farm located in the north of Córdoba province, Southern Spain (38.2° N; 4. 17° W, 700 m a.m.s.l.). The main activity of the Santa Clotilde farm is the extensive livestock production in the Dehesa agroforestry system, Fig. 1; bovines and swine are grazing rotationally the whole year. Soil texture is sandy-loam (6,7% clay, 64% sand, 29,3% loam), due to rotational grazing, the first 30 cm of soil profile holds 70% of the total carbon stock (Román-Sánchez et al., 2018). According to the Köppen-Geiger classification, the climate is Mediterranean, with an average annual rainfall of 878 mm, cold-dry winters seasons, long summers and a mean temperature of 25.4 °C (Peel et al., 2007). Since 2017 five soil moisture sensors (Campbell Scientific CS655) were installed in grassland open-field, at 3 soil depths at 5, 15, 25 cm depth) monitoring the grassland soil moisture dynamics. In this study, we used the soil moisture readings at 25 cm depth to have comparable results to Soil Water Index, which is representative of the first 20 cm of soil profile Fig.2. Precipitation data is obtained using the SM2RAIN-ASCAT satellite-based method. SM2RAIN-ASCAT is a global product obtained from Advanced SCATterometer (ASCAT) satellite through the SM2RAIN algorithm developed by Brocca et al. (2011).

Fig. 2: Study area. Santa Clotilte Dehesa farm and sensors location

The SM2RAIN algorithm allows calculating rainfall using the inverse equation of water balance to calculate rainfall using in situ or satellite-based soil moisture data (Brocca et al., 2014, 2019a). We also estimated the satellite-based Soil Water Index (SWI) at the study area. SWI of Copernicus Global Land Service (Bauer-Marschallinger et al., 2018) is acquired from measurements of near-surface soil moisture supplied by ASCAT by means of an algorithm which summarizes and exponentially weights past measurements according to the time length T, which ranges between 001 and 100 (Wagner et al., 1999). The T factor indicates how many past observations of surface soil moisture affect the current value of SWI. Conceptually, higher delay and the increasing smoothing signal detected at the soil surface, from a higher T value, is comparable to the effect of the soil water infiltration. Thus SWI is a reliable proxy of soil moisture content at 20 cm depth (Brocca et al., 2011; Paulik et al., 2014). Specifically, this study has been selected with a value of the T-parameter equal to 20 days.

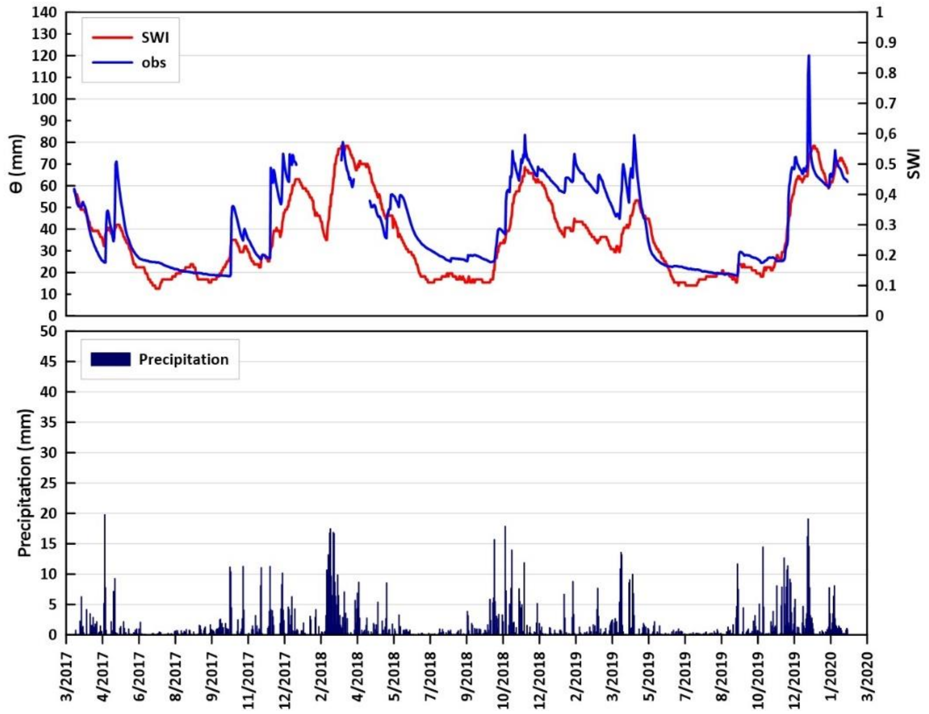

Fig. 3: Top panel – daily soil moisture information observed in the study area: the blue line is the soil moisture measured by field sensors; the red line is the Soil Water Index (SWI). Bottom panel – daily precipitation obtained using the SM2RAIN-ASCAT method.

2.2 Soil moisture model

To estimate the soil moisture dynamic of the Mediterranean permanent grassland, we modelled soil moisture using as water input the daily satellite rainfall data from ASCAT data (Brocca et al., 2019b). The soil moisture dynamic model is conceptually based on the BEACH model (Sheikh et al., 2009), which divides the soil moisture reservoir in two layers: the top layer, which depth is delimited by the root zone, water balance is determined by rainfall, evapotranspiration, runoff and deep percolation; and the passive layer where soil moisture is mainly driven by the deep percolation. In this study, we simplify the model to only represent the top layer of 25 cm depth. Irrigation input is not considered because it is a rainfed grassland.

The soil moisture model (SM25) calculates the volume of water stored in the soil (*St*; in mm) considering the stored volume from the previous day (*St*-1):

$$S_t = S_{t-1} + Rf - ET_{act} - DP \qquad (1)$$

where St is the daily soil moisture (mm); St-1 is the soil moisture of the antecedent day (mm); Rf is the net precipitation (mm); ETact is the actual evapotranspiration (mm); Dp is the deep percolation (mm).

To compute Rf that reaches the soil surface we apply a the formula proposed by Morgan and Duzant (2008):

$$Rf = R(1 - PI) \qquad (2)$$

Where R is the total daily precipitation (mm) and PI is the plant interception (mm). To calculate PI we applied the empirical equation proposed by Braden (Braden and Deutscher, 1995) as a function of the Leaf area index (LAI), the Canopy cover CC and the daily precipitation R (mm):

$$PI = aLAI\left(1 - \frac{1}{1+\frac{CCR}{aLAI}}\right) \qquad (3)$$

where PI daily plant interception (mm); a is an empirical coefficient that ranges between 0,3 (before senescence) and 0,6 (end of the senescence period) (Arnold et al., 2012); CC is canopy cover; LAI is the Leaf area index. Similar to the DREAM model (Manfreda et al., 2005) and the SWAP model (Kroese et al., 2014) the actual evapotranspiration (ETact) is calculated as a combination of the reference evapotranspiration (ET0) from the vegetated fraction (CC) and the actual evaporation from the bare soil fraction (1–CC):

$$ET_{act} = ET_{veg}\,CC + E_{soil}(1 - CC) \qquad (4)$$

where ETveg is the actual daily evapotranspiration from the vegetated fraction (in millimetres) and Esoil the actual daily evaporation of the bare soil fraction (in millimetres), shown in (5) and (6). Both ETveg and Esoil depend on the degree of water availability in the soil. The degree of water availability is expressed by actual soil moisture divided by field capacity soil moisture. This approach is based on the following assumptions (Singh, 1995) (5):

If St-1 > of water stored in the soil at field capacity:

$$ET_{veg} = ETo - PI \qquad (5)$$

$$E_{soil} = ETo \qquad (6)$$

therefore:

$$ET_{act} = (ETo - PI)CC + ETo(1 - CC) \qquad (7)$$

If St-1 < of water stored in the soil at field capacity (Sfc) and higher than at the wilting point (Swp), ETact is equal to the potential plant evapotranspiration in mm, plus the actual soil daily evaporation of the bare soil fraction in mm,(8) and (9):

$$ET_{veg} = (ETo - PI)\left(\frac{S_{t-1} - S_{wp}}{S_{fc} - S_{wp}}\right) \qquad (8)$$

$$E_{soil} = ETo\left(\frac{S_{t-1} - S_{wp}}{S_{fc} - S_{wp}}\right) \qquad (9)$$

Therefore, (10):

$$ET_{act} = ET_{veg}\ CC + E_{soil}(1 - CC) \qquad (10)$$

Deep percolation Dp was simulated by applying the same BUDGET model method (Raes et al., 2006), (11):

$$Dp = d_s\, \tau\, (\theta_{sat} - \theta_{fc})\left(\frac{e^{\theta - \theta_{fc}} - 1}{e^{(\theta_{sat} - \theta_{fc})} - 1}\right) \qquad (11)$$

Where ds is the depth of the soil A-horizon (mm); θ is the soil moisture expressed as millimetres of water depth per millimetre of soil depth; θsat is the soil moisture at saturation; θfc is the soil moisture at field capacity; τ is a drainage parameter, given by the equation (12):

$$0 \leq \tau = 0.0866^{0.8063\log_{10}(K_{sat})} \leq 1 \qquad (12)$$

where K_{sat} is the saturated hydraulic conductivity (mm d^{-1}).

2.3 Soil moisture model calibration and validation

For the soil moisture model calibration and validation, we used the observed daily soil moisture values, from 17/03/2017 to 01/02/2021. From 17/03/2017 to 23/06/2019 for the model calibration and from 24/04/2019 to 12/02/2020 for the model validation.

Model performance is evaluated using the Nash-Sutcliffe efficiency (NSE). NSE determines the relative magnitude of the residual variance compared to the observed data variance (Nash and Sutcliffe, 1970). For the model calibration computation we used the Non-dominated Sorting Genetic Algorithm (NSGA-II) and considered the following parameters: canopy cover (CC); saturated hydraulic conductivity in mm/day (K_{sat}); soil moisture ratio at wilting point in mm/mm (Swp); soil moisture ratio at field capacity in mm/mm (Sfc); soil moisture ratio at wilting saturation in mm/mm (theta_sat). NSGA-II was set to maximize NSE.

2.4. NDVI forecasting models.

The NDVI forecast model uses available NDVI, temperature and soil moisture data at present to predict NDVI values. We developed two NDVI ahead forecasting models using the selected two soil moisture products:

$NDVI_{SWI}$: NDVIt0+x~ SWI + T20 + NDVIt0

NDVI_SM25: NDVIt0 + x~ SM_25 + T20 + NDVIt0

where NDVIt0+x is the forecasted NDVI at day t0+x (7 or 30); SWI is the Soil Water index (SWI); T20 is the cumulative mean temperature of the previous 20 days; NDVIt0 is the observed NDVI value at present. They were retrieved from the Copernicus Sentinel-2 using the Google Earth Engine. Observed NDVI values are also used as a reference to compare forecasted results. Observed NDVI corresponds to an area of 21 m radius, covering a grid of approximately 3 × 3 pixels with 10-m of spatial resolution, located in the hill-plateau of Santa Clotilde farm, in open grassland avoiding the tree influence; SM_25 is the simulated soil moisture at 25 cm soil depth. Grassland's NDVI was predicted by applying the Random Forest machine learning algorithm (Breiman, 2001). This approach is composed of accumulation of singular decision trees (estimators) that allow an exceptional achievement of prediction accuracy (Zaimes et al., 2019). The training and testing of the NDVI forecast models were performed from 21/07/2015 to 30/12/2021, using 50% of the data for each one. Prediction performance was evaluated using the NSE and the Mean Bias Error (MBE). MBE is used to estimate the bias between the predicted value and the observed (Li et al., 2016). In comparison with NSE, MBE provides a view of how close the forecasts are to the measurements in absolute values, displayed respectively in (13) and (14).

$$\mathrm{NSE} = 1 - \frac{\sum_{t=1}^{n}[q_{obs}(t) - q_{sim}(t)]^2}{\sum_{t=1}^{n}[q_{obs}(t) - \bar{q}_{obs}]^2} \qquad (13)$$

$$\mathrm{MBE} = \frac{1}{n}\sum_{t=1}^{n}(q_{obs} - q_{sim}) \qquad (14)$$

Moreover, we assess the grassland vegetation response to droughts by filtering the NDVI dataset temporally. To calculate the vegetation response to environmental condition we estimate the anomalies (Z-Score) (Klisch and Atzberger, 2016) (15). Conceptually, these anomalies represent the intra-seasonal variations of NDVI in response to the fluctuation of the environmental condition (e.g. drought condition) (Kogan et al., 2003).

$$\mathrm{Z-score} = \frac{\mathrm{NDVIt} - \mathrm{NDVImean,i}}{\mathrm{NDVIstdi}} \qquad (15)$$

where $NDVI_t$ is the NDVI observed at time step t; $NDVI_{mean,i}$ is the monthly mean of the NDVI daily values; $NDVI_{std,i}$ is the monthly standard deviation of NDVI daily values. A positive or negative value of Z-Score indicates a period wetter or drier than the average, respectively. This helps us identify exceptionally dry periods which can have an important impact on grass production.

In order to evaluate the correlation between the anomalies (Z-Score) simulated by the NDVI models and the observed ones we applied Pearson's correlation (16):

$$r_{x,y} = \frac{\sum_{i=1}^{n}(x_i - \bar{x})(y_i - \bar{y})}{\sqrt{\sum_{i=1}^{n}(x_i - \bar{x})^2}\sqrt{\sum_{i=1}^{n}(y_i - \bar{y})^2}} \qquad (16)$$

where rx,y is the correlation coefficient; n is the length of the time series; i study period (in year); xi and yi are the NDVI anomaly respectively, and x and y are the mean value of NDVI. If the value of rx,y is greater than zero, it is a positive relationship; if rx,y is a negative value, it is a negative relationship; if rx,y is equal to zero there is no relationship between the two variables (Tong et al., 2017).

3. Results and Discussion

3.1 Soil moisture dynamic

The calibration and validation results of the soil moisture model yields NSE values of 0.71 and 0.70 respectively (Supplementary material). This indicates that the model is able to satisfactorily simulate the observed soil moisture. In Fig. 3 we compare the results of the soil moisture dynamic modelled over the study period. Fig. 3a displays the difference between the ground observed soil moisture and the simulated. The results show that the model generally overestimates the observed values. This can be explained due to the fact that not all the precipitation events are reflected by sensors (e.g. see dotted rectangle in Fig. 3 where wet periods are not reflected in an increase of observed soil moisture). It must be also noted that we are comparing values obtained at different spatial scales, precipitation data and hence model results are pixel values while soil moisture observations are point values. We observe differences between the two soil moisture products along the study period in how quickly they respond to precipitation. The SM25 reaches higher values of soil moisture (in mm) quicker than SWI, as it is shown in Fig. 3b. This may be due to dissimilarity issue in spatial scales between point (model) and pixel values (satellite observations) (Gruber et al., 2013).

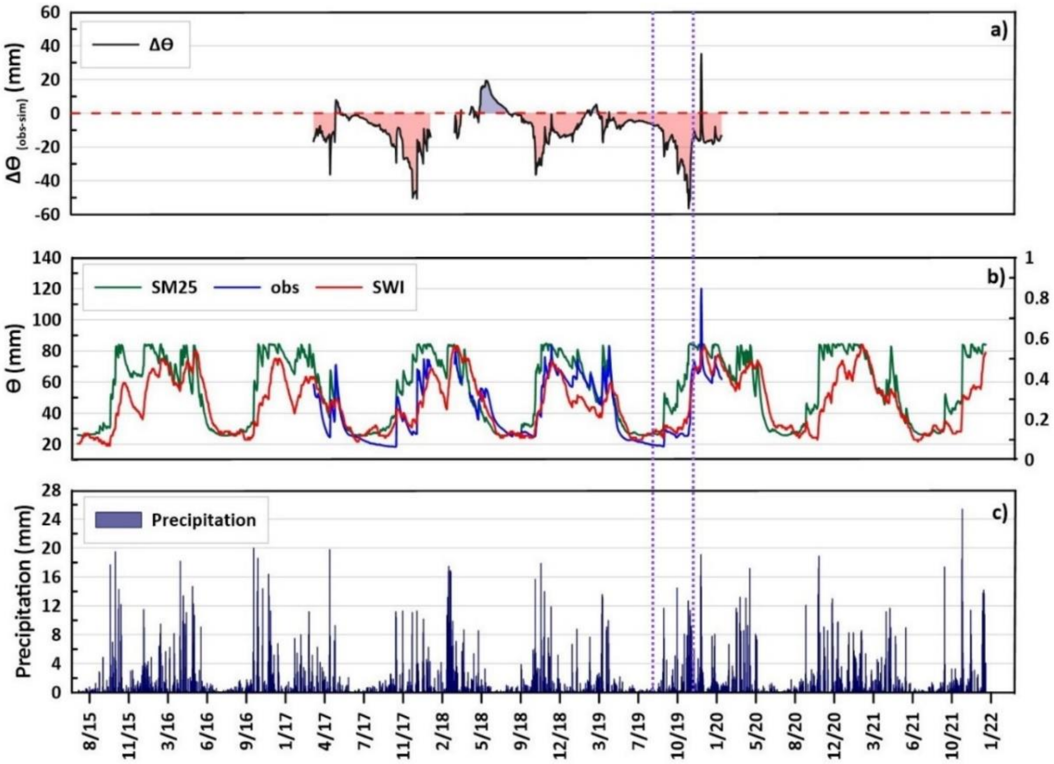

Fig.3. Soil moisture dynamic. Panel a) shows the difference between the ground-assed and modelled soil moisture dynamic. Panel b) displays with the green line the soil moisture model simulations; in the blue line, the observed soil moisture; in the red line, the Soil Water Index (SWI) dynamic of the study period. Panel c) displays the precipitation events satellite-based.

3.2 NDVI forecasting model

Fig. 4 and 5 show the results of NDVI prediction respectively at 7 and 30 days ahead. For both versions of the NDVI forecast model results are satisfactory. As expected, the 7-day lead time forecasts (NSE over 0.9 and MBE lower than 0.02) are better than the 30-day lead time forecasts (NSE over 0.80 and MBE lower than 0.01). From Fig. 4 and 5, we can break down the seasonal grassland dynamics into two main stages: the growing season (light blue area in Fig. 4 and 5, when NDVI values raise and reach the highest values and the senescence season when NDVI decreases and reaches the lowest values. In the Mediterranean climate, the grassland growing season is characterized by a fluctuation of NDVI values during the productive season (harvest or grazing season) as a result of dry periods (Chelli et al., 2016; Zavaleta et al., 2003), this underline the relationship between the phonology dynamics and soil moisture dynamic (Gómez-Giráldez et al., 2020). The peak values of NDVI obtained with both forecasting models, at both 7 and 30-day lead times, NDVI between 0.50 and 0.80, are in the order of magnitude of observed values found in the literature in arid and semi-arid climate, which range between 0.53 and 0.78 (Flynn et al., 2008; Insua et al., 2019). These results not only demonstrate the significance of soil moisture as a driver of grassland dynamics in Mediterranean climates but also show the potential use of the two proposed NDVI forecasting models to predict seasonal variations of NDVI.

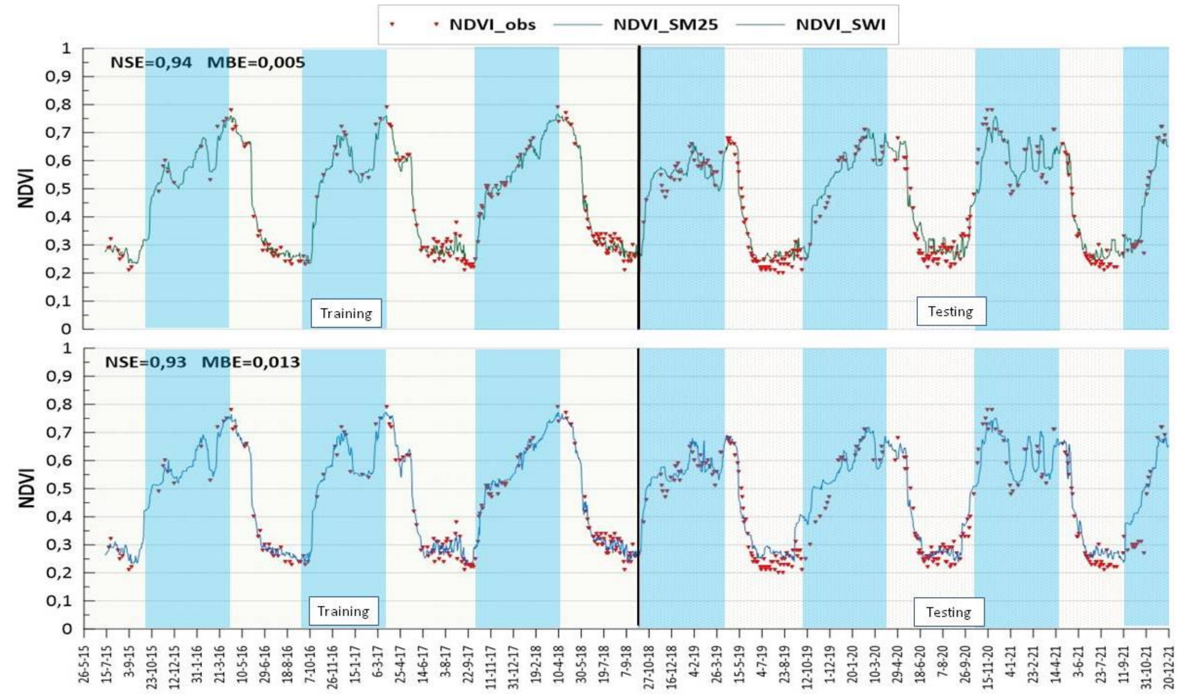

Fig.4.NDVI forecasts model results vs observations (NDVI_obs) for a 7-day lead time. The top panel displays the forecasts obtained using SWI remote sensed observation as soil moisture information (NDVI_SWI). The bottom panel displays the forecasts obtained using the soil water model (NDVI_SM25). The period corresponding to the growing season is shaded in light blue.

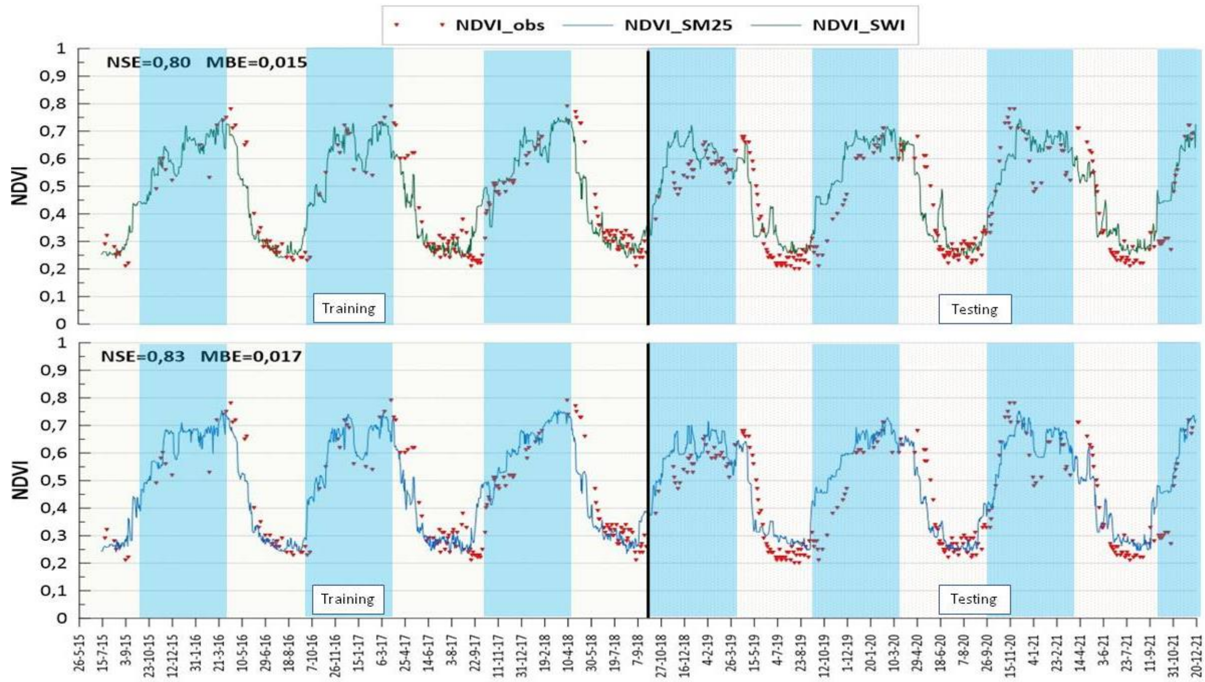

Fig.5. NDVI forecasts model results vs observations (NDVI_obs) for a 30-day lead time. The top panel displays the forecasts obtained using SWI remote sensed observation as soil moisture information (NDVI_SWI). The bottom panel displays the forecasts obtained using the soil water model (NDVI_SM25). The period corresponding to the growing season is shaded in light blue.

Regarding the intra-seasonal variations or anomalies (z-score), Fig. 6 and 7 show that both NDVI_SM25 and NDVI_SWI for 7-day lead time ($r = 0.92$, p-value < 0,05, for both models, see Table 1) perform better than for 30-day lead time ($NDVI\_SWI_{30}$ r =0,54, $NDVI\_SM25_{30}$ r=0.60, p-value < 0,05, Table 1). We observe similar result comparing performances focusing only on the growing season. Indeed, both versions of the NDVI forecast models at 7-day, $NDVI\_SWI_{7}$ and $NDVI\_SM25_{7}$, showed satisfactory performance recording a high correlation with the observed anomalies (r=0.93 and r=0.92 respectively). Instead, NDVI forecast models at 30-day, do not perform realisable NDVI anomalies during the growing periods. However, the $NDVI\_SM25_{30}$ predicts slightly better NDVI anomalies than the $NDVI\_SWI_{30}$ (respectively r=0.56 and r= 0.62). Taking as an example the growing season of 2017-2018 with 454 mm of precipitation, and the one from 2018-2019, with 1796 mm (Fig 6 and 7), we can observe how the models perform under particularly dry and wet weather conditions respectively. Under these two conditions, both 7-day forecasting models predict anomalies satisfactorily (r=0.93 for both models), in contrast, forecasting models at 30 days are weakly correlated to the observed anomalies (respectively r=0.59 for $NDVI\_SWI_{30}$ and r=0.65 for $NDVI\_SM25_{30}$). This shows the limitation of using past and present data to forecast NDVI anomalies in a mid and long term. Future work should explore the use of mid and long-range weather forecast products to improve the performance of this type of NDVI forecasting models. In particular, NDVI_SM25 may benefit by using weather forecast data because to feed the soil moisture model and thus obtain soil moisture forecasts of one or several months.

Table1: Pearson correlation (p-value < 0,05) between the observed NDVI anomalies and the forecasted at 7 and 30 days. The overall correlation takes into consideration the entire study period; the growing season correlation takes into consideration only the growing season of the study period; the 2017-2018 correlation takes into consideration the driest growing season of our study period.

PERIOD	$NDVI\_SWI_{7}$	$NDVI\_SM25_{7}$	$NDVI\_SWI_{30}$	$NDVI\_SM25_{30}$
OVERALL	**0.92**	**0.92**	**0.54**	**0.60**
GROWING SEASON	**0.93**	**0.92**	**0.56**	**0.62**
GROWING SEASON 2017-2018	**0.93**	**0.93**	**0.59**	**0.65**

Prediction of NDVI anomalies gains particular importance in the context of agricultural insurance. In Spain, agricultural insurers, under the jurisdiction of the Spanish government, use the NDVI anomaly method to assess grassland yield loss caused by drought or extreme weather events, estimating remotely the production deficit with an NDVI-based indicator called Guaranteed Vegetation Index (BOE, 2022). It is calculated using data from the last 20 years and during the guaranteed period, which corresponds to the growing season, as the 10-day mean NDVI minus 0.5-1.5 times the 10-day standard deviation multiplied by an economical estimator. This model is based on past estimations however the use of the NDVI forecasting models such as the ones presented in this study, can let both farmers and insurers to anticipate production deficits and hence compensations. However, it must be noted that their potential applicability is rather different. In the case of the SWI version, the use of satellite products, increases the potential the scalability of its use, from single pixel scales to larger areas comprising multiple pixels. In the case. The use of the soil moisture model version allows combined with seasonal weather forecast data can potentially increase the temporal scale and potentially obtain better performance for longer lead times than 7 days.

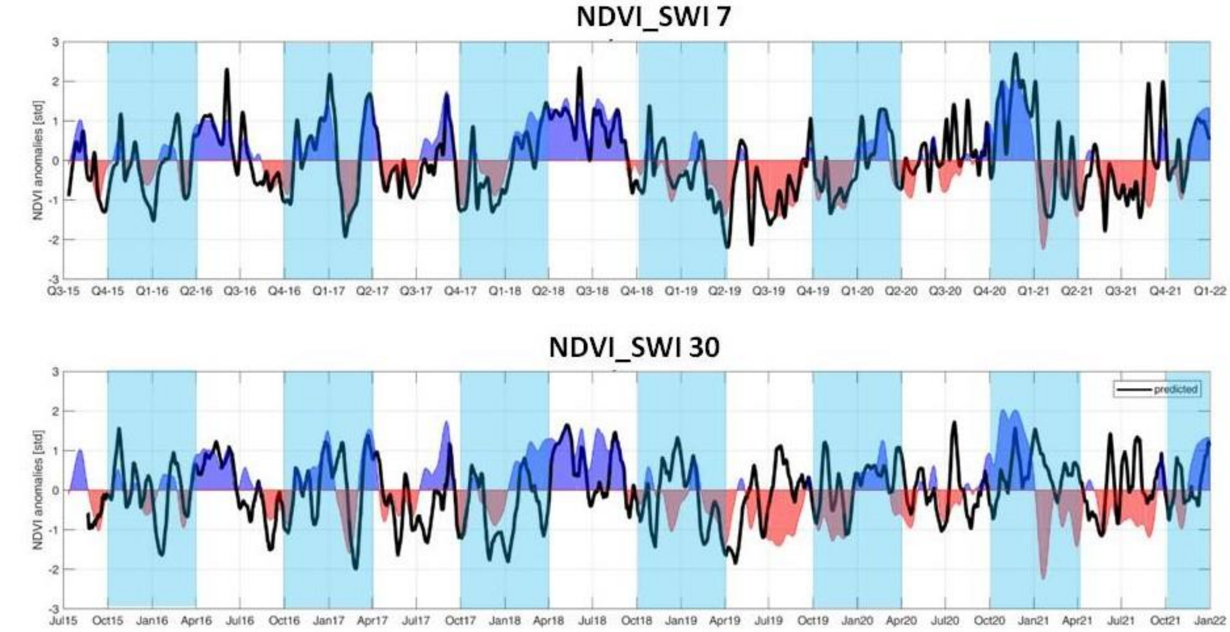

Fig.6. Forecasting model's anomalies of NDVI_SWI. Anomalies are calculated using the Z-score. The black line shows the NDVI anomalies predicted at 7 days (upper graph) and at 30 days (bottom graph). The blue shade shows the positive observed NDVI anomalies, the red shade shows the negative observed NDVI anomalies. The background light shade highlight the growing season.

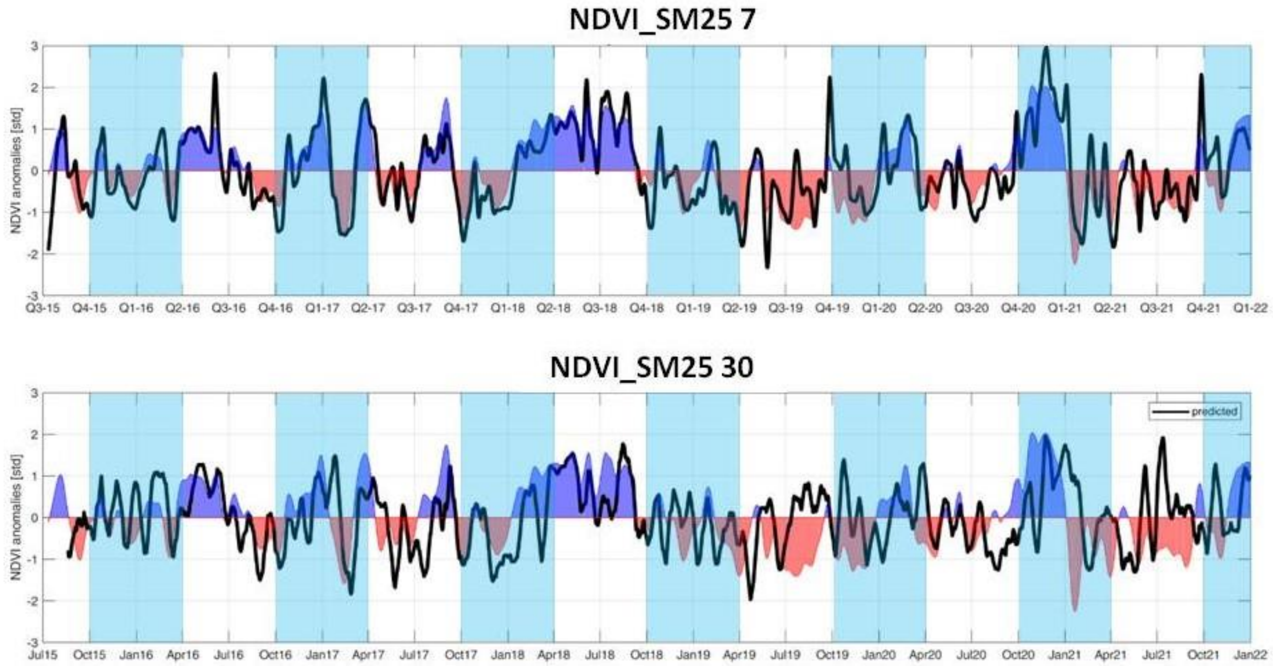

Fig.7. Forecasting model's anomalies of NDVI_SM25. Anomalies are calculated using the Z-score. The black line shows the NDVI anomalies predicted at 7 days (upper graph) and at 30 days (bottom graph). The blue shade shows the positive observed NDVI anomalies, the red

shade shows the negative observed NDVI anomalies. The background light shade highlights the growing season.

While the results of the study are promising, we recognise that there were several limitations, such as one observation point only and the use of historical meteorological data. Considering the limited literature on the use of soil moisture products as NDVI predictors, we advise further investigation into other bioregions and at larger scale. Moreover, the use of stationary weather prediction can be explored to extend the forecasting period to predict anomalies. Field NDVI assessment can be carried out to fit better the models and assess discrepancies with satellite based NDVI observation.

4.Conclusions

In this study, we present two NDVI forecasting models based on the use of machine learning and past and present weather and soil moisture data as predictors. One model, NDVI_SM25, uses simulated soil moisture values and the other, NDVI_SWI, uses satellite-based Soil Water Index (SWI) values. The performance of both models is evaluated in a Mediterranean permanent grassland in South Spain by comparing forecasted and observed NDVI daily values. Results show high reliability of models, at 7 and 30-day forecast lead times, in predicting seasonal NDVI dynamics and demonstrate the significance of soil moisture dynamics as a driver of grassland phenology in dry climates. In the case of intra-seasonal variations or anomalies, NDVI are significantly better predicted by both models at a 7-day lead time than at a 30-day lead time. These results show the potential of using NDVI forecasting models based on the of soil moisture information and machine learning to help both farmers and insurers anticipate production deficits and apply mitigation measures.

Acknowledgments

This research is funded by the European Union, under the Horizon 2020 project "Developing SUstainable PERmanent Grassland systems and policies (Super-G)", grant no. 774124. Views and opinions expressed are those of the author(s) only and do not necessarily reflect those of the European UnionEU or the European Research Council. Neither the European Union nor the granting authority can be held responsible for them. Vanwalleghem and Milazzo also acknowledge additional financial support from the Spanish Ministry of Science and Innovation, the Spanish State Research Agency, and through the Severo Ochoa and María de Maeztu Program for Centers and Units of Excellence in R&D (Ref. CEX2019-000968-M).

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Chapter 6: Conclusions

Permanent pastures, in Andalusia as in all Europe, are important ecosystems not only for the communities exploiting them for agricultural purposes, but also for the rest of the population that unknowingly uses benefits from the ecosystem services issued.

In the This study provides a deeper overview of the importance of permanent grasslands for erosion and flood mitigation in Europe, and particularly in Southern Spain. In Chapter 2 a quantitative meta-analysis evaluated four erosion and flooding-related indicators: bulk density, hydraulic conductivity, runoff and soil loss, comparing three land uses: permanent grassland, arable land and forests. In total 24 articles were fully analysed, after screening over 14,203 articles. The results showed that on the one hand, in comparison with arable land, results are often in contrast to the widespread opinion of topsoil structural amelioration of grassland. In fact, no significant differences have been reported comparing bulk density and hydraulic conductivity and soil loss, highlighting the temporary positive effect of tillage in arable land and of the local environmental conditions that can promote soil degradation (i.e. fire) in grasslands and forests. On the other hand, permanent grassland mitigates better runoff than arable land. In contrast with forest land, differences are not clear cut, suggesting that soil erosion and runoff mitigation conditions are similar between the two land uses, except for the hydraulic conductivity which is higher in forest land. However, these general indicators are limited in scope. A second, broader review showed how European permanent grasslands suffer from additional land degradation hazards. This additional review identified six processes important for soil erosion in European grasslands: trampling-induced erosion, gullying, piping, landsliding, snowmelt erosion and avalanche erosion. All these processes were documented in European grassland to have cause significant erosion problems locally. At present, their extent and regional impact is mostly unknown. These are boosted by several promoting processes related to soil management and environmental conditions: compaction, hydrophobicity and wildfires. In summary, although permanent grasslands are considered crucial for the reduction of soil loss and flooding, they are under degradation risk. Due to the complex nature and the interconnection between erosion and flooding processes, and the lack of knowledge on many of the processes involved, their assessment, understanding and modelling are still often challenging. Therefore, these processes must be studied more in detail in order to get a good view of the status of European permanent grasslands. This will help with designing a site-specific soil management strategy for European grasslands, aiming at the zero net land degradation goals promoted by the Green Deal.

In the context of climate change and increasing grassland degradation, it is essential to understand soil quality development for the resilience of the grassland ecosystems. In Chapter 3, We show the importance here of using a variety of SQIs, including physical, chemical, and biological indicators, is crucial for achieving different sustainable international goals. Soil quality preservation and maintenance, should be considered essential for environmental quality in general (Döring et al., 2015). The application of sustainable management cannot be separated from careful monitoring of soil quality development. Indeed, the assessment of the reviewed SQIs is a reliable strategy for undertaking sustainable and good management practices. However, the efforts to assess soil quality qualitatively and quantitatively are not new, and the standardization of indicators remains an ambitious task. Therefore, due to the site-specific soil quality, the SQIs threshold should be selected according to the base of the soil function of interest. Thus, the development of a SQIs assessment framework, also for limited data availability, can support grassland managers to preserve soil quality. Despite the current limitation of standardization, there are several initiatives aiming to harmonize soil quality information (e.g the Global Soil partnership, the Global Soil Biodiversity Atlas) at a different scale, that can support the management decisions. Sustainable grazing strategies can be implemented and adapted to promote soil quality and the related ecosystem services delivered, with the aim to overcome climate change effects. The test of new grassland species, drought-resistant and with desirable traits for soil protection, must be explored for the different bioregions aiming to improve grassland resilience in terms of soil protection, production, and ecosystem services delivery.

In Chapter 4, we develop a methodology to quantify the ecosystem service of erosion mitigation offered by permanent grasslands, at regional scale. Firstly, land use change was quantified on a regional scale in Southern Spain and over a period spanning more than six decades, from 1956 to 2018. Abrupt land use dynamics were observed in this period of important socio-economic changes. Conspicuous loss of permanent grassland was observed, mostly for the benefit of forest and natural areas. Permanent crops, in particular olive orchards, also grew rapidly, while cropland first increased and then decreased towards the end of the studied period. The variation of historical erosion rates was then calculated, using the observed frequency distribution of C-factor and a Monte Carlo approach for their spatial allocation. Despite the observed land use changes, the cumulative soil erosion rate on the regional scale remained constant, between 6,49 and 6,86 × 10^7 t y^{-1}. The underlying reasons for this resilient behaviour was further explored, and, was attributed to different reasons. Firstly, land use changes occurred frequently between land use classes that had reasonably similar C-factor; cropland and permanent crops on the one hand for instance and PG and forest and natural areas on the other. This occurred within 60% of the terrain. Secondly, and possibly the most important reason, is that a small fraction of the regional area (20%) is responsible for most of the total cumulative erosion rate (67%). This area has an average LS-factor ranging 2,5 and 7,5, belonging mainly to the permanent crop and forest and natural area on higher slope gradient. Finally, the importance of permanent grassland was quantified by applying four land use scenarios: one scenario of most probable change, based on a stochastic land use prediction model and three hypothetical scenarios of complete conversion. Future changes under the most likely scenario are predicted to be minor. However, this analysis shows how permanent grassland plays an important role on the regional cumulative erosion mitigation, as the permanent grassland total conversion to permanent crop and cropland can raise the regional cumulative erosion of 13% and 14%.

Finally, in Chapter 5, we present two NDVI forecasting models for Mediterranean grasslands, based on the use of machine learning and past and present weather and soil moisture data as predictors. One model, NDVI_SM25, uses simulated soil moisture values and the other, NDVI_SWI, uses satellite-based Soil Water Index (SWI) values. The performance of both models is evaluated in a Mediterranean permanent grassland in Southern Spain by comparing forecasted and observed NDVI daily values. Results show high reliability of models, at 7 and

30-day forecast lead times, in predicting seasonal NDVI dynamics and demonstrate the significance of soil moisture dynamics as a driver of grassland phenology in dry climates. In the case of intra-seasonal variations or anomalies, NDVI are significantly better predicted by both models at a 7-day lead time than at a 30-day lead time. These results show the potential of using NDVI forecasting models based on the of soil moisture information and machine learning to help both farmers and insurers anticipate production deficits and apply mitigation measures.

In addition to what arose in this thesis research, some important knowledge gaps have emerged, to which more in-depth research is proposed. For example, several shortcomings were identified in Chapter 2. After a systematic collection of the European studies published so far, there is a lack of clarity in the results, often leading to a lack of efficiency in the mitigation of erosion and flooding by permanent pastures in favour of ploughed land. Moreover, the number of scientific reports collected is quite limited, suggesting a low scientific activity of quantification in the field in comparison with other uses of soil. In addition, advanced studies on the physical dynamics of models for reliable representation in terms of processes and quantification of erosion of major erosive processes such as gullying, landslides, pipes, and avalanches are missing. Furthermore, in the geomorphological and erosion studies never specify if they study area is temporally o permanent grassland making harder the review.

Moreover, in Chapter 3, we highlight the current gaps of knowledge relegated to the standardization of soil quality and the intensity of grazing. In fact, although the former is specific to the site and the objectives to be achieved in relation to land management, the intensity of grazing is defined only by the productive capacity of the pasture and its propensity to vegetative growth. The intensity of grazing, and therefore the possibility of feeding as many animals as possible on pasture, cannot be restricted only to the nutritional needs of the animal and the biomass of the pasture but must necessarily take into account the effect of livestock on soil quality. In addition, the study of the introduction of drought-resistant plant species should be strengthened (Fernández-Habas et al., 2023) in order to have a rapid response to climate change by ensuring the reduction of soil degradation and other ecosystem services of permanent grassland. In addition, this study argues for a shift in EU policies towards prioritizing permanent grassland conservation linked to farming and livestock rearing, supporting extensive grazing management, and recognizing the importance of the extensive grazing for the recognizing the importance of extensive grazing for promoting soil quality and mitigating erosion and flooding. Further research is needed to explore and understand different levels of organization, including farming systems, landscape and territory, the food chain, and policy design, to fully realize the potential of integrated sustainable systems and their benefits for the EU (Poux et al., 2022).

The study presented in Chapter 4, however, highlights the lack of a study of the effect of the historical change of land use on the rate of erosion on a European scale, and the global quantification of the importance of permanent pastures.

The importance of the last study, in Chapter 5, shows a high correlation between soil moisture and the dynamics of the NDVI of permanent pastures in the Mediterranean climate. However, the correlation between NDVI and production of fresh and dry biomass is not fully delineated. The knowledge of this correlation could enhance this tool, providing the farmer a dual use, focused on the production of biomass and the sustainable management of his rangeland. Predicting with the use of satellite data opens up new avenues of research to scale this algorithm to greater scales, thus creating a powerful tool for policy-makers and insurance institutions.

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Acknowledgments

During my personal and academic growth, I have been fortunate to meet many people who have significantly contributed to my intellectual development. While I cannot possibly name everyone, I would like to express my gratitude to those who have supported me during difficult times and celebrated with me during joyful moments.

Firstly, I would like to thank Jesus Fernandez-Habas, who is not only a colleague but also a friend. He is a young researcher with a promising future, and I wish him all the success in the world. Secondly, I would like to express my appreciation to Vanesa García Gamero, a detail-oriented woman who is always available to listen. Thirdly, I am grateful to Andrés Peñuela, who has been like an older brother to me, guiding me through the toughest moments. Fourthly, I want to thank Juan Vicente Giráldez Cervera, from whom I learned that the best way is often the longest and that each of us will find our own Ithaca at the end of our journey. Lastly, I am thankful to Pilar Fernández Rebollo for bringing out the best in me.

Finally, I would like to express my deepest gratitude to Tom Vanwalleghem. He has been my reference point and guiding light on the darkest of nights. Tom taught me how to fight my frustrations, think critically, reflect deeply, and enhance my analytical skills. I thank him for his unwavering support and guidance as I have learned everything from him.

I also extend my sincere appreciation to the entire research group who welcomed me with open arms. They provided me with valuable advice and support throughout my doctoral journey.