Universitat Autònoma de Barcelona

SOIL CARBON TEMPORAL DYNAMICS FOLLOWING AGRICULTURAL LAND ABANDONMENT FROM FIELD TO CONTINENTAL SCALES

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PhD thesis

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Doctoral degree in Environmental Science and Technology

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Thesis Submitted For Partial Fulfillment of the Requirements for the Degree of:

Doctor of Philosophy

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"*On this sand farm in Wisconsin, first worn out and then abandoned by our bigger and better society, we try to rebuild, with shovel and axe, what we are losing elsewhere*."

– Aldo Leopold, in the Foreword to *A Sand County Almanac*.

Table of Contents

Overview

Millennia of intensive and extensive agricultural land use practices has severely depleted global soil organic carbon (SOC) stocks and drastically increased atmospheric CO_2 concentrations. Yet, for all the billions of hectares of terrestrial ecosystems converted to agriculture, there has also been hundreds of millions of hectares of agricultural land abandoned. This often-neglected land use change represents a significant opportunity for ecosystem restoration and soil carbon sequestration (SCS), especially if ecological succession occurs spontaneously. However, despite the potential these lands represent for climate change mitigation, they are still underrepresented in global change science and policy. The impacts of agricultural land abandonment (ALA) on soil carbon stocks are insufficiently calibrated in biogeochemical models. We struggle to predict which kinds of agricultural land will sequester, lose, or maintain pre-existing SOC levels following ALA, and how these trends behave at different spatial scales. This is especially true in Europe where there has not been a continental-scale synthesis, despite the widespread extent of past and ongoing ALA and the strong policy interest to increase European soil carbon stocks.

In this doctoral thesis, I generate new knowledge on the effects of ALA on soil carbon temporal dynamics, thereby contributing practical information for sustainable land management decisions involving the land carbon sink. Six major categories of proposed management strategies for abandoned agricultural lands were identified following a literature review, each with positive, negative, direct and indirect outcomes depending on site-specific factors and management objectives. Accordingly, no single strategy is ideal in all scenarios and a combination of strategies addresses multiple rural development goals concurrently. Focusing on two of the six strategies (active and passive restoration), I sampled new chronosequences of ALA to explore the effects of depth and time on SOC at the field scale and I synthesized a new dataset from peninsular Spain to identify the potential factors responsible for the high variability in post-agricultural SCS rates observed in the Mediterranean region. Chronosequence field studies indicate a highly variable process, depending on multiple environmental and land management factors. The highest rates of SOC accumulation post-abandonment in Spain can be expected on lands previously used for woody crop production featuring ~13–17 ° C MAT and ~450–900 mm MAP, with the lowest rates expected on lands previously used for annual crop production outside this climatic window. From these insights, I expanded the analysis to the continental scale and assembled the largest dataset ever collected on SOC stock changes following ALA at known times (n = 804) to investigate the potential environmental and human management factors driving SCS rates following ALA in Europe. There is a slow, but significant, rate of SOC stock increase of 1.28% yr^{-1} (0.32 Mg C ha^{-1} yr^{-1}) on abandoned agricultural lands across Europe. SOC responses were negatively correlated with initial stock, indicating a soil carbon saturation effect. Abandoned agricultural lands in biogeographical regions featuring optimal climatic windows had higher SCS rates, but human management factors can generate both positive and negative effects on SOC, resulting in several strongly divergent responses to ALA. Past croplands had a notably greater rate of SOC increase over time than sites that were previously used as pastures, likely a result of lower initial SOC stocks in croplands compared to pastures. Sites that underwent natural ecological succession exhibited a greater rate of change in SOC stock compared to sites that were actively restored or converted to new vegetation land covers, for example through tree planting practices. These findings suggest that abandoned croplands with low initial SOC stock and managed through natural succession would show the greatest SOC accrual in Europe, while fertile pastures that are actively converted (e.g., afforested) would result in the lowest increases in SOC, or even losses.

This work helps clarify some of the previous regional debates on the positive, negative, and neutral SCS potentials of post-agricultural soils, which have likely been confounded by the multiple factors identified. Overall, this PhD thesis informs ecosystem restoration policies and land management strategies on the potential soil carbon benefits, costs, and challenges of post-agricultural landscapes. The variability in SOC dynamics following agricultural land abandonment/conversion must be considered in sustainable land use planning that strives to incorporate the positive ecological and climate change mitigation implications of ALA, taking into account site-specific conditions and past and present land management factors to avoid negative impacts for soil health and lost opportunities for climate change mitigation.

List of Papers

I. Bell, S.M., Barriocanal, C., Terrer, C. and Rosell-Melé, A., 2020. Management opportunities for soil carbon sequestration following agricultural land abandonment. *Environmental Science & Policy*, 108, pp.104-111.

II. Bell, S.M., Terrer, C., Barriocanal, C., Jackson, R.B. and Rosell-Melé, A., 2021. Soil organic carbon accumulation rates on Mediterranean abandoned agricultural lands. *Science of the Total Environment*, 759, p.143535.

III. Bell, S.M., Terrer, C., Barriocanal, C., Perpiñá Castillo, C., Jackson, R., Franklin, O, Schillaci, C, Saia, S., Rosell-Melé, A. Factors driving soil carbon sequestration following agricultural land abandonment in Europe. *Manuscript in preparation*.

IV. Bell, S.M., Raymond, S.J., Yin, H., Jiao, W., Leshyk, V., Olivetti, E., Terrer, C. Recarbonizing post-agricultural landscapes. *Invited "Comment" manuscript in preparation for Nature Communications.*

List of Figures

List of Tables

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CHAPTER I: Introduction

1.1 The critical land and soil nexus

In Earth system science, human activity is now conceptualized as the Anthroposphere (Kuhn and Heckelei, 2010), influencing all other spheres (Figure 1). Unfortunately, the magnitude of current anthropogenic pressures and their impacts on terrestrial ecosystems is unprecedented: from higher temperature increases over land compared to oceans via intensified greenhouse gas emissions (IPCC, 2021), to the exploitation of net primary productivity via biomass harvesting (Haberl et al., 2014), to the alteration of biogeochemical cycles via chemical inputs and extractions (Galloway et al., 2008; Lu and Tian, 2017). Human well-being is inextricably tied to the sustainable management of land and soil resources (Isbell et al., 2017). Natural and modified landscapes serve as the foundation of human livelihoods through the provisioning of vital ecosystem services (Hoekstra and Wiedmann, 2014). Therefore, strategically managing land and soils (i.e., the pedosphere) has become a critical nexus of the Anthropocene (Lewis and Maslin, 2015), linking human activities with ecosystem stability and climate change (Foley et al., 2005; Smith et al., 2019; Turner et al., 2007).

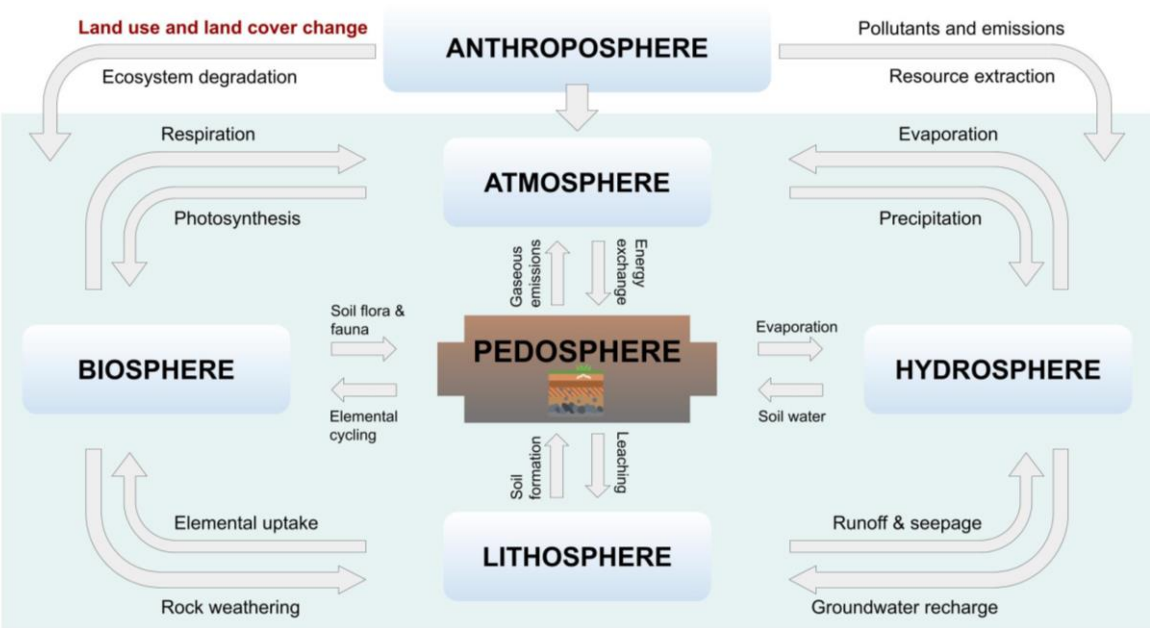

Figure 1. All spheres in Earth system science are present in soils (i.e., the pedosphere): soil air, soil water, soil mineral particles, and soil living and decaying organic matter. Soils are where the atmosphere, biosphere, hydrosphere, and lithosphere interact. These interactions are now closely connected to, and sometimes directly modulated by, human activities of the anthroposphere (e.g., land use and land cover change). Adapted from Lal et al., (1998).

Over three-quarters of Earth's ice-free land surface is under human management (Erb et al., 2017) and nearly one-third has experienced land use and land cover change (LULCC) (Winkler et al., 2021) as a result of either human activities (60%) (Song et al., 2018) or indirect drivers like climate change. LULCC implies complex interactions and trade-offs to the ecosystem services soils provide (Smith et al., 2015). Historical and ongoing patterns of LULCC have stimulated varied soil nutrient and greenhouse gas fluxes (Houghton et al., 2012), with uncertain implications for regional and global biogeochemical cycling (Peñuelas et al., 2013; Wieder et al., 2018). The land carbon sink and land carbon flux (Figure 2), for example, is a function of the interactions and contributions of LULCC, CO_2 fertilization, and nitrogen deposition (Tharammal et al., 2019). However, when the impacts of LULCC on soils are poorly quantified in Earth system modelling, our ability to predict terrestrial fluxes of the land carbon sink is significantly weakened (Eglin et al., 2010; Krause et al., 2019; Quesada et al., 2018). The impacts of LULCC on soil biogeochemical cycling must be constrained to inform appropriate responses to 21st century land challenges (IPCC, 2019; Smith et al., 2015).

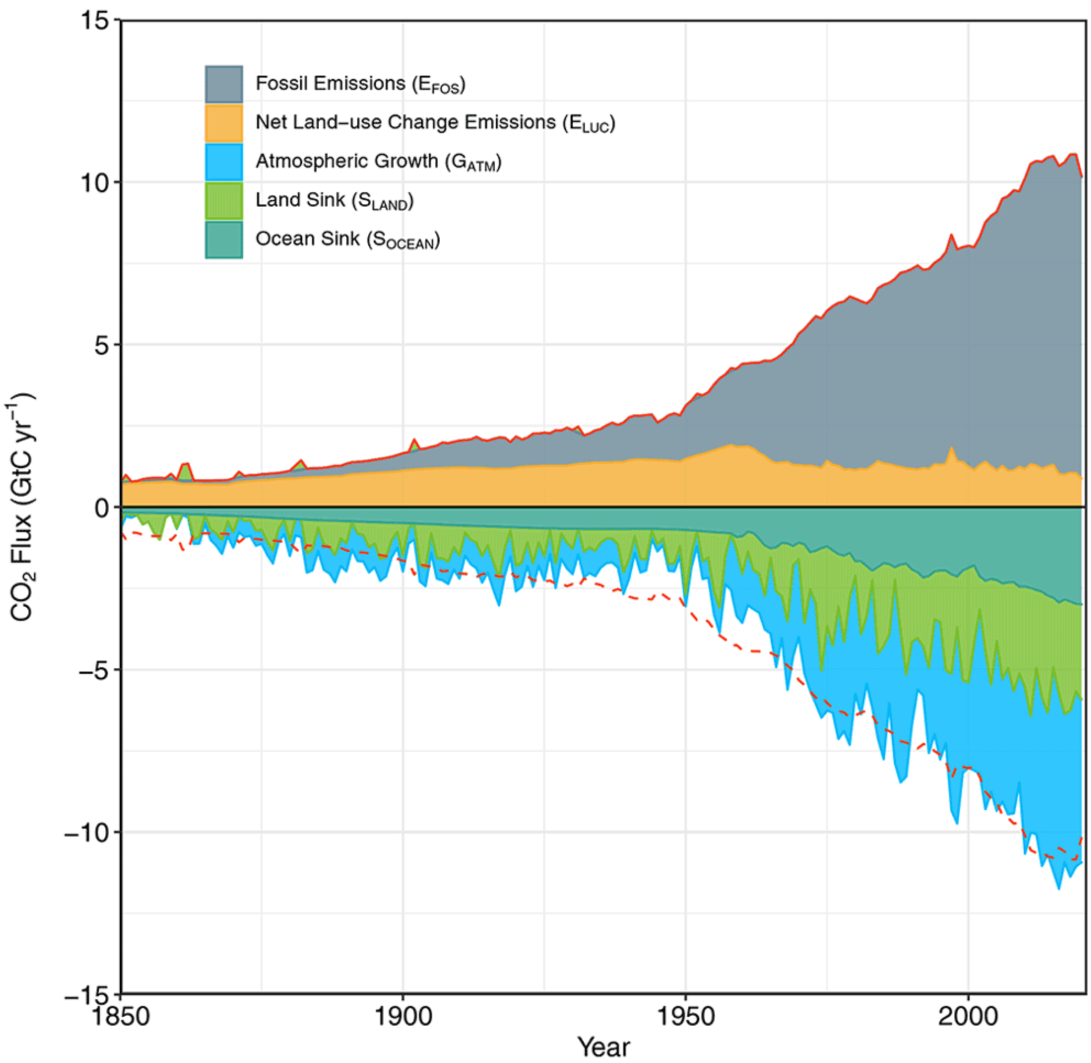

Figure 2. Annual carbon emissions (positive values) and their partitioning (negative values) as a function of time from the main components of the 2021 Global Carbon Budget (Friedlingstein et al., 2022).

Soil systems are not static following LULCC (Ryo et al., 2019); the physicochemical legacies of past disturbances can be measured for decades and longer as soil properties return, often in a non-linear fashion, to pre-disturbance levels or reach new equilibria (Beniston et al., 2014; Johannes M. H. Knops and Tilman, 2000; Marin-Spiotta et al., 2009). Despite this, statistically and geographically robust temporal response curves of soil properties like soil organic carbon (SOC) to major LULCCs like agricultural expansion and contraction (e.g., abandonment) are severely lacking in global change science. This lack of theoretical and empirical information contributes to model inaccuracies, prediction uncertainties, and, ultimately, inadequate land use policies at a time where informed, sustainable land management is needed more than ever before (Folberth et al., 2016; Hendriks et al., 2016).

1.2 Agricultural land abandonment as a global land use change

Of all the dominant land uses that significantly impact soils, agriculture (mainly croplands and pastures) is undoubtably the most pervasive. Approximately 50 million km^2 of global soils are being used for food, feed, fibre, and livestock production (Goldewijk et al., 2011), with humans claiming 38% of the world's land area for farmland and appropriating nearly 30% of global net primary productivity (Haberl et al., 2007; Ramankutty et al., 2008). However, while agriculture has steadily expanded to every corner of the globe since its advent millennia ago, the reverse process of agricultural land abandonment (ALA) has been simultaneously occurring. Even with a 9% increase in total cropland area over the last two decades, there was still 115.5±24.1 Mha of previously existing cropland that underwent abandonment or conversion (Potapov et al., 2022). Leirpoll et al., (2021) identified 83 Mha of abandoned cropland from 1992 to 2015. One of the most often cited global estimates found that between 385–472 Mha of croplands and pastures were abandoned from the years 1700 to 2000 (Campbell et al., 2008), or between a quarter to a third of global cropland area (Ramankutty et al., 2018).

Most global and regional estimates of the timing and extent of ALA vary widely, with high uncertainties for several reasons. For example, it is near-impossible to distinguish abandoned pastures from natural grasslands and short-term fallow fields using commonly employed global land cover mapping approaches with remote sensing. This is also why global maps of ALA focus primarily on abandoned croplands (Figure 3), as they are much easier to detect and monitor. Small plots sizes in heterogenous and diversely cultivated agrarian regions add more difficulties, even for cropland detection. However, there have been recent advances in methodologies enabled by higher spatiotemporal resolution imagery and more accessible cloud computing (Yin et al., 2020, 2018). As methods and computational powers improve, the first reliable global maps can be expected, offering information on not only the location of ALA, but also the timing and duration (especially in the case of cyclical recultivation following ALA). The overall situation for agricultural land extent has been summarized succinctly in a recent *Our World in Data* web article by Dr. Hannah Ritchie: while global croplands are indeed increasing, the reduction in global pastures has finally decoupled agricultural land expansion from food production, suggesting that "the world has passed peak agricultural land" (Ritchie, 2022).

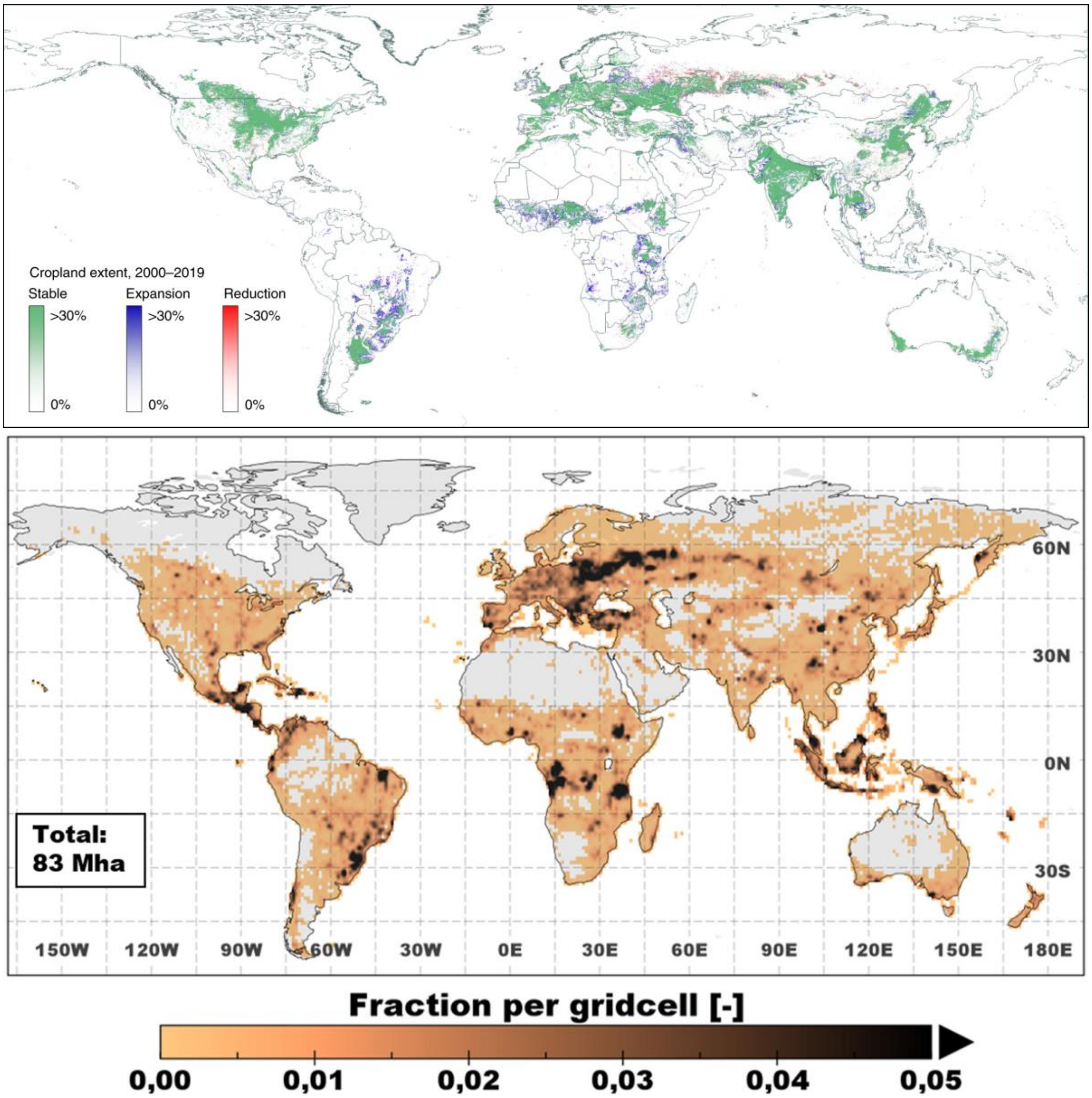

Figure 3. Global spatial estimates of abandoned croplands. Top: Cropland extent from 2000–2019, featuring stable croplands and cropland expansion and reduction (i.e., abandoned or converted) (Potapov et al., 2022). Bottom: Simplified presentation of abandoned cropland hotspots from 1992–2015. Visualized in Leirpoll et al., (2021) based on the aggregated gridcell fraction at 1° resolution. Note the post-Soviet states hotspot following the dissolution of the Soviet Union (1988-1991) (see Lesiv et al., (2018) for 10 arc second resolution map).

Abandoned agricultural lands can be conceptualized as one of several sub-categories under the umbrella of "marginal lands", overlapping primarily with degraded lands because low-productivity is a common driver of abandonment (Figure 4) (Mellor et al., 2021). There is no universally recognized definition of ALA with an agreed minimum timeframe due to the wide array of differing sociocultural and economic perspectives of abandonment as a land use change. Complication things further, alternative terminologies are also used interchangeably with "abandoned agricultural lands" in different contexts (e.g., old fields, post-agrogenic, set-aside, retired land, etc.) and abandonment is by no means a permanent nor one-off change (Prishchepov et al., 2021). ALA can be cyclical, with periods of abandonment followed by recultivation followed by abandonment, resulting in conceptual overlaps with fallow lands and shifting agriculture (Heinimann et al., 2017; Sarkar et al., 2015).

The various definitions of ALA used by land managers and stakeholders, policy makers, and researchers typically fall under five main categories: administrative, economic, social, ecological, and agronomic (Anguiano et al., 2008). Regardless of the different strengths, weaknesses, and specific targets of each category of definitions, they all follow the same basic tenant: the cessation of agricultural activities and the withdrawal of agricultural management from the land (Fayet et al., 2022). The Food and Agriculture Organization (FAO) of the United Nations defines land abandonment as "*a process, whereby human control over land (e.g., agriculture, forestry) is given up and the land is left to nature. After a number of years, depending on the ecological zones and climate, land can be considered as completely "abandoned", when either legal (e.g., forest law) or natural conditions (e.g., desertification, overgrowth with forest) render a restoration for agricultural use is impossible or too costly."* (FAO, 2006). This "number of years" is now generally considered to be at least four (Prishchepov et al., 2021).

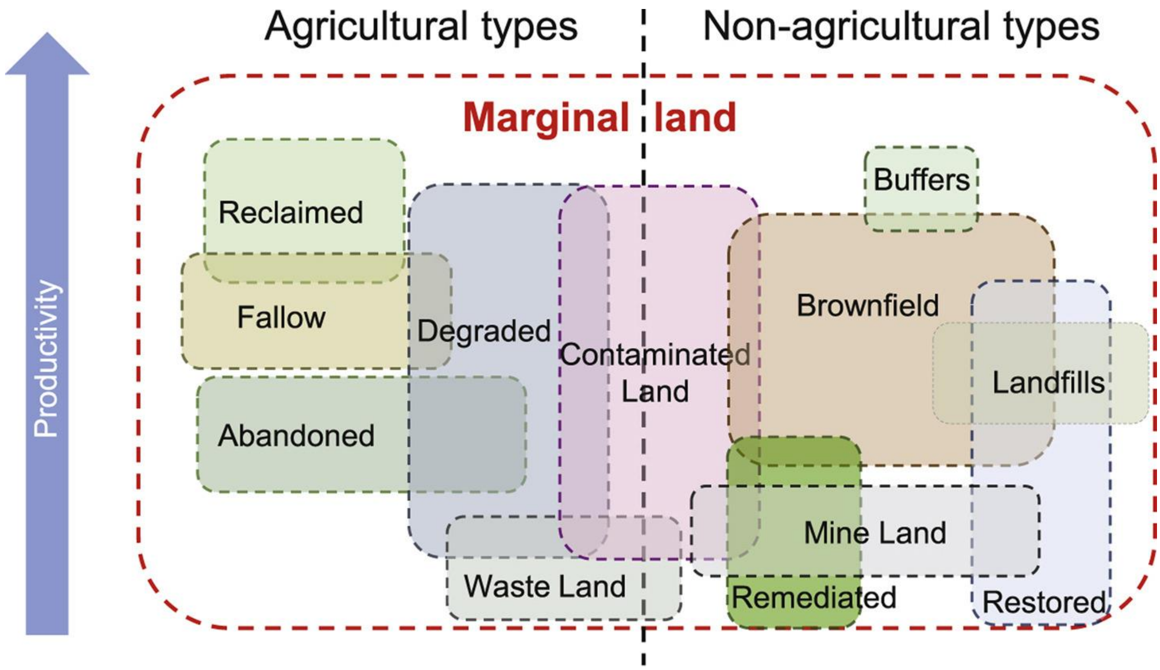

Figure 4. Common categories of marginal lands often compared to abandoned agricultural lands in the context of soil productivity and ecosystem restoration science (Mellor et al., 2021).

One of the main distinctions of ALA as a global land use change that makes it difficult to map, monitor, and manage, is the complexity and diversity of potential drivers and impacts. For example, ALA can be caused by one or more of a range of factors in different regions of the world (Fayet et al., 2022; Li and Li, 2017; Rey Benayas et al., 2007; Ustaoglu and Collier, 2018). Subedi et al., (2022) identified seven generic categories of drivers (demographic, household characteristics, farm characteristics, biophysical, economic, regulatory, and socio-political), and found that some specific drivers were present in all case studies of ALA reported in the literature: slope, soil quality, land suitability, accessibility of farm and remoteness, off-farm employment and farm income, migration and depopulation, and farmer age. Ecological and biophysical drivers include the physiographic and biological factors of the ecosystem when perceived as constraints to agricultural production, including climate change, whereas land mismanagement drivers, such as soil degradation due to over exploitation, represent the result of unsustainable agricultural production over time in a specific ecosystem. Social, economic, political, demographic, and institutional factors leading to ALA encompass the wide range of external forces affecting farming profitability, such as market incentives to abandon, migration to cities and rural depopulation, agricultural industrialization, farmer age and replacement, field accessibility, proximity to markets, etc. Conflicts and other geopolitical drivers are also an easily identifiable cause of ALA (Yin et al., 2019). Among many smaller scale national policy incentives that have been implemented in the last century to abandon croplands, notable large-scale policies or geopolitical events resulting in widespread abandonment include the breakup of the Soviet Union (Kuemmerle et al., 2011; Lesiv et al., 2018; Schierhorn et al., 2013) and the implementation of China's 1999 Conversion of Cropland to Forest Program, also known as the "Grain-for-Green" program (Deng et al., 2014a; Gutiérrez Rodríguez et al., 2015).

The impacts of ALA are as diverse and context-dependent as the drivers. Depending on the perspectives of the observer or stakeholder, ALA can have both positive and negative impacts. In many cases it is perceived positively as an opportunity for ecosystem regeneration and climate co-benefits (Navarro and Pereira, 2012; Poore, 2016; Yang et al., 2020), while for others it signals economic depression, the loss of traditional rural livelihoods, loss of biodiversity, and an increased risk of wildfires and their effects (Benjamin et al., 2008; Katayama et al., 2015; Lucas-Borja et al., 2018; Queiroz et al., 2014). The five main negative impacts identified by Rey Benayas et al., (2007) are: reduction of landscape heterogeneity and promotion of vegetation homogenisation, soil erosion and desertification, reduction of water stocks, biodiversity loss and reduced populations of adapted species (compared to biodiverse agroecosystems) and, lastly, a loss of cultural and aesthetic values. Ustaoglu & Collier (2018) lists the positive impacts as follows: natural habitat restoration (i.e., rewilding), improvement in hydrological regulation, decrease in soil erosion, and increases in water quality, soil carbon, soil fertility, biodiversity, and renewable energy potential. Subedi et al., (2022) surveyed the consequences of ALA reported in 65 studies from around the world and found further evidence of these sometimes contradicting positive, negative, and mixed impacts (Figure 5). But from the perspective of soil functions and ecosystem carbon sequestration at the global scale, ALA was found to be an almost entirely positive LULCC.

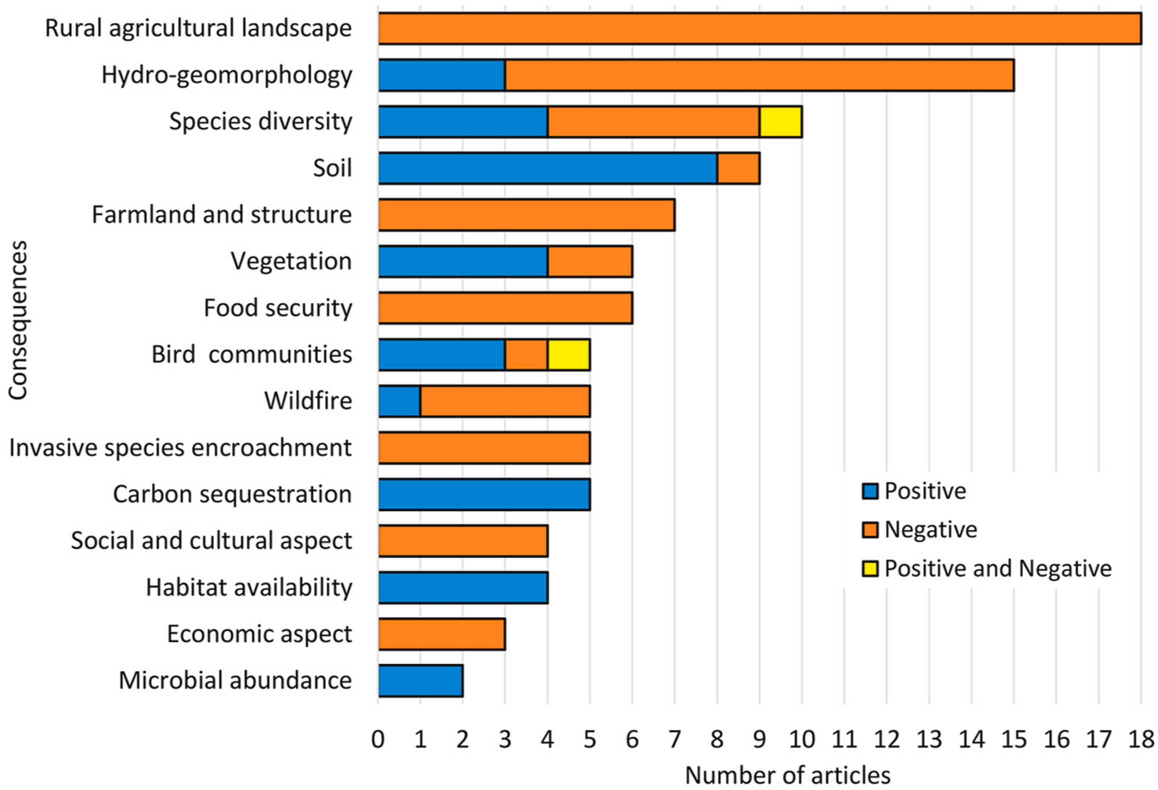

Figure 5. Summary of the consequences of ALA based on a survey of 65 studies from around the world by Subedi et al., (2022). Note the largely positive impacts of ALA on soils and carbon sequestration.

Indeed, land use determines the environmental services provided by terrestrial ecosystems, including the ability to sequester carbon and maintain biodiversity. The return of native vegetation with increased biomass and soil carbon storage makes ecosystems resilient to perturbation and supports climate change mitigation (Vilà-Cabrera et al., 2017). Unless the agricultural land is degraded beyond natural recovery, ALA typical initiates spontaneous ecosystem regeneration through ecological succession (Cramer et al., 2008; Yang et al., 2020). Because of the large spatial extent of present and historical ALA and the fact that post-agricultural soils are often severely carbon depleted, the potential of abandoned agricultural lands for significant rates of soil carbon sequestration is an important area of research in global change science.

1.3 Soil carbon sequestration for climate change mitigation

There is more carbon stored in the world's soils than in the atmosphere and all vegetation combined (Figure 6). It is the largest reactive pool of carbon in terrestrial ecosystems and may reach up to 4000 Gt (at a depth of 3 m) when permafrost is fully accounted for (Lal, 2013). Consequently, because of its large size, potential long-term residence time, and ability to be manipulated by human management, soil carbon can play a crucial role in balancing the global carbon budget. Carbon sequestration is the transfer of atmospheric CO_2 into longer-term pools that keep carbon stored in a more stable state, delaying reemission. Soil carbon sequestration (SCS) is simply this process in relation to the soil component (~2500 Gt) of the global carbon cycle, increasing both organic and inorganic soil carbon typically through specific land use management practices. The SOC pool (~1550 Gt) responds more rapidly to human management than the soil inorganic carbon pool (~950 Gt) and is therefore the main target of sequestration measures employed in sustainable agriculture and ecosystem restoration activities (Lal, 2004a). Most agroecosystems have lower than baseline SOC stocks, due to the long-term negative impacts (e.g., erosion, compaction, mineralization, leaching, etc.) of agriculture-related processes like ploughing, residue removal, monocropping, and other intensive farming practices (Lal, 2013; Sanderman et al., 2017).

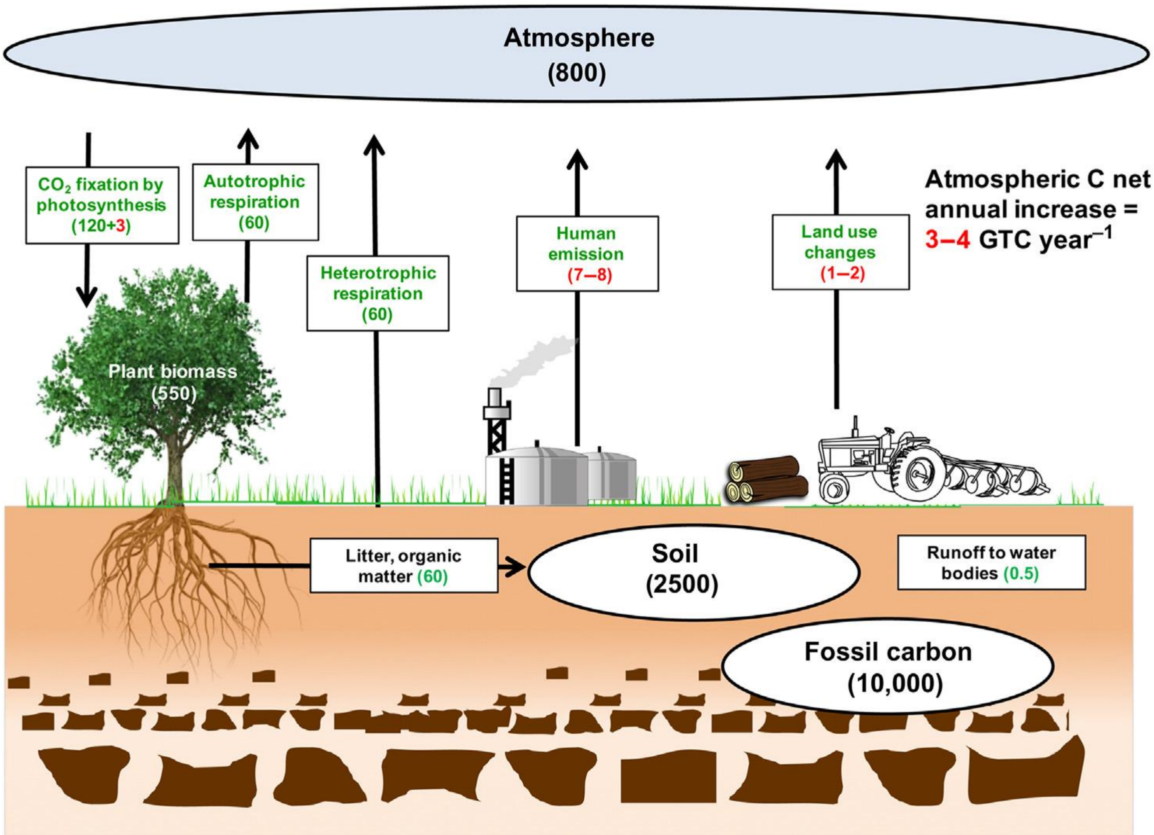

Figure 6. The terrestrial carbon cycle featuring natural fluxes (green) and human emissions (red) in Gt of carbon per year. Soils contain more carbon than the atmosphere and vegetation combined. Numbers in parenthesis represent the estimated size of the carbon pool in Gt (Trivedi et al., 2018).

Rebuilding carbon stocks in depleted agricultural soils is an effective way to promote food security and climate change mitigation. The importance of SCS through various sustainable agricultural practices is recognized globally as a key priority area for the coming years (Bossio et al., 2020; Bradford et al., 2019; Cornelia Rumpel et al., 2018; Vermeulen et al., 2019), and it has become increasingly necessary to accurately describe the underlying mechanisms and nature of soil organic matter (SOM) and SOC stabilization. Not only is SOM fundamental to soil health and fertility, it is also the largest actively cycling terrestrial carbon pool. It is the fraction of soil that is composed of living and dead organic materials and residues (i.e., plant and animal) at varying stages of decomposition representing sources of macro- and micronutrients for plant growth. Making up approximately only one tenth of total SOM, the living component consists of live roots and soil organisms generally classified as either macrofauna (e.g., earthworms, insects, spiders), mesofauna (e.g., springtails, diplura, enchytraeids), microfauna (e.g., protozoa, nematodes) or microbioata (bacteria and fungi) (Briones, 2014).

Within SOM, SOC refers only to the carbon component of organic compounds. Because SOM is difficult to measure directly, researchers prefer to measure, report, monitor, and verify SOC either through destructive, intensive, and expensive laboratory methods (e.g., physical fractionation, loss on ignition method, elemental analysis) or through proximal (e.g., VNIR-SWIR soil spectroscopy) and satellite remote sensing (Angelopoulou et al., 2020; Biney et al., 2022a, 2022b). Approximately 55–60% of SOM is soil C by mass according to the conventional understanding, and only one of either of the two properties needs to be measured in order to infer the other using the “Van Bemmelen conversion factor” (1.724 for determining soil humus content from a SOC concentration following elemental analysis). This factor has been used since the late 19th century for converting primary soil data from field sites and for standardizing large datasets comprising both SOC and SOM values (Minasny et al., 2020). Although modern review studies argue that a factor of closer to 2 is almost always more accurate (Pribyl, 2010), the original conversion factor is still regularly used to achieve consistency throughout datasets (Deng et al., 2016; Don et al., 2011; Kämpf et al., 2016; Liu et al., 2018).

Starting in the 1970–80’s (see, for example Kögel-Knabner et al., (1988)), SOM was often characterized as having four distinct fractions based on size, residence time and chemical composition: dissolved organic matter; particulate organic matter (POM, comprising of fresh residues and living components); humus; and resistant organic matter. The classification of SOM and SOC into fast, intermediate, and slow cycling pools, however, was more of a conceptual exercise, being operationally defined rather than a measurable attribute (see Supplementary materials Table 2 for an outline of the previously dominant SOM/SOC framework) (Lehmann and Kleber, 2015). Over subsequent decades and especially around the turn of the millennium, SOM researchers began to demonstrate that the compounds previously considered recalcitrant where in fact degradable under the right conditions (Gleixner et al., 2002; Rasse et al., 2006). New pools were identified largely based on the percentage of SOC that is environmentally susceptible to microbial activities (Schmidt et al., 2011), physically protected in aggregates (Tisdall and Oades, 1982), or mineral-associated organic matter (MAOM) following standard analytical techniques (Torn et al., 1997). Indeed, both SOM and SOC are now described as continuums of organic material mixtures undergoing constant back-and-forth transformation between decomposition and stabilization (Lehmann and Kleber, 2015), best defined by using mainly the POM and MAOM fractions (Lavallee et al., 2020).

Unfortunately, some of the traditional conceptions of SOM processes, which do not represent the diverse conditions of soils found across the globe, still inform many biogeochemical models. For example, model results might predict greater SOM storage and slower turnover for finely textured soils, by generally assuming high clay and silt content are good indicators of SOM-stabilizing conditions (i.e., greater aggregation, sorption, soil moisture, etc.) and by describing SOM storage based on the presence of arbitrary carbon pools with varying turnover times (Rasmussen et al., 2018). But in reality, recent findings indicate that all clay-sized particles may not have the same effect on SOM storage, and other physicochemical attributes, such as sorption and the content of extractable metals, might be more indicative of SOM stabilization potential. Because sorption (or the formation of chemical associations between minerals and organic compounds) can aid in the stabilization and protection of even labile or young compounds (Abramoff et al., 2021), it is now generally considered one of the key mechanisms building MAOM as a carbon pool and it is being increasingly adopted in SOC models (Figure 7) (Schmidt et al., 2011; Sulman et al., 2018). SOM stabilization enables long-term SCS, and according to the new paradigm recently championed by Lehmann and Kleber (2015), there are two primary mechanisms driving this process: the formation of MAOM and the formation of soil aggregates which protect SOM (Angst et al., 2021), both which are promoted following the cessation of destructive agricultural practices.

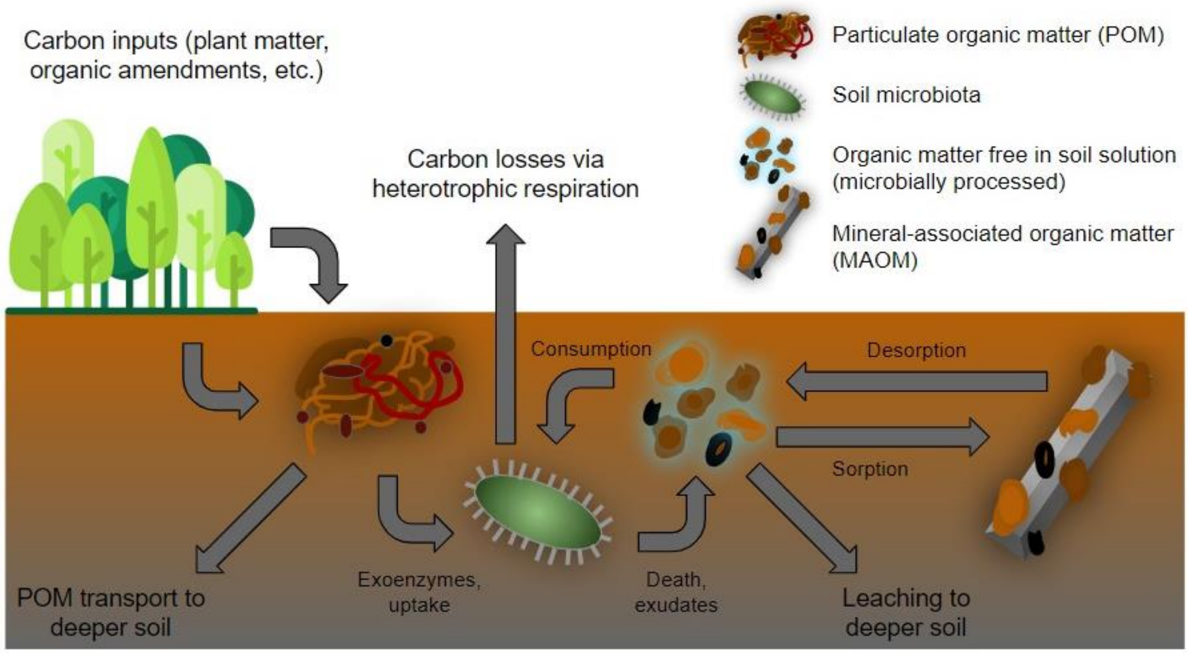

Figure 7. Pathways of SOC transport, stabilization (i.e., as MAOM), and loss through the plant-soil interface. Land use determines the quality and quantity of each pathway in human managed soil systems. Adapted from Dynarski et al., (2020).

Aside from collecting new in-situ soil data from field sites to determine SCS rates (whether long-term experimental stations or one-time sampling campaigns), modelling approaches can integrate multiple predictor factors to estimate farm-, regional-, and global-scale sequestration potentials under various management regimes. In the European Union, several different models have recently been employed to up-scale field data and project SOC stock fluctuations on active agricultural lands into the future based on potential EU and individual member state policy scenarios, including MITERRA-NL in the Netherlands, C-TOOL in Denmark, CENTURY and RothC in Spain, RothC and RothC10N in Italy, EPIC in Germany and Italy, DNDC in Poland, ICBM in Sweden, and PaSim and STICS in France (see Rodrigues et al., (2021) for individual study references).

Maximizing SOC storage is also the best strategy to improve overall soil health, while increasing the global land carbon sink (Figure 2) (FAO and ITPS, 2021). A healthy soil system has the capacity to provide multiple ecosystem services, especially soils managed sustainably with carefully selected practices (Paustian et al., 2016). In recent decades, several sustainable agricultural approaches that ensure food, feed, and fibre production while promoting SCS and other soil health measures have been developed as alternatives to intensive conventional practices. These include popular approaches like agroecology, nature-inclusive agriculture, permaculture, biodynamic agriculture, organic farming, conservation agriculture, regenerative agriculture, carbon farming, climate-smart agriculture, high nature value farming, low external input agriculture, circular agriculture, ecological intensification, and sustainable intensification (Oberč and Arroyo Schnell, 2020). Within these approaches, the specific sustainable agricultural practices employed can help promote faster rates of SCS on many of the key land uses dominating the surface of the Earth, whether croplands, pastures and grazing lands, or managed forests. At the most extreme, one strategy could simply be the conversion from a low-SOC land use into a land use with a higher SOC base level (i.e., higher SOC equilibrium). But other SCS-promoting practices are less disruptive and do not require a complete change in land use, such as promoting vegetation regimes with higher carbon inputs (e.g., crop rotation, cover crops, perennial crops, etc.), managing applied nutrient regimes (e.g., optimized fertilizer rate, type, timing, and precision, etc.), and protecting soil and water properties (e.g., reduced tillage, no-tillage, crop residue retention, reduced compaction and erosion, improved water/irrigation regimes, etc.) (Smith et al., 2019). Sustainable agricultural practices imply actions taken with a holistic understanding for the integrated agroecosystem of *soil* (e.g., no-tillage, minimum tillage, hedgerows, erosion and compaction management, etc.), *crop* (e.g., cover, intercropping, rotation, residues, legume incorporation, etc.), *water* (e.g., irrigation regime, harvesting, reuse practices, etc.), *biodiversity* (e.g., pollination management, pest management, agro-ecological measures, etc.), and *inputs* (e.g., dose size and timing of amendments like biochar, manure, compost, litter, mulching, etc.).

1.4 Abandoned agricultural lands as carbon sinks

In the global effort to mitigate the environmental and social impacts of climate change, highly complex solutions are often proposed that involve significant technical and financial intervention. Recent sentiment in the land sector, however, has been shifting towards more naturally inspired solutions, commonly known as *nature based solutions* (Cohen-Shacham et al., 2016; Keesstra et al., 2018) or *natural climate solutions* (Bossio et al., 2020; Griscom et al., 2017) depending on the objectives and approaches. For example, instead of large-scale afforestation with single species tree plantations to restore ecosystems, holistic ecosystem regeneration practices in line with natural successional processes has been shown to be a more effective strategy for achieving carbon sequestration and other overarching environmental goals (Seddon et al., 2019). Reducing atmospheric CO_2 concentrations through sequestration in the pedosphere and biosphere has many potential advantages. It serves to mitigate climate change, improve ecosystem resilience, and, when it is achieved through sustainable agricultural practices, it can increase SOC, improve soil quality, advance global food security and possibly provide another economic generation stream for farmers (i.e., carbon credits) (FAO and ITPS, 2021; Lal, 2008). As most nations have committed to limiting global average temperature rise to well below 2° C, soils are now central to global climate change mitigation efforts (Minasny et al., 2017). This has resulted in the consensus that carbon-depleted agricultural soils have the greatest potential for immediate action to promote carbon sequestration as a natural climate solution (Amelung et al., 2020).

Soils as a whole still contain more carbon than plants and the atmosphere combined (Figure 6), but the depletion of agricultural SOC stocks by intensive practices throughout human history (12 millennia of agriculture) has left a global carbon *debt* of approximately 116 Gt ("Correction for Sanderman et al., Soil carbon debt of 12,000 years of human land use," 2018; Sanderman et al., 2017), roughly equivalent to all the CO_2 emitted by the United States since 1800. Since the Industrial Revolution alone, 78±12 Pg C has been released to the atmosphere from soils used as croplands, pastures, and rangelands, together covering nearly half of the Earth's surface that can support vegetation (Bondeau et al., 2007; Foley et al., 2005; Lal, 2004b). Fortunately, carbon-depleted soils have the ability sequester SOC, either naturally or through specific SOM-friendly management practices as described in the previous section. The leading international initiative aiming to leverage the climate mitigation potential of carbon sequestration in agricultural soils is known as *"4 per 1000" Soils for Food Security and Climate*. Launched at UNFCCC COP21 by the French Ministry of Agriculture, it suggests that by increasing global soil organic matter by 0.4% per year through improved agricultural practices, 2–3 Gt C year^{-1} could be sequestered, which would effectively offset 20–35% of anthropogenic greenhouse gas emissions (Minasny et al., 2017). Although it may not be necessary nor productive to target such a specific and potentially unrealistic sequestration rate (de Vries, 2018; Poulton et al., 2018; White et al., 2018), there is no debate that SOC on active agricultural lands globally must be increased and stabilized as much as possible within ecological and socioeconomic limits (Bossio et al., 2020; Bradford et al., 2019; Vermeulen et al., 2019; Zomer et al., 2017). With that being said, the cessation of agriculture altogether (i.e., ALA) is often the most efficient way to restore ecosystems and sequester carbon in tandem and at scale.

Due to the SOC-depleted nature of agricultural lands, the widespread historical and ongoing prevalence of ALA, and the innate ability of soils to rebuild SOC stocks, abandoned agricultural lands represent some of the largest human-induced carbon sinks ever measured. In the post-Soviet states, vast expanses of forests regrew over the 62.6 Mha of croplands abandoned following the collapse of the Soviet Union (Schierhorn et al., 2019), and in this process drew enormous quantities of carbon down into the above- and belowground biomass and soils (Henebry, 2009). Estimates of the size of this sink range in the hundreds of teragrams, depending on the spatial extent considered, carbon pools included, and methods of analysis (Kuemmerle et al., 2011; Schierhorn et al., 2013; Vuichard et al., 2008), with significant annual SOC accrual rates comparable to intentional restoration initiatives (Dymov et al., 2018; Kalinina et al., 2015; Kurganova et al., 2015, 2014; Wertebach et al., 2017). Further back in human history, the deadly arrival of Europeans to the Americas tragically lead to the abandonment of an estimated 55.8 Mha, which subsequently regrew into secondary forests, sequestering 7.4 Pg C and possibly intensifying the post-medieval Little Ice Age (Koch et al., 2019). On a more practical level, ambitious regional efforts that intentionally restore active and abandoned agricultural lands, like the "Grain-for-Green" program in China and the Conservation Reserve Program in the USA, have also demonstrated the significant ecological and carbon sink co-benefits possible following agricultural cessation (Deng et al., 2014a, 2014b; Munson et al., 2012; Robles and Burke, 1998; Shi and Han, 2014).

Once agricultural activities are abandoned, there are multiple ways to measure and calculate the SCS rates present. Time-stamped data points are crucial for understanding what factors determine if and when a given plot of abandoned land will act as a carbon sink or source following agricultural cessation. The gold-standard is unquestionably repeated measurements on the same plot of land over time, ideally in a long-term experimental setting, as it will have the most reliable and representative conditions. However, global biogeochemical models including ALA are currently limited by relatively poor temporal SOC data (i.e., low quality and quantity) due to the logistical and financial challenges of long-term field sites with repeated measurements, especially in under-resourced regions. Statistically robust and geographically representative temporal SOC response curves are needed to improve model accuracy and prediction certainty, and thereby strengthen land management policies that incorporate ALA and ecosystem restoration objectives for agriculture lands.

Investing new money, time, and energy in long-term field sites may not be necessary nor practical. Previously published space-for-time substitutions, like *paired-plots* and *chronosequences* (Figure 8), are readily available alternative data sources that can be used to complement datasets of repeated field measurements because they have not yet been sufficiently synthesized at different geographic scales (mainly a lack of comprehensive continental and global syntheses) (Huggett, 1998). Paired-plots of ALA comprise of one control plot under active agricultural practices (i.e., stage 0) paired with one field plot where the same agricultural practices have since ceased at a known time in the past (years). Chronosequences of ALA are simply a series of two or more field plots differing in time since abandonment, each paired with the same control plot. All plots must be comparable in all other environmental and human management factors (i.e., similar soil type, climate, vegetation, management and cropping practices, restoration practices, etc.), such that the modulating effects of time since abandonment on the investigated variable (e.g., soil biological, chemical, and physical properties) is isolated. Due to these sampling constraints, the individual field plots of paired-plots and chronosequences are typically identified and selected in as close proximity to each other as possible to ensure the similarity of environmental and land use history factors.

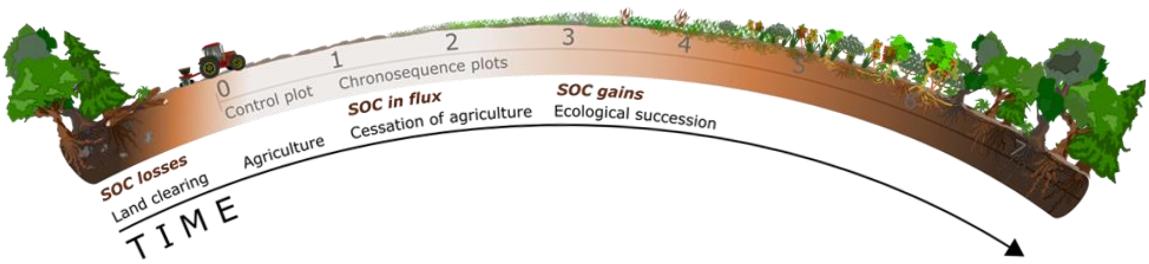

Figure 8. Typical landscape progression involving (left-to-right) land clearing from primary forest, agriculture, agricultural cessation, and ecological succession into secondary forest. Chronosequences of ALA identified in the field require multiple individual plots: a control plot under active agricultural practices (i.e., stage 0) and associated abandoned plots at known times (years) since agricultural practices (i.e., stages 1, 2, 3, etc.). When sampled collectively and analyzed sequentially, these plots represent the temporal trajectory of SOC and other parameters during ecological succession in response to historical ALA. Under normal conditions without stalled succession (i.e., if the soil is not degraded beyond recovery), ALA initiates the spontaneous regeneration of below- and aboveground biomass (i.e., carbon stocks) and the return of pre-agricultural soil physicochemical conditions. Traditional theory presumes that SOC is lost following the clearing of land and the initiation of agricultural practices, followed by a period of flux immediately after ALA and a period of gradual accumulation during ecosystem recovery until reaching saturation (i.e., a new soil carbon equilibrium).

Repeated measurements sample one field site repeatedly, meaning the samplings are separated through time but not space. As space-for-time substitutions, paired plots and chronosequences sample multiple field sites (separated in space) at the same time, with time since abandonment accounted for *a priori* through knowledge of the land use history. While they are not ideal, they are logistically superior to repeated measurements and generally informative for exploring broad ecological theories, especially when investigating processes that can surpass the career or life span of researchers (e.g., SCS) (Walker et al., 2010). By extracting, synthesizing, and repurposing time-stamped SOC data from any published study involving the use of paired-plots or chronosequences of ALA, no matter the original research angle so long as SOC is reported, we can more robustly benchmark and validate models of successional carbon dynamics.[1] Early limited attempts of this at regional and national scales have already been performed, but not robustly and not at continental nor global scales, resulting in an incomplete understanding of one of the world's major LULCCs: ALA.

To overcome spatial and inferential limitations of individual experiments, and to detect underlying driving factors for many processes present in ecology, one of the best statistical approaches is to synthesize response ratios of multiple studies and perform meta-analyses (Gurevitch et al., 2001; Hedges et al., 1999). In the case of the response of soil carbon to ALA, there has not been a dedicated, comprehensive, and statistically robust global study. Nevertheless, there have been two seminal studies exploring SOC responses that include ALA among other LULCCs, originally published over two decades ago and receiving increasing interest: Guo and Gifford (2002) and Post and Kwon (2000). As of June 2022 on *Google Scholar*, Guo & Gifford (2002) has been cited 4001 times (an increase of 2505 since January 2019), while Post & Kwon (2000) has been cited 3201 times (increase of 2083 since January 2019). Yet, despite the high interest and fact that they are global in scale, both studies are severely data sparse (e.g., < 50 observations for ALA related LULCCs, such as crop to grassland or forest conversions). Guo & Gifford (2002) found an overall 53% increase in SOC stock following crop to secondary forest; however, the authors noted that the low quantity of available data combined with the diversity of methodologies used prevented strong statistical conclusions. The authors also noted that broadleaf tree plantations on native forest land or pastures did not affect SOC stock, while pine plantations reduced C stock. This early finding in the literature supports the recent shift for more research into natural climate solutions like restoration of abandoned agricultural lands over tree plantations as SCS strategies. Post & Kwon (2000) conducted a more in-depth review of SOC dynamics during natural regeneration after agricultural cessation and found average rates of accumulation to be 33.8 and 33.2 g C m^{-1} y^{-1} for forests and grasslands, respectively. They found this level of sequestration, when considering the global land area potentially affected by ALA at the time, to be relatively small in comparison to the estimated rate of total carbon sequestration occurring in the Northern

[1] This is also an intentional exercise in repurposing past data, revaluing past investments (e.g., research funding), and recapturing the often neglected inferential and transformative potential of the "long tail of dark data" (Heidorn, 2008; Novick et al., 2018) (i.e., uncompiled, underrepresented, and unique past research at risk of disappearing due to the threat of continual data availability loss (Vines et al., 2014)).

Hemisphere in biomass and surface litter. However, as in the case of Guo & Gifford (2002), this review includes only a small number of observations (47) and uses unreliable estimates of the extent of land abandonment in the Northern Hemisphere (e.g., all sources are from before the dissolution of the Soviet Union (1988–1991)).

In the two decades since Guo & Gifford (2002) and Post & Kwon (2000), there have been hundreds of individual field studies on post-agricultural SOC dynamics, which have since been more comprehensively summarized in a few synthesis studies at regional and biome scales (Table 1). Don et al., (2011) reviewed 385 studies on LULCC in the tropics and found that a conversion of cropland to secondary forest resulted in an increase of SOC by 50.3±11.9% over an average of 32±7 years, with lesser increases for cropland to grassland (25.7±11.1% over 21±6 years) and cropland to fallow (32.2±16.1% over ≤ 7 years). Their study was restricted to the tropics and, presumably because the authors considered several other LULCC types, there were relatively few observations compiled for cropland revegetation conversions (25 for cropland to secondary forest; 16 for cropland to grassland; 21 for cropland to fallow). Additionally, the paper specifically refers to afforestation and not natural vegetation succession nor abandonment. Poeplau et al., (2011) found that for a cropland to grassland LULCC in the temperate zone, SOC accumulated at a rate of 40±11% over 20 years and 128±23% over 100 years based on the model predictions. This was the highest gain among the LULCCs considered in their study. For cropland to forest, SOC increased at a rate of 16±7% over 20 years and 83±39% over 100 years. This paper also included a limited number of observations: 89 for cropland to grassland and 70 for cropland to forest. The authors also specifically use the term afforestation, so it is unclear how many, if any, of the 15 observations of the cropland to forest LULCC category were natural vegetation succession (i.e., reforestation), which is the trajectory most representative of ALA. Kämpf et al., (2016) assessed the potential of temperate agroecosystems for SCS under different climatic and edaphic conditions. Their study looked at global temperate soils and identified, among other LULCCs, 54 observations of SCS as a result of ALA from 17 publications. Over an average of 14 years since abandonment, SOC stocks increased by 18%, with a sequestration rate of 0.72 t C ha^{-1} y^{-1}. The authors also found that SCS may not be limited by low primary productivity within the temperate zone, providing incentive for more comprehensive analyses at different spatial scales and comparing multiple biomes. W. Li et al., (2018) compiled a global dataset of 836 observations of grassland-land related LULCC, including 194 observations of cropland to grassland conversion, and the subsequent changes in SOC stock. The cropland to grassland LULCC category resulted in a

SCS rate of 4.3 kg C per m^2 (median) over 79 years (median), a relative increase of 46%. While their study offers a useful dataset, it unfortunately does not include the key LULCC of cropland to forest because the authors specifically targeted only grassland-related changes. Furthermore, the authors used satellite-based net primary productivity (NPP) observations as a proxy for carbon input.

Table 1. Summary of large geographic scale meta-analyses of SCS and LULCCs related to ALA.

Authors	Year	Region	LULCC	N. of Observations	N. of Studies	SCS	Years
Guo & Gifford	2002	Global	C to F	9	?	53%	?
Post & Kwon	2000	Global	C to F	47	28	33.8g C m^{-2} y^{-1}	n/a
Laganière, et al.	2010	Global	C to F	92	?	26%	?
Don, et al.	2011	Tropical	C to F	?	25	50.3±11.9%	32
Don, et al.	2011	Tropical	C to G	?	16	25.7±11.1%	21
Poeplau, et al.	2011	Temperate	C to F	70	15	16±7%	20
Poeplau, et al.	2011	Temperate	C to F	70	15	83±39%	100
Poeplau, et al.	2011	Temperate	C to G	89	24	39.8±11%	20
Poeplau, et al.	2011	Temperate	C to G	89	24	128.4±23.2%	100
Kämpf, et al.	2016	Temperate	C to G	54	17	18%	14
W Li, et al.	2018	Global	C to G	194	?	46%	79
Deng, et al.	2016	Global	C to F	31	?	*Not significant*	n/a
Deng, et al.	2016	Global	C to G	57	?	0.30 Mg $ha^{-1}yr^{-1}$	n/a

(C to F: cropland to forest; C to G: cropland to grassland; ? : not indicated)

Elsewhere in the world, individual studies that include temporal SOC data following ALA can be found on every continent, and national-scale meta-analyses synthesizing SOC dynamics after ALA and similar post-agriculture LULCCs have been undertaken in China, Russia, and Northern Europe. Deng, Liu, et al., (2014) synthesized 135 studies, which included 844 observations at 181 sites, to determine the effects of China's "Grain-for-Green" Program. The authors found a SCS rate of 0.33 Mg ha^{-1} yr^{-1} in the top 20 cm of soil. While Hong et al., (2020) is one of the most recent and data-rich SOC studies with 619 afforested paired-plots, it only covers northern China, and mixes post-agricultural soils with other non-forested soils like barren land, grassland, natural forest and riparian sand land. Kurganova et al., (2014) compiled a database of 116 paired plots from 45 sites across Russia and calculated an average SCS rate of 0.96 Mg ha^{-1} yr^{-1} in the top 20 cm of soil over the first 20 years since abandonment. In their meta-analysis of SCS following afforestation in Northern Europe, Bárcena et al., (2014) did not consider natural forest regrowth, but found the largest increase in SOC occurred on former croplands (compared to former grasslands, heathlands, and barren lands) at 20% for the 0–10 cm depth based on 51 observations. Studies such as these provide useful databases and reference lists that can be included into larger continental or global meta-analyses.

Most meta-analyses indicate a positive effect on SOC stocks after conversion of croplands to forests globally. However, there still remains significant data and knowledge gaps and a need for more robust and specific continental and global meta-analyses explicitly dedicated to SCS following ALA, as a globally relevant LULCC. Indeed, ALA has occurred anywhere in the world where natural land has been converted into agricultural use, and has the potential to continue to occur as long as environmental, social, and economic factors remain dynamic. When viewed as a climate change mitigation strategy, it is arguably the only strategy that does not require any additional resources to initiate and is possible to implement anywhere in the world when conditions permit (e.g., when conflicts with food production, agrobiodiversity, or land rights, etc., are not present). It is now clear we must consider integrating ALA and protecting existing abandoned agricultural lands whenever possible to diversify the global land carbon sink and support the UN Decade of Ecosystem Restoration (2021-2030) (Abhilash, 2021; Aronson et al., 2020). However, ALA is poorly represented in terrestrial carbon models, both spatially as a land classification and temporally as carbon sinks. The lack of temporal SOC data in particular has resulted in severe uncertainties on the intensity, the longevity, and the modulating factors of SCS following ALA. This hinders our ability to monitor, quantify, and leverage ALA processes strategically, precisely when we need to most.

1.5 Statement of the problem

1. **Demand for policy support:** Despite the potential abandoned agricultural lands represent for climate change mitigation through SCS, they are still underrepresented in global change science and policy. ALA is a global land use change present in every agricultural region of the world. Due to its contentious sociocultural aspects, there is no standard framework on how abandoned agricultural lands should be managed for ecosystem and climate-related goals. In Europe, not only is ALA expected to be a significant driver in land use dynamics in the coming decades, but most EU climate policies dealing with land explicitly call for dedicated attention given to rebuilding the continent's soil carbon pool. There is currently a strong need for increased scientific research efforts to support policy decisions concerning the SCS implications of processes like ALA.
2. **Logistical barriers and data limitations:** One of the most important analytical dimensions needed to be able to fully evaluate the carbon sink potential of ALA at large geographic scales and over longer policy horizons is still severely lacking: namely, time. Current state-of-the-art biogeochemical models that incorporate the effects of ALA on soil carbon stocks are limited by relatively poor temporal data (i.e., low sample sizes) due to the logistical and financial challenges of long-term field studies with repeated measurements, especially in under-resourced regions of the world. To produce robust datasets considering the full temporal dynamics of SOC following ALA, alternative methodologies that are faster, more accessible, and less expensive are needed. Fortunately, there exists thousands of published studies with data that provides the necessary temporal resolution, spatial coverage, and combinations of soil type, crop type, and management practices. Instead of resource intensive repeated measurements, these studies utilize chronosequences (i.e., simultaneously sampled soil plots sharing key environmental and experimental conditions but differing in time since the implementation of the investigated practice). Chronosequence data has not yet been comprehensively extracted, standardized, and analyzed to assess the temporal dynamics of SOC following ALA.
3. **Unknown soil carbon sequestration rates following ALA:** Insufficient calibration of temporal soil carbon trends directly results in lost opportunities to prepare for the long-term impacts of ALA and to properly account for its role as a driver or mitigator of climate change. SCS rates following abandonment are either statistically weak or non-

existent in many regions of Europe, let alone the world. We lack the ability to accurately predict whether any given plot of agricultural land will sequester, lose, or maintain pre-existing SOC levels following abandonment, and we do not know how any of these SOC trends will behave through time or at different geographic spatial scales.

4. **Unknown modulating factors of soil carbon sequestration rates following ALA:** Due to the issues mentioned above, the current conceptual framework explaining the differences in sequestration rates observed worldwide is underdeveloped. This is especially true in Europe where there has not been a continental-scale analysis, despite the widespread extent of past and ongoing ALA and the clear motivation to increase European soil carbon stocks. Researchers have not reached a theoretical consensus on the driving factors behind different sequestration rates that may explain why abandoned croplands may lose SOC in one area of Europe while they gain SOC in other areas. Site-specific conditions and human management factors have not been comprehensively explored in the literature; they are still poorly quantified due to the complexity of possible interactions.

These problems and research gaps hinder decisionmakers and land managers from promoting (or deterring) ALA in rural areas as a component of regional climate initiatives—inconsistent with sustainable land management objectives. Guidance on management pathways based on accurate representation of the temporal responses of SOC to ALA is a top-priority.

1.6 Aim & research questions

The overall aim of this PhD thesis is to **generate new knowledge on the effects of agricultural land abandonment on soil carbon stocks**, thereby contributing practical information for sustainable land management decisions involving climate change mitigation. I investigate the temporal dynamics of SOC following the cessation of agricultural activities at the field (i.e., Province of Barcelona), regional (i.e., Spain), and continental (i.e., Europe) scales. Ultimately, this thesis provides novel insights into the capacity of European agricultural soils to *recarbonize* through ecological succession, whether natural or assisted. The following research questions (RQ) are addressed:

RQI. *What are the main categories of sustainable land management options for abandoned agricultural lands proposed by researchers, and how do they impact soil carbon stocks?*

I conduct a literature review to identify the main categories of sustainable land management strategies that have been proposed for existing abandoned agricultural lands or for active agricultural lands directly converted. Following this review, I then compare the SCS rates reported in the published literature for these categories.

RQII. *How does soil organic carbon in Spain respond to agricultural land abandonment, and what are the modulating factors?*

I undertake field work in the Province of Barcelona, Spain to explore the effects of depth and time on SCS following agricultural land abandonment. I then synthesize published chronosequence data across peninsular Spain to identify the potential factors responsible for the high variability in post-agricultural SCS rates observed in the Mediterranean region.

RQIII. *How do soil organic carbon across Europe respond to agricultural land abandonment, and what are the modulating factors?*

I compile and synthesize published chronosequence data of SOC change over time on post-agricultural lands in European countries, exploring potential environmental and human management factors driving SCS. I investigate the conditions by which the highest and lowest sequestration rates may be expected, and provide new insight into the capacity of European abandoned agricultural lands to serve as carbon sinks into the future.

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1.8 Supplementary materials

Table 2. Typically used delineation of SOM fractions and SOC pools according to the previous SOM conceptual paradigm.

SOM fractions	**Residence Time**	**Characteristics**
Dissolved	Minutes to days	• < 45µm • Root exudates, simple sugars, by-products of decomposition • < 5% of SOM
Particulate	2–50 years	• 53µm–2mm • Fresh and decomposing plant and animal matter • 2–25% of SOM
Humus	10s–100s years	• < 53µm • Longer-lasting organic compounds under slow decomposition • > 50% of SOM
Resistant	100s–1000s years	• < 53µm–2mm • Comparatively inert, chemically resistant remnant organic materials • < 10% of SOM
SOC pools	**Residence Time**	**Characteristics**
Fast (labile or active)	1–2 years	• Highly unstable and subject to decomposition
Intermediate	10–100 years	• Microbially processed and partially stabilized on mineral surfaces and protected within soil aggregates
Slow (recalcitrant or stable)	100– > 1000 years	• Highly stabilized • Includes pyrogenic carbon

CHAPTER II: Management opportunities for soil carbon sequestration following agricultural land abandonment

2.1 Overview

The widespread historical and ongoing abandonment of agricultural lands worldwide presents important opportunities for promoting climate change mitigation through carbon sequestration. The default management outcome of abandonment is natural regeneration through ecological succession. However, several different management strategies and new land uses for abandoned agricultural lands have been recommended by the scientific community in recent years. This paper reviews the foremost proposed strategies and compares their soil carbon sequestration potentials. Six major categories have been proposed globally. Each proposal has positive and negative outcomes depending on site-specific factors and management objectives. Accordingly, no single strategy is ideal in all scenarios and a combination of strategies addresses multiple rural development goals concurrently. A combination of passive and active management techniques is the most effective approach for maximizing soil carbon sequestration over large geographic scales, while other strategies can be designed to also promote low-carbon land use practices and fossil fuel substitution. The implications of each proposal highlighted here demonstrates the positive role that abandoned agricultural lands can serve in climate change mitigation efforts, supporting policymakers tasked with planning the future of regions undergoing abandonment.

2.2 Introduction

Vast expanses of previously cultivated lands around the world are currently undergoing the processes of natural regeneration as a result of agricultural land abandonment (ALA) (Cramer et al., 2008; Queiroz et al., 2014; Rey Benayas et al., 2007). Upwards of 472 Mha, or over half of the land area of the USA, is estimated to have been abandoned over the last three centuries globally (Campbell et al., 2008). With a further 280 Mha of shifting agriculture currently undergoing cyclical abandonment (Heinimann et al., 2017), ALA as a land use change (LUC) presents both important management challenges and opportunities.

While scientists and society debate whether or not to intervene with the regeneration process post-abandonment to control the negative impacts of revegetation and promote desired outcomes (Lasanta et al., 2015; Rey Benayas et al., 2008), alternative options are receiving increasing attention (Abolina and Luzadis, 2015; Campbell et al., 2013; Hall, 2018; Knoke et al., 2014; Pace Ricci and Conrad, 2018; Schröder et al., 2018; Smaliychuk et al., 2016). This growing body of research has gone beyond describing ALA solely from a LUC perspective, and has instead proposed management strategies to guide sustainable transitions into new land use classifications. The recognized importance of land management for climate change mitigation efforts (IPCC, 2019), alongside increasing demand for land resources, provides ample reason to explore what these proposals are, the development goals they can meet, and their implications for climate change.

The default management strategy post-abandonment is simply natural regeneration through ecological succession, usually triggering both aboveground (i.e., plant biomass) and belowground (i.e., soil) carbon accumulation (Silver et al., 2001). Considering that agroecosystems are often depleted of soil organic carbon (SOC) (Lal et al., 2015), quantifying the soil carbon sequestration potentials of each of these proposals is especially important for assessing their climate change mitigation relevance. With recent advances in our understanding of the consequences of ALA over the last two decades, it is also an opportune moment to review how managing abandoned lands can contribute towards replenishing the soil carbon pool.

In light of these issues, this paper identifies and compares the most commonly proposed post-abandonment management strategies globally through a review of the ALA literature. First, the ecological and rural development features of each proposal are discussed in support of policymakers tasked with planning the future of regions undergoing ALA. Second, the proposals with soil carbon sequestration rates reported on abandoned or converted agricultural lands synthesized over large geographic scales are compared to determine their relative potential contributions towards climate change mitigation. With the reduced importance of local environmental factors and differing agricultural histories at larger scales, active management through restoration is hypothesized to produce the highest rates of SOC accumulation post-LUC.

2.3 Methods

The following terms in various combinations were searched using ISI Web of Science, Scopus, and Google Scholar: “agricultural land abandonment”, “farmland abandonment”, “cropland abandonment”, “ex-arable lands”, “old fields”, “land abandonment”, “marginal land”, “degraded land”, “land use change”, and “land management”. Search results were limited to English language studies published over the last two decades. Each potential study was assessed first by title to determine suitability, then by abstract to determine potential for land management proposals, and lastly by main text to extract the most relevant information. Reference lists were also reviewed and relevant grey literature was consulted when applicable. As this is a global analysis, studies from all countries were considered. Management strategies and new land uses for abandoned agricultural lands that were proposed by several authors globally with case studies in at least two continents were selected for analysis. A second literature search was performed to collect SOC data following implementation of each of the selected management strategies. SOC sequestration rates were synthesized based on study area, resulting in three classifications of large geographic scales featured in this review: country, climatic zone, and global. All studies that were found reporting rates at the climatic zone or global scale of any of the proposed management strategies on abandoned or directly converted agricultural lands were included. Studies reporting data at the country scale were only included if they featured comparably large sampling sizes with significant geographic extent. Rates applying to the first three decades post-LUC and taken from topsoils (0–30 cm) where preferentially selected.

2.4 Results & Discussion

2.4.1 Proposed management strategies

Although most of the ALA literature focuses primarily on drivers (Baumann et al., 2011; Díaz et al., 2011; Osawa et al., 2016; Zhang et al., 2014), impacts (Cramer et al., 2008; Hanaček and Rodríguez-Labajos, 2018; Lasanta et al., 2015), and spatial distribution (Estel et al., 2015; Schulp et al., 2018; Verburg and Overmars, 2009; Wang et al., 2018; Yin et al., 2018), researchers have also discussed potential management strategies and new land uses for abandoned lands that take into account the unique social, environmental, and economic conditions and needs of the stakeholders involved. Our literature review revealed six major categories proposed by the scientific community globally (Figure 9). Three higher-tier categories were identified based on the primary desired function of the new land use in which all proposals could be further grouped (i.e., Naturalness, Multifunctionality, and Productivity).

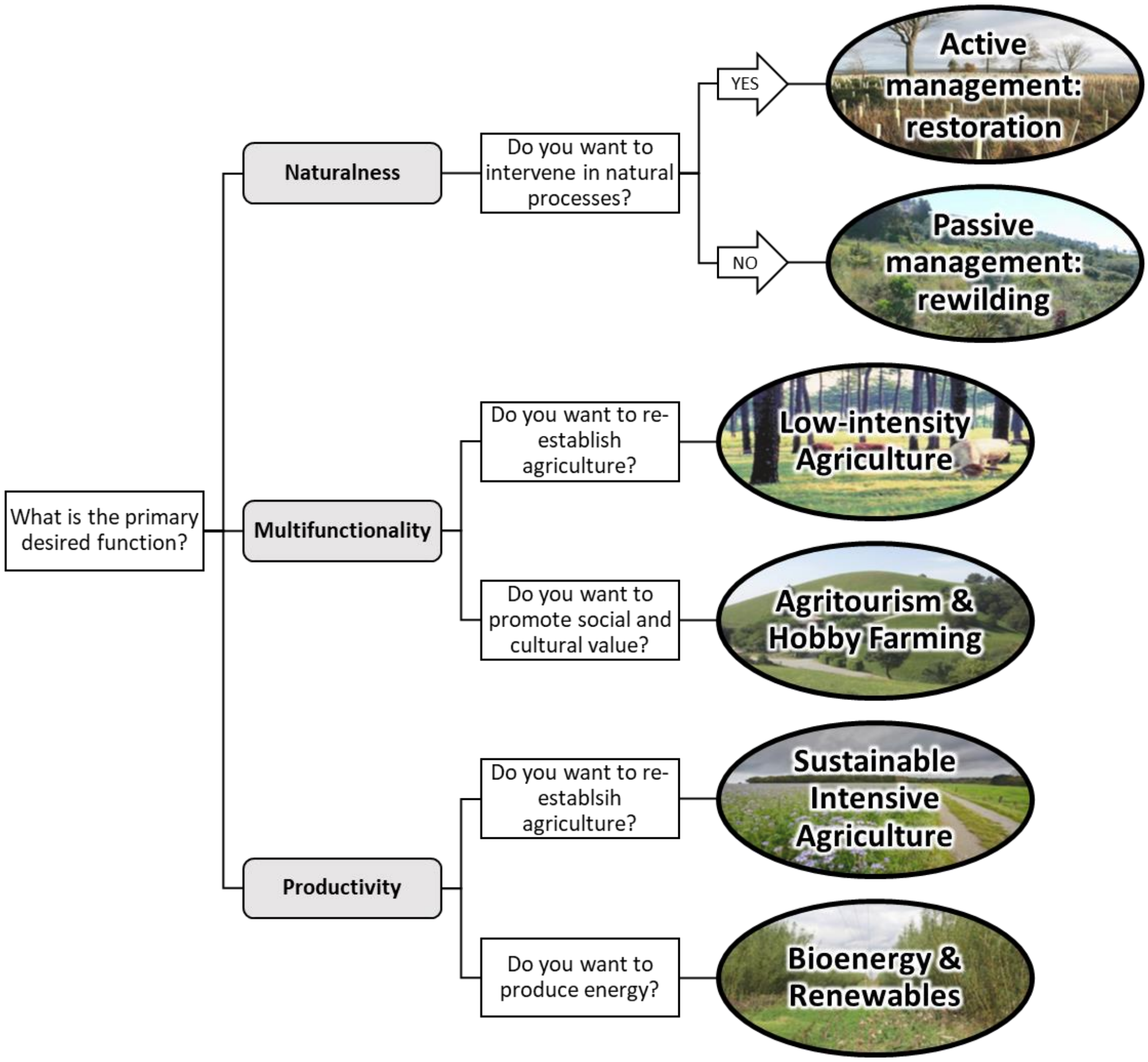

Figure 9. Decision tree highlighting the major categories of proposed management strategies and potential new land uses for abandoned agricultural lands globally. See Table 3 for a summary.

At the farm and landscape scale, abandoned land is often perceived negatively and devalued, motivating landowners and decision-makers to consider new management options (Abolina and Luzadis, 2015; Benjamin et al., 2007; Ruskule et al., 2013; van der Zanden et al., 2018). At the regional scale, a patchwork of several different adaptive management strategies likely offers the best compromise for balancing ecological, social, and economic interests through time (Merckx and Pereira, 2015; Pelorosso et al., 2011; Ruskule et al., 2013). Researchers acknowledge that implementing the most suitable post-abandonment strategy will be based on numerous variables, requiring increased efforts to address information gaps in the decision-making process (Meli et al., 2017). To that end, the ecological and rural development features of the six proposals are discussed below and summarized in Table 3.

2.4.1.1 Naturalness

Promoting naturalness on abandoned lands can be achieved with intensive human intervention (i.e., active management) or without (i.e., passive management). Assisted afforestation and grassland restoration via seeding and planting are commonly proposed active management practices. On the other end of the spectrum, passive management practices aim to restore naturalness through ecological succession (e.g., rewilding) (Perino et al., 2019).

Various factors play a significant role in the efficacy of both strategies, resulting in occasionally differing impacts. The agricultural history, ecosystem characteristics, and local lithology of abandoned lands may influence the rate at which vegetation recolonizes, biodiversity returns to native states, and soil properties improve (Robledano-Aymerich et al., 2014). For example, passive management on abandoned pastures in the Atlantic Forest region of Brazil resulted in low aboveground biomass and low biodiversity, suggesting the need for enrichment plantings, nucleation techniques (e.g., plantation in islands), and other forms of active management (Sansevero et al., 2017). In the Amazon on the other hand, (Rocha et al., 2016) specifically proposed natural regeneration as the ideal restoration strategy for abandoned pastures due to the speed and intensity of revegetation observed.

Climate is indeed a key factor on whether active or passive techniques deliver intended management outcomes. Throughout the tropics, rewilding has been found to outperform active management for restoring biodiversity and vegetation structure, challenging the view that natural regeneration has inferior conservation value in these ecosystems (Crouzeilles et al., 2017). In semi-arid climates, afforestation of abandoned lands produced semi-natural soil conditions after 40 years and formed important natural resource islands in the landscape, while natural succession resulted in minor non-linear improvements leaving the soil prone to erosion (Zethof et al., 2019). However, rewilding on abandoned lands is generally sufficient for faunal, floral, and biogeochemical recovery across most climates globally (Meli et al., 2017).

The local differences in efficacy between rewilding and active restoration are not always easy to discern. The variability of impacts of both strategies implies that decision-makers sometimes need to consider different solutions for seemingly similar situations. For instance, slower tropical forest regrowth rates on abandoned lands observed in Uganda, in comparison to South America, has led researchers to propose silviculture plantations to create new sources of income while allowing regeneration processes to occur under tree shelter (Chapman and Chapman, 1999). In the species-rich grasslands of Czech Republic, where ALA has been a significant regional LUC over the last few decades, passive management for grassland restoration is appropriate if the desired species are already present in ancient grasslands close by (Sojneková and Chytrý, 2015). In situations where this might not be the case, specific mixtures of regional seeds and hay transfer are recommended (Albert et al., 2019; Prach et al., 2014).

One solution often suggested is to employ both passive and active management within the same region, usually involving localized restoration techniques rather than widespread rewilding initiatives (Robledano-Aymerich et al., 2014). This would allow for the best of both worlds and is ideal for climates where natural regeneration of forests and grasslands post-abandonment would benefit from some level of human intervention. In the Mediterranean, for example, tailor-made strategies at the farm or landscape scale (e.g., focusing planting efforts locally in small groups of native trees to jumpstart natural regrowth) incorporated into regional mosaics of differing landscapes may support biodiversity and ecosystem goals more efficiently (Plieninger et al., 2014; Rey Benayas et al., 2008). From a productivity standpoint, rural development subsidies for both rewilding on marginal lands and active biodiversity protection with sustainable intensive agriculture on more productive lands is a practical approach (Merckx and Pereira, 2015). Rewilding has been especially recommended in areas where the socioeconomic activities of the landscape are no longer sustainable (Cerqueira et al., 2015).

2.4.1.2 Multifunctionality

Some of the proposals address a wider range of ecological and rural development goals, reducing the need to combine strategies within the same landscape. The most often proposed management strategies in the ALA literature in this category are forms of low-intensity agriculture, agritourism and hobby farming. Low-intensity agriculture provides sustainable food production and several other benefits valued in regions with marginal and abandoned lands (Mander et al., 1999). In Europe, for instance, silvopastoral systems are proposed both for their potential to provide beef while sequestering carbon (Hall, 2018) and their suitability as low-intervention tools for limiting potential negative impacts of unmanaged revegetation (e.g., wildfire risk) (Lasanta et al., 2015). In Central and South America, although government policies have promoted natural regeneration to increase biodiversity, numerous studies instead propose returning to, or conserve existing, low-intensity agricultural practices such as cacao and coffee agroforestry and the traditional *milpa* system to achieve additional benefits (De Beenhouwer et al., 2013; García-Frapolli et al., 2007; Jezeer et al., 2017; Londoño et al., 2017; Queiroz et al., 2014; Santos et al., 2019; Tscharntke et al., 2011).

Further proposals to reclaim or maintain existing traditional low-intensity agroecosystems in regions experiencing ALA have been made across diverse cultural and environmental settings globally, such as the *dehesa* and *montado* of the Iberian Peninsula (Arosa et al., 2017; Garrido et al., 2017) and the *satoyama* of Japan (Morimoto, 2011). Reclaiming traditional agricultural landscapes such as these can also support local agritourism due to the improved landscape aesthetics (i.e., tourist preferences for active over abandoned lands) (Zagaria et al., 2018), as has been proposed for the Noto Peninsula in Japan (Chen et al., 2016) and Mediterranean mountainous regions (Sayadi et al., 2009). Agritourism can be sustainable in regions experiencing ALA while building socioeconomic resilience to climate induced threats to rural livelihoods (Sanagustín-Fons et al., 2011; Valdivia and Barbieri, 2014).

Another approach with sociocultural benefits for rural development involves promoting hobby and community social farming (Knapik, 2018), especially on abandoned lands located near urban areas with large populations of interested citizens and engaged landowners (Pace Ricci and Conrad, 2018). Further away from urban areas, hobby farming can also overcome hereditary barriers associated with abandoned lands by bringing in new families to farm the land and revive rural community life (Varotto and Lodatti, 2014).

2.4.1.3 Productivity

The third higher-tier category of proposals, which includes sustainable intensive agriculture and bioenergy and renewables, involves generating economic value on abandoned lands through producing goods (i.e., food and energy) sustainably. Calls for sustainable agricultural intensification have been made worldwide to address increasing land pressures and demand for food. Complementary to efforts to transition conventional agricultural lands, abandoned and marginal agricultural lands are now receiving attention in this regard (Schröder et al., 2018). Reclamation of abandoned lands in Ecuador, for example, has been proposed as a way to increase food production, improve land allocation, decrease food prices, and prevent forest conversion to agriculture (Knoke et al., 2013). For global dryland regions, reestablishment of agriculture on abandoned or marginal lands inherently entails incorporating best management practices from integrated soil fertility management and conservation/climate smart agriculture (Shahid and Al-Shankiti, 2013). The greatest potential for sustainable intensification possibly lies in post-Soviet states where significant recent reclamation has already occurred (Schierhorn et al., 2013).

The basis of proposing bioenergy crops and renewables on abandoned lands is the argument that competition for land with food production, one of the central constraints of large-scale implementation, would no longer be a factor. These two energy production methods are already promoted as ways to reduce greenhouse gas emissions and reliance on fossil fuels while creating new development opportunities in the agricultural sector. Consequently, researchers argue that their integration on marginal and abandoned lands greatly increases their attractiveness and large-scale viability (Campbell et al., 2008; Edrisi and Abhilash, 2016; Fargione et al., 2008; Milbrandt et al., 2014; Nijsen et al., 2012; Valentine et al., 2012; Vuichard et al., 2009). Campbell et al., (2008) estimated the global potential of bioenergy on abandoned lands, and found that it could easily satisfy the national energy demand of some African nations, while reaching less than 10% of the demand for most countries in North America, Europe, and Asia. Subsequent studies have argued that this option can still meet a meaningful proportion of the primary energy demand of several countries across these continents (Campbell et al., 2013; Liu et al., 2017; Nijsen et al., 2012).

Renewable energy technologies established on marginal and abandoned lands are also worthy of consideration. In the USA, such installations could significantly increase domestic energy supply even if only a fraction of the potential is realized, with solar power offering the highest return (Milbrandt et al., 2014). Importantly, large-scale implementation of bio- or renewable energy production on marginal and/or degraded lands is considered by some to be too costly due to accessibility and soil productivity issues, implying that abandoned lands may be a more suitable option (Baxter and Calvert, 2017; Bryngelsson and Lindgren, 2013; Shortall, 2013; Swinton et al., 2011).

Table 3. Summary of proposed management strategies and potential new land uses for abandoned agricultural lands globally.

Proposal	Purpose	Reasons for proposal	Ideal landscape conditions	Relevant studies
Naturalness				
Active Management: Restoration	Ecosystem services through managed restoration	• Increased aboveground carbon sequestration • Reduced soil erosion • Potential return of large fauna • Strategic management of forest regrowth	• Degraded • High ecological value • Low potential for natural regeneration	(Knoke et al., 2014; Prach et al., 2014; Rey Benayas et al., 2008; Tomaz et al., 2013; Tremblay and Ouimet, 2013; Zethof et al., 2019)
Passive Management: Rewilding	Ecosystem services naturally without intervention	• Increased belowground carbon sequestration • Recovery of soil functions • Potential return of large fauna • Increased biodiversity • Low-cost	• Degraded • High ecological value • High potential for natural regeneration	(Basualdo et al., 2019; Crouzeilles et al., 2017; Jiao et al., 2011; Meli et al., 2017; Navarro and Pereira, 2012; Regos et al., 2016; Scott and Morgan, 2012)
Multifunctionality				
Low-intensity Agriculture	Sustainable food production and ecosystem services	• Increased habitat connectivity, biodiversity, above- and belowground carbon sequestration (vs. conventional agriculture) • Recovery of soil functions • Reduced wildfire risk (with grazing) • Sustainable food production • Rural development opportunities	• High ecological value • Arable soils	(Arosa et al., 2017; Hall, 2018; Hombegowda et al., 2016; Jezeer et al., 2017; Lasanta et al., 2015; Londoño et al., 2017; Morimoto, 2011; Santos et al., 2019; Smith et al., 2013; Torralba et al., 2016)

Proposal	Purpose	Reasons for proposal	Ideal landscape conditions	Relevant studies
Agritourism & Hobby Farming	Traditional/small-scale food production and rural development	• Social/ecological resilience to climate threats • Rejuvenate lands under inactive ownership • Preserve cultural and aesthetic landscapes • Rural development opportunities	• Association with traditional agricultural practices • Low ecological value • Near large populations	(Chen et al., 2016; Pace Ricci and Conrad, 2018; Sanagustín-Fons et al., 2011; Sayadi et al., 2009; Valdivia and Barbieri, 2014; Varotto and Lodatti, 2014; Zagaria et al., 2018)
Productivity				
Sustainable Intensive Agriculture	Sustainable large-scale food production	• Alleviate land pressure/avoid deforestation (through use of abandoned lands) • Recovery of some soil functions • Improved food supply	• Low ecological value • Arable soils	(Benjamin et al., 2007; Griffiths et al., 2013; Knoke et al., 2013; Schröder et al., 2018; Shahid and Al-Shankiti, 2013; Smaliychuk et al., 2016)
Bioenergy & Renewables	Sustainable low-carbon energy production	• Avoid competition with land-based food production (through use of abandoned lands) • Reduce emissions and reliance on fossil fuels • Supplement domestic energy supplies • Rural economic opportunities	• Degraded • Low ecological value • Low potential for natural regeneration/restoration	(Abolina and Luzadis, 2015; Baxter and Calvert, 2017; Calvert and Mabee, 2015; Campbell et al., 2008; Fargione et al., 2008; Liu et al., 2017; Milbrandt et al., 2014; Nijsen et al., 2012; Tilman et al., 2006; Vuichard et al., 2009)

2.4.2 Soil carbon sequestration rates of proposals

Recognized for its role in climate change mitigation, soil carbon sequestration removes and stores atmospheric carbon for longer periods of time compared to other terrestrial carbon pools while providing valuable ecosystem co-benefits (Lal et al., 2015). Efforts to increase SOC stocks of active agricultural lands, such as the international initiative "4 per 1000", centre on leveraging these features to offset anthropogenic greenhouse gas emissions (Minasny et al., 2017). Abandoned agricultural lands therefore offer additional opportunities for large-scale sequestration initiatives. Several global and biome-wide syntheses of LUC have shown that biomass and SOC accumulation is most pronounced on lands previously cultivated that are regenerating under passive or active management due to the tendency of these lands to be SOC depleted (Deng et al., 2016; Don et al., 2011; Guo and Gifford, 2002; Kämpf et al., 2016; Laganière et al., 2010; Li et al., 2018; Poeplau et al., 2011; Post and Kwon, 2000). In addition to these studies, researchers have also reported soil carbon sequestration rates following conventional agriculture conversion to low-intensity agriculture and bioenergy production. This allows for a comparison of rates for these proposals synthesized over large geographic scales (Figure 10).

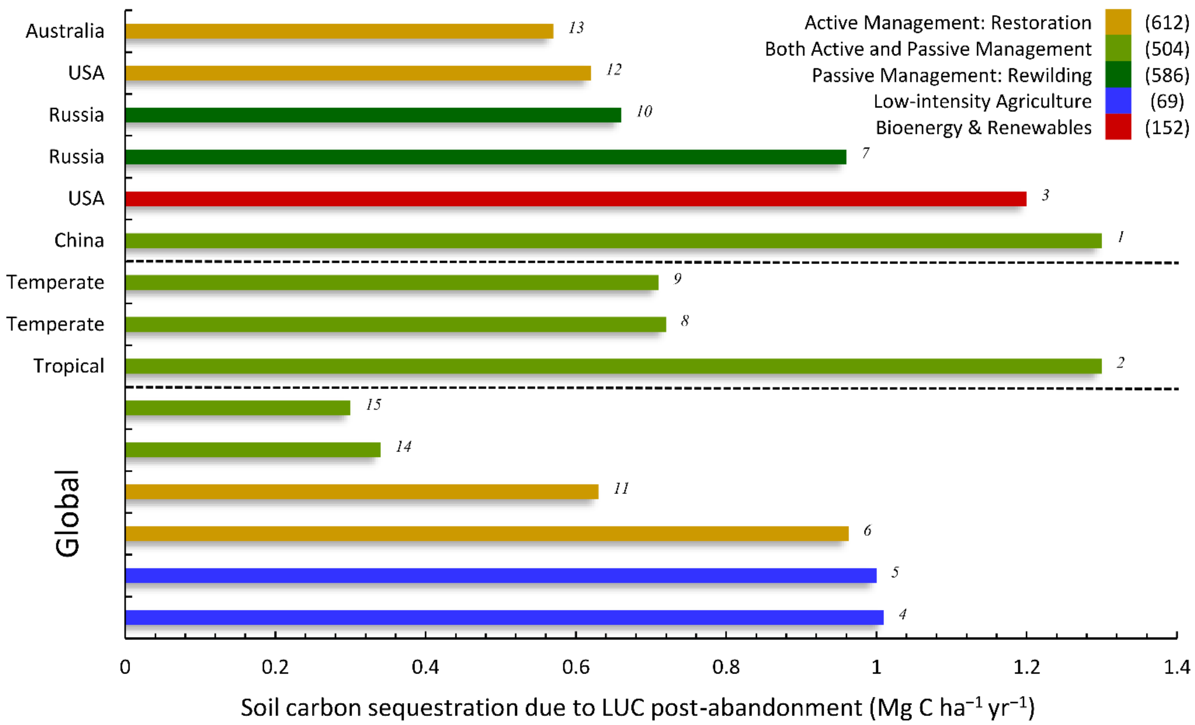

Figure 10. Soil carbon sequestration rates of proposals implemented on abandoned or converted agricultural lands synthesized at the country, climatic zone, and global scales. Combined number of observations from studies of each proposal indicated in parentheses.

Numbers next to bars indicate studies in order from highest to lowest sequestration: [1]Deng et al., (2014), 11-30 years post-abandonment; [2]Silver et al., (2000), 20 years post-abandonment; [3]Tilman et al., (2006), first 10 years, derived from net life cycle CO2 sequestration, includes roots; [4]Conant et al., (2001), cropland conversion to pasture; [5]Feliciano et al., (2018), cropland conversion to agrisilviculture; [6]Li et al., (2012), mixed species plantations; [7]Kurganova et al., (2014), 20 years post-abandonment; [8]Kämpf et al., (2016), various years post-abandonment; [9]Poeplau et al., (2011), 20 years post-abandonment; [10]Wertebach et al., (2017), 20 years post-abandonment; [11]Shi et al., (2013), 20 years post-abandonment; [12]McLauchlan et al., (2006), mixed species plantations; [13]England et al., (2016), mixed species plantations; [14]Post and Kwon, (2000), various years post-abandonment; and [15]Deng et al., (2016), cropland conversion to grassland.

All the proposals with reported sequestration rates had a positive impact on SOC over the first two to three decades. The most effective approach is to combine passive rewilding and active restoration techniques, enabling soil carbon sequestration rates as high as 1.30 Mg C ha^{-1} yr^{-1} across the tropical zone and in China (Deng et al., 2014; Silver et al., 2001). When synthesized globally, however, active management through restoration alone has a higher maximum reported sequestration potential compared to studies considering both active and passive management techniques. By weighted average of all study observations, restoration at the global scale sequestered 0.87 Mg C ha^{-1} yr^{-1} (Li et al., 2012; Shi et al., 2013) while restoration combined with rewilding sequestered 0.34 Mg C ha^{-1} yr^{-1} (Deng et al., 2016; Post and Kwon, 2000).

Rewilding alone has higher reported rates than restoration at the country scale, although still slightly lower than restoration at the global scale. The widespread ALA in the former Soviet Union allowed for passive rewilding of 58 Mha in Russia and Kazakhstan (Kurganova et al., 2015). The studies from this region included in Figure 10 estimate a weighted average sequestration rate of 0.72 Mg C ha^{-1} yr^{-1} (Kurganova et al., 2014; Wertebach et al., 2017). This exact rate has been observed across the entire temperate zone for grasslands regeneration post-abandonment (Kämpf et al., 2016).

The high variability in rates reported between active, passive, and combined management strategies is likely attributed to contrasting study parameters, including past (e.g., cropland vs. pasture) and new (e.g., grassland vs. forest) land use, ecosystem characteristics, time since ALA, geographic scale, sampling depth, and management approach and intensity. In Australia, for example, tree plantings established on abandoned pastures in the northern wet tropics resulted in no consistent soil carbon accumulation (Lewis et al., 2019), whereas southern and eastern Australian plantations increased SOC stocks by 0.57 Mg C ha^{-1} yr^{-1} with more gains on abandoned croplands than pastures (England et al., 2016).

Low-intensity agriculture results in substantial SOC accumulation, although with notably fewer observations globally (Figure 10). The amount and quality of carbon input from aboveground vegetation, both trees and crops, determines the rate and longevity of belowground accumulation (Nair et al., 2009). For example, tree species diversity in agroforestry systems implemented on previously cultivated lands and grasslands in India was positively linked to increased sequestration, eventually producing near natural SOC levels (Hombegowda et al., 2016).

Bioenergy production also features some of the highest rates of sequestration, in addition to facilitating fossil fuel substitution. However, the carbon intensive production methods of many sources of biofuel can limit their net positive impact. Only a few production methods, namely grassland species grown specifically on abandoned lands, do not incur a carbon debt (Fargione et al., 2008). When mixtures of grassland species are used, soil carbon sequestration can exceed production emissions (Tilman et al., 2006) and outperform alternative management strategies such as rewilding (Vuichard et al., 2009).

Considering the positive effect of combining active and passive management techniques reported at some country and climatic zone scales, restoration initiatives may be most beneficial for soil carbon sequestration when supplementing ongoing rewilding. Slow ecological succession processes in degraded landscapes can be boosted from tree and shrub plantings (Rey Benayas et al., 2008; Zethof et al., 2019). However, the wide range of sequestration rates reported when combining restoration and rewilding around the world (0.30–1.30 Mg C ha^{-1} yr^{-1}) illustrate the critical importance of site-specific factors in determining what approach or combination of specific techniques to use. Even in cases where plantations clearly outperform natural succession in terms of aboveground carbon sequestration, rewilding is still considered an attractive option due to lower costs, higher plant biodiversity, and generally higher belowground sequestration in the 0–30 cm depth (Tremblay and Ouimet, 2013).

These proposals have direct SOC implications. They need to be designed to promote their above- and belowground accumulation potentials well beyond the first few decades (Vuichard et al., 2009). Passive management and bioenergy production are also two of the most significant climate change mitigation strategies to implement on active croplands directly (Albanito et al., 2016). Policymakers providing mitigation incentives over large rural areas with or without ALA should ensure landowners can reasonably consider one or more of these strategies based on site-specific factors.

2.5 Conclusions

Establishing new land uses on abandoned agricultural lands is becoming increasingly attractive as global demand for land, food, and energy intensifies. This presents notable opportunities for carbon sequestration co-benefits. All six categories of proposals highlighted here have lasting implications (directly or indirectly) for climate change mitigation efforts. From a soil carbon sequestration perspective across large geographic scales, a combination of passive and active management achieves the highest reported rates on abandoned lands.

Site-specific factors play a significant role in the efficacy of each proposal in each agricultural region of the world, resulting in high variability among soil carbon sequestration rates. To better quantify what past and present ALA implies for climate change mitigation, long-term experiments featuring a variety of agricultural practices and crop types abandoned in different bioclimates should be considered. This will require overcoming the challenge of gathering soil data at the decadal scale and longer. Chronosequences of ALA and paired-plots substitutes should be established extensively to supplement databases of observed sequestration rates. This space-for-time substitution is a widely accepted method for determining the environmental impacts of LUC over time (Walker et al., 2010).

The ecological and rural development implications of each management strategy and new land use highlighted here informs policymakers tasked with planning the future of rural areas experiencing ALA. Mitigating the detrimental impacts of climate change by returning carbon to depleted soils will require exploring all available avenues, including leveraging abandoned agricultural lands.

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CHAPTER III: Soil organic carbon accumulation rates on Mediterranean abandoned agricultural lands

3.1 Overview

Secondary succession on abandoned agricultural lands can produce climate change mitigation co-benefits, such as soil carbon sequestration. However, the accumulation of soil organic carbon (SOC) in Mediterranean regions has been difficult to predict and is subject to multiple environmental and land management factors. Gains, losses, and no significant changes have all been reported. Here, I compiled chronosequence data (n = 113) from published studies and new field sites to assess the response of SOC to agricultural land abandonment in peninsular Spain. I found an overall SOC concentration accumulation rate of +2.3% yr^{-1} post-abandonment. SOC dynamics are highly variable and context-dependent. Minimal change occurs on abandoned cereal croplands compared to abandoned woody croplands (+4% yr^{-1}). Accumulation is most prevalent within a Goldilocks climatic window of ~13–17 ° C and ~450–900 mm precipitation, promoting > 100% gains after three decades. Our secondary forest field sites accrued 40.8 Mg C ha^{-1} (+172%) following abandonment and displayed greater SOC and N depth heterogeneity than natural forests demonstrating the long-lasting impact of agriculture. Although changes in regional climate and crop types abandoned will impact future carbon sequestration, abandonment remains a low-cost, long-term natural climate solution best incorporated in tandem with other multipurpose sustainable land management strategies.

3.2 Introduction

Agricultural land abandonment (ALA) has been a part of Mediterranean landscape dynamics for millennia (Butzer, 2005). Increases in key drivers such as agricultural intensification, rural depopulation, and soil degradation have fuelled surges of widespread abandonment over the course of the last century (Lasanta et al., 2017; Rey Benayas et al., 2007; Ustaoglu and Collier, 2018). While ALA is often associated with negative economic and sociocultural impacts for landowners, decision-makers and rural tourists alike (Benjamin et al., 2007; Faccioni et al., 2019; Ruskule et al., 2013; van der Zanden et al., 2018), the potential for Mediterranean ex-arable lands to act as valuable carbon sinks though above- and belowground carbon accumulation during ecological succession has become a research and policy focus (Chiti et al., 2018; Novara et al., 2017; Pellis et al., 2019; Vilà-Cabrera et al., 2017).

Spain is one of the most vulnerable countries to ALA; its agricultural land area has contracted by a fifth over the last six decades (FAO, 2019). Compared to the European Union average of three percent, five percent of Spanish agricultural lands are projected to be abandoned by 2030 with over five million hectares (ha) at severe risk of abandonment (Castillo et al., 2020). Mountainous agricultural landscapes in Spain have commonly experienced over 50% abandonment during the 20th century (Lasanta et al., 2017), with some catchments reaching near total (> 95%) reductions (Arnaez et al., 2011; Cohen et al., 2011). Considering such extensive historical and projected coverage, understanding the ecosystem implications of ALA is crucial for supporting rural development planning and improving national carbon inventories (Padilla et al., 2010; Pausas and Millán, 2019).

Conventional agricultural practices are known to both deplete soil organic carbon (SOC) stocks over time and emit substantial quantities of greenhouse gases (Carlson et al., 2017; Lal, 2013). In contrast, the spontaneous recovery of SOC following the cessation of cultivation has been identified as a potential avenue for climate change mitigation through soil carbon sequestration (SCS) (Kämpf et al., 2016; Kurganova et al., 2014; Laganière et al., 2010; Wertebach et al., 2017). While the SCS potential of most agricultural soils transitioning to grasslands, shrublands, and forests has been established (Deng et al., 2014a; Guo and Gifford, 2002; Post and Kwon, 2000), its generality as a universal and guaranteed response to abandonment worldwide is unclear (Breuer et al., 2006; Hoogmoed et al., 2012). In semi-arid landscapes in particular, intensive agriculture can often degrade soils that then require much longer periods of natural or assisted restoration to overcome stalled vegetation recovery (Bonet, 2004; Garcia-Franco et al., 2014; Robledano-Aymerich et al., 2014).

Several authors have observed slow or negligible SOC accumulation post-abandonment across Spain: in the southeast under semi-arid conditions (Lesschen et al., 2008; Ruiz-Navarro et al., 2009; Segura et al., 2020; Zethof et al., 2019); in the central restrictive gypsum steppes (Eugenio et al., 2012; Martinez-Duro et al., 2010); and in the northeastern humid Pyrenees (Nadal-Romero et al., 2016; Navas et al., 2012). However, there have also been reports of original SOC concentrations increasing substantially on abandoned orchards and vineyards in both northeastern (Dunjó et al., 2003; Emran et al., 2012; Pardini et al., 2003) and southeastern Spain (De Baets et al., 2012; Rodrigo-Comino et al., 2018; Romero-Díaz et al., 2017), with similar results observed in northern (Chiti et al., 2018) and southern Italy (Badalamenti et al., 2019; Novara et al., 2013). This dichotomy in SOC response during Mediterranean post-agricultural secondary succession presents challenges for decision-makers involved in the management of regions undergoing ALA. It also demonstrates the need for increased research efforts to clarify the key factors driving positive or negative responses.

Here, I compiled chronosequence data from published and new field sites to assess the response of SOC to ALA in peninsular Spain. After first reviewing published study sites, I sampled three new chronosequence locations in an underrepresented region of northeastern Spain to investigate changes in SOC and soil N stocks at different stages of secondary succession and soil depths. I then compared these field results with published data to determine if ALA has generally led to increases or decreases in SOC concentrations across a range of Mediterranean environments. In doing so, I explored the drivers modulating the rate of SOC gain or loss over time. I hypothesize increasing SOC concentrations during secondary succession on average. This paper aims to shed new light on the debate surrounding the long-term SCS value of Mediterranean ALA.

3.3 Methods

3.3.1 Field sites

Three new chronosequences featuring six field sites of abandoned olive groves and vineyards and six control sites were selected in Barcelona Province of Catalonia, Spain (NE of the Iberian Peninsula), hereafter referred to as Font-rubí, Torrelavit, and Cànoves (Table 4). In this region, just over six percent of all agricultural land is currently classified as abandoned, with ALA and reforestation among the four most important land pressures (Baśnou et al., 2013; Funes et al., 2019). All the field sites fall within a transitional zone between *Csa* (Temperate, dry and hot summer) and *Cfb* (Temperate, no dry season, hot summer) Köppen–Geiger climates (Beck et al., 2018), and are located within the Catalan Coastal Depression between the Mediterranean

Sea and the Catalan Pre-Coastal Range. The sites share similar underlying lithologies of sediments and sedimentary rocks originating in the Neogene Period. Font-rubí and Torrelavit are both located within the Penedès wine producing region while Cànoves is located in the Vallès Oriental region just south of the Montseny Massif, a UNESCO Biosphere reserve. Their soils are characterized by high calcium carbonate contents but support a diverse assemblage of herb and shrub vegetation. Tree species dominating the forested sites include *Pinus halepensis*, *Quercus ilex*, and *Quercus cerrioides*, with an occasional presence of *Pinus pinea*, *Arbutus unedo* and *Rhamnus alaternus*. Occasional legacy growth of *Olea europaea* (European olive) and *Vitis vinifera* (common grapevine) is also present (Figure 11.c). Management of olive groves and vineyards in this region involves periodic ploughing, branch pruning and removal, and moderate manure, herbicide, and mineral fertilizer application (García et al., 2007).

Table 4. Geographical characteristics and soil classification of the chronosequence field sites.

Chronosequence	**Location**	**MAP (mm)**	**MAT (° C)**	**e.a.s.l. (m)**	**Soil Type (WRB, 2014)**	**Soil Temperature (USDA Soil Taxonomy)**	**Soil Moisture (USDA Soil Taxonomy)**
Font-rubí	41°25'35.5"N, 1°36'33.5"E	650	15.0	420–500	Calcaric Leptosols and Petric Calcisols	Thermic	Xeric
Torrelavit	41°26'07.5"N, 1°43'49.4"E	600	15.5	250–320	Calcaric Leptosols and Petric Calcisols	Thermic	Xeric
Cànoves	41°40'55.4"N, 2°18'52.3"E	696	15.1	300–370	Calcaric Cambisols and Haplic Calcisols	Thermic	Ustic

MAP: mean annual precipitation; MAT: mean annual temperature; e.a.s.l.: elevation above sea level.

3.3.1.1 Site selection and sampling

Near each location in Table 4, one natural control (late-stage forest), one pre-abandonment control presently under cultivation (cropland), and two secondary forests previously cultivated (early- and mid-stage forests) were selected to create three chronosequences of ALA (Figure 11). Our chronosequences therefore do not include stalled vegetation recovery, nor grassland or shrubland landcovers. Historical orthophotography databases (SignA, Spanish National Geographic Institute) and Landsat imagery (SatCat, Catalonia Satellite Image Server, CREAF-UAB) were compared at different time intervals to determine the approximate date of abandonment and if the subsequent secondary forest development was uninterrupted until present (Breuer et al., 2006; Gabarrón-Galeote et al., 2015b; García et al., 2007; Lesschen et al., 2008). Late-stage forests were classified based on having continuous forest coverage since at least 1956 to represent soils that were presumably never cultivated. Further age verification was conducted by consulting landowners of selected or adjoining properties and through ground truthing during field visits. In each chronosequence, at least two of the four fields were directly adjoining and all fields were within a 1.5 km radius to ensure similar environmental conditions. Sampling was undertaken in spring 2019. As all plots were < 5 ha, three sampling sites were chosen 50 m apart from each other based on similar aspect and slope in each successional stage of each chronosequence (Stolbovoy et al., 2007). To reduce any affect of field boundaries, a distance of approximately 50 m was also maintained between sampling sites and the borders of other land covers. The sampling sites featured generally flat, shallow soils approximately 30–50 cm deep. At each sampling site, the litter layer was removed and a trench was dug in 10 cm increments to a depth of 30 cm. This depth was chosen according to IPCC guidelines for the determination of SOC inventories following land use change (IPCC, 2006), however ALA can also have notable impacts deeper in the soil profile especially after several decades (Beniston et al., 2014). At each increment, 500 g of soil was sampled resulting in a total of 108 samples. Undisturbed samples were also collected at each sampling site using a 100 cm^3 steel ring for bulk density determination.

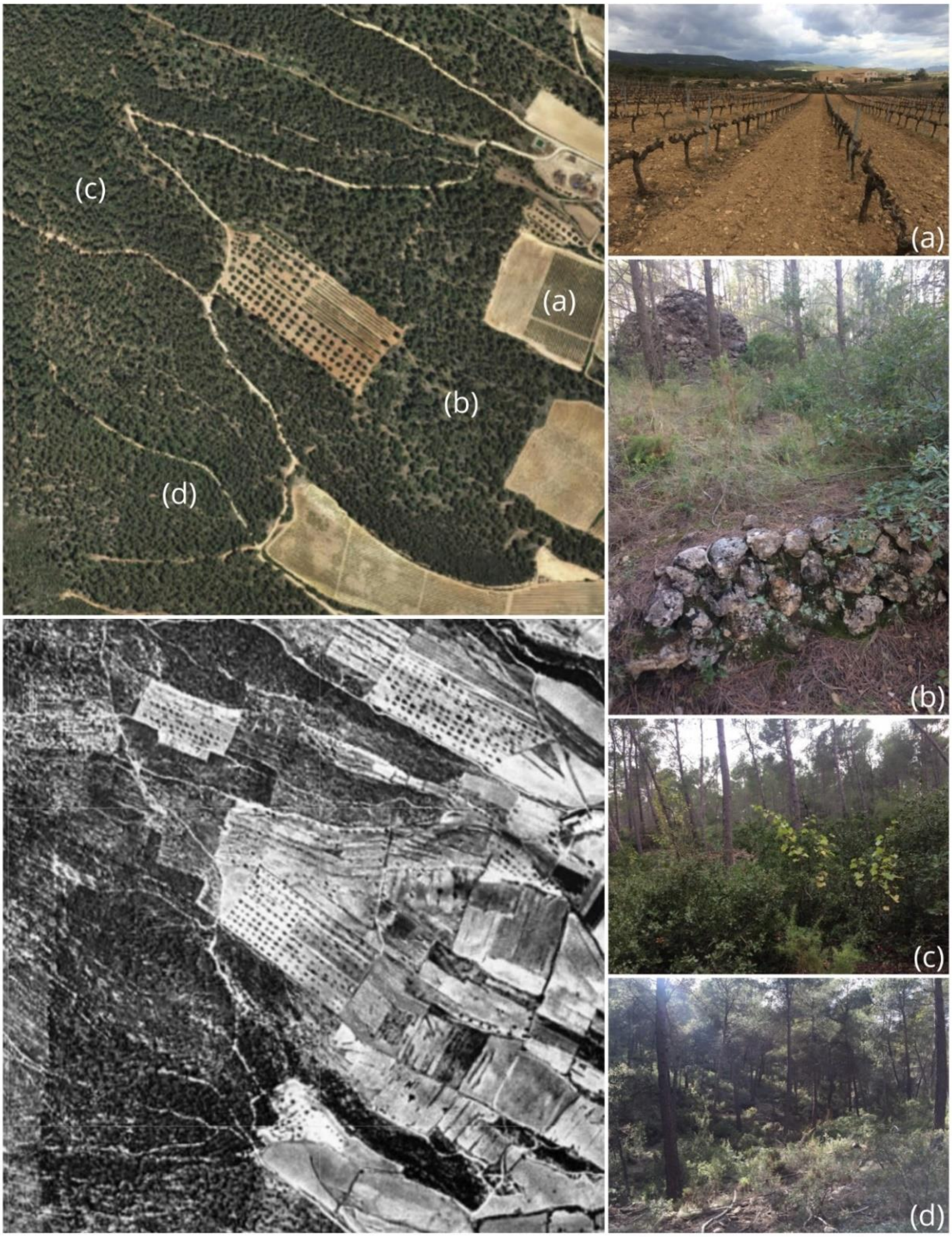

Figure 11. Orthophoto comparison of the Font-rubí chronosequence between 2019 (upper left) and 1956 (lower left): (a) active cropland; (b) early-stage forest; (c) mid-stage forest; and (d) late-stage forest. Ground evidence of past agricultural land use includes remnants of stone terrace walls and abandoned dry stone huts (b), and legacy grapevines growing amongst colonizing shrubs (c).

3.3.1.2 Sample analysis

The main set of samples were airdried for 48 hours and then sieved to procure the fine earth fraction (< 2 mm) for analysis. Soil pH and electroconductivity were determined via multimeter probe (1:2.5 deionized water) (MM 40+, Crison Instruments, Spain). All grinding and sample preparation equipment was rinsed with acetone between each sample to reduce residual C and isotopic signals. After pulverization and homogenization of the fine fraction with a mixer mill, the samples were fumigated with HCl 32% to remove inorganic carbon contents and analyzed for SOC, total nitrogen (N), and the ratio of stable isotopes ^{13}C:^{12}C ($\delta^{13}C$) and ^{15}N:^{14}N ($\delta^{15}N$) via elemental analyzer (FlashEA 1112, Thermo Fisher Scientific, USA) coupled to an isotope ratio mass spectrometer (DELTA V Advantage, Thermo Fisher Scientific, USA). To determine total carbon (TC) and soil inorganic carbon (SIC), another complete set of samples was analyzed without acid fumigation treatment. The natural abundance of stable isotopes was expressed in delta (δ) notation with values in parts per mille (‰), relative to the international Vienna Pee Dee Belemnite standard (IAEA, Austria) for ^{13}C and atmospheric N for ^{15}N (values provided in Supplementary materials Table 6). The undisturbed (bulk density) samples were oven dried at 105° C for 24 hours and weighed for fine soil and coarse rock fractions (Blake and Hartge, 1986). Bulk density values for the fine soil fraction produced from equation (1) were used to calculate SOC and N stock, based on an approximated rock fragments density of 2.6 g cm^{-3} (Don et al., 2007), using equation (2):

$$BD = \frac{M_s - M_{rf}}{V_s - \frac{M_{rf}}{\rho_{rf}}} \qquad (1)$$

$$SOCst = SOC \times BD \times D \times (1 - rff) \qquad (2)$$

where *BD* is the bulk density (g cm^{-3}), M_s is the sample mass (g), V_s is the sample volume (cm^3), M_{rf} is the sample's rock fragment mass (g), ρ_{rf} is the rock fragment density (g cm^{-3}), *SOCst* is the SOC (or N) stock (Mg ha^{-1}), *SOC* is the concentration of SOC (or N) (%), *D* is the soil depth investigated (cm), and *rff* is the gravimetric rock fragments fraction (vol. %/100).

3.3.2 Published data synthesis

Published chronosequence and paired plot studies undertaken in peninsular Spain investigating the impacts of ALA on grassland, shrubland, and forest succession were compiled for analysis following a literature search. While repeated measurements are the most ideal approach for determining the effects of land use change over time, chronosequences and paired plots are proven alternatives commonly employed (Breuer et al., 2006; Walker et al., 2010). Key terms were searched using ISI Web of Science, Scopus, and Google Scholar with results limited to English language studies conducted within peninsular Spain published since 1990. For an individual study to be included, the time since abandonment (years) and the SOC or soil organic matter (SOM) concentration (various units) within the topsoil (0–30 cm) of each chronosequence stage and paired plot must have been provided. The following necessary secondary criteria were either extracted from the studies themselves or determined through other sources: mean annual precipitation (mm), mean annual temperature (° C), past crop type (woody (e.g., *Prunus dulcis*, *Olea europaea*, *Vitis vinifera*) or annual (e.g., *Hordeum vulgare*, *Triticum aestivum*)), and sampling site coordinates. Furthermore, each study included must have additionally provided all the required data for a suitable agricultural control field (actively cultivated, representing 0 years since abandonment). Studies were excluded if they used active pastures or meadows as a control, if the past crop type was not explicitly stated, or if the data points did not span at least more than 10 years post-abandonment. A total of 24 published studies were identified under these criteria ranging from 1997–2020 (Supplementary materials Table 8). With our field sites included (n = 12), the final dataset featured 113 examples of ALA and 64 agricultural and natural control plots (Figure 12). SOC and SOM data were extracted from tables, text, or by digitizing graphs (GetData Graph Digitizer, v.2.26, Russia). When only SOM was provided (4 studies), SOC values were calculated from the Van Bemmelen conversion factor following (Guo and Gifford, 2002)). This is permissible since only the relative change in SOC over time is relevant and not the absolute SOC values. To standardize the effect of time since abandonment on SOC content between chronosequences (La Mantia et al., 2013), all SOC data points were plotted in reference to their paired agricultural control SOC values according to equation (4):

$$\Delta SOC = \frac{SOC_{ab} - SOC_{ag}}{SOC_{ag}} \times 100 \qquad (4)$$

where ΔSOC is the relative change in SOC concentration (%), SOC_{ab} is SOC concentration of the chronosequence stage investigated (%), and SOC_{ag} is the SOC concentration of the associated agricultural control (%).

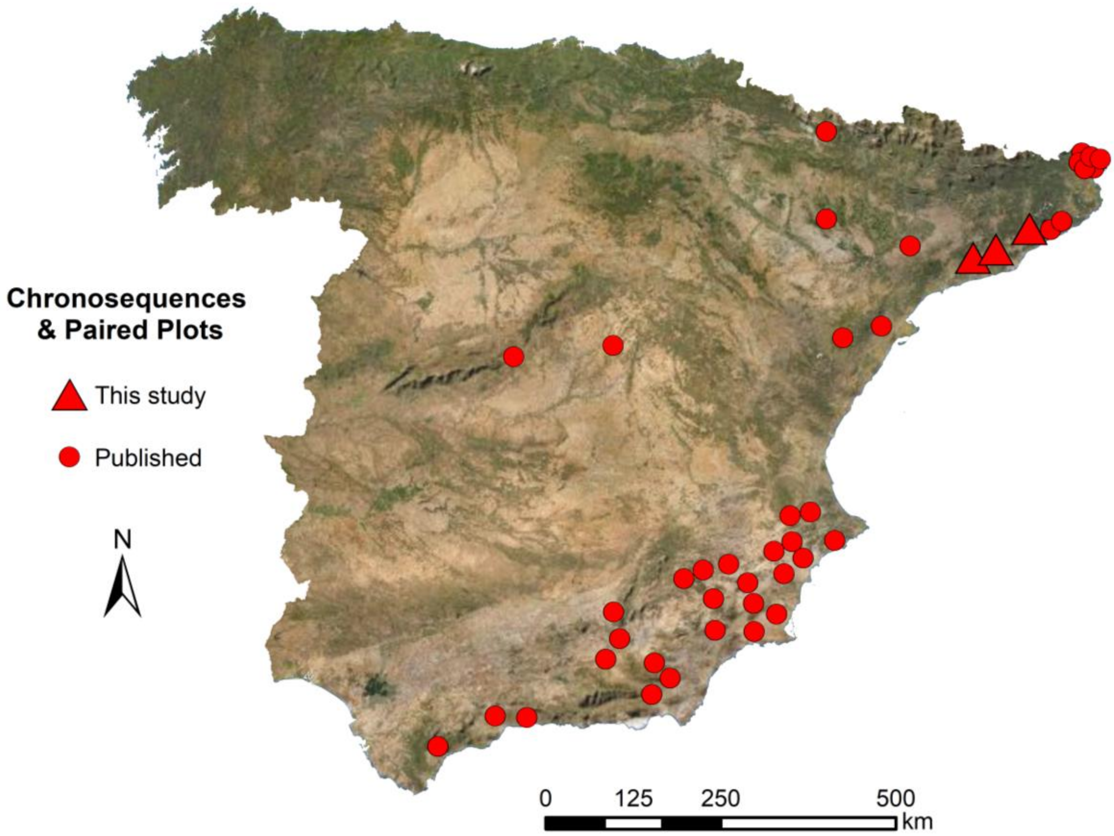

Figure 12. Locations of the published chronosequences and paired plots (circles) and our chronosequences (triangles), representing 351 individual field plots in peninsular Spain.

3.3.3 Statistical analysis

Homoscedasticity of the data was verified with Levene's test and normality with the Shapiro–Wilk test. Pearson correlation coefficients were calculated to identify relationships between the soil variables. The effect of abandonment in the entire soil profile and per depth was assessed for each of the chronosequences through one and two-way analysis of variance (ANOVA). Non-normality in Torrelavit N stock data required a non-parametric Kruskal–Wallis test. When statistical differences among means were apparent, a post-hoc multiple comparison test was performed (Tukey-HSD). Data from all Spanish chronosequences were modelled (generalized linear model with Akaike information criterion (AICc)) and fit to linear regressions to determine the influence and interactions of time since abandonment, past crop type, MAT, and MAP on SOC accumulation. Significance was assumed at $p < .05$ and all calculations and visualizations were performed using R statistical software (R Core Team, 2020) and Grapher (Golden Software, v.15, USA).

3.4 Results

3.4.1 SOC and N concentrations by successional stage and depth

Soil C concentration in the entire soil profile (0–30 cm) increased significantly from croplands to either mid-stage or late-stage forests in all three chronosequences (Table 5). The mid-stage forests, representing the most advanced stages of forest development post-abandonment in our chronosequences, contained 358% (p = .005) and 148% (p = .004) higher concentrations of SOC compared to the cropland control sites for Font-rubí and Cànoves, respectively, while late-stage forests (the natural controls) contained 386% (p =.003) more SOC for Font-rubí and 120% (p = .021) for Torrelavit. ALA across all three chronosequences resulted in an average SOC increase of 157% from croplands (1.10% SOC) to the mid-stage forests (2.82% SOC).

Soil N concentration also demonstrated an increasing trend over time following abandonment, although not in all cases. The mid-stage forest for Font-rubí contained a 157% (p = .046) higher N concentration compared to the cropland control site, while the same stage for Cànoves did not differ significantly from the cropland control but contained 79% (p = .017) more N than the early-stage forest. The stages of Torrelavit did not show any significant differences amongst each other. On average, soil N concentrations increased by 51% from croplands (0.11% N) to mid-stage forests (0.16% N).

Table 5. Mean soil chemical characteristics of the field sites (0–30 cm). Values in parentheses represent standard error (±). Letters in columns indicate significant differences between stages within each chronosequence ($p < 0.05$). Values per depth increment provided in Table 7.

Chronosequence	**Stage**	**SOC (%)**	**SIC (%)**	**TC (%)**	**N (%)**	**EC (μS cm⁻¹)**	**pH**
Font-rubí	Cropland	0.75 (0.14)a	6.57 (0.58)	7.32 (0.44)	0.07 (0.01)a	165	8.14
	Early-stage	1.83 (0.21)ab	5.13 (1.05)	6.95 (0.86)	0.12 (0.01)ab	202	7.96
	Mid-stage	3.42 (0.44)bc	6.21 (1.39)	9.63 (1.26)	0.18 (0.02)b	250	7.79
	Late-stage	3.63 (0.56)c	5.86 (1.33)	9.80 (0.53)	0.19 (0.04)b	249	7.7
Torrelavit	Cropland	1.63 (0.16)a	7.91 (0.66)	9.59 (0.70)	0.12 (0.00)	227	8.03
	Early-stage	1.67 (0.17)a	5.34 (0.86)	7.02 (0.70)	0.14 (0.01)	225	7.9
	Mid-stage	2.47 (0.25)ab	5.68 (0.36)	8.14 (0.55)	0.13 (0.04)	253	7.74
	Late-stage	3.58 (0.62)b	6.29 (0.82)	9.87 (0.99)	0.18 (0.03)	273	7.64
Cànoves	Cropland	1.04 (0.10)a	1.41 (0.31)	2.45 (0.39)	0.13 (0.01)ab	190	7.62
	Early-stage	1.36 (0.15)a	3.26 (1.22)	4.62 (1.26)	0.09 (0.01)a	157	7.72
	Mid-stage	2.58 (0.33)b	1.78 (0.64)	4.36 (0.97)	0.17 (0.02)b	187	7.65
	Late-stage	1.73 (0.20)ab	1.25 (0.26)	2.98 (0.19)	0.11 (0.00)ab	149	7.99

Within the soil profile, the rates of decrease of SOC and N concentrations with increasing sampling depth increased following ALA in all three chronosequences (Figure 13). Mid-stage forests featured the highest average decrease for both SOC and N at 0.68 and 0.05 g kg^{-1} cm^{-1}, respectively. Conversely, cropland fields showed the smallest decrease with depth for SOC and

N at 0.17 and 0.02 g kg^{-1} cm^{-1}, respectively. Differences in N concentrations among the stages of succession in the surface soil (0–10 cm) rapidly converge down the soil profile, displaying similar values in the 20–30 cm depth regardless of time since abandonment. This trend is also noticeable to a lesser degree for SOC. The magnitude of both SOC and N changes as croplands transitioned to mid-stage forests were highest at the soil surface, increasing by 188% and 99% respectively. The lowest increases occurred in the 20–30 cm depth (125% for SOC and 15% for N). Of all the soil variables among the three chronosequences, only SOC concentration in Cànoves exhibited a significant interaction effect between stage and depth (p = .024).

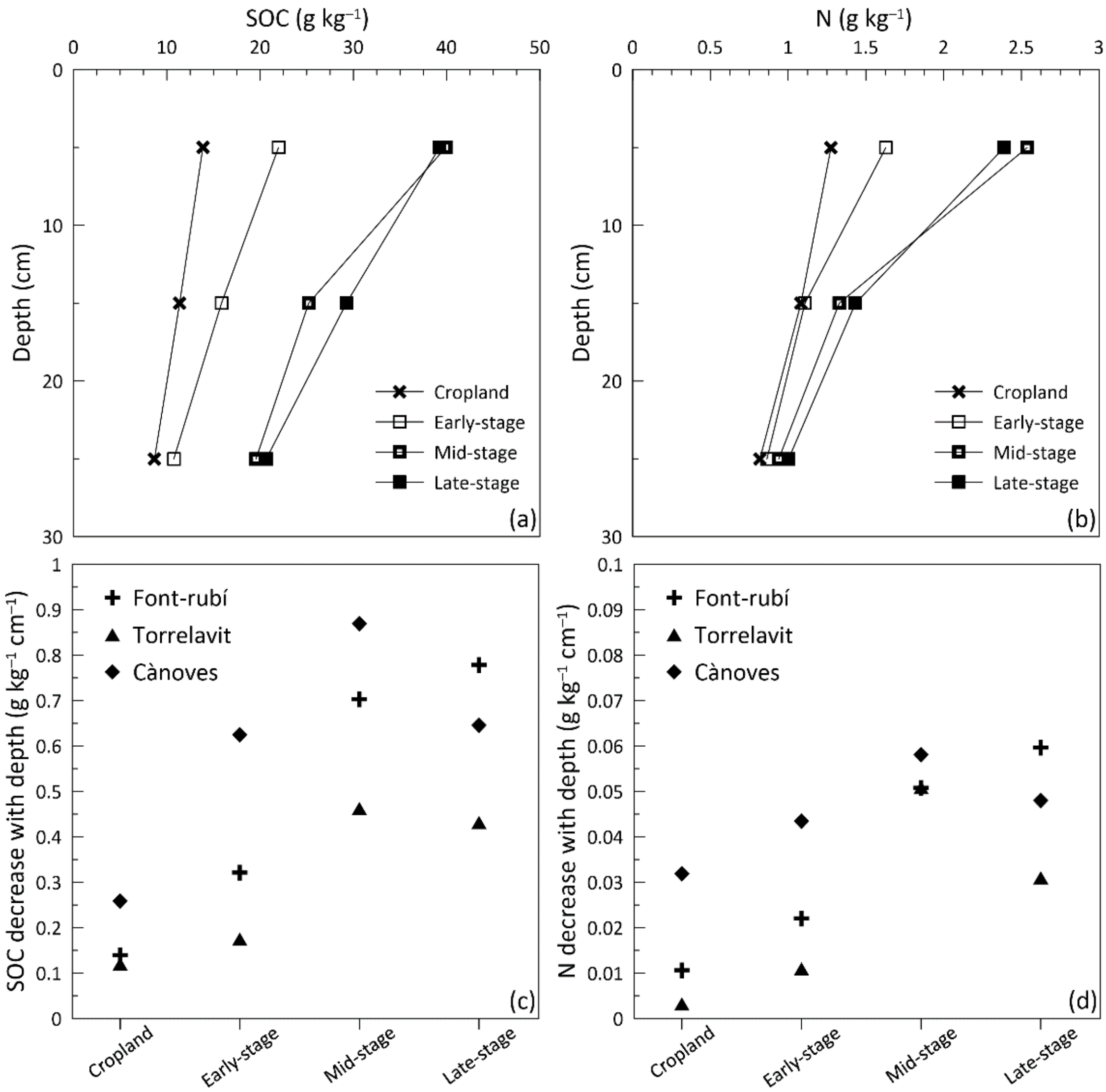

Figure 13. Effect of soil depth (0–30 cm): (a) average change in SOC (g kg^{-1}) by stage for all three chronosequences; (b) average change in N (g kg^{-1}) by stage for all three chronosequences; (c) rate of decrease of SOC (g kg^{-1} cm^{-1}) down the soil profile by stage for each chronosequence; (d) rate of decrease of N (g kg^{-1} cm^{-1}) down the soil profile by stage for each chronosequence.

3.4.2 SOC and N stocks by successional stage and depth

Similar to SOC concentration, calculated SOC stocks in the entire soil profile increased from cropland to mid-stage and late-stage forests in all three chronosequences (Figure 14). SOC stock increased from cropland (23.7 Mg ha^{-1}) to mid-stage forests by an average of 40.8 Mg ha^{-1} (+172%) as a result of ALA, with significant increases for Font-rubí (+51.6 Mg ha^{-1}, p = .039) and Cànoves (+34.9 Mg ha^{-1}, p = .013). Late-stage forests, representing the natural control, featured an average SOC and soil N stock of 64.3 Mg ha^{-1} and 3.5 Mg ha^{-1}, respectively.

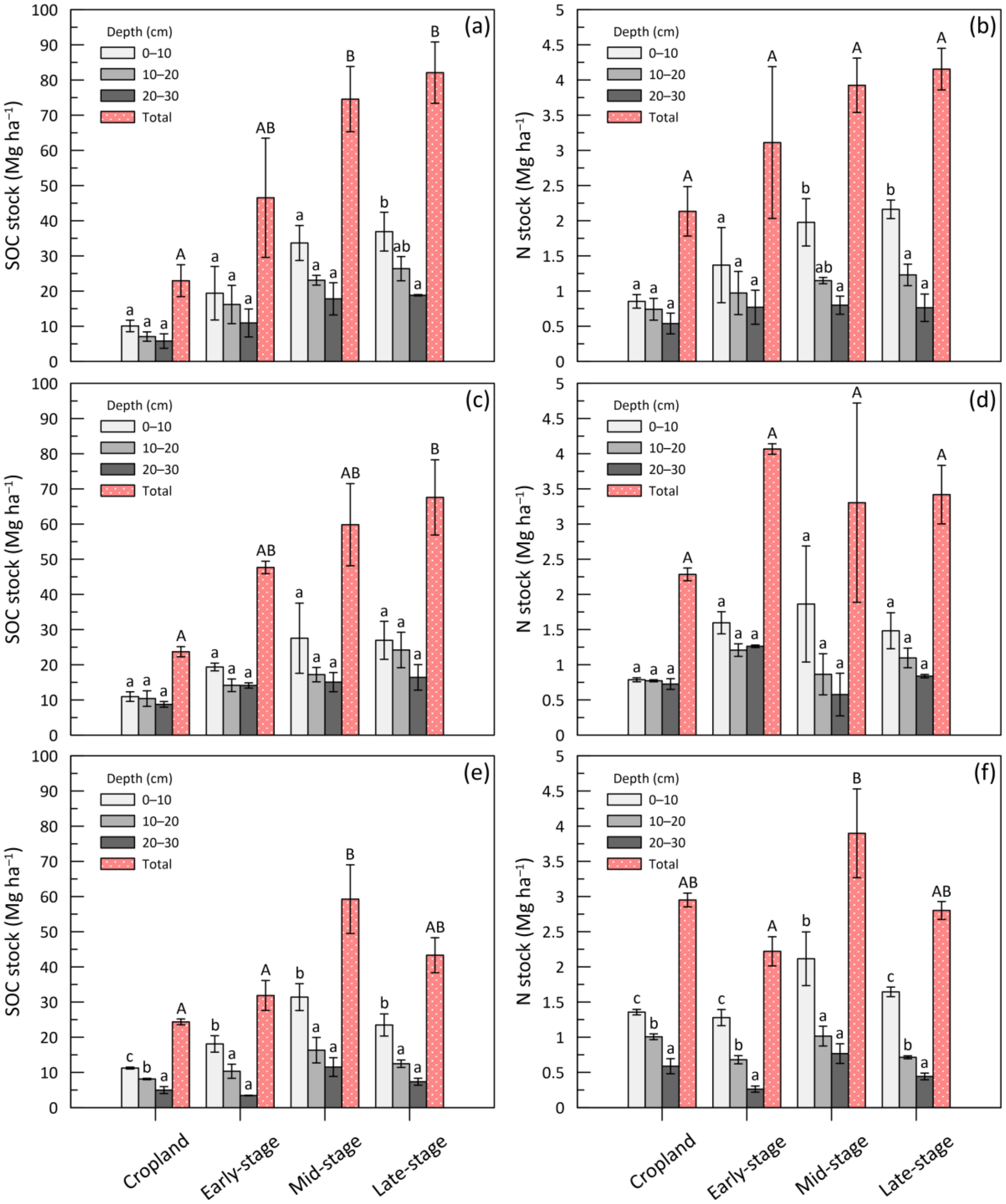

Figure 14. Change in SOC and N stock (Mg ha^{-1}) for each chronosequence: (a, b) Font-rubí; (c, d) Torrelavit; (e, f) Cànoves. Error bars represent standard error (±). Lowercase letters above bars represent significant differences between depths within each stage, while uppercase letters indicate statistical differences between stages within each chronosequence ($p < 0.05$).

Within the soil profile, SOC and N stock decreased with increasing sampling depth. Soil depth played a significant role in Font-rubí on SOC stock only in the late-stage forest, and in both the mid-stage and late-stage forests for N stock. Cànoves featured significant SOC and N differences within the soil profiles of each successional stage, in contrast to Torrelavit. The highest SOC and N stock values across all stages of all three chronosequences were observed closest to the soil surface (0–10 cm), while the lowest values were virtually all found at the lowest sampling depth (20–30 cm).

3.4.3 Soil C:N ratios by successional stage

Font-rubí and Cànoves showed an increasing soil C:N ratio from cropland to mid-stage forests following abandonment (Figure 15). Cànoves exhibited significant increases in comparison to croplands for each successional stage, with values of 14.2 (p = .001) for early-stage forests, 15.6 (p = .0004) for mid-stage forests, and 16.1 (p = .0002) for late-stage forests. Forest regrowth had no significant effect on the C:N ratio across all chronosequence stages of Torrelavit. The average C:N ratio of the three chronosequences increased from 10.5 (croplands) to 18.4 (mid-stage). Average late-stage forests had a C:N ratio of 19.6.

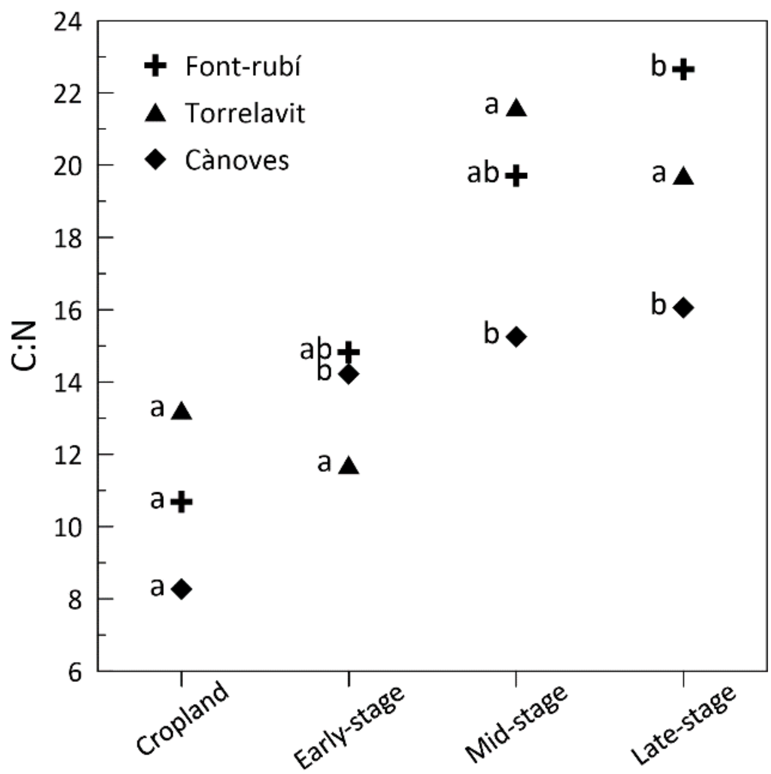

Figure 15. C:N ratio of different chronosequence stages. Letters indicate significant differences between stages within the same chronosequence (p < 0.05). Values per depth increment provided in Supplementary materials Table 7.

3.4.4 Synthesis of chronosequences in Spain

Data from published chronosequences and paired plots, incorporated with the field data presented above, indicates that ALA in Spain has had a net positive impact on promoting SCS. The average rate of SOC accumulation following ALA is +2.3% yr^{-1} (R^2 = 0.14, p < .0001) (Figure 16.a) relative to cropland control fields, requiring nearly four and a half decades of ecological succession before the first doubling of SOC concentration can be observed. The synthesis featured 113 examples of ALA (after excluding cropland and natural controls) with nearly 80% indicating a positive change. The average age across all sites is 24 years post-abandonment and the average change in SOC is positive, at +69%. However, even after several decades negative values have been reported due to various factors preventing a universally positive trend over time for all categories of sites.

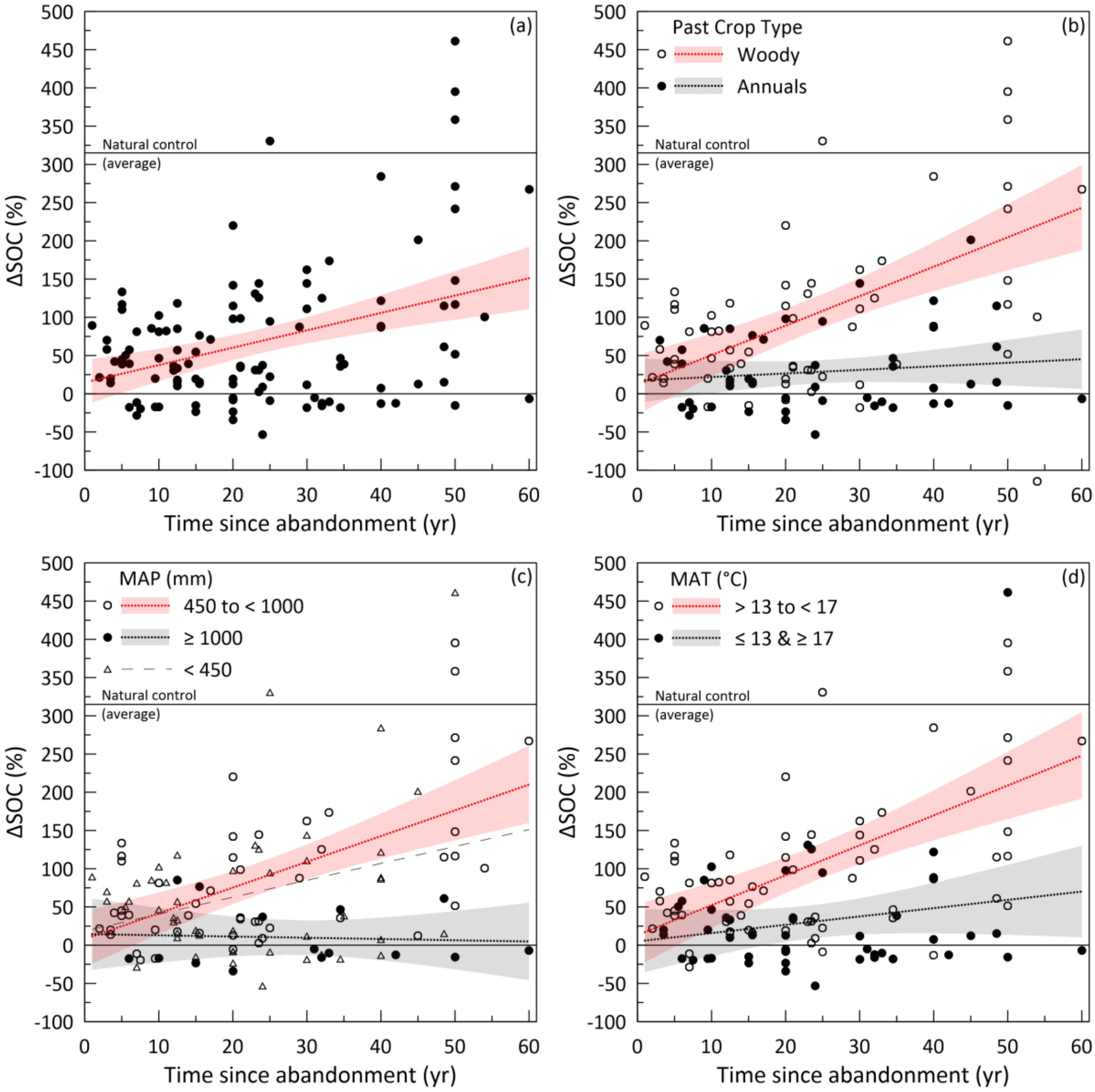

Figure 16. Relative change in SOC concentration (%) with time since abandonment (yr): (a) all Spanish chronosequence sites; (b) Past crop type with linear regressions for sites previously used for woody and annual crop production; (c) MAP (mm) with linear regressions for < 450 mm, 450 to < 1000 mm, and ≥ 1000 mm sites; (d) MAT (° C) with linear regressions for sites within and outside a temperature range of 13–17 ° C. Shaded areas represent 95% confidence interval. Horizontal line labelled "Natural control" represents average relative difference in SOC concentration between paired natural and agricultural control sites (i.e., horizontal line at 0).

Previous crop type is an important factor in SOC accumulation post-abandonment, with annual increases of 4% for abandoned perennial woody croplands (R^2 = 0.33, p < .0001, n = 61) compared to annual cereal croplands (not significant, n = 48) (Figure 16.b). Less than 10% of abandoned woody cropland sites demonstrated a negative change in SOC, compared to 40% of annual cropland sites (Supplementary materials Figure 17). After three decades post-abandonment, all woody cropland sites reported a gain in SOC. Similarly, sites below 1000 mm MAP displayed a positive trend in SOC accumulation over time, while sites at or above this threshold exhibited no relationship (Figure 16.c). After two decades post-abandonment, all observed losses of SOC were found in drier sites below 450 mm MAP (n = 48) and in more humid sites ≥ 1000 mm MAP (n = 15). Within this precipitation range (n = 50), over 90% of sites reported a positive change (reaching 100% after two decades) with an annual accumulation rate of +3% (R^2 = 0.36, p < .0001). Sites between 13 and 17 ° C (n = 64) also demonstrated a greater SOC accumulation rate of +3% yr^{-1} (R^2 = 0.27, p < .0001) compared to sites above or below this range (not significant, n = 49) (Figure 16.d). Sites with observed SOC losses were mainly outside this range (Supplementary materials Figure 17). The most important variables for predicting SOC accumulation (model-averaged importance > 0.8) were time since abandonment, past crop type, MAT and the variable interactions of past crop type with both time and MAT (adj. R^2 = 0.49, $F_{7,105}$ = 16.47, p < 0.0001). Accumulation post-abandonment was significantly correlated (bivariate) with time since abandonment (r = 0.38, p < .0001) and past crop type (r = 0.32, p < .001) but not significantly with MAT (r = 0.10) nor MAP (r = -0.13).

3.5 Discussion

3.5.1 Post-agricultural SOC and N changes with succession and depth

All our abandoned field sites exhibited higher SOC concentrations and stock values than active cropland sites, as expected and observed across diverse ecosystems (Chiti et al., 2018; Deng et al., 2016a; Gabarrón-Galeote et al., 2015a; Hu et al., 2018; Spohn et al., 2016). Our mid-stage forests displayed a SOC stock gain of 40.8 Mg ha^{-1} (+172%) compared to the cropland controls. This is higher than the 120% increases observed in the central Apennine range of Italy (+66.5 Mg ha^{-1}) (Chiti et al., 2018) and in southwestern China under notably higher precipitation (> 1500 mm) (+50.6 Mg ha^{-1}) (Yang et al., 2016). Differences in study parameters considered, such as time since abandonment, climatic conditions, and restoration methods help explain the range of SOC changes reported. With afforestation for example, SOC increases of 116±54% have been reported in the temperate zone (Poeplau et al., 2011) and 190% across arid and semiarid regions (on abandoned and barren lands) (Liu et al., 2018). Intensive practices employed in olive groves and vineyards across the Mediterranean are generally not conducive for SCS. The cessation of crop residue removal (e.g., branches taken for firewood) and periodic soil disturbances (e.g., tillage) can promote SOC accumulation (Debasish-Saha et al., 2014; Laganière et al., 2010; Romero-Díaz et al., 2017). Revegetation reduces soil temperature, water evaporation, and erosion while increasing the quantity and quality of organic matter inputs to compensate losses from decomposition (Serpa et al., 2015). Post-abandonment increases in fine root biomass also supports new SOC rhizodeposition (Novara et al., 2014), while fresh plant residues promote macro-aggregate formation and therefore SOC stabilization and accumulation (Nadal-Romero et al., 2016).

Although all three of our chronosequences showed positive SOC accumulation trends along successional trajectories, significant changes from cropland values were not found until the mid-stage forests. Similar lag times (~30–50 years) between post-agricultural revegetation and significant SOC and N increases have been identified in subtropical soils of southwestern China (Hu et al., 2018; Yang et al., 2016) and following afforestation in a global perspective (Li et al., 2012). The mechanisms that regulate SOC and N accumulation are different (McLauchlan, 2006), but N availability is considered a key factor in both the process of secondary succession and the potential for simultaneous long-term SCS in soils post-abandonment (Johannes M H Knops and Tilman, 2000; Li et al., 2012; Luo et al., 2004). The lack of any significant differences in soil N stock between our chronosequence stages and the clear accumulation of SOC throughout the successional process makes the presence of a strong N limiting effect unlikely (Hu et al., 2018; Wen et al., 2016). Immediate above- and belowground plant biomass production following abandonment may be permitted in the early stages of succession with available N from remnant agricultural fertilizers (Spohn et al., 2016). The increasing soil C:N ratio with each progressive successional stage has also been observed in other ALA chronosequences studies (Deng et al., 2013; Li et al., 2012; Spohn et al., 2016). Because both SOC and N increased from cropland to mid-stage forests, the positive correlation between C:N and SOC accumulation can be explained by higher rates of C input than N during forest regrowth (Deng et al., 2016a; Li et al., 2012).

Similar to other long-term ALA chronosequence studies (Zhao et al., 2015), SOC accumulation also decreased with each successive stage in Font-rubí and Torrelavit. This might indicate a SOC saturation effect influencing the sequestration capacity of mid- and late-stage forests (Stewart et al., 2007). Conversely, it may also be a result of a proportionately higher contribution of new SOC in early stages of succession (e.g., from new vegetation on soils previously barren), while later stages exhibit a more balanced SOC budget resulting in a lower SCS rate (i.e., steady inputs contribute proportionately less over time as SOC accumulates). Soil depth exhibited a noticeable effect on SOC and N for all stages of succession of each chronosequence. Converging soil N concentrations by the 20–30 cm depth among all stages of succession indicates minimal depth penetration over time. As reported in other studies, the highest concentrations of SOC and N in our sites were all in the 0–10 cm depth, where nutrient accumulation throughout the successional process was highest (Hu et al., 2018; La Mantia et al., 2013; Nadal-Romero et al., 2016). While cropland soils displayed the most homogenous profiles due to tillage, SOC and N decreased with depth faster in mid-stage forests than late-stage forests (natural controls presumably never tilled), indicating greater rates of surface accumulation than depth saturation in regenerating post-agricultural soils. An opposite legacy of tillage (i.e., distinct homogeneity) is also possible when excluding the effect of time since abandonment (Sulman et al., 2020). Nutrient modelling efforts over regions containing both previously tilled and never tilled revegetated soils should account for their potential differences in depth distributions (e.g., custom pedotransfer functions) (Fernández-Ugalde and Tóth, 2017).

3.5.2 Post-agricultural SOC changes in Spain

Our study reveals that ALA has an overall net positive impact on promoting SCS in peninsular Spain. The average SOC concentration accumulation rate of +2.3% y^{-1} varies depending on site-specific conditions, as shown by negative rates reported even after several decades of secondary succession. Nearly four and a half decades would be required before the first doubling of SOC with a much longer period needed before reaching pre-agricultural levels. This supports the view that SCS post-abandonment in Mediterranean environments can be a slow, long-term process (Chiti et al., 2018; Lesschen et al., 2008; Nadal-Romero et al., 2016; Segura et al., 2020). Spanish SOC stocks likely reach equilibrium well after the default 20-year transition period assumed in carbon stock calculations following land use and land cover changes (IPCC, 2006; Segura et al., 2020). In comparison, Paul et al., (2002) calculated a relative SOC accumulation rate of 1.9% y^{-1} in topsoils following active restoration (i.e., afforestation) on former croplands globally, while Shi and Han, (2014)) calculated SOC accumulation rates across China between 1.2–8.8% yr^{-1} during passive restoration (i.e., natural succession), 10.7% yr^{-1} during natural grassland succession, and 0.8–4.7% yr^{-1} during afforestation.

Mean annual precipitation at or above a threshold of approximately 1000 mm limited the overall positive effect of time since abandonment on SOC concentration. High precipitation inducing drop-offs in SCS have also been reported across Italy: Alberti et al., (2011) found SOC accumulation below 900 mm MAP and losses above, while La Mantia et al., (2013)) found SOC gains at 650 mm and losses at 1100 mm during pasture-forest transitions. At or above 1000 mm MAP in Spain, SOC losses have been reported in mountainous abandoned agricultural lands in the north (Navas et al., 2012), and no gains after two decades in the south (Gabarrón-Galeote et al., 2015b). High precipitation can result in N leaching and decreases in aggregate protected SOC alongside increases in less protected particulate organic matter fractions (Alberti et al., 2011; Guidi et al., 2014). Although long-term positive changes in SOC post-abandonment at around 1000 mm MAP have also been observed in Mediterranean environments (Chiti et al., 2018), precipitation and SOC accumulation during woody secondary succession generally correlate negatively at the global scale (Jackson et al., 2002). In the tropics, post-agricultural forests with a MAP below 1000 mm accumulate SOC faster than forests with between 1000–2500 mm MAP, while no change can be expected in sites with above 2500 mm MAP (Silver et al., 2001).

Accumulation of SOC on abandoned pastures and grazing lands transitioning to grasslands also correlates positively with temperature and negatively with precipitation (Kämpf et al., 2016; La Mantia et al., 2013; Pellis et al., 2019). The highest relative SCS rates have been in fact observed within semi-arid climates (Kämpf et al., 2016; Liu et al., 2018). However, precipitation levels closer to the lower limit of semi-arid conditions (i.e., < 450 mm MAP) can also severely limit net primary productivity and therefore SOC accumulation through reduced organic matter inputs (Figure 16.c) (Bonet, 2004; Gabarrón-Galeote et al., 2015b; Robledano-Aymerich et al., 2014). Our results indicate a Goldilocks climate window of ~450–900 mm MAP and ~13–17 ° C MAT for SCS during secondary succession post-abandonment in Mediterranean environments. Effectively all of the sites in our synthesis within this window showed net positive SOC changes, with increases of 100% and greater expected after three decades. Our results also support the view that temperature plays a more dominant role in post-abandonment SCS than previously thought (La Mantia et al., 2013; Pellis et al., 2019; Poeplau et al., 2011), with precipitation playing more of an indirect role through its influence on other factors (i.e., vegetation dynamics) (Wang et al., 2020).

Soil C stocks in Spain are also a function of past and present land management and land cover, especially at local and regional scales (Hontoria et al., 1999; Muñoz-Rojas et al., 2015).

(Willaarts et al., 2016)) found a greater effect of land classification than MAT and MAP in the south of Spain, with more SOC stock in croplands than in forests and shrublands because of the flatter and deeper soil profiles of lands typically allocated for cultivation. During secondary succession after ALA, our results indicate that the dominant historical crop type plays a significant role in the rate of SOC accumulation. The high SCS capacity of abandoned woody croplands has been observed throughout the Mediterranean region and elsewhere (Atallah et al., 2015; Badalamenti et al., 2019; Romero-Díaz et al., 2017; Spohn et al., 2016). Soils used for grain cultivation in semiarid regions of Spain may respond better to active restoration compared to natural succession (Cuesta et al., 2012). Twenty-two years after abandoning cereal fields in Andalusia, SOC gains were higher in afforested soils compared to naturally regenerating soils (Segura et al., 2020). Our analysis featured very few active restoration sites which may have limited the amount of positive values for semi-arid annual croplands (Garcia-Franco et al., 2014); although this is not certain (Ruiz-Navarro et al., 2009).

Differences in the initial stock between woody and annual croplands at the time of abandonment is likely the main reason for their different SOC accumulation rates. Vineyards and orchards suffer from reduced SOM inputs (e.g., pruned branch removal) and are typically allocated on marginal lands with lower quality soils (i.e., sloping, shallow, and stony) where they grow better than cereal crops (García et al., 2007; Jebari et al., 2018; Pardini et al., 2003). Cereal croplands may also receive SOM friendly management practices (e.g., regular manure inputs, periodic stubble grazing and seed fallowing) which may entail SOC losses in the first few years after their cessation (Navas et al., 2012; Ruecker et al., 1998). Recent estimations indicate that topsoils of annual croplands have on average 6–7 Mg ha^{-1} more SOC than soils of woody croplands in Spain (Calvo de Anta et al., 2020; Rodríguez Martín et al., 2016). (Rodríguez-Murillo, 2001)) reported even greater differences between Spanish olive/vineyard soils (40–43 Mg ha^{-1}) and soils of irrigated/dryland croplands (51–58 Mg ha^{-1}). The average SOC concentrations for the woody croplands control sites in our synthesis were 10% lower than the annual croplands. Fields with lower initial SOC are presumably farther from reaching saturation and therefore have a higher carbon sink capacity during succession (Stewart et al., 2007). Initial SOC stock and SCS potential correlate negatively during post-agricultural succession globally (Deng et al., 2016b; Kämpf et al., 2016). In the Mediterranean and the temperate zone, the sequestration potential of agricultural soils with > 50 Mg C ha^{-1} is limited (Novara et al., 2017) or non-existent (Atallah et al., 2015; Kämpf et al., 2016).

3.5.3 Managing post-agricultural SOC accumulation under a changing climate

Protecting and replenishing SOC stocks represents one quarter of the mitigation potential of all land-based climate solutions (Bossio et al., 2020). Abandoned agricultural lands can be managed in several ways that promote SCS while achieving other socioeconomic and ecological goals (CHAPTER II; García-Ruiz et al., 2020; Yang et al., 2020). However, Mediterranean precipitation and temperature regime changes will influence SOC accumulation rates during secondary succession. In the south of Spain, where climate is recognized as one of the most important drivers of ALA (Alonso-Sarría et al., 2016), drylands are expected to expand (Gao and Giorgi, 2008) and precipitation to decline (-15%) (García-Ruiz et al., 2011). Drier and hotter agricultural regions, such as in the southeast, face three future challenges: greater losses of existing SOC stock (Jebari et al., 2018), greater risk of abandonment (Castillo et al., 2020), and lower rates of SOC accumulation post-abandonment according to our results. At the same time, predicted increases in precipitation intensity will exacerbate soil-plant water stress (Rocha et al., 2020). This will impact abandoned land regeneration in addition to irrigation agriculture which will already require more water following future infrastructure modernization (Eekhout et al., 2018; Fader et al., 2016). Although abandonment typically occurs on unproductive or difficult to access plots (i.e., marginal lands) (Rey Benayas et al., 2007), a large percentage of abandoned land is highly suitable for forest growth and SOC accumulation under drought conditions relative to undisturbed soils. This is due to their relatively deep profiles with high available water capacities previously selected and favored for tillage, and the possibility of legacy fertilizers buffering nutrient deficiencies during periods of water stress (Willaarts et al., 2016). For example, secondary forests appearing in the second half of the 20th century throughout Spain have been found to have higher growth rates than pre-existing forests in drier regions (Vilà-Cabrera et al., 2017). However, higher evapotranspiration resulting from the development of new forests on abandoned lands will further reduce surface and sub-surface water resources.

The spontaneous regeneration of plant biomass and SOC following ALA implies multiple climate change mitigation co-benefits in addition to removing atmospheric CO_2 (Serpa et al., 2015). Rural development strategies that intend to leverage ALA need to consider the high variability of SOC responses and any potential risks that can offset intended benefits. As in Italy (Novara et al., 2017), much of central Spain has experienced policy-driven and financially incentivized abandonment of degraded cereal fields (Boellstorff and Benito, 2005). While the intention was to improve soil conditions as seen in other parts of Europe, it has in some cases increased erosion and therefore SOC losses, demonstrating the complexity and variability of ALA impacts on Mediterranean soils (Rodrigo-Comino et al., 2018). Widespread unmanaged forest regeneration in Mediterranean environments can also raise the risk of wildfires due to increased plant homogeneity, biomass fuel, and forest connectivity (Viedma et al., 2006). Planned or climate-induced crop conversions and land use changes, such as cereal production in drier regions converted to bioenergy crops or left to regenerate into shrublands (Serpa et al., 2015), presents additional opportunities for SCS that require further research efforts. The ecological legacy of ALA in Spain and its potential for promoting land degradation neutrality and climate change mitigation should be considered in rural development planning and policy-making (van Leeuwen et al., 2019).

3.6 Conclusions

Agricultural land abandonment has produced divergent increases in SOC concentrations in peninsular Spain. Chronosequence field studies indicate an average SOC accumulation rate of +2.3% yr^{-1} post-abandonment. It is a highly variable process, depending on multiple environmental and land management factors. The highest rates of SOC accumulation post-abandonment can be expected on lands previously used for woody crop production featuring ~13–17 ° C MAT and ~450–900 mm MAP, with the lowest rates expected on lands previously used for annual crop production outside this climatic window. Our secondary forest field sites accrued 40.8 Mg C ha^{-1} (+172%) following abandonment but displayed greater SOC and N depth heterogeneity than natural forests, demonstrating the long-lasting impact of agriculture. By altering the SOC accumulation rates of existing secondary forests and influencing the locations and crop types of future ALA, precipitation and temperature changes in the Mediterranean region will determine the SCS potential and ecological value of abandoned agricultural lands. Regional climate change mitigation policies in Mediterranean and semi-arid environments can consider ALA as a low-cost but long-term option best incorporated in tandem with other multipurpose sustainable land management strategies.

3.7 References

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3.8 Supplementary materials

Table 6. Mean soil 13C and 15N stable isotope values of the field sites (0–30 cm). Values in parentheses represent standard error (±). Letters in columns indicate significant differences between stages within each chronosequence ($p < 0.05$).

Chronosequence	Stage	$\delta^{13}C$ (‰)	$\delta^{15}N$ (‰)
Font-rubí	Cropland	-24.17 (1.39)	4.76 (0.34)
	Early-stage	-26.05 (0.12)	4.41 (0.94)
	Mid-stage	-26.49 (1.13)	2.15 (1.30)
	Late-stage	-25.50 (1.03)	2.17 (0.14)
Torrelavit	Cropland	-20.45 (2.18)	5.25 (0.35)c
	Early-stage	-25.30 (0.06)	3.48 (0.19)b
	Mid-stage	-23.01 (2.25)	1.05 (0.49)a
	Late-stage	-24.00 (0.97)	2.17 (0.05)ab
Cànoves	Cropland	-25.67 (0.03)b	9.03 (0.35)c
	Early-stage	-26.25 (0.29)ab	0.15 (0.38)a
	Mid-stage	-26.34 (0.16)ab	1.72 (0.12)ab
	Late-stage	-26.55 (0.13)a	4.71 (1.43)b

Table 7. Mean soil chemical characteristics of the field sites by depth (n = 3).

		0–10 cm					10–20 cm					20–30 cm				
Chronosequence	**Stage**	**SOC (%)**	**SIC (%)**	**TC (%)**	**N (%)**	**C:N**	**SOC (%)**	**SIC (%)**	**TC (%)**	**N (%)**	**C:N**	**SOC (%)**	**SIC (%)**	**TC (%)**	**N (%)**	**C:N**
Font-rubí	Cropland	0.98	6.41	7.39	0.08	12.17	0.70	6.58	7.28	0.07	9.62	0.56	6.73	7.29	0.05	10.29
	Early-stage	2.26	5.22	7.48	0.16	14.15	1.93	4.83	6.76	0.12	16.48	1.29	5.32	6.62	0.09	13.84
	Mid-stage	4.56	6.17	10.73	0.27	17.16	3.26	5.20	8.46	0.16	20.07	2.45	7.25	9.71	0.11	21.90
	Late-stage	4.80	6.34	11.15	0.29	16.88	3.54	5.60	9.14	0.17	21.52	2.47	6.66	9.13	0.11	33.17
Torrelavit	Cropland	1.75	8.45	10.20	0.13	14.07	1.68	7.98	9.52	0.12	18.85	1.39	6.83	9.05	0.12	18.26
	Early-stage	2.02	5.51	7.53	0.17	12.28	1.51	5.07	6.58	0.13	11.65	1.49	5.45	6.94	0.13	11.20
	Mid-stage	3.30	6.37	9.67	0.22	15.78	2.19	5.32	7.51	0.10	25.40	1.91	5.34	7.25	0.07	20.15
	Late-stage	4.14	7.19	11.33	0.23	18.06	3.74	5.83	9.57	0.18	21.60	2.85	5.85	8.70	0.14	19.49
Cànoves	Cropland	1.43	1.19	2.62	0.17	8.29	1.03	1.56	2.59	0.13	8.10	0.65	1.47	2.12	0.08	8.42
	Early-stage	2.32	3.24	5.56	0.16	14.05	1.32	3.20	4.52	0.09	14.96	0.44	3.34	3.78	0.03	13.68
	Mid-stage	4.11	3.05	7.16	0.28	15.28	2.11	1.07	3.18	0.13	15.75	1.50	1.24	2.74	0.10	14.72
	Late-stage	2.82	2.32	5.14	0.20	14.21	1.49	0.59	2.08	0.09	17.40	0.88	0.86	1.73	005	16.56

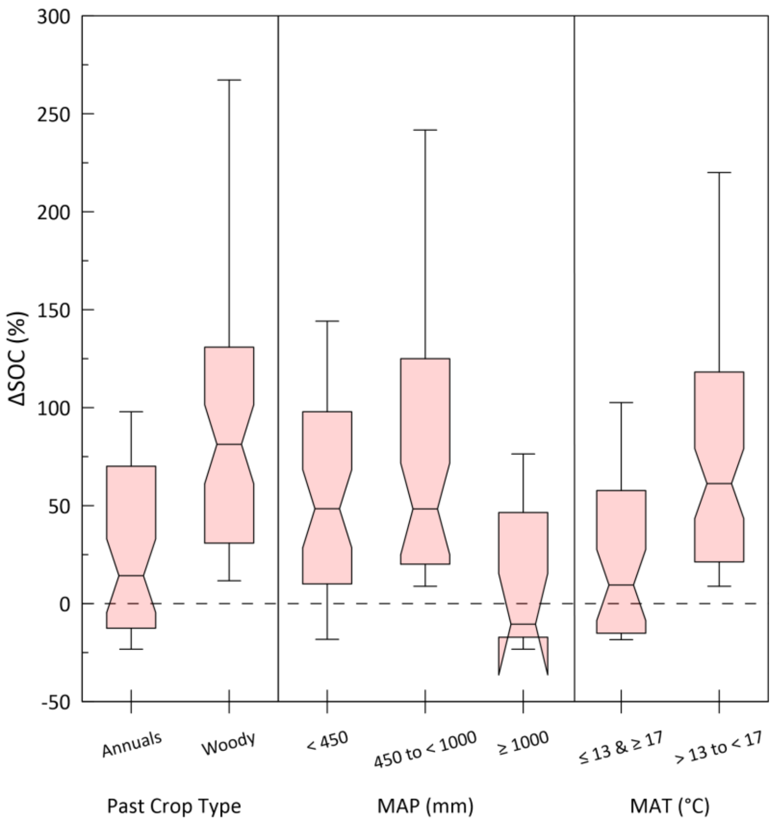

Figure 17. Notched boxplots for past crop type, MAP, and MAT variables considered in the synthesis of chronosequence data. Middle line represents median and whiskers represent upper (90%) and lower (10%) percentiles.

Table 8. List of papers included in the Spanish synthesis.

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9.	García, H., Tarrasón, D., Mayol, M., Male-Bascompte, N., Riba, M., 2007. Patterns of variability in soil properties and vegetation cover following abandonment of olive groves in Catalonia (NE Spain). Acta Oecologica 31, 316–324. https://doi.org/10.1016/J.ACTAO.2007.01.001
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	abandonment in a semi-arid ecosystem. J. Arid Environ. 72, 2082–2092. https://doi.org/10.1016/J.JARIDENV.2008.06.006
12.	Martinez-Mena, M., Lopez, J., Almagro, M., Boix-Fayos, C., Albaladejo, J., 2008. Effect of water erosion and cultivation on the soil carbon stock in a semiarid area of South-East Spain. Soil Tillage Res. 99, 119^{-1}29. https://doi.org/10.1016/J.STILL.2008.01.009
13.	Moreno, J.L., Torres, I.F., García, C., López-Mondéjar, R., Bastida, F., 2019. Land use shapes the resistance of the soil microbial community and the C cycling response to drought in a semi-arid area. Sci. Total Environ. 648, 1018^{-1}030. https://doi.org/10.1016/J.SCITOTENV.2018.08.214
14.	Navas, A., Gaspar, L., Quijano, L., López-Vicente, M., Machín, J., 2012. Patterns of soil organic carbon and nitrogen in relation to soil movement under different land uses in mountain fields (South Central Pyrenees). Catena 94, 43–52. https://doi.org/10.1016/j.catena.2011.05.012
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17.	Rodrigo-Comino, J., Martínez-Hernández, C., Iserloh, T., Cerdà, A., 2018. Contrasted Impact of Land Abandonment on Soil Erosion in Mediterranean Agriculture Fields. Pedosphere. https://doi.org/10.1016/S1002-0160(17)60441-7
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20.	Romero-Díaz, A., Ruiz-Sinoga, J.D., Robledano-Aymerich, F., Brevik, E.C., Cerdà, A., 2017. Ecosystem responses to land abandonment in Western Mediterranean Mountains. Catena. https://doi.org/10.1016/j.catena.2016.08.013
21.	Ruecker, G., Schad, P., Alcubilla, M.M., Ferrer, C., 1998. Natural regeneration of degraded soils and site changes on abandoned agricultural terraces in Mediterranean Spain. L. Degrad. Dev. 9, 179^{-1}88.
22.	Valverde-Asenjo, I., Diéguez-Antón, A., Martín-Sanz, J.P., Molina, J.A., Quintana, J.R., 2020. Soil and vegetation dynamics in a chronosequence of abandoned vineyards. Agric. Ecosyst. Environ. 301. https://doi.org/10.1016/j.agee.2020.107049

23.	Zethof, J.H.T., Cammeraat, E.L.H., Nadal-Romero, E., 2019. The enhancing effect of afforestation over secondary succession on soil quality under semiarid climate conditions. Sci. Total Environ. 652, 1090–1101. https://doi.org/10.1016/J.SCITOTENV.2018.10.235
24.	Zornoza, R., Mataix-Solera, J., Guerrero, C., Arcenegui, V., Mataix-Beneyto, J., 2009. Comparison of soil physical, chemical, and biochemical properties among native forest, maintained and abandoned almond orchards in mountainous areas of Eastern Spain. Arid L. Res. Manag. 23, 267–282. https://doi.org/10.1080/15324980903231868

CHAPTER IV: Factors driving soil carbon sequestration following agricultural land abandonment in Europe

4.1 Overview

Agricultural land abandonment (ALA) is a prominent land use change throughout Europe, with several notable implications for soil health and ecosystem restoration. In particular, the cessation of intensive agricultural practices often induces an increase in soil organic carbon (SOC) and can potentially support land-based climate change mitigation efforts. However, large uncertainties on the variability of post-abandonment soil carbon sequestration (SCS) rates and the absolute storage potentials across Europe hinders the development of dedicated policies leveraging the ecological benefits of both planned and unplanned ALA. In this chapter, I collected and synthesized SOC stock changes following ALA derived from field sites in European countries using published chronosequence/paired plot data (804 observations, 546 soil profiles). In doing so, I determined how rates of soil carbon accumulation during ecological succession differ in space and time. I found a slow, but significant, rate of SOC stock increase across Europe of 1.28% yr^{-1}, and an absolute rate of 0.32 Mg ha^{-1} yr^{-1}. The average relative and absolute changes are +32.1% and +10.5 Mg ha^{-1}, respectively, with an average time since abandonment of 34 years. SOC responses were negatively correlated with initial SOC stock, indicating a soil carbon saturation effect. Low initial stock (< 25 Mg ha^{-1} at 0–30 cm depth) exhibited a significantly higher SCS rate than all the other initial stock classes, accumulating SOC at 1.95% yr^{-1}. Abandoned agricultural lands in biogeographical regions featuring optimal climatic windows showed greater SOC sequestration rates, with mean soil carbon change ranked from highest to lowest as Pannonian > Mediterranean > Atlantic > Continental > Boreal > Alpine. However, climatic conditions and human management factors can have both positive and negative effects on SOC, resulting in several strongly divergent responses to ALA. Past croplands had a notably greater rate of SOC increase over time relatively (1.52% yr^{-1}) and absolutely (0.38 Mg ha^{-1} yr^{-1}) than sites that were previously used as pastures, likely a result of lower initial SOC stocks in croplands compared to pastures. Sites that underwent natural ecological succession exhibited a greater rate of change in SOC stock relatively (1.59% yr^{-1}) and absolutely (0.35 Mg ha^{-1} yr^{-1}) compared to sites that were actively restored or converted to new vegetation land covers, for example through tree planting practices. These results provide some clarity on previous regional debates surrounding the positive, negative, and neutral SCS potential of post-agricultural soils, which have likely been confounded by the factors investigated here. Abandoned croplands with low initial SOC stock and managed through natural succession can be expected to show the greatest SOC accrual in Europe, while fertile pastures that are actively converted (e.g., afforested) would result in the lowest increases in SOC, or even losses. This variability in post-abandonment/conversion SOC dynamics must be considered in sustainable land use planning that strives to incorporate the positive ecological and climate change mitigation implications of ALA, taking into account site-specific conditions and past and present land management regimes to avoid negative impacts for soil health and lost opportunities for climate change mitigation.

4.2 Introduction

Agricultural land abandonment (ALA) is a prominent global land use change. But despite its ubiquity across agricultural regions (Campbell et al., 2008; Li and Li, 2017), it is difficult to accurately measure and monitor at large geographic scales, which can lead to large uncertainties (Yin et al., 2020). This is due to both its multiple definitions as a land use change and its rather ephemeral nature as a land use classification, often undergoing recultivation after short periods (e.g., overlapping with shifting agriculture or unreported and informal fallowing practices) (Benjamin et al., 2007; Heinimann et al., 2017). The lack of incentives and interest to report genuine ALA also produces inaccurate land use inventories and low-resolution mapping in under-resourced regions. In the European Union however, efforts to monitor, measure, and map ALA have achieved a comparatively higher level of success due to the incorporation of multiple sources of predictive variables and model parameters (i.e., the LUISA modelling platform, see Lavalle et al., (2020)). These efforts suggest that more than 5.6 Mha is predicted to be abandoned in the EU and the UK by 2030, or 3.6% of total agricultural land (Perpiña Castillo et al., 2021). Indeed, most of the +1.4% total forest area increase across the continent from 1992 to 2015 has occurred on former agricultural lands (Palmero-Iniesta et al., 2021).

In non-degraded agricultural landscapes, the cessation of intensive agricultural practices typically results in the spontaneous recovery of ecosystem properties towards pre-agricultural levels (Cramer et al., 2008). Ecosystem health indicators for vegetation, soil, and wildlife can all improve from the ensuing ecological succession following ALA. These trajectories depend significantly on site-specific conditions (e.g., the level of degradation, the suitability for restoration, and the level of biodiversity one would find there naturally compared to the level of biodiversity maintained by the active agroecosystem before ALA) (Beilin et al., 2014; Plieninger et al., 2014; Queiroz et al., 2014). Although conventional agricultural practices are known to continuously deplete soil carbon (Carlson et al., 2017; Lal, 2013), one of the most important benefits of the natural recovery of post-agricultural soils is the regeneration of soil carbon stocks (Deng et al., 2014; Laganière et al., 2010; Wertebach et al., 2017).

As the largest terrestrial carbon pool that can be effectively influenced by human efforts, soil organic carbon (SOC) represents a critical carbon sink for climate change mitigation efforts (Lal, 2004). The natural ability of post-agricultural soils to reabsorb carbon until, presumably, reaching pre-agricultural levels has therefore received increasing attention, especially for its implications for sustainable land management (CHAPTER II; Lasanta et al., 2015; Schröder et al., 2018). Unfortunately, there remains several uncertainties surrounding ALA's potential as a climate change mitigation tool via soil carbon sequestration (SCS) at a regional, continental, and global scale, despite notable instances in history of large-scale sequestration (e.g., following the collapse of the former Soviet Union (Kuemmerle et al., 2011; Schierhorn et al., 2013; Wertebach et al., 2017), or even following the mass die-off of pre-colonial South America (Koch et al., 2019)). Not all landscapes accumulate carbon at the same intensity (i.e., amount of stock increase) and speeds (i.e., rate of stock increase) following ALA (Breuer et al., 2006; Hoogmoed et al., 2012; Nadal-Romero et al., 2016), and under specific conditions some can even lose soil carbon (Martinez-Duro et al., 2010; Segura et al., 2020).

In general, new policies on sustainable land management are expected to include stipulations for protecting and replenishing SOC stocks whenever possible (Amelung et al., 2020; Bossio et al., 2020; Bradford et al., 2019). The ability to calculate the rates and amounts of post-agricultural SCS across large geographies is pivotal for the planning, implementation, and assessment of land management policies that incorporate ecosystem restoration aspects (Xie et al., 2020). This is made even more necessary by the fact that ALA is a continuous (i.e., both historically and currently relevant) and often unplanned land use change (LUC) that is already influencing soil carbon stocks, for better or worse, and needs to be more accurately quantified. Therefore, the combination of robust datasets that produce region- and management-specific rates of SCS with accurate and detailed maps of ALA in that region (i.e., the spatial body on which to apply the rates) creates the possibility to support effective and climate-smart land management policies (Cook-Patton et al., 2020; Vermeulen et al., 2019).

Europe, where 11% of total greenhouse gas emissions stem from agriculture (EU NIR, 2021), represents an ideal combination of data resources and land use histories for an integrated, large-scale study on the SCS potential and implications of ALA. This is due to the widespread historical and ongoing ALA (Estel et al., 2015; Lasanta et al., 2017; Levers et al., 2018; Ustaoglu and Collier, 2018), the recent political and scientific push for effective, efficient, and accessible SCS strategies (Gardi et al., 2016; Montanarella and Panagos, 2021; Navarro and Pereira, 2012; Rodrigues et al., 2021; Schröder et al., 2018); the availability of published studies on soil properties and ALA (i.e., chronosequence and paired-plot data); and the detailed, robust, and up-to-date land use/cover inventory keeping that has produced high-quality spatial projections of ALA (Lavalle et al., 2020; Perpiña Castillo et al., 2021). The continental coverage of published chronosequence and paired-plot data, in particular, allows for the quantification of total soil carbon stock changes following ALA, the determination of clusterspecific SCS rates, and the elucidation of the modulating factors on post-agricultural SCS. Despite the increasing focus on this topic in recent years, there remains much uncertainty on the direction of soil carbon response to ALA (i.e., increase, decrease, or no change), the intensity (i.e., how much change), and the duration (i.e., how long will the change last) in the various biogeographical regions of the EU and its neighbouring countries, likely due to the complex interactions of the modulating factors and their confounding effects. Afterall, the EU's Thematic Strategy for Soil Protection has listed SOC protection and enhancement as a necessary goal for member states, and SOC loss as one of the eight soil threats on the continent (EC, 2012).

In light of these challenges and opportunities for both research advancement and policy support, here I synthesized published chronosequence and paired-plot data from field sites within Europe and explored the variability in SOC responses to ALA and direct conversion from agriculture to re-naturalized landscapes. I conducted a literature search to combine all previously synthesized data at different geographical scales with never-before synthesized published studies and I categorized each chronosequence/paired-plot collected based on several key factors that may influence SOC stock dynamics (i.e., climate, biogeographical region, past land use, past crop type, and present land management). I expect a noticeable increase in SOC across Europe following abandonment/conversion, but with high variability in sequestration rates. These results are intended to provide important context on the soil carbon implications of land use change in Europe, particularly from an ecosystem restoration perspective.

4.3 Methods

4.3.1 Literature search and data collection

Published chronosequence and paired-plot studies undertaken in Europe investigating the impacts of ALA (or direct land use conversions from agriculture) on grassland, shrubland, and forest succession were compiled for analysis following a literature search. While repeated measurements are the most ideal approach for determining the effects of land use change over time, chronosequences and paired-plots are proven alternatives commonly employed (Breuer et al., 2006; Walker et al., 2010).

The literature search and data collection process comprised of two stages. In stage one, an initial dataset of relevant studies was established by identifying any individual studies that included European sampling sites from the databases and lists of references of previously published synthesis studies on thematically related topics (i.e., land use changes and soil properties) at any geographic scale (i.e., regional syntheses within Europe, syntheses of Europe, and global syntheses including Europe) (Table 9). This allowed me to collate all previously synthesized, time-stamped, post-agricultural SOC data in Europe into one combined dataset.

Table 9. Results of the first stage of data collection process. Previously published synthesis studies thematically related to the present study (i.e., land use changes and soil properties) at any geographic scale (i.e., regional syntheses within Europe, syntheses of Europe, and global syntheses including Europe) were surveyed for relevant individual studies. Relevant synthesis studies that were found to contain zero relevant individual studies are not included (e.g., Guo and Gifford, (2002)).

Synthesis study	**Spatial extent**	**Number of relevant individual studies identified within**	**Final number of relevant individual studies retained after exclusion criteria**	**Final number of data-pairs extracted from remaining individual**
Li et al., (2018)	Global	85	4	66
Deng et al., (2016)	Global	49	3	21
Bárcena et al., (2014)	Europe (Northern)	2	1	21
Li et al., (2012)	Global	49	1	9
Post and Kwon, (2000)	Global	7	0	0
Laganière et al., (2010)	Global	6	0	0
Paul et al., (2002)	Global	13	0	0
Conant et al., (2001)	Global	3	0	0
Shi et al., (2013)	Global	5	0	0
Kämpf et al., (2016)	Temperate	12	0	0
Poeplau et al., (2011)	Temperate	11	0	0
TOTAL		242	9	117

The second stage of the literature search targeted all new and/or previously un-synthesized individual studies with relevant data. The following key terms were searched in November 2020 using ISI Web of Science with results limited to English language studies published in any year: (plough* OR till* OR crop* OR farm* OR agri* OR cultivat* OR *field OR pasture OR meadow OR grazing OR range*) AND (*forest* OR grassland OR shrubland OR natural

OR secondary OR recover* OR plantation* OR conver* OR abandon* OR old* OR regenerat* OR *aside OR restor* OR succession* OR fallow OR revegetat*) AND (chronosequence OR pair*) AND (soil OR carbon). This initial search resulted in 4718 hits, which were then sorted by geographic region, producing a subset of studies that had related terms either in their title, abstract, or keywords (i.e., "Europe" related terms, European country names, and geographic feature names that indicated potential sampling sites in Europe (e.g., Alps, Nordic, Mediterranean)). Another ISI Web of Science search was conducted in early 2022, following the same procedure, to update the dataset with relevant studies published since the original search date in 2020. In addition, relevant studies discovered through snowballing reference lists and "recently cited by" lists were included to further supplement the dataset.

The final list of relevant papers from these two stages of the literature search were then subjected to inclusion/exclusion criteria before data extraction. For an individual study to be included, the time since abandonment/conversion (years) and the SOC or soil organic matter (SOM) concentration or stock (various units) of the mineral soil at any depth for chronosequence stage or paired-plot must have been provided. Each chronosequence and paired-plot must have featured one agricultural control field (i.e., actively cultivated, representing 0 years since abandonment/conversion) to compare the treatment field(s) to (i.e., abandoned or converted from agriculture). The following secondary criteria for both the control and treatment fields were either extracted from the studies themselves, provided by authors upon request, or determined through either online sources or inferred empirically during data processing (see below): sample bulk density (BD), sampling site coordinates (latitude, longitude), mean annual precipitation (mm), mean annual temperature (° C), past land use (cropland, pasture), past crop type (woody, annual), post-ALA/conversion land management system (natural, assisted, occasionally grazed), vegetation type restored (forest, shrubland, grassland), and biogeographical region as per the European Environment Agency classification system (EEA, 2016), and the soil sample depth (upper and lower), sample size (number of samples), and error estimates for SOC/SOM/bulk density values reported (standard deviation or standard error). Studies were excluded if they were in locations outside of the biogeographical region coverage of Europe, or if they failed to provide a means to determine any of the previously outlined criteria.

Soil carbon concentration and stock data in either SOC or SOM were extracted from tables, text, supplementary files, graphs/figures by digitizing (GetData Graph Digitizer, v.2.26, Russia), or by request to the corresponding authors. To ensure both accuracy and comparability of the data, all values were collected directly from the original published source or author, and never from secondary datasets contained in the synthesis studies listed in Table 9.

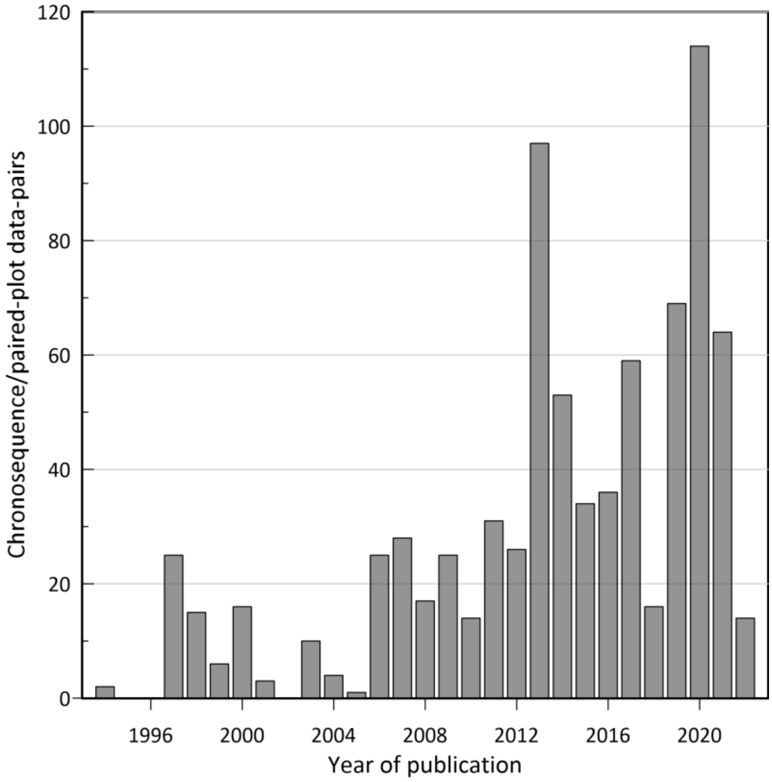

Figure 18. Distribution of chronosequence/paired-plot data-pairs in the dataset according to the year of publication of the original studies.

The final dataset from both stages of the literature and data collection process featured studies from most of the EU27 member states, in addition to the United Kingdom (UK), Switzerland (CH) and Norway (NO). A total of 102 studies published from 1994 to 2022 were identified under the inclusion/exclusion criteria (Figure 18, Supplementary materials Table 12), representing 804 time-stamped data-pairs of control and abandoned/converted soils sampled throughout the EU27+UK+CH+NO (Figure 19) at any depth. The first stage of data collection resulted in 117 data-pairs (Table 9), while the second stage resulted in 687. The dataset ranges in time since abandonment/conversion from 1 to 193 years.

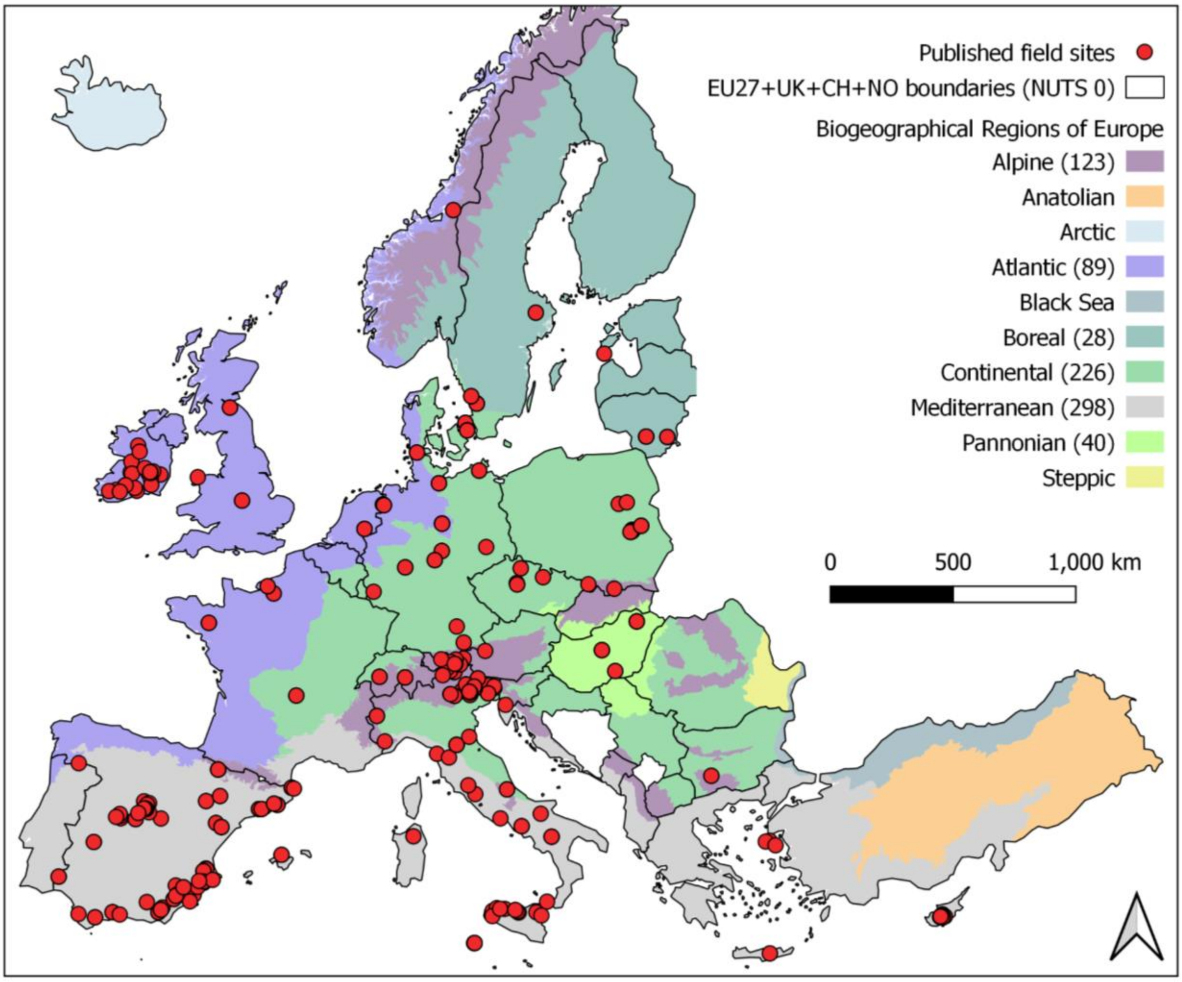

Figure 19. Map of the sampling sites of the 804 data-pairs from the 102 published studies located within the extent of the Biogeographical Regions of Europe (EEA, 2016).

4.3.2 Data processing and analysis

Within the full dataset, there were 438 data-pairs with SOC stock already reported and collected (54% of the dataset). To calculate the SOC stock of the remaining 366 data-pairs, several steps were needed. When only SOM data was provided (n = 89), SOC concentration values were calculated from the revised Van Bemmelen conversion factor (0.5 instead of 0.58) following Cook-Patton et al., (2020), based on Pribyl, (2010) using Eq. (1):

$$SOC = SOM \times 0.5 \quad (1)$$

All SOC concentration values for the 366 data-pairs were then standardized to the same concentration unit (%). To calculate the SOC stock from these values, BD is necessary and was already reported for 134 of the remaining data-pairs. When BD was not reported (n = 232), it was estimated based on the available SOC data according to the pedotransfer function of Manrique and Jones, (1991) shown in Eq. (2):

$$BD_e = 1.660 - 0.318(SOC_C)^{1/2} \tag{2}$$

where BD_e is the estimated bulk density (g cm^{-3}) and SOC_c is the reported SOC concentration (%). The reported and estimated BD values from Eq. (2) where then used to calculate SOC stocks in Mg ha^{-1} at the respective sampling depth for each of the 366 data-pairs according to Eq. (3):

$$SOC_{st} = SOC_c \times BD_{m/e} \times D \tag{3}$$

where SOC_{st} is the SOC stock (Mg ha^{-1}), SOC_c is the SOC concentration (%), $BD_{m/e}$ is the measured or estimated bulk density (g cm^{-3}), and D is the soil depth sampled (cm) based on the upper and lower depth of the sample. Because not all of the studies reported BD for each of the soil depths sampled, it was not possible to correct the entire dataset to a common soil mass, although this limitation is generally accepted in large-scale syntheses as it is not expected to result in a significant bias in SOC stock estimates following land use change (Deng et al., 2016; Guo and Gifford, 2002; Laganière et al., 2010).

The full dataset (n = 804) of reported and calculated SOC stocks at all depths was then standardized to a depth of 30 cm to increase the comparability of the study sites. Each soil profile was combined to produce a total stock per profile at a known maximum depth from the surface, reducing the dataset to 546 data-pairs. The SOC stock in the top-30 cm of each profile was then estimated following the methodology of Deng et al., (2016), using the SOC depth distribution function developed by Jobbágy and Jackson, (2000), according to Eq. (4) and (5):

$$Y = 1 - \beta^d \tag{4}$$

$$X_{30} = \frac{1 - \beta^{30}}{1 - \beta^{d0}} \times X_{d0} \tag{5}$$

where Y is the cumulative proportion of the SOC stock from the soil surface to depth d (cm); β is the relative rate of decrease in the SOC stock with soil depth; X_{30} denotes the SOC stock in the upper 30 cm; $d0$ denotes the original soil depth from the single or combined soil profile (cm); and X_{d0} is the original SOC stock from the single or combine soil profile (Mg ha^{-1}). The lack of significant differences detected between SOC depth distribution functions analysed in global biomes in Jobbágy and Jackson, (2000), and the large biogeographic scale of the present study, allows for the adoption of the global average SOC depth distribution β value (0.9786), which is commonly employed in studies with similar approaches, data, and large spatial scales (Deng et al., 2016; Li et al., 2012). Although this estimation introduces uncertainties in the data accuracy (e.g., the fact that control and treatment sites undoubtably have different SOC depth distributions in reality, as well as the treatment sites at different successional stages (see, for example, CHAPTER II, Figure 13)), it is not expected to skew overall trends of SOC dynamics during revegetation when analysing this kind of data at large geographic scales for the determination of generalized land use change effects (Li et al., 2012; Yang et al., 2011).

To standardize the effect of time since abandonment/conversion on SOC stocks between the various chronosequences and paired-plots of all the studies at the 30 cm depth, all profiles were plotted as the absolute and relative difference from their paired agricultural control SOC values, according to Eq. (5) and (6):

$$\Delta SOC_{st\_abs} = SOC_{st\_post} - SOC_{st\_pre} \tag{5}$$

$$\Delta SOC_{st\_rel} = \frac{SOC_{st\_post} - SOC_{st\_pre}}{SOC_{st\_pre}} \times 100 \tag{6}$$

where ΔSOC_{st_rel} is the relative change in SOC stock (%), ΔSOC_{st_abs} is the absolute change in SOC stock (Mg ha^{-1}), SOC_{st_post} is the SOC stock after abandonment/conversion (Mg ha^{-1}) (i.e., the treatment), and SOC_{st_pre} is the SOC stock before abandonment/conversion (Mg ha^{-1}) (i.e., the control). The relative change in SOC stock data were fit to linear regressions with 95% confidence intervals to determine the general directional responses of SOC to time since abandonment considering various climatic factors, biogeographical regions, past land uses, past crop types, and management factors (assuming significance at $p < .05$ using Grapher (Golden Software, v.15, USA)). By dividing the ΔSOC_{st_abs} or ΔSOC_{st_rel} by the age (time since abandonment), the absolute and relative SOC sequestration rates can be determined for each soil profile at its respective point in time, while the slope of the linear regressions provide more generalized SOC sequestration/loss rates for the entire category of data modelled. The overall changes in SOC stock (sequestration or loss) for each of the main categories within each overall factor, irrespective of time, were summarized for simplified comparison in forest plots with 95% confidence intervals, based on Eq. (7) and (8):

$$SE_{total} = \sqrt{\frac{V_s}{N}} \tag{7}$$

$$95\%\ CI = 1.96 \times SE_{total} \tag{8}$$

Where SE_{total} is the standard error of the change in SOC stocks per category, V_s is the variance, and N is the number of samples. Due to a Spearman rank correlation test revealing a weak significant correlation between sample size and effect size (*rho* = –0.08, *p* = 0.04) in the full profile dataset (n = 804) using R statistical software (R Core Team, 2020), a funnel plot was used to graphically inspect the possibility of skewed data (Figure 20).

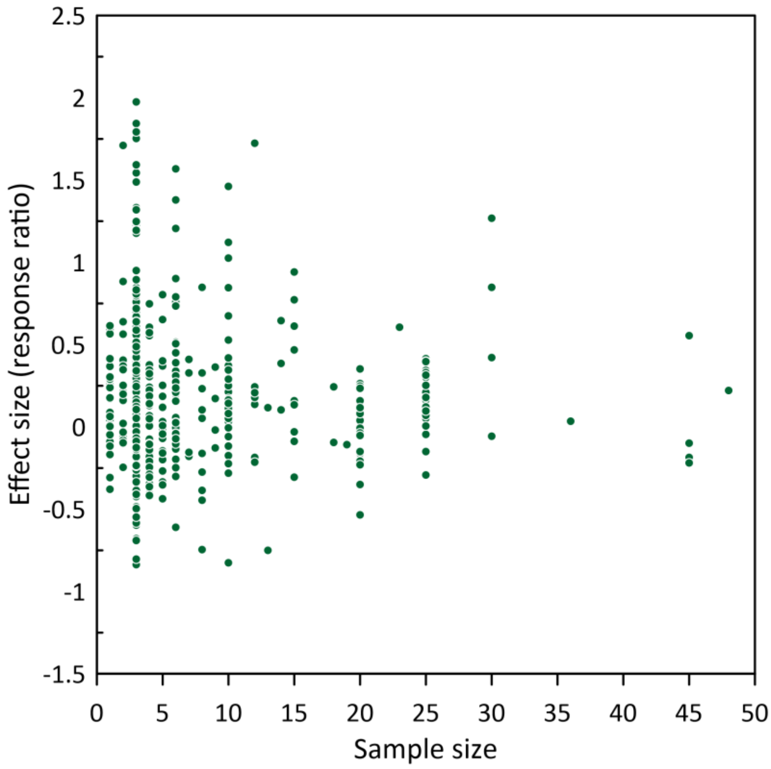

Figure 20. Funnel plot of the full dataset (n = 804)

The presence of publication bias is considered high if the funnel shape is irregular; for example, with the narrow end of the funnel closer to the y-axis than the wide end of the funnel. Figure 20 indicates low likelihood of publication bias, with a regular funnel distribution as seen in other large SOC datasets built from 50+ individual studies (see, for example, Figure 1 in Kämpf et al., (2016)). The impact of each predictor variable in the dataset on SCS, and all their potential pairwise interactions, was explored through the R package 'gmulti' (Calcagno and Mazancourt, 2010), whereby the best fitting generalized linear model was determined (candidate screening parameters set to "exhaustive", and ranking criteria set to Akaike information criterion (AICc)). The best fitting model's performance was not improved when run as a generalized additive model (Gaussian), and was therefore assumed to be linear. As analysis of variance (ANOVA) requirements for data distribution normality were not met

(Shapiro–Wilk test), non-parametric Kruskal–Wallis rank sum tests were also performed in R statistical software to further test for significant differences of the factor effects on soil carbon changes (significance at $p < 0.05$) (R Core Team, 2020).

4.4 Results & Discussion

4.4.1 Overall factors driving soil carbon sequestration following ALA

The average absolute SOC change amongst the 546 data-pairs is 10.5±2.48 Mg ha^{-1}, with an average time since abandonment of 32 years. Each of the groups of factors contained significant differences between at least one pair of categories (Supplementary materials Table 10). The overall SOC stock trends for each of the dominant non-human related factors (i.e., site-specific) and human-related factors (i.e., management) are illustrated in Figure 21 and Figure 22, respectively. The data-pairs were classified based on their time since abandonment/conversion as age classes being either young (≤ 10 years), early-stage (> 10 to ≤ 20 years), middle-stage (> 20 to ≤ 40 years), or late-stage (> 40 years) succession. All stages have positive mean SCS values, and the overall trend was as expected, with young sites having the lowest increase (3.07±2.66 Mg ha^{-1}, n = 105), followed by early-stage (4.71±4.56 Mg ha^{-1}, n = 108), middle-stage (9.66±4.53 Mg ha^{-1}, n = 153) and late-stage (19.07±5.41 Mg ha^{-1}, n = 180) sites. In many cases, SOC can be lost in the first ~5–10 years following abandonment/conversion from agriculture as vegetation regrowth begins and SOC remains in flux, followed by a return to pre-abandonment/conversion levels and then a net increase in the following decades (Deng et al., 2014). In cases of afforestation of abandoned agricultural lands, Paul et al., (2002) identified a 3–35 year period of initial decrease, after which net SCS can be expected from ~30 years. An initial period of flux can be expected as both SOM stabilizing (e.g., manure and organic amendment application) and SOM disrupting (e.g., ploughing, crop residue removal) practices cease. This implies the importance of time since abandonment generally, however the unclassified and independent effect of time is less clear (e.g., Supplementary materials Figure 28).

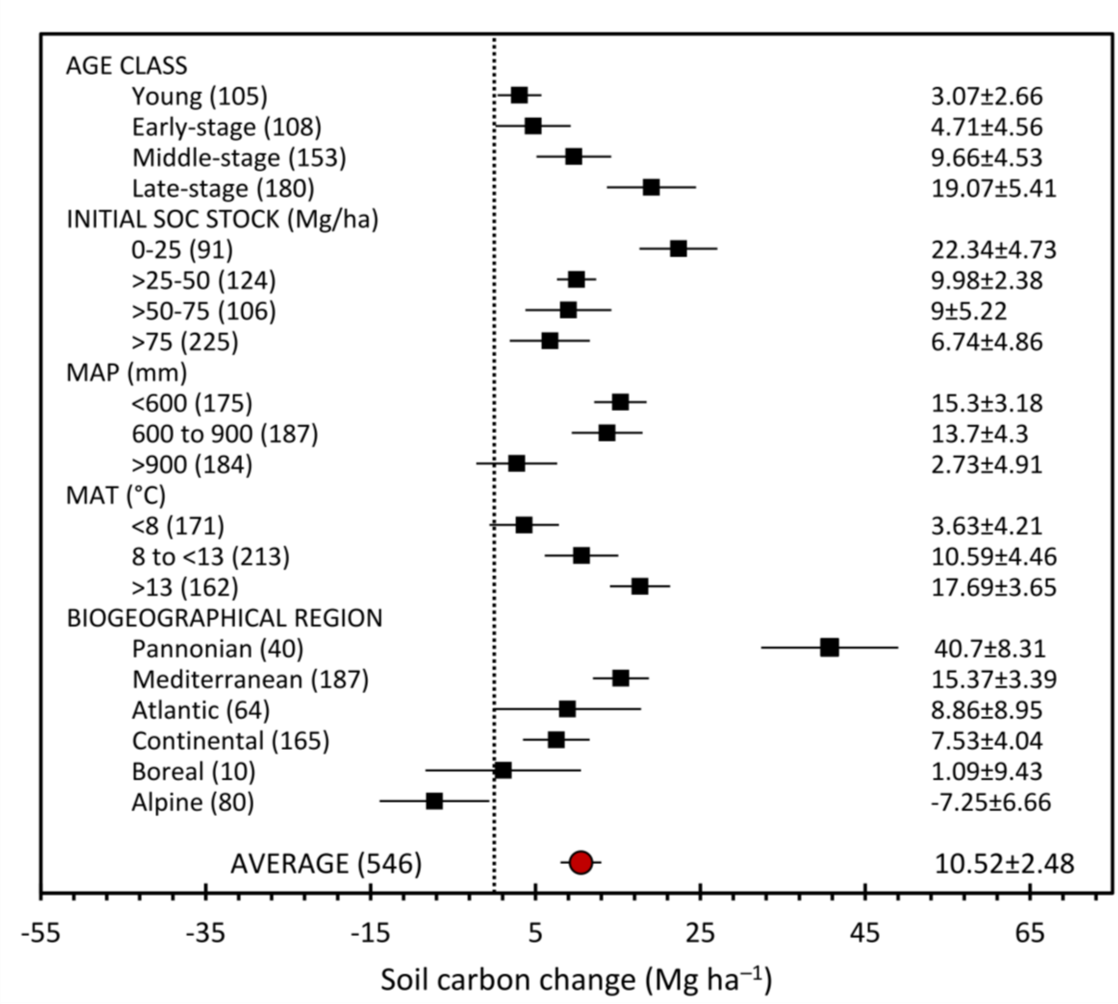

Figure 21. Effects of dominant non-human-related factors on soil carbon change (Mg ha^{-1}) following abandonment/conversion. Age classes are young (≤ 10 years), early-stage (> 10 to ≤ 20 years), middle-stage (> 20 to ≤ 40 years), or late-stage (> 40 years) successional sites. Symbols and error bars indicate mean ±95% CI. Dashed line indicates x = 0. Numbers in parentheses indicate number of observations.

Nearly all categories showed a significantly positive overall effect on SOC change. The only categories with insignificant changes in this dataset, based on 95% confidence intervals, were sites with > 900 mm MAP (2.73±4.91 Mg ha^{-1}, n = 184), < 8 ° C MAT (3.63±4.21 Mg ha^{-1}, n = 171), and the associated Biogeographical regions closer to these climatic thresholds: Atlantic (8.86±8.95 Mg ha^{-1}, n = 64), Boreal (1.09±9.43 Mg ha^{-1}, n = 10), and Alpine (-7.25±6.66 Mg ha^{-1}, n = 80). The effects of the initial SOC stock at the time of abandonment/conversion are also clearly evident in Figure 21, exhibiting the expected inverse relationship with SOC change. The lowest initial stock category (< 25 Mg ha^{-1}, n = 91) responded the greatest to abandonment/conversion, increasing by an average of 22.34±4.73 Mg ha^{-1}, followed by increasingly larger initial stocks categories of > 25-50, > 50-75, and > 75 Mg ha^{-1}, increasing by 9.98±2.38 (n = 124), 9±5.22 (n = 106), 6.74±4.86 Mg ha^{-1} (n = 225), respectively.

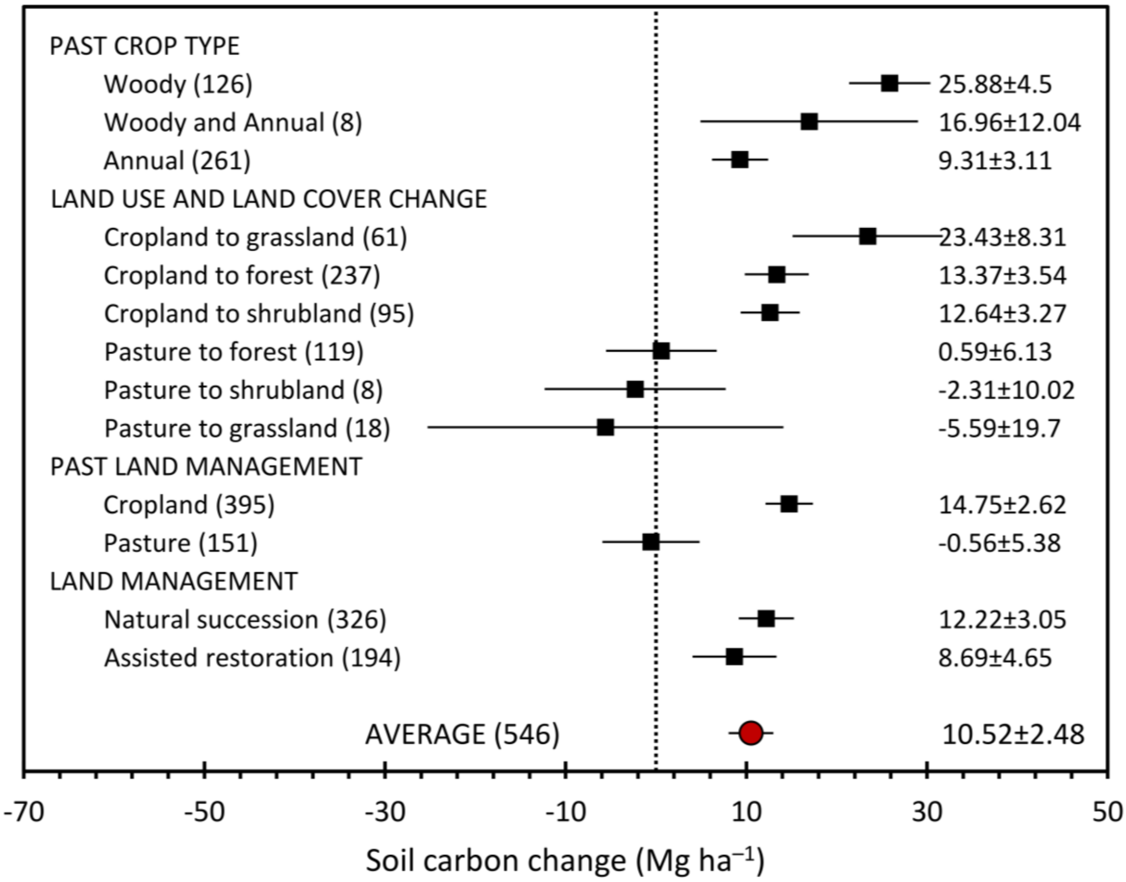

Figure 22. Effects of dominant human-related factors on soil carbon change (Mg ha^{-1}) following abandonment/conversion. Symbols and error bars indicate mean ±95% CI. Dashed line indicates x = 0. Numbers in parentheses indicate number of observations.

The categories within the human-related factors also exhibited several expected trends identified in previous studies covering different geographic extents (Deng et al., 2016; Kämpf et al., 2016). Although all past croplands exhibited a positive increase (14.75±2.62 Mg ha^{-1}, n = 395), there is a significant difference between past woody croplands (e.g., vineyards, olives groves, orchards) at 25.88±4.5 Mg ha^{-1} (n = 126) and past annual croplands (e.g., cereals) at 9.31±3.11 Mg ha^{-1} (n = 261). On the other hand, lands that were previously used as pastures did not demonstrate a significant response to abandonment/conversion (-0.56±5.38 Mg ha^{-1}, n = 151). Accordingly, all land use and land cover change categories involving abandonment/conversion of pastures to more naturalized vegetation communities (i.e., grassland, shrubland, or forest) did not exhibit statistically significant SOC responses, while all categories involving croplands produced significant positive responses in SOC, ranging from 12.64–23.43 Mg ha^{-1}. The categories of land management regimes post-ALA/conversion, interestingly, did not differ as significantly as the categories within the other driving factors (MGMT in Supplementary materials Table 10). However, natural succession (n = 326), also known as passive management, did result in a higher overall soil carbon change than assisted restoration (n = 194), also known as active management (12.22±3.05 Mg ha^{-1} compared to 8.69±4.65 Mg ha^{-1}).

4.4.2 Overall SOC dynamics following ALA

Despite notable variability in responses, abandonment/conversion from agricultural practices across Europe results in a slow, but significant, relative rate of SOC stock increase of 1.28% yr^{-1} (n = 546, $R^2 = 0.19$, $p < 0.0001$) (Figure 23.a) and an absolute rate of 0.32 Mg ha^{-1} yr^{-1} (n = 546, $R^2 = 0.09$, $p < 0.0001$) (Figure 23.c). This absolute rate is quite comparable to other large geographic scale studies, like Post and Kwon (2000) who found global croplands to forest conversions sequestered 0.34 Mg ha^{-1} yr^{-1} (n = 47), and Deng et al., (2016) who calculated a sequestration rate of 0.30 Mg ha^{-1} yr^{-1} for global croplands to grassland conversions (n = 57). Deng et al., (2014) also reported another highly similar rate of 0.33 Mg ha^{-1} yr^{-1} based on 844 observations at 181 sites from China's "Grain-for-Green" Program. However, at the biome and regional scales, several other synthesis studies have reported larger rates (see Figure 10 in CHAPTER II). For arable land to managed or unmanaged grasslands conversions in the temperate zone, Kämpf et al., (2016) found a SCS rate of 0.72 Mg ha^{-1} yr^{-1} (n = 54), which may be overestimated due to the comparatively shorter average time since abandonment/conversion (14 years). SCS rates even as high as 1.30 Mg C ha^{-1} yr^{-1} have been estimated across the tropical zone and in China (Deng et al., 2014; Silver et al., 2001). In another example, the weighted average rate of two studies exploring the SCS potential of active restoration on at the global scale is 0.87 Mg C ha^{-1} yr^{-1} (Li et al., 2012; Shi et al., 2013). It is important to remember that directly comparing reported SCS rates between synthesis studies is challenging due to differences in sample site distribution and study parameters (i.e., time range since abandonment/conversion, soil depth considered, management practices included or excluded, etc.).

On a logarithmic scale, the positive correlation between time and SOC stock change is more noticeable, reaching a clearer direct relationship at the X,Y extremes (i.e., arrowhead converging to a 1:1 correlation at longer time scales) (Figure 23.b,d). These results provide new insight into some of the previous regional debates on the positive, negative, neutral SCS potential of post-agricultural soils (Bárcena et al., 2014; CHAPTER III), which have likely been confounded by other key factors examined here. The model-average importance of each predictor factor explored is summarized in Supplementary materials Figure 28 with the results of the multi-model analysis and best model equation provided in Supplementary materials

Table 11. Time since abandonment alone was not among the most important predictor factors, indicating the highly complex interactions at-play following abandonment/conversion, at least over the distribution of time periods available in this dataset (Q1 of 14 years, median of 27 years, mean of 32 years, and Q3 of 41 years). The overwhelming variability of SOC responses in the first several decades demonstrates the importance of considering all potential factors in addition to the time-scale reported. Long-term land management scenarios must therefore be detailed enough with these factors to adequately capture the uncertainty in SOC responses, especially in the first few decades. The distribution of time since abandonment in this dataset may not be sufficient to fully constrain the isolated effect of time over stronger cofounding interactive effects of human management and environmental factors.

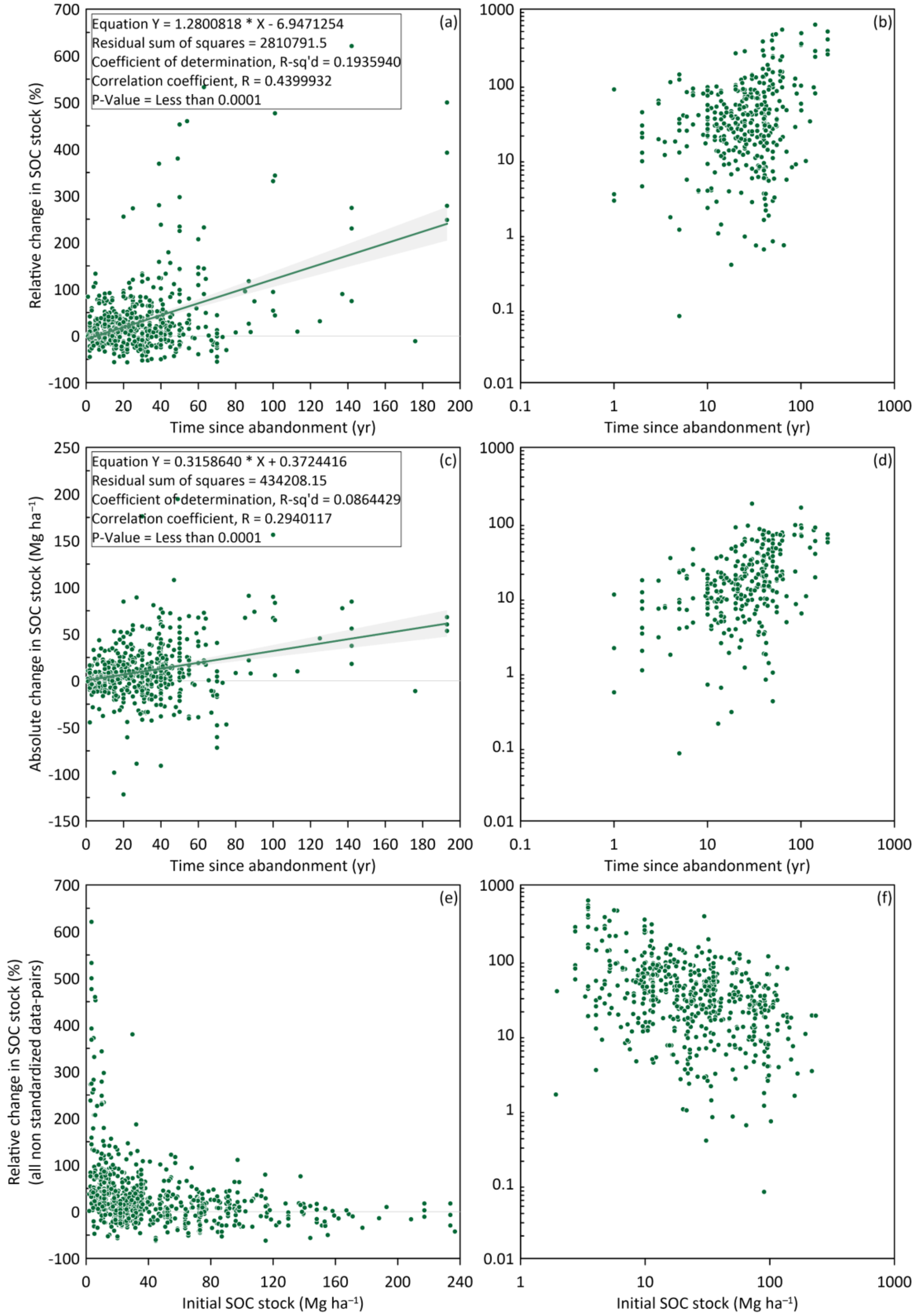

Figure 23. Relative and absolute change in SOC stock over time since abandonment/conversion (yr) on a linear scale with regression results shown in the insert (a, c) and on a logarithmic scale (b, e). Relative change in SOC stock (%) against initial SOC stock (Mg ha^{-1}) of the full non-standardized dataset (n = 804) on a linear scale (e) and on a logarithmic scale (f). Shaded area represents 95% confidence interval.

The influence of the initial SOC stock at the time of abandonment/conversion is also evident in Figure 23. The ability of post-agricultural soils with high initial SOC stocks to accrue new carbon were limited, whereas the soils with the highest relative increases in SOC stock were exclusively ones that had very low initial SOC stock (Figure 23.e). However, many soils with low initial SOC stock also had very low or even negative SOC responses to abandonment/conversion. Overall, the relationship between initial SOC stock and relative SOC increase is negative (Figure 21, Figure 23.f). This relationship is to be expected based on classical soil carbon saturation theory, with the soils with greater initial stock presumably closer to their saturation limit and therefore with less capacity to accrue new SOC (Stewart et al., 2007). Amongst the different soil depth classifications examined in the full dataset (n = 804), the relative change in SOC stock followed expected patterns (Figure 24.b). The highest rates of increase were found in the top-soil at < 5 cm (2.52% yr^{-1}, n = 79, $R^2 = 0.51$, $p < 0.0001$), followed by the 5-15 cm depth (1.00% yr^{-1}, n = 366, $R^2 = 0.16$, $p < 0.0001$). The > 15 cm depth exhibited no statistically significant change over time (n = 359, $p = 0.436$), with the potential of SOC losses after several decades. The effect of initial SOC stock is also present when combined with depth (Figure 24.a) with higher initial stock soils (> 50 Mg ha^{-1}) exhibiting slower rates of SOC change than lower initial stock (< 50 Mg ha^{-1}) soils for each maximum depth grouping of complete soil profiles measured (0–10 cm and 0–30 cm).

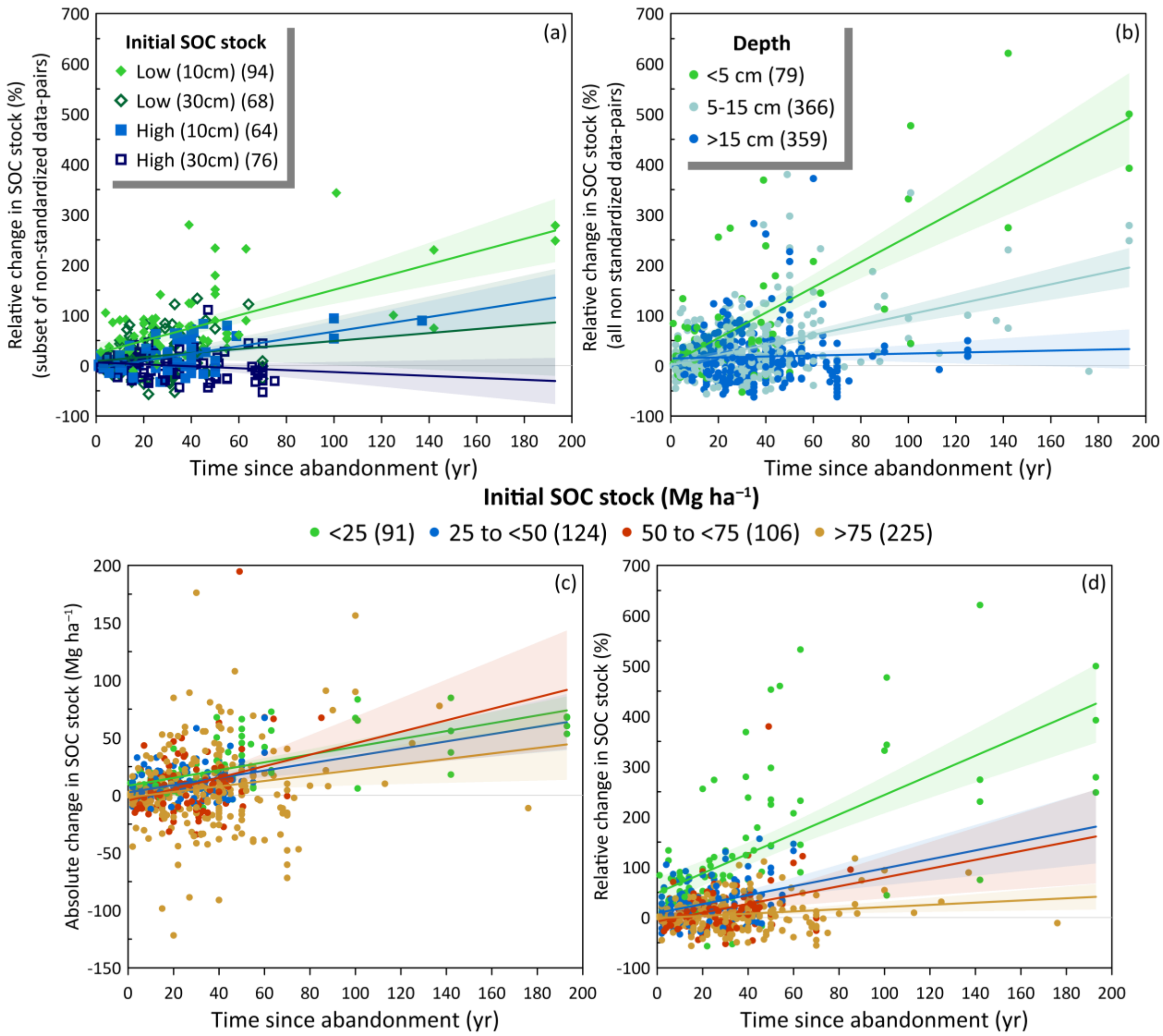

Figure 24. (a) Relative change in SOC stock (%) over time since abandonment/conversion (yr) for different initial SOC stocks found in complete soil profiles with different maximum soil depths (0–10 or 0–30), representing a subset of the total dataset. L indicates low initial stock (< 50 Mg ha^{-1}) and H indicates high initial stock (> 50 Mg ha^{-1}). Shaded areas represent respective 95% confidence intervals of linear regressions. (b) Relative change in SOC stock (%) over time since abandonment/conversion (yr) for different soil depths (average) within the total dataset (n = 804). (c, d) Relative and absolute change in SOC stock of different initial SOC stock classes (Mg ha^{-1}) in the standardized dataset (n = 546), plotted over time since abandonment/conversion (yr) on a linear scale. Numbers in parenthesis indicate sample sizes.

Soil nutrients accumulation is known to be highest closer to the surface during secondary succession, where plant litter inputs are present and there is comparatively more biochemical processes and exchanges occurring (Cramer et al., 2008; Hu et al., 2018; La Mantia et al., 2013; Nadal-Romero et al., 2016). Although post-agricultural soil profiles may demonstrate a distinct legacy of tillage in having a lasting homogeneity (e.g., Sulman et al., (2020)), it is also possible that the ability of new SOC to saturate deeper into the previously homogenized and SOC-depleted top/mid-soils is outpaced by the accrual of new SOC at the surface resulting in greater rates of SOC increases over time (CHAPTER III, Figure 13). In absolute terms across the standardized dataset (n = 546), the effects of initial stock over time since abandonment are less clear than in relative terms (Figure 24.c,d). Low initial stock (< 25 Mg ha^{-1}) exhibited a significantly higher SCS rate than all the other initial stock classes (1.95% yr^{-1}, n = 91, R^2 = 0.41, $p < 0.0001$), while the highest initial stock (> 75 Mg ha^{-1}) exhibited a significantly lower sequestration rate than all the other classes (0.22% yr^{-1}, n = 225, R^2 = 0.03, $p < 0.007$).

4.4.2.1 Climatic and biogeographical factors on SOC dynamics

The climatic regime present at the sampling sites had a noticeable influence on the rates of SOC change following abandonment/conversion from agricultural practices. Similar to the results of a synthesis of natural succession post-agricultural chronosequences and paired-plots in peninsular Spain (CHAPTER III), the sampling sites distributed across Europe were subject to complex temperature (Figure 25.top) and precipitation (Figure 25.bottom) windows for post-agricultural SOC accumulation.

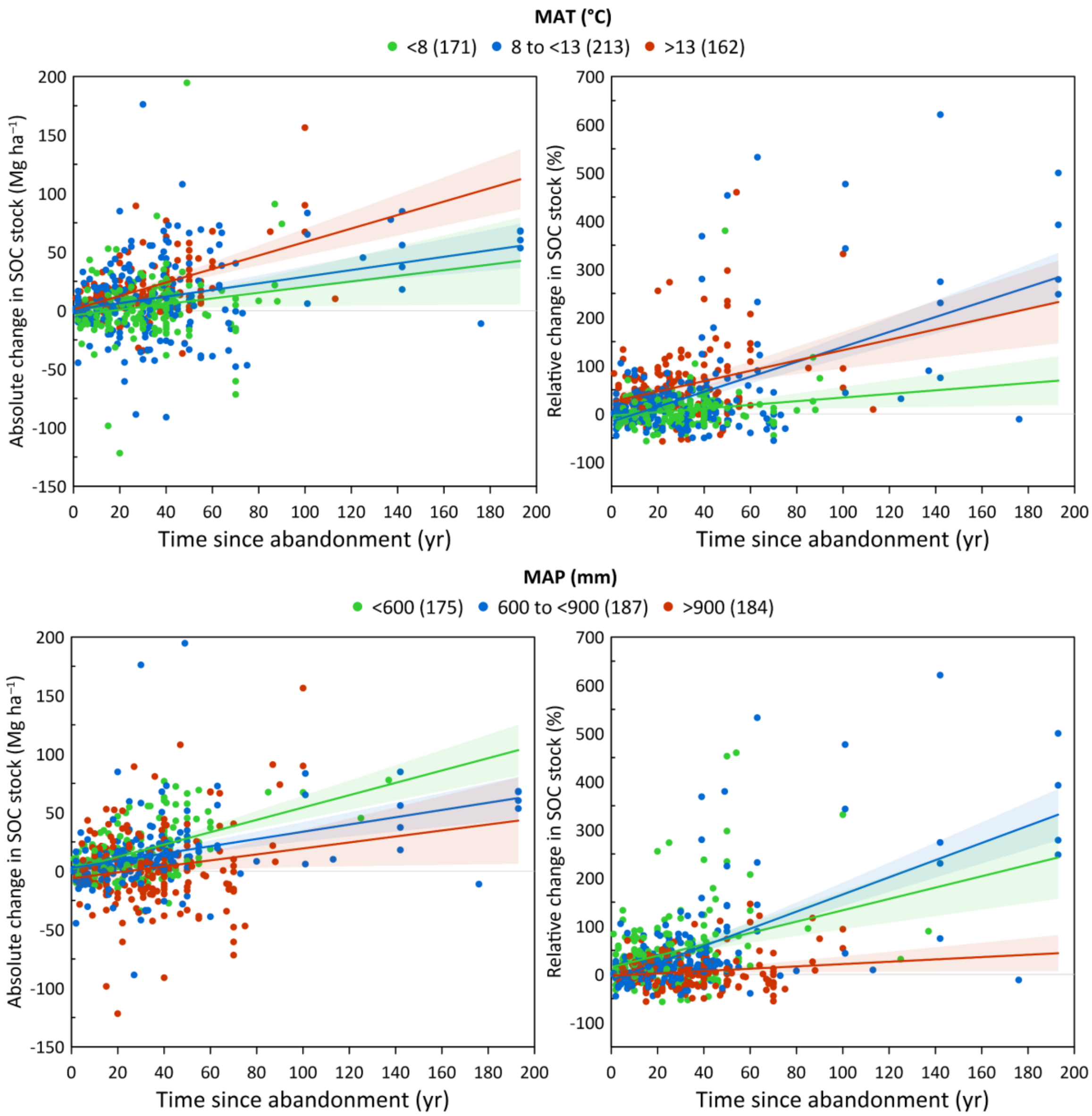

Figure 25. Absolute (Mg ha^{-1}) and relative (%) change in SOC stock over time since abandonment/conversion (yr) for mean annual temperature (MAT, ℃, top panels) and mean annual precipitation (MAP, mm, bottom panels) and their linear regressions. Shaded areas represent respective 95% confidence intervals. Numbers in parenthesis indicate sample sizes.

SOC stock change depends on high precipitation and temperature for organic matter input through the increased net primary productivity. However, in many regions of Europe that is represented in this dataset, confounding climatic effects are present (e.g., cold and wet Alpine pastures, hot and dry Mediterranean vineyards). This climatic complexity in SCS following ALA can also be seen through the lens of biogeographical region classifications, which take into account specific vegetation, biodiversity, and climate interactions (Figure 26), and are linked closely with soil processes (Ibáñez et al., 2013). Interestingly, while cold and wet climates are normally associated with higher SOC accumulation for most land use changes, it may not hold true for ALA (Gardi et al, 2016; Guo and Gifford, 2002). For ALA, the apparent limiting effect of precipitation at levels above 900–1100 mm per year that has been reported across the Mediterranean is likely the result of precipitation induced N leaching, decreases in aggregate protected SOC, and increases in less protected particulate SOM fractions (Alberti et al., 2011; Gabarrón-Galeote et al., 2015b; Guidi et al., 2014; Navas et al., 2012). At the global scale, precipitation and SOC accumulation during ecological succession generally correlate negatively (Jackson et al., 2002), although dry conditions found in semi-arid climates (e.g., < 450 mm yr^{-1}) can also limit net primary productivity (NPP) and therefore limit organic matter inputs that promote SOC accumulation (Bonet, 2004; Gabarrón-Galeote et al., 2015b; Robledano-Aymerich et al., 2014). Conversely, while increased soil carbon cycling can be linked with increasing temperature, high SOM decomposition rates limiting SCS in sermi-arid climates are unlikely because microbial activity would also be drought limited (Moreno et al., 2019). For grasslands on previously managed pastures, SOC also correlates positively with temperature and negatively with precipitation (Kämpf et al., 2016; La Mantia et al., 2013; Pellis et al., 2019). As a partial representation of climatic conditions, the European biogeographical regions also display a wide variety of rates of change for SOC stock following abandonment/conversion (Figure 26). These biogeographical regions are based on biota, unlike biomes, and emphasize endemic and/or spatially distinct and limited taxa and communities (Morrone, 2018).

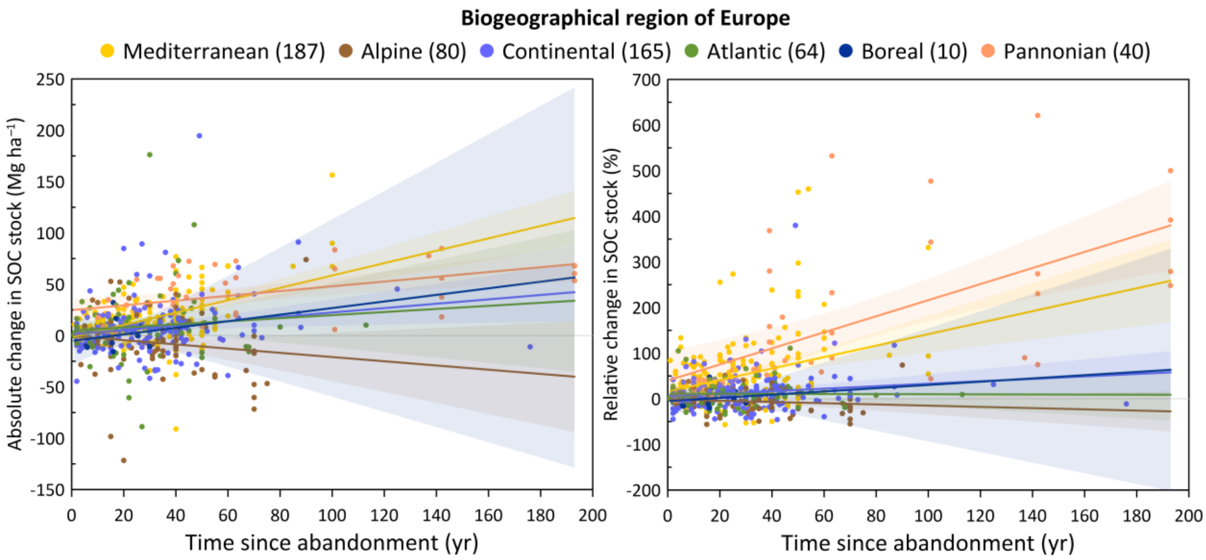

Figure 26. Absolute (left, Mg ha^{-1}) and relative (right, %) change in SOC stock (%) over time since abandonment/conversion (yr) for the Biogeographical Regions of Europe and their linear regressions. Shaded areas represent respective 95% confidence intervals. Numbers in parenthesis indicate sample sizes.

Some of the highest relative rates of SOC stock increase were from soils within the Mediterranean biogeographical region (1.26% yr^{-1}, n = 187, R^2 = 0.10, $p < 0.0001$), likely attributed to the higher relative contribution of new organic matter production and inputs post-abandonment/conversion compared to the lower significance of such additions in landscapes with greater NPP (i.e., in more temperate zones). In other words, the act of ALA/conversion can have a much more dramatic impact on Mediterranean agroecosystems in terms of SOM than in other regions. In accordance with this, there is a much lower relative rate of SOC stock increase in Continental soils (0.27% yr^{-1}, n = 165, R^2 = 0.02, p = n.s.), and even a negative rate (i.e., steady SOC loss) in soils within Alpine regions (-0.13% yr^{-1}, n = 80, R^2 = 0.01, p = n.s.), as the site conditions differ greatly from Mediterranean conditions. However, these trends are highly variable and non-significant at $p < 0.05$, especially in absolute terms. The only region with greater relative rates of SOC change than the Mediterranean was the Pannonian region (1.76% yr^{-1}, n = 40, R^2 = 0.38, $p < 0.0001$), where warm, wet, dry, and cold fronts from the Mediterranean, Alps, and Carpathians all converge in the sheltered basin. It is also worth noting that the Mediterranean relative SCS rate reported here is much smaller than the rate identified in CHAPTER III as a result of the inclusion of abandoned pastures, the data standardization to full 0–30 cm profiles, and the quantification and analysis of SOC stock instead of concentration.

4.4.2.2 Land use and management factors on SOC dynamics

Aside from abiotic factors across Europe influencing SOC response to ALA, such as climate and topography, human driven factors also play a critical role especially at smaller spatial scales (i.e., the plot or landscape). In this study, I explored the influence of past land use classification (whether cropland or pasture), post-abandonment/conversion land management regimes (whether natural succession or assisted restoration), and past crop type (whether woody or annual) (Figure 27). Sites that were croplands (n = 393) before abandonment/conversion had a notably greater rate of SOC increase over time relatively (1.52% yr^{-1}, n = 393, R^2 = 0.27, $p < 0.0001$) and absolutely (0.38 Mg ha^{-1} yr^{-1}, n = 393, R^2 = 0.18, $p < 0.0001$) than sites that were previously used as pastures (n = 145, p = n.s.). Pastures are expected to have greater initial SOC stocks than croplands at the time of abandonment/conversion, resulting in a lower or negative relative changes in SOC stock as indicated in Figure 27.d. Croplands, on the other hand, may receive more intensive agricultural practices than pastures, including significant biomass removal and regular tillage, which depletes SOC stocks and allows for greater positive relative changes in SOC stock following the cessation of these practices (i.e., abandonment/conversion) (García et al., 2007). Although there is variation among cropland types, with cereal cultivation receiving SOM friendly management practices akin to pastures (e.g., manure application, stubble grazing, seed fallowing) compared to woody croplands that receive poor SOM management practices (e.g., pruned branch losses), croplands as a whole are generally under more SOM degrading management systems than pastures and meadows (Navas et al., 2012; Ruecker et al., 1998). Pasture and grassland plants also have a longer growing season and are more efficient at allocating carbon to soil through their roots than crops, both of which helps them overcome their lower efficiency in converting CO_2 into organic matter and further explains their greater SOC stock (Kuzyakov & Domanski, 2000).

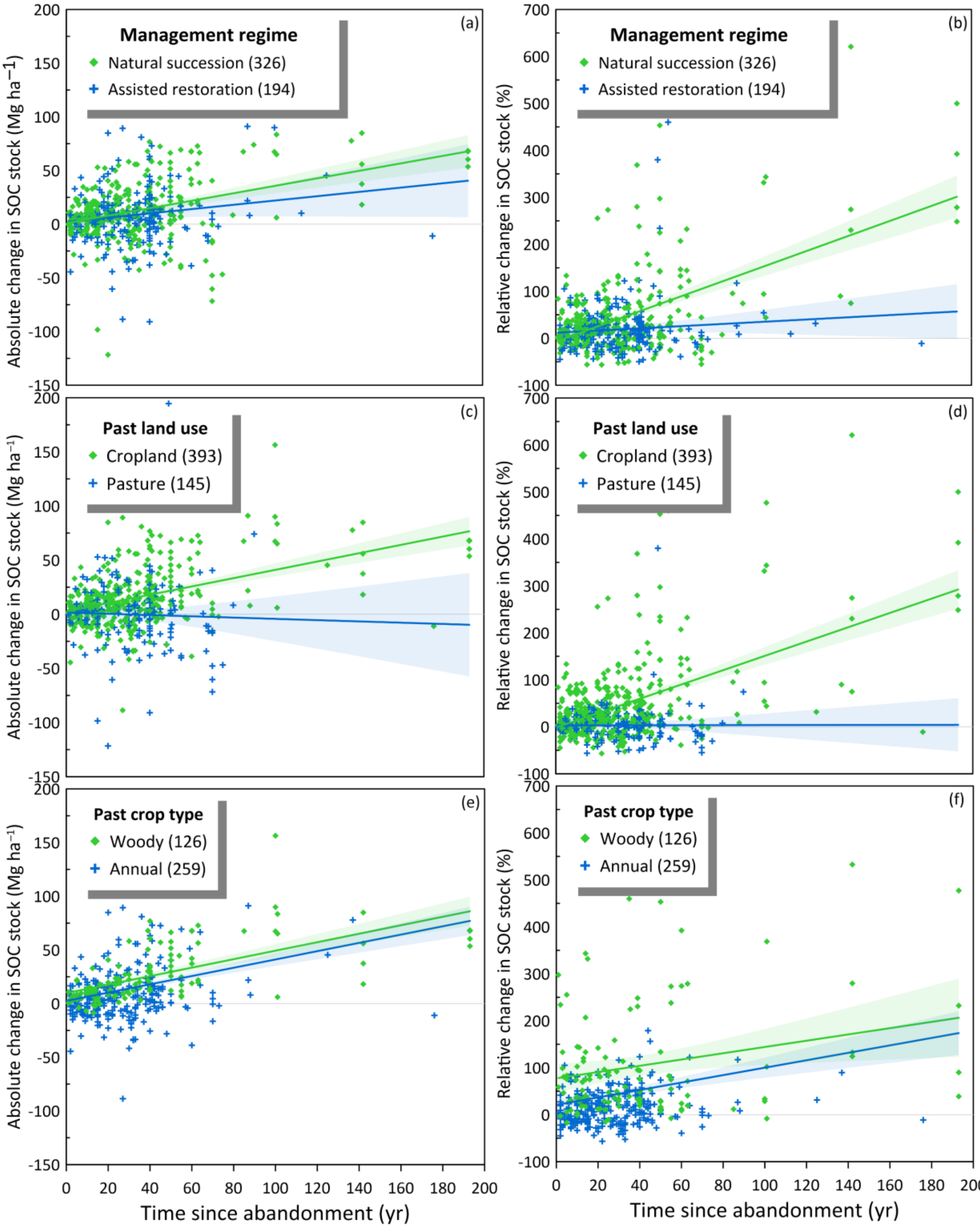

Figure 27. Absolute (left, Mg ha^{-1}) and relative (right, %) change in SOC stock over time since abandonment/conversion (yr) and their linear regressions for post-agricultural sites that: underwent natural succession or assisted restoration after abandonment/conversion (a,b); were either croplands or pastures before abandonment/conversion (c,d); had woody or annual crops (if previously used as croplands) (e,f). Shaded areas represent respective 95% confidence intervals. Numbers in parenthesis indicate sample sizes.

Similar to past land use, the influence of post-abandonment/conversion management systems also produced divergent SOC responses in our dataset (Figure 27.a.b). Sites that were abandoned from agriculture and left to undergo spontaneous ecological succession (n = 326) exhibited a greater rate of change in SOC stock relatively (1.59% yr^{-1}, $R^2 = 0.29$, $p < 0.0001$) and absolutely (0.35 Mg ha^{-1} yr^{-1}, $R^2 = 0.14$, $p < 0.0001$) compared to sites that were actively restored or converted to new vegetation land covers (n = 194, p = n.s.), for example through tree planting practices. The potential of each management approach for SOC accrual certainly depends on conditions at the site of abandonment/conversion. For example, the long land use history involving intensive agricultural in the Mediterranean biogeographical region often requires specific forms of active restoration to overcome stalled vegetation recovery that natural succession may lead to (Garcia-Franco et al., 2014; Ruiz-Navarro et al., 2009; Segura et al., 2020, 2016). While past woody croplands exhibited an overall higher increase in SOC across the whole dataset than past annual croplands (25.88±4.5 and 9.31±3.11 Mg ha^{-1}, respectively) (Figure 22), the SCS rate of both past crop types are similar (Figure 27.e.f). In absolute terms, past woody (n = 126) and annual (n = 385) croplands sequestered SOC at rates of 0.40 Mg ha^{-1} yr^{-1} ($R^2 = 0.40$, $p < 0.0001$) and 0.39 Mg ha^{-1} yr^{-1} ($R^2 = 0.19$, $p < 0.0001$) respectively, while in relative terms the respective rates were 0.67% yr^{-1} ($R^2 = 0.05$, $p = 0.0143$) and 0.79% yr^{-1} ($R^2 = 0.07$, $p < 0.0001$).

The results synthesized in this study represent one of the most comprehensive assessments of the positive or negative impacts of ALA on SCS at a continental scale. However, due to the diversity in the original research aims of all the studies providing data and the methodologies used here, there remains uncertainty in many of the sequestration rates estimated. For example, uneven distribution in time since abandonment/conversion, geographic location (site coordinates), and human management factors may skew the overall trends towards more positive or negative rates. This is in addition to the fact that ALA is already biased towards more degraded soils that may be less conducive for SCS than highly productive agricultural soils that have the potential for positive legacy fertilizer effects if agricultural practices cease and the landscape is restored. Furthermore, because many of the collected studies did not provide important variables like bulk density, coarse material content, SOC concentration at each depth examined, the necessary standardization steps used here have undoubtably reduced accuracy. And lastly, neglect of subsoil samples in many studies and the standardization to the 0–30 cm depth for the entire dataset leaves subsoil SOC stocks unquantified during ALA. The trends observed in topsoils cannot be assumed to hold true in subsoils. Further efforts should focus on improving the dataset representativeness for all predictor factors and sample variables (e.g., bulk density), and incorporating deeper SOC stocks for total soil carbon assessments in Europe, where many soils reach far beyond 30 cm.

4.5 Conclusions

The widespread historical and ongoing agricultural land abandonment found across Europe has resulted in slow, but steady increases in soil organic carbon stocks at an overall rate of 1.28% yr^{-1}, sequestering 0.32 Mg C ha^{-1} yr^{-1}. However, large variabilities in rates are apparent, with some post-agricultural landscapes losing SOC stock over time. In general, sites with low initial stock had greater potential for SOC accumulation while sites with high initial stock are presumably closer to SOC saturation and unlikely to exhibit large relative increases post-abandonment/conversion. Climatic conditions and biogeographical regions influence the likelihood of an abandoned/converted agro-landscape to accumulate SOC, with specific combinations likely driving the observed divergent SOC stock accumulation/loss rates. Past land use (cropland vs. pasture) and post-abandonment/conversion land management strategy employed (natural vs. assisted) also produced divergent responses in SOC change, implying that croplands managed through natural succession would show the greatest SOC accrual while pastures that are actively converted (e.g., afforestation) would result in the lowest increases in SOC, or even losses. The high variability and divergencies in post-abandonment/conversion SOC dynamics must be considered in sustainable land use planning that strives to incorporate the ecological and climate change mitigation benefits of agricultural land abandonment, taking into account site-specific conditions and past and present land management histories to avoid detrimental impacts for soil health and lost opportunities for ecosystem restoration.

4.6 References

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4.7 Supplementary materials

Table 10. Results from Kruskal–Wallis tests of significance for factor effects on SOC stock change.

Factor	df	Chi-square	p-value
CTRL	379.95	240	2.144e-08
AGE	166.5	95	8.101e-06
PLU	34.657	1	3.932e-09
MGMT	10.678	3	0.0136
CROP	65.589	2	5.721e-15
BIOGEO	91.092	5	< 2.2e-16

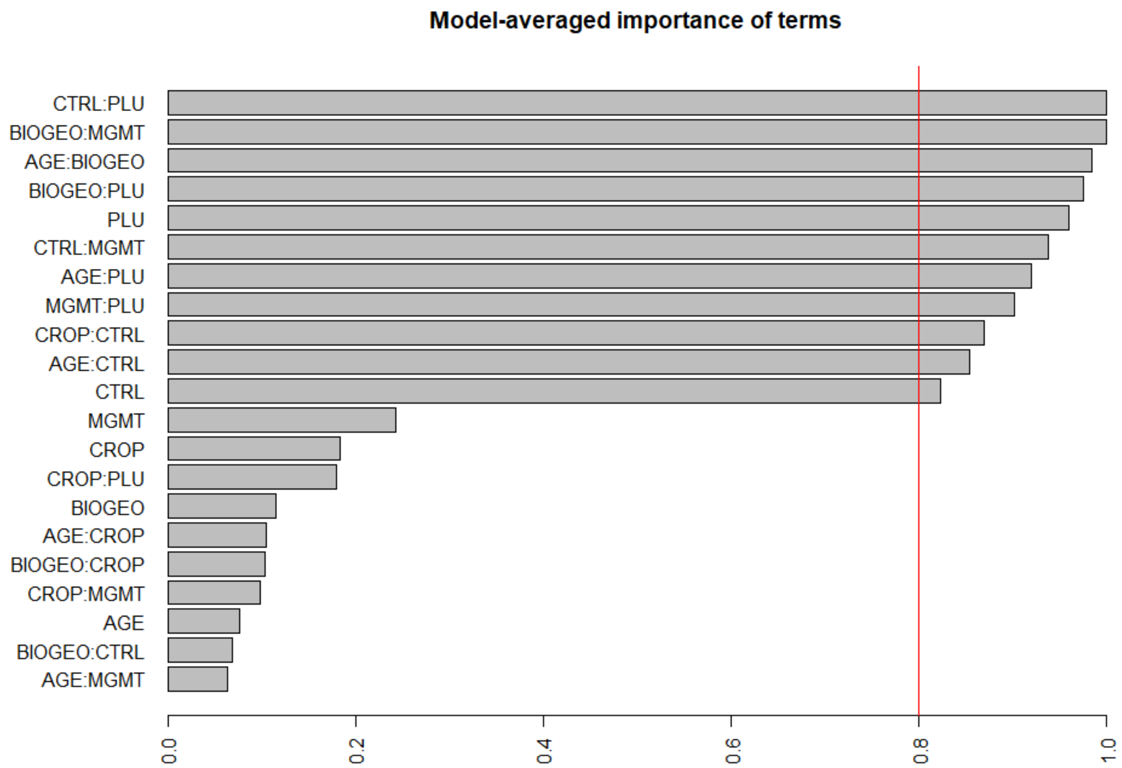

Figure 28. Model-averaged importance of terms following multi-model analysis (generalized linear model). Information criterion set to AICc, and search set to "exhaustive". Factors exceeding the red line have the most importance among all models. SCS, soil carbon sequestration; MGMT, management regime; PLU, past land use; CROP, past crop type; CTRL, initial SOC stock; BIOGEO, biogeographical region; AGE, time since abandonment/conversion.

Table 11. Summary of the best-fitting linear model following multi-model analysis. 43 models were needed to reach 95% of evidence weight, and convergence achieved after 540 generations. SCS, soil carbon sequestration; MGMT, management regime; PLU, past land use; CROP, past crop type; CTRL, initial SOC stock; BIOGEO, biogeographical region; AGE, time since abandonment/conversion.

```
Call:
lm(formula = SCS ~ 1 + PLU + CTRL + MGMT:PLU + BIOGEO:PLU + BIOGEO:MGMT +
    AGE:PLU + AGE:BIOGEO + CTRL:PLU + CTRL:MGMT + CTRL:CROP +
    CTRL:AGE, data = mydata)

Residuals:
     Min       1Q   Median       3Q      Max
-106.950  -13.264   -2.055    9.922  172.555

Coefficients:
              Estimate Std. Error t value Pr(>|t|)
(Intercept) -45.979389   9.321979  -4.932 1.09e-06 ***
PLU          60.669681  11.356469   5.342 1.36e-07 ***
CTRL          0.190686   0.106405   1.792 0.073687 .
PLU:MGMT     -8.405442   2.253538  -3.730 0.000212 ***
PLU:BIOGEO   -3.458415   1.047793  -3.301 0.001029 **
MGMT:BIOGEO   2.028733   0.489941   4.141 4.02e-05 ***
PLU:AGE      -0.220855   0.097794  -2.258 0.024325 *
BIOGEO:AGE    0.092376   0.016198   5.703 1.95e-08 ***
PLU:CTRL     -0.336329   0.060965  -5.517 5.39e-08 ***
CTRL:MGMT     0.059367   0.029477   2.014 0.044514 *
CTRL:CROP     0.080555   0.052516   1.534 0.125640
CTRL:AGE      0.001654   0.001056   1.566 0.117894
---
Signif. codes:  0 '***' 0.001 '**' 0.01 '*' 0.05 '.' 0.1 ' ' 1

Residual standard error: 25.36 on 534 degrees of freedom
Multiple R-squared:  0.2773,    Adjusted R-squared:  0.2624
F-statistic: 18.62 on 11 and 534 DF,  p-value: < 2.2e-16
```

Table 12. List of papers included in the European synthesis.

1	Alberti, G., Leronni, V., Piazzi, M., Petrella, F., Mairota, P., Peressotti, A., Piussi, P., Valentini, R., Gristina, L., Mantia, T.L., Novara, A., Rühl, J., 2011. Impact of woody encroachment on soil organic carbon and nitrogen in abandoned agricultural lands along a rainfall gradient in Italy. Reg. Environ. Chang. https://doi.org/10.1007/s10113-011-0229-6
2	Alberti, G., Peressotti, A., Piussi, P. & Zerbi, G. Forest ecosystem carbon accumulation during a secondary succession in the Eastern Prealps of Italy. Forestry 81, 1–11 (2008).
3	Alías, J.C., Mejías, J.A. and Chaves, N., 2022. Effect of Cropland Abandonment on Soil Carbon Stock in an Agroforestry System in Southwestern Spain. Land, 11(3), p.425.
4	Apostolakis, A., Panakoulia, S., Nikolaidis, N.P., Paranychianakis, N. V., 2017. Shifts in soil structure and soil organic matter in a chronosequence of set-aside fields. Soil Tillage Res. 174, 113–119. https://doi.org/10.1016/J.STILL.2017.07.004
5	Ashwood, F., Watts, K., Park, K., Fuentes-Montemayor, E., Benham, S. and Vanguelova, E.I., 2019. Woodland restoration on agricultural land: long-term impacts on soil quality. Restoration Ecology, 27(6), pp.1381-1392.
6	Badalamenti, E., Battipaglia, G., Gristina, L., Novara, A., Ruhl, J., Sala, G., Sapienza, L., Valentini, R., La Mantia, T., 2019. Carbon stock increases up to old growth forest along a secondary succession in Mediterranean island ecosystems. PLoS One 14. https://doi.org/10.1371/journal.pone.0220194
7	Bárcena, T.G., Gundersen, P. and Vesterdal, L., 2014. Afforestation effects on SOC in former cropland: oak and spruce chronosequences resampled after 13 years. Global Change Biology, 20(9), pp.2938-2952.
8	Bell, S.M., Terrer, C., Barriocanal, C., Jackson, R.B., Rosell-Melé, A., 2021. Soil organic carbon accumulation rates on Mediterranean abandoned agricultural lands. Sci. Total Environ. 759. https://doi.org/10.1016/j.scitotenv.2020.143535
9	Biasi, R., Farina, R. and Brunori, E., 2021. Family Farming Plays an Essential Role in Preserving Soil Functionality: A Study on Active Managed and Abandoned Traditional Tree Crop-Based Systems. Sustainability, 13(7), p.3967.
10	Black, K., Byrne, K.A., Mencuccini, M., Tobin, B., Nieuwenhuis, M., Reidy, B., Bolger, T., Saiz, G., Green, C., Farrell, E.T. and Osborne, B., 2009. Carbon stock and stock changes across a Sitka spruce chronosequence on surface-water gley soils. Forestry, 82(3), pp.255-272.
11	Blanco-Moure, N., Gracia, R., Bielsa, A.C., López, M.V., 2016. Soil organic matter fractions as affected by tillage and soil texture under semiarid Mediterranean conditions. Soil Tillage Res. 155, 381–389. https://doi.org/10.1016/J.STILL.2015.08.011
12	Blanco-Moure, N., Moret-Fernández, D. and López, M.V., 2012. Dynamics of aggregate destabilization by water in soils under long-term conservation tillage in semiarid Spain. Catena, 99, pp.34-41.
13	Boecker, D., Centeri, C., Welp, G. and Möseler, B.M., 2015. Parallels of secondary grassland succession and soil regeneration in a chronosequence of central-Hungarian old fields. Folia Geobotanica, 50(2), pp.91-106.
14	Bonet, A., 2004. Secondary succession of semi-arid Mediterranean old-fields in south-eastern Spain: insights for conservation and restoration of degraded lands. J. Arid Environ. 56, 213–233. https://doi.org/10.1016/S0140-1963(03)00048-X
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17	Cerdà, A., Ackermann, O., Terol, E. and Rodrigo-Comino, J., 2019. Impact of farmland abandonment on water resources and soil conservation in citrus plantations in eastern Spain. Water, 11(4), p.824.
18	Chiti, T., Blasi, E., Pellis, G., Perugini, L., Chiriacò Maria, V., Valentini, R., 2018. Soil organic carbon pool's contribution to climate change mitigation on marginal land of a Mediterranean montane area in Italy. J. Environ. Manage. https://doi.org/10.1016/j.jenvman.2018.04.093
19	Company, J., Valiente, N., Fortesa, J., García-Comendador, J., Lucas-Borja, M.E., Ortega, R., Miralles, I. and Estrany, J., 2022. Secondary succession and parent material drive soil bacterial community composition in terraced abandoned olive groves from a Mediterranean hyper-humid mountainous area. Agriculture, Ecosystems & Environment, 332, p.107932.
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22	De Baets, S., Van Oost, K., Baumann, K., Meersmans, J., Vanacker, V., Rumpel, C., 2012. Lignin signature as a function of land abandonment and erosion in dry luvisols of SE Spain. CATENA 93, 78–86. https://doi.org/10.1016/J.CATENA.2012.01.015
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26	Don, A., Scholten, T. and Schulze, E.D., 2009. Conversion of cropland into grassland: Implications for soil organic-carbon stocks in two soils with different texture. Journal of Plant Nutrition and Soil Science, 172(1), pp.53-62.
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31	Fernández-Calviño, D., Nóvoa-Muñoz, J.C., López-Periago, E. and Arias-Estévez, M., 2008. Changes in copper content and distribution in young, old and abandoned vineyard acid soils due to land use changes. Land degradation & development, 19(2), pp.165-177.
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38	Guidi, C., Vesterdal, L., Gianelle, D. and Rodeghiero, M., 2014. Changes in soil organic carbon and nitrogen following forest expansion on grassland in the Southern Alps. Forest ecology and management, 328, pp.103-116.
39	Hamer, U., Makeschin, F., Stadler, J. & Klotz, S. Soil organic matter and microbial community structure in set-aside and intensively managed arable soils in NE-Saxony, Germany. Appl. soil Ecol. 40, 465–475 (2008).
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CHAPTER V: Conclusions and future perspectives

5.1 Conclusions

The overall aim of this PhD thesis is to generate new knowledge on the effects of ALA on soil carbon stocks by exploring the temporal dynamics of soil organic carbon following the cessation of agricultural activities at the field, regional, and continental scales. This thesis provides novel insights into the capacity of European agricultural soils to recarbonize through ecological succession, informing international ecosystem restoration policies and land management strategies on the potential carbon benefits, costs, and challenges of post-agricultural landscapes.

Establishing new land uses on abandoned agricultural lands is becoming increasingly attractive as global demand for land, food, and energy intensifies. This presents important opportunities for carbon sequestration co-benefits. A literature review was conducted to identify the foremost proposed management strategies for abandoned agricultural lands and compare their reported soil carbon sequestration rates, at any temporal or spatial scale (RQI). Six major categories were identified, each with positive and negative, direct and indirect outcomes for climate change mitigation efforts depending on site-specific factors and management objectives (Table 3). Accordingly, no single strategy is ideal in all scenarios and a combination of strategies can address multiple rural development goals concurrently. These site-specific factors also play a significant role in the soil carbon sequestration efficacy of each proposal in each agricultural region of the world, resulting in high variability among rates (Figure 10). A combination of passive and active management techniques is the most effective approach for maximizing soil carbon sequestration over large geographic scales, while other strategies can be designed to also promote low-carbon land use practices and fossil fuel substitution. The ecological and rural development implications of each management strategy and new land use highlighted in CHAPTER II informs policymakers tasked with planning the future of rural areas experiencing ALA. To better quantify what past and present ALA implies for climate change mitigation, temporal analyses featuring a variety of agricultural practices and crop types abandoned in different bioclimates should be considered. However, this requires overcoming the challenge of gathering soil data at the decadal scale and longer.

One of the global hot-pots of ALA over the last century has been the Mediterranean region of Europe, with significant rates of ALA projected to continue. While secondary succession on abandoned agricultural lands globally can be expected to promote SCS, the accumulation of SOC in Mediterranean countries has been difficult to predict and is subject to multiple competing factors. Gains, losses, and no significant changes have all been reported. Therefore, field work was conducted to explore the effects of depth and time on SCS following ALA in typical Mediterranean agricultural and secondary forest environments, and a new dataset of chronosequences of ALA and paired-plots from peninsular Spain was generated and synthesized to identify the potential factors responsible for the high variability in post-agricultural SCS rates observed in the Mediterranean region (RQII). Chronosequence field studies indicated an average SOC concentration accumulation rate of +2.3% yr^{-1} post-abandonment (Figure 16); but it is a highly variable process, depending on multiple environmental and land management factors. The highest rates of SOC accumulation post-abandonment can be expected on lands previously used for woody crop production featuring ~13–17 ° C MAT and ~450–900 mm MAP, with the lowest rates expected on lands previously used for annual crop production outside this climatic window. Interestingly, the secondary forest field sites accrued 40.8 Mg C ha^{-1} (+172%) following abandonment but displayed greater SOC and N depth heterogeneity than natural forests (Figure 13), demonstrating the long-lasting impact of agricultural practices. Overall, the findings highlighted in CHAPTER III demonstrated that ALA has produced divergent increases in SOC concentrations in peninsular Spain. By altering the SOC accumulation rates of existing secondary forests and influencing the locations and crop types of future ALA, precipitation and temperature changes in the Mediterranean region will determine the SCS potential and ecological value of abandoned agricultural lands. Regional climate change mitigation policies in Mediterranean and semi-arid environments can consider ALA as a low-cost but long-term option best incorporated in tandem with other multipurpose sustainable land management strategies.

At the continental scale, ALA is also a prominent land use change throughout Europe, with several notable implications for soil health, ecosystem restoration, and transboundary rural development planning. However, large uncertainties on the variability of post-abandonment SCS rates (as evident in Spain, CHAPTER III) and the absolute storage potentials across Europe hinders the development of dedicated policies leveraging the restoration benefits of both intentional (i.e., managed restoration and direct conversions) and unintentional, unplanned ALA. To provide critical information and new insights in support of the next generation of European land use and climate-related policies seeking to leverage post-agricultural soils as carbon sinks, the largest dataset ever collected on SOC stock changes specifically following agricultural land abandonment/conversion at a continental scale was produced and synthesized. By extracting over 800 data-pairs from published chronosequences and paired plots and estimating 546 individual soil profiles, the potential environmental and human management factors driving SCS rates following ALA in Europe were investigated (RQIII). There is a slow, but significant, rate of SOC stock increase across Europe, at 1.28% yr^{-1}, with an absolute rate of 0.32 Mg ha^{-1} yr^{-1} (Figure 29). The mean relative and absolute increase amongst the data-pairs is 32.1% and 10.5 Mg ha^{-1}, respectively, with an average time since abandonment of 34 years.

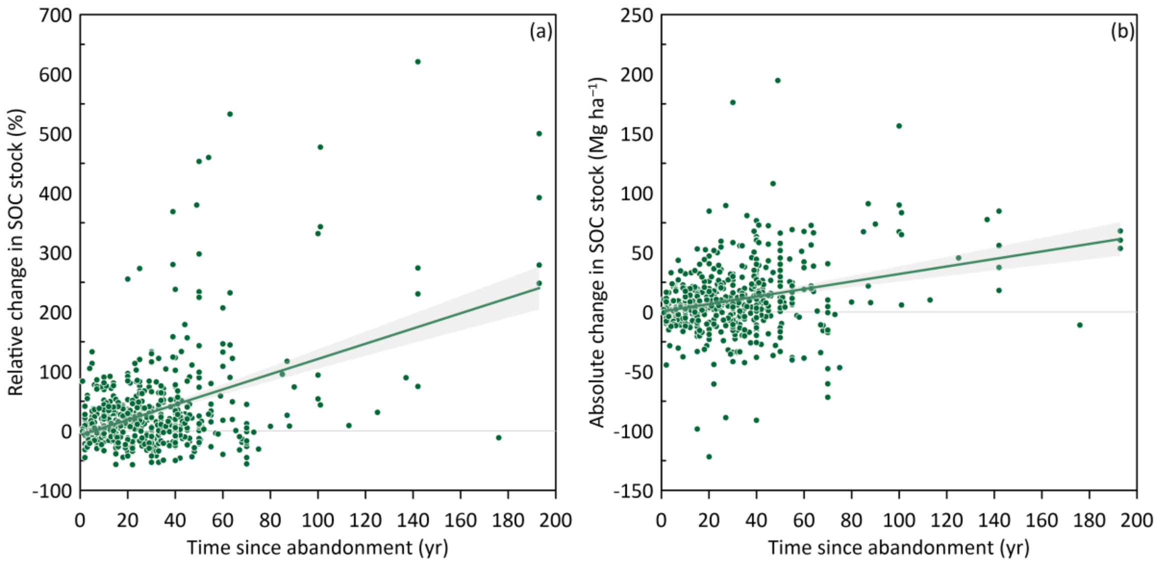

Figure 29. Relative (a) and absolute (b) change in SOC stock (Mg ha^{-1}) over time for the full European dataset of ALA.

These results provide some explanation behind the regional debates on the positive, negative, neutral SCS potential of post-agricultural soils, which have likely been confounded by other key factors. In general, sites with low initial stock had greater potential for SOC accumulation while sites with high initial stock are presumably closer to SOC saturation and unlikely to exhibit large relative increases post-abandonment/conversion. Abandoned agricultural lands in biogeographical regions featuring optimal climatic windows had higher SOC sequestration rates, but human management factors can produce both positive and negative effects on SOC, resulting in several strongly divergent responses to ALA (Figure 27). Past croplands had a notably greater rate of SOC increase over time than sites that were previously used as pastures, likely a result of lower initial SOC stocks in croplands compared to pastures. Sites that underwent natural ecological succession exhibited a greater rate of change in SOC stock relatively compared to sites that were actively restored or converted to new vegetation land covers, for example through tree planting practices. CHAPTER IV suggests that abandoned croplands with low initial SOC stock and managed through natural succession would show the greatest SOC accrual in Europe, while fertile pastures that are actively converted (e.g., afforested) would result in the lowest increases in SOC, or even losses. The variability in post-abandonment/conversion SOC dynamics must be considered in sustainable land use planning that strives to incorporate the positive ecological and climate change mitigation implications of ALA, taking into account site-specific conditions and past and present land management histories to avoid detrimental impacts for soil health and lost opportunities for climate change mitigation.

Overall, the findings presented in this PhD thesis helps inform ecosystem restoration policies and land management strategies on the potential soil carbon benefits, costs, and challenges of post-agricultural landscapes. While the reported SCS rates on abandoned/converted agricultural lands are generally positive for all proposed land management strategies (CHAPTER II), this thesis demonstrated an overarching trend defining the temporal responses of SOC stocks to ALA during revegetation over large geographic scales: divergencies depending on initial carbon stock, past land use, past crop type, restoration management regime, and climate/biogeographical variables (CHAPTER III & IV). The high variability in post-abandonment/conversion SOC temporal dynamics must be considered in sustainable land use planning that strives to incorporate the positive ecological and climate change mitigation implications of ongoing ALA at regional and continental scales, taking into account site-specific conditions and past and present land management factors.

5.2 Future perspectives

Maximizing natural climate solutions is becoming increasingly critical as atmospheric CO_2 concentrations continue to rise (Friedlingstein et al., 2022; Griscom et al., 2017). As another an indication of the level of urgency, Matthews et al., (2022) argues it is no longer even a question of carbon sequestration permanence: even temporary and later reemitted nature-based carbon removal should be pursued to keep global temperature rise below 2° C. Aside from emissions reductions, sustainable and climate-smart land management is one of our most important objectives (IPCC, 2019). Mitigating the negative impacts of climate change by returning carbon to depleted soils, in particular, will require exploring all available avenues (Bossio et al., 2020).

The combined results from this thesis present a promising outlook for leveraging ongoing ALA processes for soil carbon sequestration, and also ecosystem restoration more broadly (Yang et al., 2020). But despite calls for protection (Poore, 2016), unprotected abandoned agricultural lands often lead to recultivation and lost opportunities for climate change mitigation (Crawford et al., 2022). I argue we must further explore integrating strategic post-agricultural landscapes into land use planning, while protecting existing abandoned agriculture lands whenever feasible (i.e., avoiding conflicts with food production and land rights by focusing especially on uncontested abandoned lands (Xie et al., 2020)). This can help diversify the global land carbon sink and support the objectives of the UN Decade of Ecosystem Restoration (2021-2030) (Abhilash, 2021; Aronson et al., 2020). To that end, I propose three next-steps:

1. **Expanding the dataset to all continents.** The inferential potential of assembled chronosequences and paired-plots has not yet been realized. Just like in Europe, there exists thousands of un-synthesized, disparate data-pairs of post-agricultural sites in other continents, with hundreds more published every year. A large-scale global synthesis is needed along the lines of Cook-Patton et al., (2020) or Veldkamp et al., (2020), both of which featured large collections of time stamped data-pairs on soil properties following land use change from agriculture. Special attention must be paid to underrepresented areas in global change science, namely Africa, South America, and most of Asia. The importance of this task is twofold: 1) chronosquences/paired-plots are logistically superior to long-term experimental plots with repeated measurements and should therefore be exploited as much as possible to advance urgent global change science; and, 2) archiving this kind of data for future research lowers the risk of data loss, repurposes valuable past investments, and increases data source diversity which

improves model accuracy (i.e., avoiding data "siloing" by repeatedly analyzing the most popular or available datasets).

2. **Improving post-agricultural soil carbon sequestration modelling.** The high variability in SOC temporal responses found across Europe are a function of the complexity of soils and their interactions with the environment and human activities. Nevertheless, dedicated efforts need to be made to better predict how SOC stocks of a given plot of land will respond to ALA, expanding from linear regression analyses to more appropriate estimations of SOC saturation functions. Just as it is important to produce more accurate soil carbon sequestration rates for different categories of abandoned agricultural lands, it is also necessary to predict their specific equilibrium parameters (i.e., time to equilibrium, equilibrium SOC stock value). In this exercise, it will also be important to account for the different SOC pools (e.g., MAOM and POM), which will have different rates of change than SOM as a whole. The rates of change for these pools are normally assumed in equilibrium models because they are not represented well in most datasets. By extracting, synthesizing, and repurposing time-stamped SOC data from any published study of ALA, no matter the original research angle, we can more robustly benchmark and validate models of successional carbon dynamics.
3. **Combining more robust, data-informed models with next-generation ALA spatial estimates.** After improving temporal SOC data quality and quantity, the lack of accurate, high-resolution ALA spatial estimates at the global scale should be addressed. Current state-of-the-art maps only cover the past few decades, and struggle to capture abandoned pastures and smaller-sized field plots. Fortunately, new sensors are now available with higher spatial, temporal, and spectral resolution allowing for improved monitoring of active agricultural lands, which by extension improves our abilities to identify and map post-agricultural landscapes such as abandoned lands. For example, Sentinel-1 and Sentinel-2, together with Landsat, provide a denser time series at the global scale than what has been previously available. There is also new, very high-resolution (e.g., < 5m) commercial satellite imagery now available that can be used to isolate and process smaller agricultural plots (e.g., PlanetScope). The large volumes of data generated by these new sensors can also now be processed more rapidly and with more accessibility than ever before using free cloud computation resources like *Google Earth Engine*. Ongoing agricultural classifications via ground truthing at the global scale will help validate these newly produced, high-quality agricultural land cover

maps. And with the continual improvement of machine learning approaches at our disposal, we can anticipate the first reliable and detailed global maps of past and present abandoned agricultural lands, and their durations of existence, in the coming years.

I encourage a concerted effort by researchers, policy-makers, and land managers to constrain the impact of ALA on the global land carbon sink as soon as possible. This PhD thesis has served as a springboard for the first global data collection initiative focused on the temporal effects of ALA on soil carbon stocks. Our team of data collectors, led by myself for the past 1.5 years, have expanded the European dataset to every inhabited continent of the world (Figure 30). Our goal is to provide an antithesis publication to the excellent work by Sanderman et al., (2017) by quantifying how ALA has helped us pay back agriculture's longstanding soil carbon debt, putting carbon back where it belongs.[2]

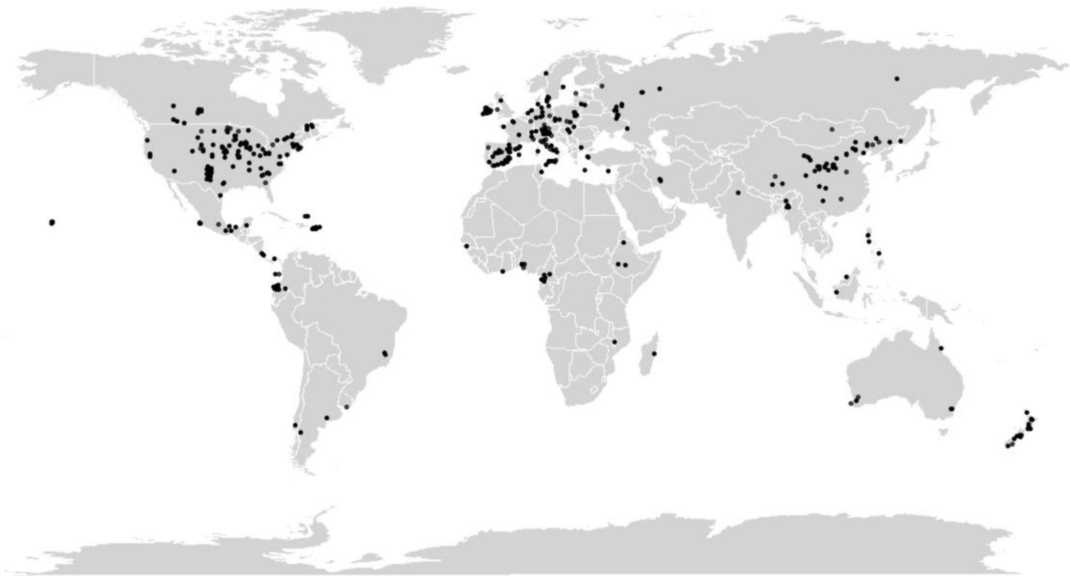

Figure 30. Global distribution of chronosequences and paired-plot data-pairs (n = 3460) collected by myself and our team of MSc and PhD student colleagues in the inter-university project created from this PhD thesis.

[2] www.ted.com/talks/stephen_bell_retiring_farmlands_to_put_carbon_back_where_it_belongs

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