



UNIVERSITÀ
DEGLI STUDI DELLA
TUSCIA

Università degli Studi della Tuscia di Viterbo

DIBAF Department for Innovation in Biological, Agro-Food and Forest Systems

Corso di Dottorato di Ricerca in

SCIENCE, TECHNOLOGIES, AND BIOTECHNOLOGIES FOR SUSTAINABILITY- XXXVII cycle

**ASSESSING THE RISK OF LAND DEGRADATION AND DESERTIFICATION UNDER
CLIMATE CHANGE IN THE MEDITERRANEAN THROUGH TWO CASE STUDIES**

Settore scientifico-disciplinare
(s.s.d. AGRI-03/B)

Tesi di dottorato di:

Dott. Francesca Adducci

Coordinatore del corso

Prof. Andrea Vannini

Tutore

Prof. Giuseppe Scarascia Mugnozza

Co-tutore

Dott. Ettore D'Andrea

Co-tutore

Dott. Giorgio Matteucci

A.A. 2024/25

Sommario

1	Introduction	8
1.1.	Definition	8
1.2.	Socio-political context	8
1.3.	The situation in the Mediterranean basin	9
1.4.	The Importance of Soil and the Impact of Various Drivers	11
1.4.1.	Soil Respiration and Fire-Induced Microclimatic Alterations	13
1.5.	Nature-Based Solutions and the EU Restoration Law for Ecosystem Recovery	14
1.6.	Climate Change and Anthropogenic Drivers of Forest Degradation in the Mediterranean	15
1.6.1.	Forest fires and climate change	16
1.6.2.	Impact of grazing on degraded agroforestry system	21
2	PhD Thesis Objectives and Methodology Rationale	23
3	Material and Methods	25
3.1.	Description and Significance of the Study Sites	25
3.1.1.	Monte Morrone – Parco Nazionale della Majella	27
3.1.2.	Companhia das Lezírias – Lisbon, Portugal	31
3.2.	Environmental and Microclimate Monitoring in a Degraded Mediterranean Forest Ecosystem	35
3.2.1.	Meteorological Data Collection	35
3.3.	Satellite Remote Sensing in Monitoring the Regeneration of Natural Vegetation in Fire-Affected Areas	36
3.3.1.	Spectral index processing and burned area mapping methodology	36
3.3.2.	Post-Fire Vegetation Monitoring and LAI Trend Analysis	38
3.4.	Study Sites and Experimental Setup	39
3.4.3.	Monte Morrone Site	39
3.4.4.	Companhia das Lezírias Site	42
3.5.	Soil CO ₂ Efflux	44
3.5.1.	Estimation of Temperature Sensitivity (Q ₁₀) and Basal Respiration	47
3.6.	Soil Analysis	48
3.6.1.	Soil Sampling	49
3.6.2.	Physical and Chemical Soil Analysis	50
3.6.2.1.	Monte Morrone: Methodologies and Analyses	50
3.6.2.2.	Companhia das Lezírias: Methodologies and Analyses	50
3.7.	Microbiological analysis	52
3.8.	Statistical Analysis	54
4	Results	57
4.1.	Microclimate Data Analysis for Monte Morrone Monitoring Site	57
4.2.	Spectral Index Processing and Burned Area Mapping	59
4.2.1.	Results and Analysis: Monte Morrone Wildfire 2017	59
4.2.2.	Results and Analysis of the Monte Morrone Wildfire 2023	60
4.2.3.	Results and Analysis of Vegetation Recovery Following the 2017 and 2023 Monte Morrone Wildfires	62
4.3.	Soil CO ₂ Efflux Measurement in Relation to Environmental Factors and Soil Properties: A Comparative Study Across Different Stress Drivers	64
4.3.1.	Introduction	64
4.3.2.	Soil CO ₂ Efflux in Monte Morrone Site	65

4.3.2.1.	General Trends of Soil CO ₂ Efflux in Relation to Temperature Across Fire and Control Sites.....	65
4.3.2.2.	Temperature Dependency of SR.....	67
4.3.2.3.	Influence of Soil Moisture on Soil CO ₂ Efflux in Burned Site.....	70
4.3.2.4.	Influence of Soil Temperature and Water Content Interaction on Soil CO ₂ Efflux ..	72
4.3.2.5.	Q ₁₀ and Basal Respiration (BR)	74
4.3.2.6.	Statistical Modeling of SR Drivers	75
4.3.3.	Soil Properties in Monte Morrone Site	77
4.3.3.1.	Soil Properties: Texture, Bulk Density, and pH.....	77
4.3.4.	Companhia das Lazirias Site.....	77
4.3.4.1.	Temporal Variations in Soil Respiration and Temperature	77
4.3.4.2.	Temperature Dependency of SR	79
4.3.4.3.	Influence of Soil Moisture on Soil CO ₂ Efflux	83
4.3.4.4.	Soil Temperature Differences Between Degraded Pastures and Oak Forests in Portugal	85
4.3.4.5.	Temperature Sensitivity (Q ₁₀) and Basal Soil Respiration (BR) in Companhia das Lezírias	85
4.3.4.6.	Statistical Modelling and Environmental Controls on Soil Respiration	87
4.3.5.	Soil Properties in Companhia das Lezírias Monitoring Site.....	89
4.3.5.1.	Introduction.....	89
4.3.5.2.	Soil Property Comparison between Degraded and Control Sites	89
4.3.5.3.	Influence of Soil Properties on Soil Respiration.....	90
4.3.5.4.	Ecological Implications and Comparative Insights	91
4.3.5.5.	Summary of Soil Property Differences	91
4.4.	Soil Microbial Community Composition and Functional Groups in Burned and Control Soils	92
4.4.1.	Bacterial Community Composition	92
4.4.1.1.	Relative Abundance of Bacterial Families.....	92
4.4.1.2.	Depth-Specific Bacterial Distribution.....	95
4.4.1.3.	Effect of Nature-Based Solutions on Bacterial Communities	97
4.4.2.	Fungal Community Composition.....	98
4.4.2.1.	Relative Abundance of Fungal Families.....	98
4.4.2.2.	Depth-Specific Fungal Distribution	100
4.4.2.3.	Effect of Nature-Based Solutions on Fungal Communities.....	102
5	Discussion.....	103
5.1.	Microclimate Dynamics and Their Implications for Post-Fire Soil and Ecosystem Recovery	103
5.2.	Remote Sensing Indicators and Post-Fire Vegetation Recovery in Monte Morrone.....	104
5.3.	Soil CO ₂ Efflux and Environmental Controls in Post-Fire Ecosystems: Functional Responses and Constraints at Monte Morrone	105
5.4.	Integrated Responses of Soil CO ₂ Efflux and Soil Properties to Degradation in Companhia das Lezírias	107
5.5.	Soil Microbial and Fungal Community Responses to Wildfire and Environmental Conditions at Monte Morrone (Abruzzo)	110
6	Conclusion.....	112
7	Future Objectives.....	115
8	References	117

ASSESSING THE RISK OF LAND DEGRADATION AND DESERTIFICATION UNDER CLIMATE CHANGE IN THE MEDITERRANEAN THROUGH TWO CASE STUDIES

Francesca Adducci (francesca.adducci@unitus.it)

Department for Innovation in Biological, Agro-food and Forest systems, Tuscia University,
Viterbo, Italy

Tutor: Prof. Giuseppe Scarascia Mugnozza

CoTutor: Dott. Ettore d'Andrea

CoTutor: Dott. Giorgio Matteucci

Abstract

This research explores the impact of wildfires and environmental stressors on soil degradation and recovery processes in Mediterranean ecosystems, focusing on two contrasting pilot sites: Monte Morrone in Italy and Companhia das Lezírias in Portugal. These two sites represent different degradation dynamics: Monte Morrone has been affected by wildfires in 2017 and 2023, whereas Companhia das Lezírias experiences stress from grazing pressure and land-use changes. This diversity enabled the application of an integrated methodological framework to assess, quantify, and better understand degradation, vulnerability, and recovery processes.

At Monte Morrone, specific Nature-Based Solutions (NBS), such as the strategic placement of burned logs, were tested to improve soil stability, enhance microbial biodiversity, and accelerate natural recovery processes. This approach was systematically applied to treated plots and compared to untreated areas, providing a critical evaluation of NBS effectiveness in fire-degraded ecosystems.

Soil properties—including soil respiration, nutrient availability, organic matter content, and microbial composition analyzed through advanced bioinformatics—were combined with climate data and remote sensing indices like dNBR, RdNBR, dLAI, and dkNDVI. These tools enabled mapping of fire severity and vegetation loss over time. Results indicated that areas burned in 2017 exhibit a more advanced recovery stage compared to those burned in 2023, which remain in the early phases of regeneration.

At the Portuguese site of Companhia das Lezírias, the study focused on soil property analysis, soil respiration, and the effects of stressors such as grazing and land-use changes. Results revealed critical issues, including soil compaction, reduced vegetation cover, and the depletion of microbial diversity, which are key factors in driving desertification processes. Climate data contextualized the observed stressors, offering a comprehensive understanding of their role in soil functionality and resilience.

The primary objective of this research was to analyze the complex interactions between environmental stressors and soil recovery processes, filling knowledge gaps regarding desertification and degradation in Mediterranean ecosystems. Using synthetic indicators, the study quantified vulnerability and identified priority intervention areas. Findings demonstrate that distinct stress drivers—wildfires at Monte Morrone and grazing at Companhia das Lezírias—significantly influence degradation and regeneration processes.

Looking forward, this study proposes a scalable and adaptable methodological framework for monitoring soil degradation across broader Mediterranean contexts. By integrating soil physical and chemical analyses, remote sensing, bioinformatics, and vegetation structural studies, the approach provides a robust tool for ecological monitoring and sustainable management of degraded lands. In summary, this research delivers a multidisciplinary and innovative approach to understanding degradation and recovery processes in Mediterranean ecosystems. It offers concrete solutions to improve soil functionality, promote biodiversity, and develop adaptive strategies to combat desertification and ensure ecosystem resilience for land managers, policymakers, and conservation practitioners.

VALUTAZIONE DEL RISCHIO DI DEGRADAZIONE E DESERTIFICAZIONE DA CAMBIAMENTO CLIMATICO NEL MEDITERRANEO ATTRAVERSO DUE CASI STUDIO

Francesca Adducci (francesca.adducci@unitus.it)

Department for Innovation in Biological, Agro-food and Forest systems, Tuscia University,
Viterbo, Italy

Tutor: Prof. Giuseppe Scarascia Mugnozza

CoTutor: Dott. Ettore d'Andrea

CoTutor: Dott. Giorgio Matteucci

Abstract

Questa ricerca approfondisce l'impatto degli incendi boschivi e dei fattori di stress ambientali sui processi di degrado e recupero degli ecosistemi mediterranei, focalizzandosi su due siti pilota caratterizzati da dinamiche di degrado contrastanti: il Monte Morrone in Italia e la Companhia das Lezírias in Portogallo. Il primo è stato colpito da incendi nel 2017 e nel 2023, mentre il secondo è soggetto a stress legati al pascolo e ai cambiamenti nell'uso del suolo. Questa diversità ha consentito di testare l'efficacia di un quadro metodologico integrato basato su analisi fisiche, chimiche e biologiche del suolo, per comprendere e quantificare i processi di degrado, vulnerabilità e recupero. Nel Monte Morrone, sono state sperimentate specifiche Nature-Based Solutions (NBS), come la collocazione strategica di tronchi bruciati, al fine di migliorare la stabilità del suolo, promuovere la biodiversità microbica e accelerare i processi naturali di rigenerazione. Questa metodologia è stata applicata in aree trattate e confrontata con zone non trattate, offrendo una valutazione critica sull'efficacia delle NBS in ecosistemi degradati dagli incendi.

L'analisi ha integrato parametri del suolo (respirazione, disponibilità di nutrienti, contenuto di sostanza organica, composizione microbica ottenuta tramite bioinformatica avanzata) con dati climatici e indici di telerilevamento (dNBR, RdNBR, dLAI, dkNDVI) per mappare la severità degli incendi e la perdita di copertura vegetale. I risultati mostrano che le aree bruciate nel 2017 presentano uno stadio di recupero più avanzato rispetto a quelle colpite nel 2023, ancora nelle fasi iniziali della rigenerazione.

Nel sito portoghese della Companhia das Lezírias, la ricerca si è focalizzata sull'analisi delle proprietà del suolo, della respirazione e degli effetti del pascolo e dei cambiamenti d'uso del suolo. Le analisi hanno evidenziato criticità quali compattazione, ridotta copertura vegetale e impoverimento della biodiversità microbica, fattori che accelerano i processi di desertificazione. I dati climatici raccolti

hanno fornito un quadro più ampio del ruolo delle pressioni ambientali nella funzionalità e resilienza del suolo.

L'obiettivo principale dello studio è analizzare le interazioni complesse tra stress ambientali e processi di recupero del suolo, colmando le lacune conoscitive sulla desertificazione negli ecosistemi mediterranei. Attraverso indicatori sintetici, sono state quantificate la vulnerabilità e le aree prioritarie di intervento. I risultati dimostrano che i diversi driver di stress – incendi al Monte Morrone e pascolo nella Companhia das Lezírias – influenzano in modo significativo i processi di degrado e rigenerazione.

Guardando al futuro, lo studio propone un quadro metodologico scalabile e adattabile per il monitoraggio del degrado del suolo in contesti mediterranei più ampi. L'approccio integrato – che combina analisi fisico-chimiche, telerilevamento, bioinformatica e studio della struttura vegetale – costituisce uno strumento robusto per il monitoraggio ecologico e la gestione sostenibile dei territori degradati. In sintesi, la ricerca fornisce un approccio multidisciplinare e innovativo per comprendere il degrado e il recupero degli ecosistemi mediterranei, offrendo soluzioni concrete a supporto della pianificazione territoriale e delle politiche ambientali, per favorire la biodiversità e garantire la resilienza degli ecosistemi.

1 Introduction

1.1. Definition

Desertification is increasingly recognized as a global issue. It is defined as "the process that leads to an irreversible reduction in the capacity of the soil to produce resources and services" (FAO UNEP-UNESCO, 1979). The urgency to combat desertification is well established, not only in regions with extensive desert areas but also in Mediterranean countries, which face direct consequences from land degradation processes and indirect impacts stemming from socio-economic imbalances in emerging nations (Cherlet et al., 2018). This phenomenon affects ecosystems and human livelihoods by diminishing soil productivity, biodiversity, and ecosystem services (Millennium Ecosystem Assessment, 2005).

The concept of desertification has evolved over time, integrating new elements that aid in identifying the process more clearly. The United Nations International Convention on Drought and Desertification (UNCCD, 1994) defines desertification as "land degradation in arid, semi-arid, and dry sub-humid areas resulting from various factors, including climatic variations and human activities" (Middleton & Thomas, 1997). This definition emphasizes both human influence and climatic factors in driving land degradation.

Soil degradation, a precursor to desertification, involves the reduction or loss of biological or economic productivity of the land, often exacerbated by human activity such as unsustainable land use practices (Lal, 2015). Factors such as soil cover loss, erosion, and a decline in soil organic carbon are used to assess soil degradation (Montanarella et al., 2016). Moreover, soil degradation not only affects productivity but also impairs biodiversity and ecosystem services (Peñuelas et al., 2017), which are essential for the functioning and resilience of ecosystems.

The concept of neutrality in terms of land degradation, as introduced by the UNCCD, is defined as "a state where the quantity and quality of land resources necessary to support ecosystem functions and services, and to strengthen food security, remain stable or increase within specified temporal and spatial scales" (Orr et al., 2017). When degradation processes intensify and surpass the land's capacity to recover, soil desertification can occur (Huang et al., 2020).

1.2. Socio-political context

In 2006, the European Commission adopted the Thematic Strategy for Soil Protection, emphasizing the risk of desertification due to ongoing soil degradation. The strategy aimed to promote sustainable land use, prevent further degradation, and restore soil functionality, structured around awareness,

policy integration, research, and a proposed Soil Protection Framework Directive (DQPS) (Montanarella, 2015; European Commission, 2006). However, the legislative proposal was withdrawn in 2014, though its principles remain relevant.

In 2013, the EU launched the Climate Change Adaptation Strategy, urging Member States to implement measures to mitigate climate impacts, including desertification (European Commission, 2013). That same year, the EU adopted the General Action Programme on the Environment, aiming for sustainable land management and soil protection by 2020 (OECD, 2014). Forests also play a key role in desertification mitigation, as highlighted in the 2013 EU Forestry Strategy, which underscores sustainable forest management to prevent soil erosion and land degradation (European Commission, 2013; O'Connell et al., 2015).

Globally, the UNCCD promotes sustainable land management to combat desertification, while the 2030 Agenda for Sustainable Development includes SDG 15, which aims to restore degraded lands and achieve land degradation neutrality (UNCCD, 2018; UN, 2015; Leal Filho et al., 2019).

1.3. The situation in the Mediterranean basin

The spread of sensitive and vulnerable areas to desertification in the North-Mediterranean region—referred to as areas at risk of desertification—is attributable to several factors. As illustrated in Figure 1, these areas are increasingly affected by arid and semi-arid climatic conditions, where seasonal droughts and extreme rainfall events dominate, causing “climatic aggression” (Corona, 2005; García-Ruiz et al., 2013). The region’s complex terrain, marked by frequent steep slopes, further contributes to erosion and land degradation (Greco et al., 2005). Additionally, soils in these areas are often thin and poor, particularly those derived from calcareous and clay-sandy formations, making them highly susceptible to erosion (Edeh & Kefas, 2016).

Frequent forest fires, exacerbated by hotter and drier conditions, pose a significant threat to forest ecosystems and their ability to recover, contributing further to soil degradation (Pausas & Fernández-Muñoz, 2012). Another critical factor is the widespread abandonment of arable land, leading to the loss of traditional agricultural structures like terraces, which historically mitigated soil erosion. The resulting set-aside of land has increased vulnerability to fires and erosion (Cerdà et al., 2010). Over-

exploitation of water resources, particularly in coastal areas, has resulted in salinization of aquifers, affecting both coastal ecosystems and agricultural productivity (Greco et al., 2005; Edeh et al., 2016).

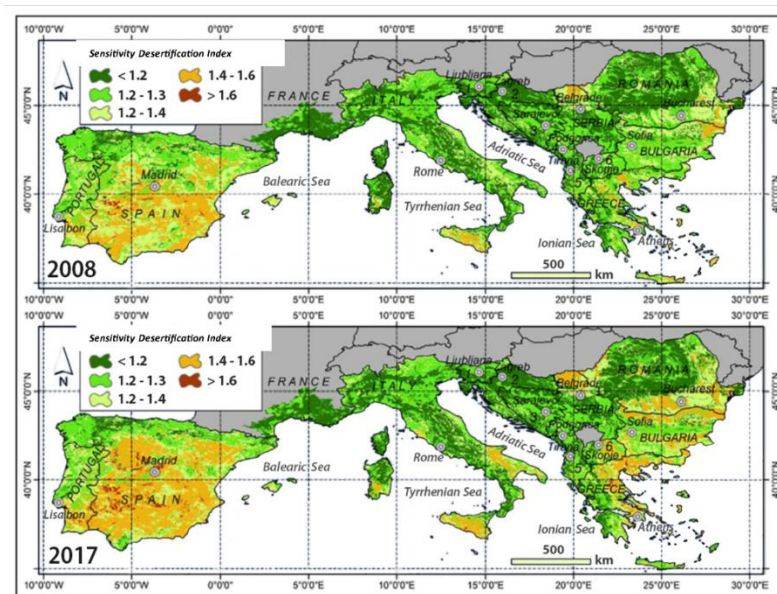


Figure 1 Sensitivity Desertification Index in the EU for 2008 and 2017
(Source Průvčalík et al., 2017.)

While the EU does not yet have a established strategy for addressing desertification, many of the contributing factors are addressed under other spending strategies and programs, particularly within the context of sustainable land management and climate adaptation (European Commission, 2020). Desertification and soil degradation are complex phenomena influenced by numerous interdependent factors. Despite the lack of consensus on how best to measure these factors, the UNCCD suggests using indirect indicators such as soil productivity, organic carbon content, land cover, and changes) soil properties to assess the extent of soil degradation (Liu et al., 2017). The various processes of soil degradation contributing to land productivity loss are visually summarized in Figure 2.

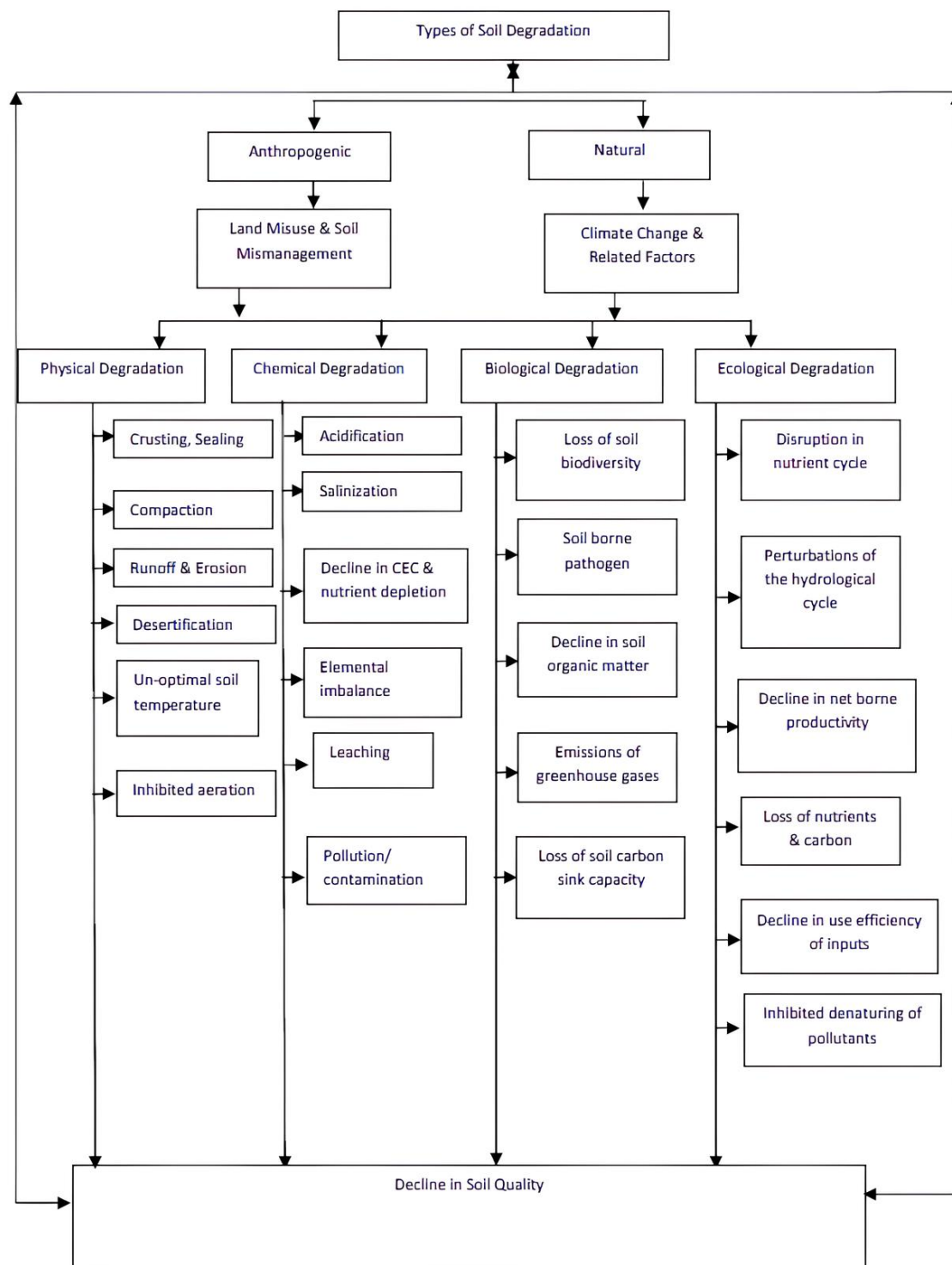


Figure 2 Various types of soil degradation
 (Source: Edeh and Kefas; IJPSS, 11(6): 1-16, 2016; Article no. IJPSS.27182)

1.4. The Importance of Soil and the Impact of Various Drivers

Soil is a vital part of terrestrial ecosystems, playing a key role in essential processes like carbon storage, nutrient cycling, water purification, and supporting plant growth (Lal, 2004; Nannipieri et

al., 2003). Beyond its ecological functions, soil serves as a dynamic resource that underpins biodiversity and helps buffer environmental changes, making it crucial for both the natural world and human well-being (Bardgett & van der Putten, 2014). Unfortunately, soil is increasingly under threat from a range of environmental and human-driven factors that influence its structure and ability to function (Montanarella et al., 2016; Smith et al., 2015; Verheijen et al., 2009).

-Climate Change

Climate change represents one of the most significant threats to soil stability, disrupting moisture levels and nutrient dynamics. Warmer temperatures speed up the decomposition of organic matter, leading to potential reductions in soil carbon stores (Davidson & Janssens, 2006). Droughts and extreme weather events, becoming more frequent with climate change, further deplete soil moisture, impair plant growth, and reduce the soil's ability to sequester carbon (Smith et al., 2008). Drought stresses soil by diminishing organic matter and microbial activity, weakening its resilience (Schimel et al., 2007). Over time, this lack of moisture can lead to desertification, which exacerbates erosion and makes soil less productive, particularly in areas prone to extended dry periods (Sivakumar, 2007). In extreme cases, these changes can flip soil from being a carbon sink to a source, worsening the effects of climate change (Dunne, 2019; Wang et al., 2014).

-Anthropogenic

Human activities like deforestation, urbanization, and intensive farming have significant effects on soil condition. Clearing forests removes vegetation that stabilizes soil, leaving it more prone to erosion and nutrient loss (Brinkmann et al., 2019; García-Ruiz et al., 2015). Urbanization seals off soil with concrete and asphalt, reducing its permeability, altering water flow, and harming biodiversity (Scalenghe & Marsan, 2009; Lorenz & Lal, 2009). Farming practices like monoculture and excessive fertilizer use deplete soil nutrients, compact its structure, and lower its organic matter, ultimately reducing its productivity and ability to store carbon (Matson et al., 1997; Pimentel et al., 2005).

-Wildfires

Wildfires severely disrupt soil ecosystems by burning away organic matter and nutrients. After a fire, soil is left bare and exposed, increasing erosion and lowering its water-holding capacity. The structural damage caused by fire makes it harder for the soil to recover and support plant regrowth, especially in fire-prone environments (CarbonBrief, 2019).

-Additional Threats: Flooding, Invasive Species, and Windstorms

Flooding can compact soil, wash away nutrients, and cause erosion, all of which weaken soil health and fertility. Invasive pests and pathogens harm native soil biodiversity, disrupt nutrient cycles, and affect plant growth. Windstorms, by stripping away vegetation cover and eroding the topsoil, leave soil more vulnerable to further degradation and desertification (Dunne, 2019; Green et al., 2021).

Together, these challenges create a complex web of pressures that threaten soil health. They often interact and amplify each other's effects, making it crucial to adopt sustainable management practices and robust environmental policies to protect soil ecosystems for the future.

-Microbiological condition

Microorganisms play a vital role in maintaining soil health. They break down organic matter, recycle nutrients, and respond quickly to changes in soil conditions. Their activity reflects the balance of factors influencing nutrient cycling and organic matter decomposition. However, measuring microbial parameters can be challenging since microorganisms are highly sensitive to seasonal variations and adapt to changing environmental conditions.

1.4.1. Soil Respiration and Fire-Induced Microclimatic Alterations

Microclimatic variations within forest ecosystems play a fundamental role in regulating biological processes, including soil respiration (SR), a key indicator of soil health, microbial activity, and carbon cycling (Davidson & Janssens, 2006).

Soil respiration represents the largest flux of CO₂ from terrestrial ecosystems to the atmosphere, originating from both autotrophic (root respiration) and heterotrophic (microbial decomposition) components (Ryan & Law, 2005). It plays a crucial role in the carbon balance of forest ecosystems, regulating the amount of carbon stored in soils versus released into the atmosphere. Any disturbance, such as wildfire, drought, or changes in land use, can significantly alter soil respiration rates, affecting ecosystem recovery, nutrient cycling, and long-term carbon sequestration (Bond-Lamberty et al., 2018).

Following a wildfire, soil respiration dynamics change due to several interconnected factors:

- Increased CO₂ flux immediately post-fire, due to rapid microbial decomposition of dead organic material and combustion residues (Certini, 2005).

- Reduced respiration rates in the medium to long term, as fire-induced soil degradation leads to loss of organic matter, microbial biomass, and soil structure, limiting microbial activity and root respiration (Dooley & Treseder, 2012).
- Altered carbon sequestration potential, as fire modifies soil organic carbon pools and their stabilization mechanisms, which can either enhance or decrease long-term carbon storage (Knicker, 2007).

Understanding how soil respiration evolves after a fire is essential for assessing forest resilience and the effectiveness of post-fire recovery strategies, such as Nature-Based Solutions (NBS) (Certini et al., 2021). The continuous monitoring of SR, alongside microclimatic factors like soil moisture and temperature, helps quantify the interactions between climate stressors, soil processes, and ecosystem carbon balance, ultimately providing critical insights into forest regeneration under climate change scenarios (Reichstein et al., 2013).

1.5. Nature-Based Solutions and the EU Restoration Law for Ecosystem Recovery

Nature-Based Solutions (NBS) provide sustainable approaches to combat land degradation, desertification, and climate change by leveraging natural processes to enhance biodiversity and ecosystem resilience (Eggermont et al., 2015; Cohen-Shacham et al., 2016). Rooted in ecological principles, NBS interventions, such as retaining burned logs and applying organic amendments, facilitate soil stabilization, reduce erosion, and support microbial recolonization, ultimately improving nutrient cycling and water retention (Lehmann et al., 2011; Pausas & Keeley, 2014). These approaches align with global sustainability targets, including the UN Sustainable Development Goals (SDG 13 and SDG 15), offering cost-effective, scalable solutions for regions prone to climatic stress, such as the Mediterranean Basin (UN, 2015; FAO, 2019; Pereira et al., 2020).

The EU Restoration Law represents a crucial step toward reversing soil and forest ecosystem degradation. This legislative framework aims to restore at least 20% of the EU's terrestrial and marine ecosystems by 2030 and achieve full ecosystem recovery by 2050 (European Commission, 2023). Given the threats posed by deforestation, wildfires, and unsustainable land-use practices, the law focuses on rehabilitating soil structure, increasing organic matter content, and restoring hydrological functions (European Environment Agency, 2020; Smith et al., 2023). Key interventions include reintroducing native plant species to prevent soil erosion and integrating sustainable forest management practices such as controlled reforestation and afforestation to enhance carbon storage and ecosystem resilience (Montanarella & Panagos, 2021; Poeplau et al., 2020). To ensure effectiveness, restoration initiatives are continuously monitored through advanced remote sensing

and in-situ assessments, allowing adaptive management strategies to respond to evolving environmental conditions (IPCC, 2022).

Effective restoration projects rely on multidisciplinary research and comprehensive methodologies that integrate environmental, economic, and social dimensions (Geist & Lambin, 2004; Sivakumar & Ndiang'ui, 2007). Identifying vulnerable areas and harmonizing methodologies are crucial to developing coherent frameworks for land degradation mitigation, yet challenges remain due to inconsistent data availability and methodological variability (Ceccarelli, 2006). The Millennium Ecosystem Assessment (2005) predicts increasing desertification trends by 2050, highlighting the need for proactive management approaches that prevent degradation rather than merely addressing its consequences (Lal, 2003; Maysoon, 2014). Understanding the underlying causes of ecosystem decline is essential to implementing targeted interventions, mitigating stress factors, and fostering long-term ecological stability (Hill et al., 2008). Despite these efforts, assessing the full impact of environmental changes remains complex due to the difficulty of acquiring complete and simultaneous data across different scales and regions.

By integrating NBS principles, the EU Restoration Law, and adaptive methodologies, restoration strategies can effectively address both the causes and effects of ecosystem degradation. These initiatives are essential for restoring ecosystem functionality, enhancing soil health, and strengthening the resilience of European landscapes against future environmental challenges.

1.6. Climate Change and Anthropogenic Drivers of Forest Degradation in the Mediterranean

The Mediterranean Basin is one of the most altered and degraded landscapes globally, primarily due to a combination of excessive logging, deforestation, and unsustainable land-use practices. Over the centuries, extensive forest exploitation, agricultural expansion, and increasing urbanization have significantly altered the natural balance of these ecosystems, leading to soil degradation, biodiversity loss, and desertification (Scarascia-Mugnozza et al., 2000; Blondel, 2006). The interaction between climate change and anthropogenic pressures has further exacerbated these impacts, necessitating a comprehensive understanding of historical forest management practices and the primary drivers contributing to the current conditions.

The Mediterranean region is widely recognized as a climate change hotspot, with rising temperatures and declining precipitation leading to significant ecological shifts. Climate projections indicate that the region is likely to experience some of the steepest reductions in rainfall globally, with expected declines ranging from 10% to 60%, depending on future climate scenarios (Cramer et al., 2018). This decline is further amplified by the region's geographical characteristics, where land heats more

rapidly than the adjacent sea, reinforcing atmospheric pressure gradients that suppress precipitation (Cook et al., 2016). Such climatic changes have already led to observable decreases in precipitation across North Africa and the Levant, intensifying water stress, accelerating soil degradation, and increasing the risk of desertification (Guiot & Cramer, 2016).

In addition to climatic factors, anthropogenic drivers, including deforestation, overgrazing, and poor land management, play a significant role in forest degradation. In Mediterranean agroforestry systems such as the Montado in Portugal, excessive grazing pressure has led to severe soil compaction, reduced vegetation cover, and a decline in biodiversity, ultimately weakening ecosystem resilience to climate variability (Pereira et al., 2020; Godinho et al., 2016). The increasing frequency and severity of wildfires further exacerbate these vulnerabilities, impairing soil structure, reducing its capacity to retain moisture and nutrients, and driving shifts in vegetation composition toward more fire-adapted shrublands (Cherlet et al., 2018).

Wildfires, particularly in semi-arid and marginal lands, act as a major catalyst of irreversible landscape transformations, contributing to desertification and biodiversity loss (Turco et al., 2017). The cumulative socio-economic and ecological consequences—such as reduced water availability, diminished agricultural productivity, and increased health risks—underscore the urgent need for adaptive and sustainable forest management strategies. Conservation-oriented approaches, including rotational grazing, sustainable water resource management, and targeted reforestation, are crucial for mitigating ongoing degradation and ensuring the long-term stability of Mediterranean forests (Godinho et al., 2016; Pereira et al., 2020).

By integrating microclimatic data with soil respiration and microbial dynamics, these findings emphasize the necessity of restoration strategies tailored to Mediterranean ecosystems. In particular, Nature-Based Solutions (NBS) have emerged as effective interventions to counteract the detrimental effects of fire and accelerate ecological recovery processes (Eggermont et al., 2015; Cohen-Shacham et al., 2016).

1.6.1. Forest fires and climate change

Wildfires are complex combustion processes that require three fundamental elements: fuel, an oxidizer, and an ignition source. In natural ecosystems, vegetation acts as the primary fuel, atmospheric oxygen serves as the oxidizer, and ignition sources include both natural factors (such as lightning) and anthropogenic activities (Keeley, 2009; Pyne, 2010). The characteristics of available fuel—including its flammability, combustibility, and moisture content—play a critical role in

determining fire behavior, along with environmental factors such as temperature and topography (Gill & Zylstra, 2005; Pausas & Keeley, 2019).

Fire spread is driven by a combination of topography, vegetation, and meteorological conditions. This is clearly visualized in Figure 3, which shows the Fire Weather Index (FWI) for 19 July 2023, highlighting high to extreme fire danger conditions across southern Europe and North Africa. Wildfires progress through distinct phases, ranging from the smoldering combustion of organic material to the active flaming of surface and canopy layers. The fire type—whether subterranean, surface, or crown—determines the ecological consequences, which may include soil heating, root combustion, and extensive vegetation loss (Rein, 2016; Scott & Reinhardt, 2001). Beyond immediate destruction, wildfires significantly alter soil properties, increase erosion susceptibility, and release substantial amounts of carbon dioxide, thereby disrupting nutrient cycling and water infiltration processes (Certini, 2005; Doerr & Shakesby, 2013).

The long-term consequences of these disturbances reshape ecosystem structure and function, promoting fire-adapted species while reducing overall biodiversity. Such shifts have profound implications for ecosystem resilience, carbon storage, and nutrient dynamics (Pausas & Keeley, 2014; Myers et al., 2000).

In recent decades, wildfires have increased in frequency and intensity across multiple regions, including the Mediterranean, the west coast of the United States, Australia, and Siberia (Bowman et al., 2020; Pausas & Keeley, 2019). In Mediterranean landscapes, fire is a natural disturbance, historically ignited by lightning or volcanic eruptions. However, in Italy, human activities have now become the primary ignition sources, including land abandonment, overgrazing, and inadequate forest management (Pausas & Keeley, 2014; Turco et al., 2017). These anthropogenic pressures, combined with prolonged droughts and rising temperatures, have altered fire regimes, increasing the frequency of high-intensity wildfires (San-Miguel-Ayanz et al., 2021).

In the Mediterranean Basin, wildfire activity has risen substantially since the 1960s, with more than 500,000 hectares affected annually across the European Union (Marchetti & Ascoli, 2018). The year 2023 saw a particularly severe wildfire season, with over 430,000 hectares burned in Europe within the first six months alone, far exceeding the decade-long average (EFFIS, 2023).

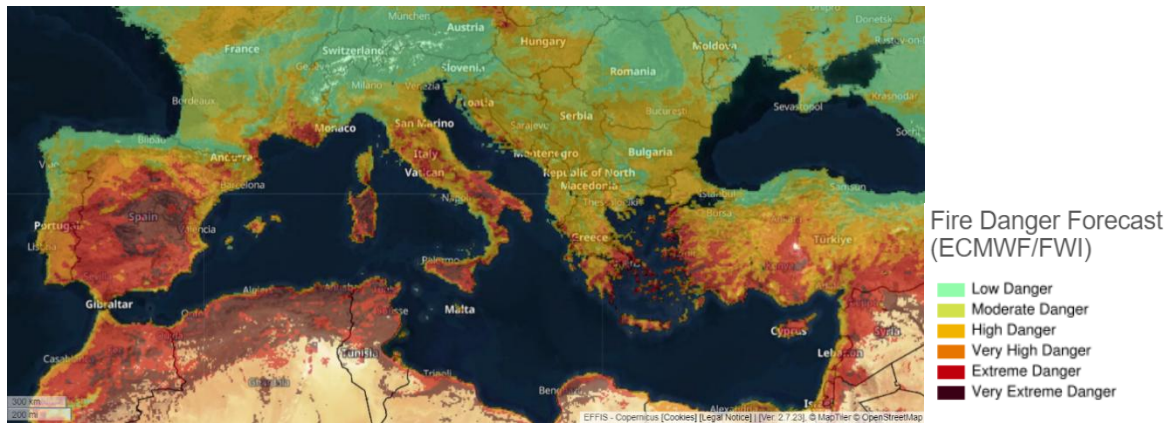


Figure 3 Fire Weather Index (FWI) fire hazard map at 8 km spatial resolution for 19 July 2023, showing high to extreme fire danger conditions across southern Europe and North Africa. Source: *EFFIS – European Forest Fire Information System*, Copernicus Emergency Management Services (<https://effis.jrc.ec.europa.eu>).

Abruzzo reflects broader wildfire trends in the Mediterranean, standing out as a hotspot for fire vulnerability in Italy. Between 2012 and 2022, the region experienced 745 wildfires, burning a total of 17,523 hectares, including 10,127 hectares of forested land. The 2017 fire season was particularly severe, with 8,576 hectares burned, accounting for more than half of the total area burned over the decade. This peak in fire activity was driven by extreme climatic conditions, including intense heatwaves and prolonged droughts (Turco et al., 2018; Moreira et al., 2020).

Table 1 Data on forest fires in Abruzzo (2012-2022). This chart shows the number of fires, and the total area burned in hectares for each year. (Source: Piano AIB Abruzzo 2023-2025)

YEAR	NUMBER OF FOREST FIRES	FORESTED AREA (ha)	NON- FORESTED AREA (ha)	TOTAL AREA (ha)	AVERAGE FORESTED AREA PER EVENT (ha)	AVERAGE TOTAL AREA PER EVENT (ha)
2012	158	672	876	1,548	4.25	9.80
2013	44	91	232	323	2.07	7.34
2014	21	31	19	50	1.48	2.38
2015	84	501	583	1,085	5.96	12.92
2016	41	110	46	156	2.68	3.80
2017	138	5,890	2,686	8,576	42.68	62.14
2018	18	81	111	192	4.50	10.67
2019	47	128	253	381	2.72	8.11
2020	62	1,299	406	1,705	20.95	27.50
2021	89	1,158	2,062	3,220	13.01	36.18
2022	43	166	121	287	3.86	6.67
TOTAL	745	10,127	7,395	17,523	-	-
Annual Average (2012- 2022)	67.73	920.64	672.27	1,593.00	13.59	23.52

A closer examination of fire trends between 2012 and 2022, as presented in Table 1, reveals considerable year-to-year variability, with 2017 marking an extreme event in terms of burned area. The annual trend highlights an increasing frequency of large wildfires, consistent with broader Mediterranean patterns.

One major factor exacerbating wildfire risk in Abruzzo is its unique topography, characterized by steep slopes and dense plantations dominated by black pine (*Pinus nigra*). While originally planted for reforestation to reduce soil erosion, over the years they have not been managed and nowadays they are a problem as they represent a source of fuel.

Fire prevention and monitoring strategies are critical, particularly in human-influenced landscapes, where urban expansion increasingly intersects with fire-prone forests. Legislative measures, such as the Forest Fire Framework Law (Law 353/2000), set regulations for fire prevention, forecasting, and suppression. However, challenges remain in coordinating fire management across agencies. For instance, the reorganization of responsibilities under Legislative Decree No. 177/2016 has sometimes led to gaps in fire suppression coordination, reducing overall efficiency (Marchetti & Ascoli, 2018). These trends in Abruzzo highlight the broader Mediterranean fire crisis, driven by climate variability, land-use changes, and socio-economic shifts. The increasing prevalence of large-scale wildfires (>500 hectares) emphasizes the urgent need for targeted interventions to mitigate fire risks and improve post-fire ecosystem recovery.

Integrating Nature-Based Solutions (NBS), such as fire-resistant reforestation, fuel breaks, and sustainable grazing management, could reduce wildfire vulnerability and enhance ecosystem resilience (Pereira et al., 2020; Cherlet et al., 2018).

Forest fires represent one of the primary disturbances affecting forest and agro-pastoral ecosystems globally, with varying degrees of severity depending on the ecosystem type, fire intensity, and recurrence frequency. Fire events reshape ecosystem structure and functionality, influencing resilience mechanisms and long-term recovery dynamics (Bovio et al., 2004; Pausas et al., 2008).

Beyond immediate vegetation loss, wildfires cause indirect ecological and socio-economic damage, including the disruption of ecosystem services such as hydrological regulation, carbon sequestration, biodiversity conservation, and soil stabilization (Moreira et al., 2020). In human-dominated landscapes, particularly in Mediterranean Europe, wildfires threaten agricultural systems, infrastructure, and rural communities, leading to substantial economic losses.

The direct effect of fire on vegetation is the destruction of biomass, with ecosystem recovery largely dependent on fire intensity, plant functional traits, and soil properties (Keeley et al., 2012). Heat stress during fires can lead to irreversible physiological damage in trees, particularly when cell temperatures exceed 44°C for one hour or 48°C for several minutes (Stephens et al., 2002).

Plant response to fire varies widely depending on species-specific adaptations. Passive pyrophytes—such as *Quercus suber* (cork oak)—develop thick, insulating bark to withstand high temperatures (Pausas & Keeley, 2014). In contrast, active pyrophytes—including *Pinus halepensis*, *Pinus pinaster*, *Cistus spp.*, and *Thymus capitatus*—exhibit fire-stimulated germination or vegetative resprouting mechanisms (Keeley et al., 2012).

Fire alters soil chemical, physical, and biological properties, influencing long-term fertility and resilience (Bovio et al., 2014; Certini, 2005). The combustion of organic matter accelerates mineralization, releasing nutrients such as nitrogen and phosphorus but also increasing the risk of post-fire leaching and erosion (González-Pérez et al., 2004).

The depth of fire impact varies with intensity. Low-intensity surface fires typically affect only the upper 15 cm of soil, while crown or barrier fires penetrate deeper, causing more profound changes in soil structure and microbiota (Certini, 2005). Hydrophobicity induced by high temperatures exacerbates water runoff and erosion (Letey, 2001; Doerr & Shakesby, 2013).

Wildfires impact fauna through direct mortality during combustion and indirect effects linked to habitat loss and food scarcity (Smith et al., 2018). The severity of these impacts depends on species mobility, fire intensity, and post-fire habitat conditions. Species with low mobility or those dependent on dense vegetation (e.g., ground-nesting birds, amphibians, and small mammals) are particularly vulnerable (Moreira et al., 2020).

1.6.2. Impact of grazing on degraded agroforestry system

Mediterranean agroforestry systems are characterized by the integration of tree cover, pasturelands, and livestock grazing. While well-managed grazing plays a vital role in biodiversity conservation, wildfire prevention, and nutrient cycling, overgrazing remains a primary driver of land degradation in many regions (Pereira et al., 2020; Blondel, 2010).

Overgrazing has been identified as a key factor in land degradation, indeed continuous livestock trampling compacts the soil, reducing infiltration rates and water retention capacity, making the land more susceptible to drought (Godinho et al., 2016). Moreover, overgrazing depletes perennial grass and shrub cover, limiting natural regeneration of native species (Otero & Boada, 2021), and leads to the selective removal of palatable species, altering species composition and reducing overall ecosystem complexity (Pereira et al., 2020). The reduction of vegetation cover exposes the soil surface, leading to higher temperatures and drier conditions, which increase wildfire susceptibility (Moreira et al., 2020).

The combined effects of overgrazing and climate change accelerate desertification processes, leading to declines in primary productivity and ecosystem resilience (Cherlet et al., 2018).

These impacts are particularly evident in Mediterranean agroforestry systems, where traditional land-use practices have shaped landscapes for centuries. The Portuguese Montado system, representing a typical agro-silvo-pastoral ecosystem of the Mediterranean basin, exemplifies both the benefits and challenges of livestock grazing. Characterized by scattered oak woodlands (*Quercus suber* and *Quercus rotundifolia*) interspersed with pastures, the Montado balances tree cover, grazing, and agricultural activities. However, in recent decades, climate variability, land abandonment, and intensive grazing practices have led to profound structural changes, exacerbating soil degradation and threatening long-term ecosystem stability (Pereira et al., 2020).

As a result, many regions now exhibit severe signs of degradation, requiring adaptive grazing management strategies to prevent further loss of ecosystem functionality.

To mitigate the impact of grazing and enhance ecosystem resilience, several adaptive land management approaches have been proposed. One effective strategy is rotational grazing, which involves alternating grazing areas to allow vegetation recovery and prevent soil degradation (Pereira et al., 2020). By limiting the time livestock spend in a particular area, this practice reduces soil compaction, preserves root systems, and enhances plant regrowth, ultimately maintaining soil structure and fertility.

Another crucial measure is maintaining optimal stocking rates, ensuring that livestock density does not exceed the ecosystem's carrying capacity (Godinho et al., 2016). Overgrazing exerts excessive pressure on vegetation and soil, leading to erosion, nutrient depletion, and biodiversity loss. By adjusting stocking rates based on seasonal forage availability and ecosystem conditions, land managers can prevent long-term degradation and promote sustainable pasture productivity.

Additionally, promoting silvo-pastoral systems has been identified as a key approach to improving ecosystem stability. This strategy integrates tree planting within grazing areas to enhance soil stabilization, increase carbon sequestration, and provide additional forage and shelter for livestock (Otero & Boada, 2021). The presence of trees not only reduces soil erosion but also moderates microclimatic conditions, offering shade that helps mitigate the impacts of extreme temperatures on both animals and vegetation.

While grazing remains a culturally and economically significant practice, unsustainable management accelerates land degradation and biodiversity loss. Implementing evidence-based grazing policies and adaptive management strategies is essential to restoring agroforestry landscapes in Portugal and across the Mediterranean Basin (Cherlet et al., 2018). Through a combination of responsible grazing practices, ecological restoration efforts, and long-term planning, it is possible to balance livestock production with environmental conservation, ensuring the resilience and sustainability of these valuable landscapes.

2 PhD Thesis Objectives and Methodology Rationale

Mediterranean ecosystems encompass a diverse range of landscapes, including forests, shrublands, and agro-silvo-pastoral systems, all of which are increasingly vulnerable to degradation due to a combination of climatic stressors and anthropogenic pressures. These ecosystems are characterized by hot, dry summers and mild, wet winters, with vegetation adapted to withstand seasonal droughts and fire disturbances (Blondel et al., 2010; Pausas & Keeley, 2014). Forested Mediterranean landscapes include broadleaved evergreen forests dominated by *Quercus ilex* and *Quercus suber*, mixed with deciduous oak and pine species, while agro-silvo-pastoral systems, such as the Montado in Portugal, integrate tree cover with extensive livestock grazing and cereal cultivation (Pinto-Correia & Mascarenhas, 1999).

In this context, degradation and desertification are dynamic and multifaceted processes influenced by multiple environmental factors, including wildfires, prolonged droughts, overgrazing, and land-use changes (Cherlet et al., 2018; Geist & Lambin, 2004). These disturbances alter soil structure, reduce vegetation cover, and disrupt biogeochemical cycles, leading to a decline in ecosystem functionality and productivity (Certini, 2005; Lal, 2009). Understanding how different Mediterranean land-use types respond to these stressors is crucial for developing sustainable management strategies that mitigate degradation risks while enhancing ecosystem resilience.

This research aims to analyze two major drivers of land degradation—wildfire and overgrazing—and their impacts in two Mediterranean landscapes: Monte Morrone, a fire-affected forest in Majella National Park (Italy), and Companhia das Lezírias, an agroforestry system in Portugal impacted by grazing. Given the ecological differences between these two sites, the research objectives were distinct and tailored to each system.

In the fire-affected site (Monte Morrone), the study investigates the consequences of wildfire on soil properties, microbial communities, and CO₂ fluxes. Wildfires drastically alter soil structure, reducing organic matter content and modifying microbial communities, leading to shifts in decomposition rates and nutrient cycling (Certini, 2005; Pausas & Keeley, 2014). Another key hypothesis is that microbial functional redundancy—the presence of taxa with overlapping roles in nutrient cycling and organic matter decomposition—can buffer ecosystem processes against fire-induced stress (Lucas-Borja et al., 2016). To test this, we will evaluate how microbial communities shift in response to fire disturbance and whether specific functional groups contribute to soil recovery.

Then, the retention of burned logs as a Nature-Based Solution (NBS) is tested to assess its effectiveness in promoting post-fire recovery by stabilizing soil structure, reducing erosion, and

enhancing microbial recolonization (Pereira et al., 2020; Godinho et al., 2016). The research integrates field data, laboratory analyses, and remote sensing techniques to monitor the degradation and recovery processes (Lucas-Borja et al., 2016; Shen et al., 2016).

In the grazed site (Companhia das Lezírias), we hypothesize that overgrazing leads to soil compaction, which lead to lower water retention capacity, reduces porosity and oxygen availability, consequently decreasing soil respiration and microbial-mediated nutrient cycling (Rey et al., 2011; Cherlet et al., 2018). Hence, the study focuses on the impact of overgrazing on soil CO₂ fluxes and nutrient availability. The findings will inform management strategies for sustainable grazing practices aimed at minimizing degradation while maintaining ecosystem functionality (Godinho et al., 2016).

3 Material and Methods

3.1. Description and Significance of the Study Sites

The study sites include ecologically and socio-environmentally significant areas that provide valuable evidence on land degradation (Cherlet et al., 2018; Geist & Lambin, 2004; Moreira et al., 2020).: Monte Morrone in the Majella National Park, Italy, and Companhia das Lezírias in Portugal were the study sites. These sites represent different types of Mediterranean landscapes affected by distinct environmental stressors—wildfires in Monte Morrone and grazing pressure in Companhia das Lezírias.

However, due to logistical and financial constraints, different methodological approaches were applied at each site. At Monte Morrone, a comprehensive monitoring framework was implemented, including:

- Soil respiration measurements
- Soil physical-chemical properties analysis
- Microbial DNA analysis
- Remote sensing data collection
- Climate monitoring

In contrast, at Companhia das Lezírias, research focused primarily on soil respiration monitoring and chemical-physical soil analysis to assess grazing-induced stressors. Remote sensing data, climate datasets, and microbial DNA analyses were not included in the Portuguese site due to resource limitations.

Despite methodological differences, the two sites offer complementary insights into how wildfire and grazing stressors impact Mediterranean ecosystems. A visual overview of the sites is provided in Figure 4 (Monte Morrone) and Figure 5 (Companhia das Lezírias). A comparative summary is provided in Table 2.

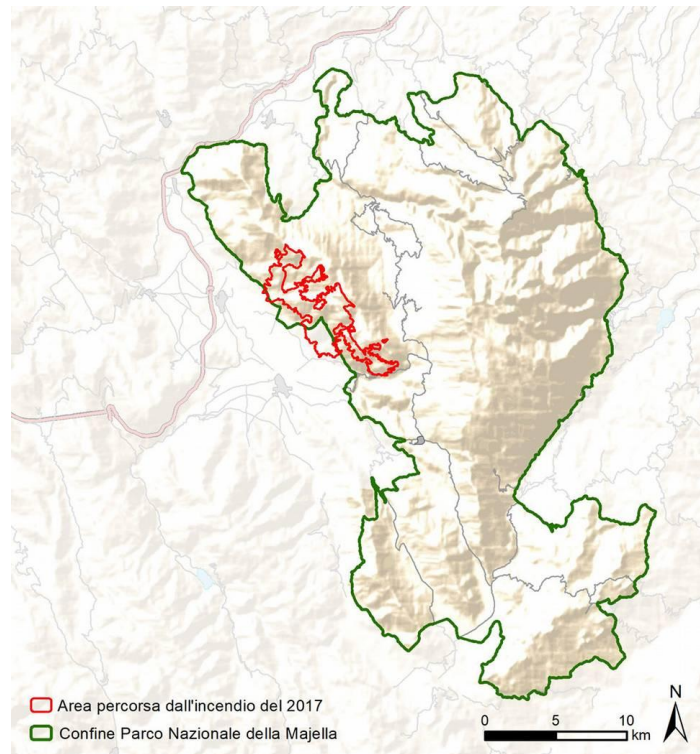
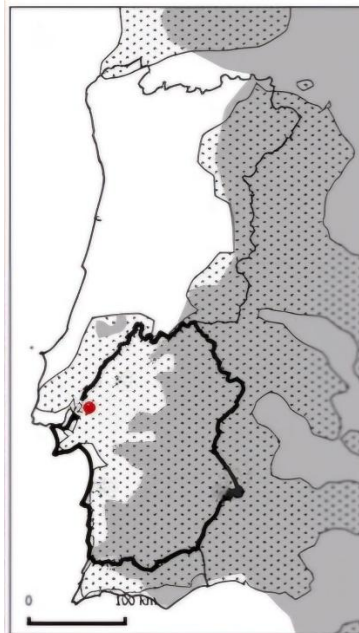


Figure 4 Study site in Monte Morrone, Majella National Park, central Italy. The area was impacted by large wildfires in 2017 and 2023.

■ Burned area from the 2017 wildfire

■ Majella National Park boundary



■ Black boundary: Montado region boundary

■ Red dot: LTER site in Portugal

Figure 5 Location of the Companhia das Lezírias site, near Lisbon, Portugal, within a Montado agro-silvo-pastoral system. (Deosaran et al., 2024)

Table 2 Comparative overview of the study sites, including coordinates, altitude, climate, and key environmental stressors.

<i>Site</i>	<i>Coordinates</i>	<i>Altitude (m)</i>	<i>Mean Temperature (°C)</i>	<i>Annual Precipitation (mm)</i>	<i>Environmental Stressors</i>
Monte Morrone (Italy)	42.1°N, 13.9°E	650	11.7	988	Wildfire, drought
Companhia das Lezírias (Portugal)	38.9°N, 8.9°W	53	16.3	662	Grazing pressure, drought

3.1.1. Monte Morrone – Parco Nazione della Majella

Monte Morrone is located within the Majella National Park, in the Abruzzo Apennines (central Italy). This mountain range, extending between the Peligna Valley, the Aterno River, and the Maiella massif, is composed mainly of limestone and dolomitic formations, with calcareous-clayey soils overlaying Cretaceous rocks. The highest peak, Monte Morrone (2061 m a.s.l.), defines the region's ecological and geomorphological features.

The site exhibits a Csb Mediterranean climate (Köppen classification), with dry, hot summers and wet, mild winters (Zohner et al., 2016). As elevation increases, the climate transitions to temperate montane conditions, influencing vegetation structure and soil dynamics.

The climate diagram for Monte Morrone (Figure 6) illustrates this variability, highlighting seasonal droughts and humid periods during the year 2023. The site received 988 mm of rainfall, with an annual mean temperature of 11.7 °C. The highest rainfall occurs in autumn and winter (Majella National Park, 2015). The summer months, particularly July and August, are characterized by low precipitation levels, leading to drought conditions. Summer droughts significantly affect vegetation, with increasing evidence that extreme weather events such as heatwaves and dry spells further stress forest ecosystems.

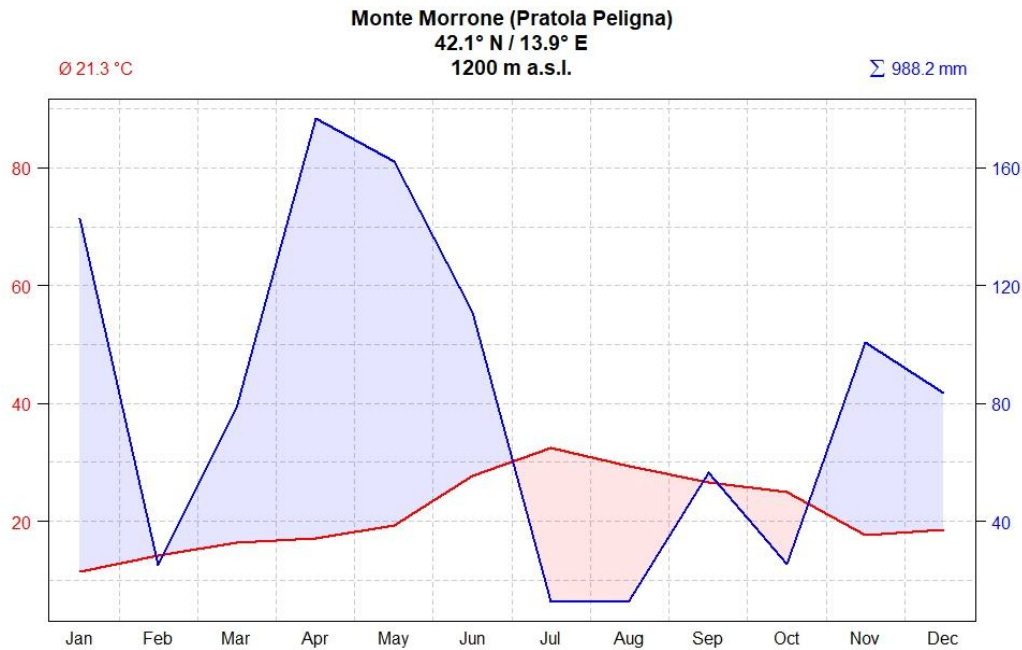


Figure 6 Walter-Lieth climate diagram (2023) for Monte Morrone (Pratola Peligna, 42.1° N, 13.9° E, 1200 m a.s.l.). Red line: mean monthly temperature (°C); blue bars: monthly precipitation (mm). Blue-shaded areas indicate humid periods; red-shaded areas indicate arid periods.

Vegetation Composition and Forest Structure

Monte Morrone's vegetation is stratified into four distinct phytoclimatic belts:

- **Mediterranean Zone (<500 meters):** Dominated by coniferous plantations of *Pinus nigra* (black pine) and *Pinus halepensis* (Aleppo pine), with small patches of holm oak (*Quercus ilex*). The flora here is adapted to dry conditions, with scrublands and low vegetation.
- **Sub-Mediterranean Zone (500–1000 meters):** Primarily composed of deciduous oaks (*Quercus pubescens*), maples (*Acer spp.*), ash trees (*Fraxinus ornus*), and hop-hornbeam (*Ostrya carpinifolia*), interspersed with black pine reforestation areas.
- **Montane Zone (900–1600 meters):** A mesophilic forest dominated by European beech (*Fagus sylvatica*), hornbeam (*Carpinus betulus*), and oak (*Quercus spp.*), hosting a variety of shrubs and herbaceous plants under more favorable moisture conditions.
- **Subalpine Zone (>1700 meters):** Characterized by beech (*Fagus sylvatica*) forests interspersed with meadows and patches of juniper (*Juniperus communis*). This zone experiences harsh winters, limiting tree growth.

Invasive species such as *Robinia pseudoacacia* and *Ailanthus altissima* have been observed in lower elevation disturbed areas, often taking advantage of post-fire conditions.

The overall forest structure in Monte Morrone is shaped by both natural forest dynamics and human intervention. In the lower altitudes, where the monitoring plot were established, particularly around

the Mediterranean and Sub-Mediterranean zones, large areas have been subject to reforestation efforts, particularly with black pine (*Pinus nigra*), which has become widespread since the early 20th century (Blasi et al., 2013). These monoculture plantations are characterized by high tree density and reduced structural complexity, leading to low plant species richness and limited vertical stratification in the understory. The consequences of this forest management style have been observed in terms of increased susceptibility to disturbances, such as fire and drought, which stress the relatively homogenous forest structure (Pausas & Keeley, 2014).

In contrast, the higher altitudes host a more biodiverse range of species, with native hardwood forests (*Quercus spp.*, *Carpinus betulus*, *Fagus sylvatica*) that provide a rich habitat for a variety of fauna (Blasi et al., 2013). However, these areas too have been impacted by human activities, particularly overgrazing and logging, which have altered the regeneration cycles of native species.

Wildfire Events in the Majella National Park

Monte Morrone was severely affected by wildfires in 2017 and 2023. The 2017 event burned over 2,000 hectares, primarily conifer forests and grasslands, causing extensive vegetation loss and changes to soil structure (Figure 7).



Figure 7 Burned landscape at Monte Morrone following the 2017 wildfire, illustrating extensive vegetation loss and surface exposure.

Five years later, vegetation recovery was observed (Figure 8), with early successional shrub and herbaceous species colonizing the site. However, the 2023 wildfire (approx. 400 ha burned) re-

affected the same areas, reversing part of the ecological recovery and increasing stress on soil and plant communities (Majella National Park, 2024) (Figure 9).



Figure 8 egeneration in the monitoring area five years after the 2017 fire, with dominant presence of pioneer shrubs and herbaceous species.



Figure 9 Effects of the 2023 wildfire on the same monitoring site, showing re-burned plots with substantial vegetation loss and delayed regrowth.

3.1.2. Companhia das Lezírias – Lisbon, Portugal

Companhia das Lezírias (CL) is one of six Long-Term Ecological Research (*LTER*) sites within the Montado region, a distinctive agro-silvo-pastoral landscape in Portugal. Located in the municipality of Benavente, approximately 50 km from Lisbon, CL spans 20,000 hectares across Benavente, Vila Franca de Xira, and Salvaterra de Magos. The area includes distinct regions such as the Lezíria of Vila Franca de Xira, Pauis, and Charneca do Infantado, where much of this research was conducted. CL follows sustainable land management principles, integrating forestry, livestock farming, viticulture, and tourism. Managed since 1989 by Companhia das Lezírias, S.A., a publicly owned company, it adheres to international standards for sustainable forest management, including certification by the Forest Stewardship Council (FSC) (Companhia das Lezírias, n.d.).

Climate and Soil Characteristics

CL experiences a Mediterranean climate with hot, dry summers and mild, wet winters. The average annual temperature is 16.3 °C, with total annual precipitation of 662 mm, concentrated between November and February (Companhia das Lezírias, n.d.), showing seasonal droughts and moderate winter precipitation, which strongly influence soil water dynamics and vegetation activity (Figure 10). The site is located on deep Miocene sandy deposits, with prevalent regosoils and podzols, and low elevation (1–53 m a.s.l.) with gentle slopes.

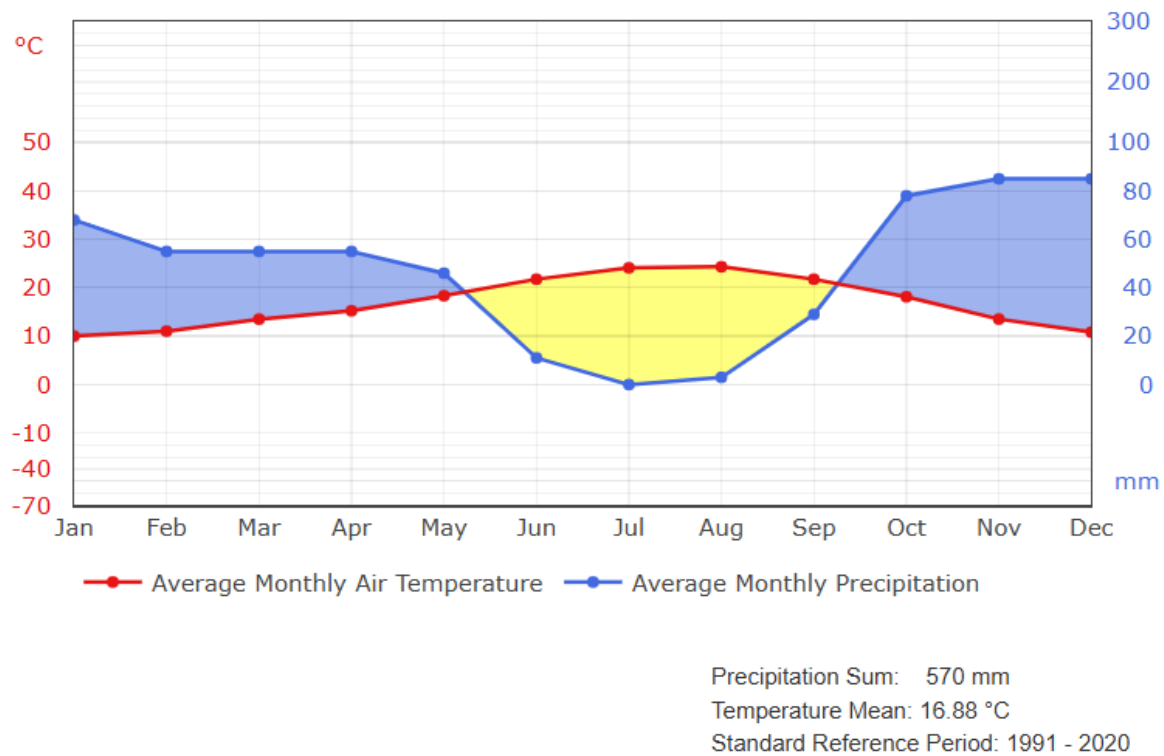


Figure 10 Monthly climate data for Companhia das Lezírias.
 Red line represents air temperature (°C, left y-axis); blue bars show precipitation (mm, right y-axis).
 (Aubard et al., 2019.)

Vegetation Composition and Land Use

At CL (Figure 11-13), forest cover is dominated by cork oak (*Quercus suber*) and holm oak (*Quercus ilex*), which account for 78% of the forested area (~6,725 ha). Other species include maritime pine (*Pinus pinaster*), stone pine (*Pinus pinea*), and eucalyptus (*Eucalyptus globulus*), which collectively cover ~1,900 ha.

Cork production represents a key economic activity, with an average yield of 986 kg/ha every nine years. Livestock grazing is also widespread, with an average stocking rate of 0.47 animal units/ha, integrated with rice farming, viticulture, and tourism (Moreira et al., 2020).

The main environmental threats at CL include:

- Prolonged droughts, which affect tree regeneration and soil moisture.
- Grazing pressure, which inhibits oak recruitment and increases soil compaction.
- Podzol soils, which have limited water retention capacity, exacerbating drought effects (Pausas & Keeley, 2019).



Figure 11 Figure 11 Degraded plot at Companhia das Lezírias under active grazing pressure. Vegetation cover is sparse and compacted due to livestock trampling.



Figure 12 Control plot in summer at Companhia das Lezírias, dominated by mature cork oak woodland with dense shrub and herbaceous understory.



Figure 13 Same control plot during winter, illustrating seasonal variation in vegetation structure and cover.

3.2. Environmental and Microclimate Monitoring in a Degraded Mediterranean Forest Ecosystem

3.2.1. Meteorological Data Collection

Meteorological data in the selected experimental areas (Monte Morrone) were collected using fixed weather stations equipped with ECH2O EC-5 sensors (Decagon Device Instruments) to monitor key environmental variables relevant to post-fire recovery. The measured parameters included:

- Air temperature and humidity
- Soil temperature and humidity
- Precipitation (via a pluviometer with 0.2 mm resolution)

Two weather stations were installed, one in the burned site and one in the control site, to enable comparative analysis of meteorological conditions and their influence on soil temperature, moisture, and respiration rates. The stations remained in fixed positions throughout the entire monitoring period (January 2022 – September 2024).

The selection of installation sites followed specific criteria to ensure comparability between burned and control areas.

- In the burned site, the weather station was randomly placed but within a plot that shared similar soil characteristics, vegetation type, slope inclination, and exposure with the surrounding experimental area.
- In the control site, the station was installed in the most comparable location possible, ensuring that soil properties and vegetation cover resembled those of the burned area before the fire.

The two weather stations were positioned at a distance of approximately 1.57 km from each other, allowing for an accurate assessment of microclimatic differences between burned and unburned conditions.

Data were recorded every 15 minutes and stored in the ECH2O system, allowing for direct field downloads. The weather stations included:

- EHT RH/Temperature sensor
- 5TM Soil Moisture/Temperature sensors
- ECRN-100 Precipitation sensor

Continuous environmental monitoring provided essential insights into ecosystem recovery dynamics, particularly regarding soil moisture and temperature variations, which strongly influence plant regeneration, microbial activity, and nutrient cycling (Gregoretti & Dalla Fontana, 2007).

3.3. Satellite Remote Sensing in Monitoring the Regeneration of Natural Vegetation in Fire-Affected Areas

Satellite remote sensing provides a powerful tool to assess vegetation changes after wildfires. In this study, Sentinel-2 satellite images were used to map burned areas and monitor post-fire vegetation recovery in Monte Morrone, Italy.

3.3.1. Spectral index processing and burned area mapping methodology

The satellite data were sourced from the Copernicus Sentinel-2 constellation and downloaded from the CDSE catalogue. All images were processed using the MACCS-ATCOR Joint Algorithm (MAJA; Lonjou et al., 2016) to apply atmospheric correction, orthorectification, and terrain flattening. Images were resampled to 20 m resolution, and topographic shadows were masked using sun angle and slope criteria to improve classification accuracy.

Five spectral indices (SIs) were calculated to detect burned areas, covering visible (VIS), red-edge, near-infrared (NIR), and short-wave infrared (SWIR) wavelengths. To ensure that all indices increased in value with fire impact, the formulas for the NDVI and NBR were adapted accordingly. The selected indices included:

- **NDVI (Normalized Difference Vegetation Index):** Represents vegetation health. Higher values indicate healthier vegetation (Yan et al., 2024; USGS, 2021).

$$NDVI = \frac{(NIR - RED)}{(NIR + RED)}$$

Sentinel-2 bands used: B8A (NIR) and B4 (Red).

- **NBR (Normalized Burn Ratio):** Sensitive to fire impact and effective in assessing burn severity (San-Miguel-Ayanz et al., 2019).

$$NBR = \frac{(NIR - SWIR)}{(NIR + SWIR)}$$

Sentinel-2 bands used: B8A (NIR) and B12 (SWIR).

Change detection was performed by computing the difference between post-fire and pre-fire images:

- $dNDVI = NDVI_{post} - NDVI_{pre}$
- $dNBR = NBR_{post} - NBR_{pre}$

These difference indices were used to map burned areas through thresholding. The optimal thresholds were selected using a spectral separability analysis based on the separability index (M):

Equation 1:

$$M = \frac{\mu_b - \mu_u}{\sigma_b + \sigma_u}$$

(1)

Where:

- μ_b and μ_u : mean values of burned and unburned pixels,
- σ_b and σ_u : standard deviations.

M values > 1 indicate good separability.

Empirical thresholds were derived from polygons defined on the most effective index (i.e., with the highest M). Regression equations were then used to derive thresholds for the other indices.

To combine binary classifications from different indices, an Agreement Index (AIX) was computed:

$$\text{Equation 2: } AIX = \sum SI(P) / N$$

Where:

- $SI(P)$ = classification result for pixel P from each spectral index (1 = burned, 0 = unburned),
- N = number of indices used.

The final burned area map was obtained by selecting the combination of indices (n out of N) that optimized overall classification accuracy and minimized omission and commission errors.

To evaluate performance, an Agreement Index Score (AIS) was calculated:

$$\text{Equation 3: } AIS = ((1 - OE) + (1 - CE)) \times OA$$

Where:

- OE = omission error,
- CE = commission error,
- OA = overall accuracy.

The workflow for the full processing and classification pipeline is illustrated in Figure 14, which shows the sequence from data pre-processing to burned area identification.

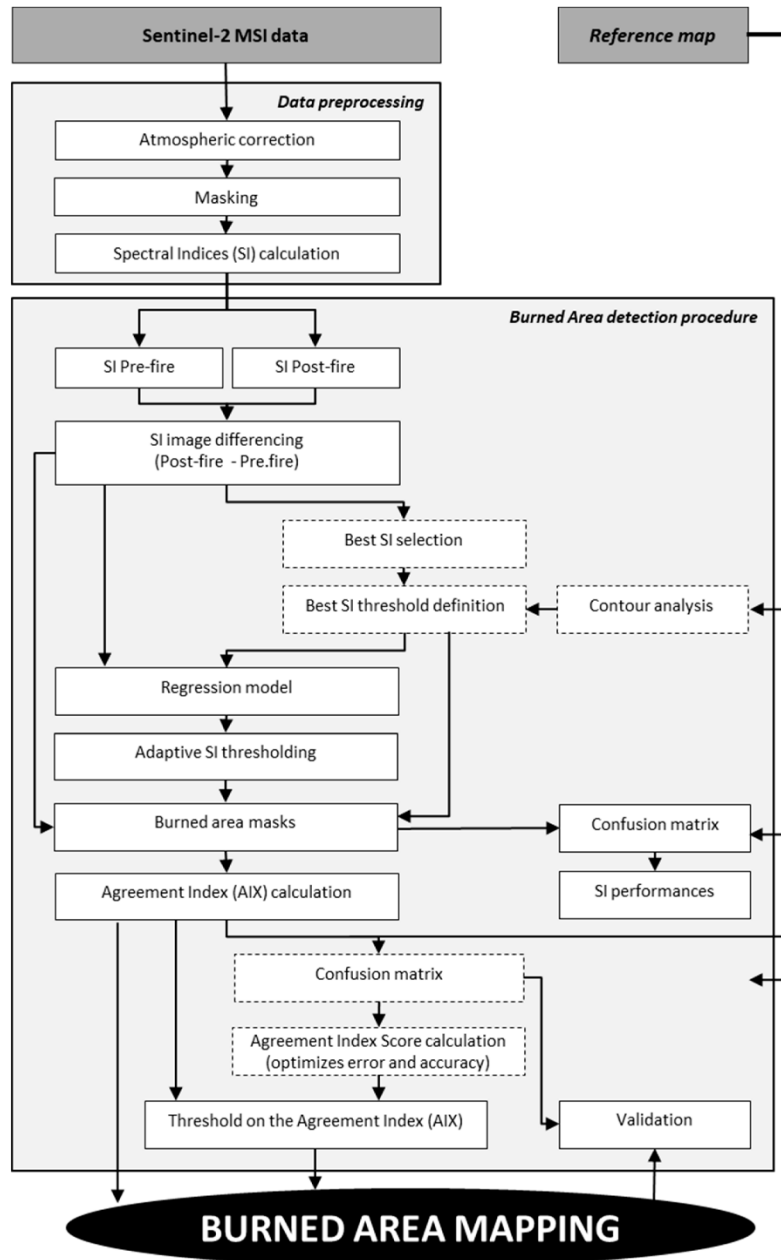


Figure 14 Workflow of the main phases of the procedure for burned area mapping using multiple spectral indices.

Adapted from Samiraglia et al. (2020).

3.3.2. Post-Fire Vegetation Monitoring and LAI Trend Analysis

Time series of NDVI, NBR, and Leaf Area Index (LAI) were generated to monitor vegetation dynamics following the 2017 wildfire. A burn severity assessment was performed using the dNBR values, classified according to USGS thresholds (USGS, 2021; Yan et al., 2024):

- Very Low: < 0.1
- Low: $0.1\text{--}0.27$

- Moderate: 0.27–0.44
- High: 0.44–0.66
- Very High: > 0.66

CLC 2018 land cover data were used to distinguish between vegetation types (e.g., coniferous and broad-leaved forests) and elevation bands (e.g., 300–600 m, 600–1100 m, >1100 m) within the burned area.

To evaluate regeneration patterns, areas affected exclusively by the 2017 fire were isolated by subtracting the 2023 burned layer. Within this zone, changes in LAI values from 2016 to 2023 were analyzed using linear regression to determine the direction of vegetation change:

- Positive slope: vegetation regrowth
- Negative slope: degradation or loss

This analysis allowed the classification of post-fire pixels into three main categories:

- Regrowth (positive trend)
- Burned (no recovery or negative trend)
- Unburned (reference)

The spectral and temporal analyses provided insight into both fire damage severity and ecosystem resilience over time, supporting the interpretation of recovery dynamics in Mediterranean forest systems.

3.4. Study Sites and Experimental Setup

3.4.3. Monte Morrone Site

In the Monte Morrone site, to assess these effects, nine experimental plots were established, classified into three categories, covering a range of burned and unburned conditions:

- **Burned plots without Nature-Based Solutions (NBS) interventions:**
 - **Plot 1C:** 42°06'01.71"N, 13°55'23.40"E, elevation 551 m a.s.l.
 - **Plot 2C:** 42°06'02.38"N, 13°55'25.84"E, elevation 578 m a.s.l.
 - **Plot 3C:** 42°06'00.47"N, 13°55'25.84"E, elevation 554 m a.s.l.
- **Burned plots with NBS interventions:**
 - **Plot 1A:** 42°06'01.37"N, 13°55'23.47"E, elevation 550 m a.s.l.
 - **Plot 2A:** 42°06'02.19"N, 13°55'25.94"E, elevation 576 m a.s.l.
 - **Plot 3A:** 42°06'00.28"N, 13°55'25.87"E, elevation 553 m a.s.l.
- **Control plots (unburned reference sites):**
 - **Plot 4C:** 42°06'34.05"N, 13°54'38.39"E, elevation 544 m a.s.l.
 - **Plot 5C:** 42°06'35.54"N, 13°54'35.27"E, elevation 555 m a.s.l.
 - **Plot 6C:** 42°06'37.75"N, 13°54'33.00"E, elevation 571 m a.s.l.

Three experimental plot types were established at Monte Morrone (Figure 15- 17): burned without NBS, burned with NBS, and unburned control plots. Burned plots with NBS interventions had three deadwood logs placed by the Majella National Park in 2017, positioned along the slope contours to mitigate erosion and enhance ecosystem recovery. Soil respiration collars were placed at the central log in each NBS plot, ensuring that measurements captured microclimatic and microbial activity differences. To minimize the impact of environmental condition between NBS-treated and untreated burned plots, collars in the NBS plots were positioned at 2–3 meters from their non-NBS counterparts. Each plot was 10 m × 10 m, ensuring a standardized area for measurement. The study design ensured that all plots had similar slope inclination, vegetation type, and exposure. Each plot contained three PVC collars, providing three replicates per treatment.



Figure 15 Burned plot without Nature-Based Solutions (NBS) interventions at Monte Morrone. The plot exhibits sparse post-fire vegetation recovery and exposed soil surfaces.



Figure 16 Burned plot with NBS interventions at Monte Morrone. Deadwood was placed along the slope by forest authorities to reduce erosion and enhance soil recovery.



Figure 17 Control plot at Monte Morrone, showing undisturbed vegetation and intact soil structure in an area unaffected by wildfire.

3.4.4. Companhia das Lezírias Site

The Companhia das Lezírias (CL) site, located in Portugal, was selected to investigate the effects of drought and grazing on soil respiration in a Mediterranean agro-silvo-pastoral system. This site experiences high seasonal variability in precipitation, making it an ideal location for assessing how climate variability and land management influence Soil CO₂ Efflux dynamics.

Six experimental plots were selected, categorized as:

- **Degraded plots (continuous grazing, no tree cover, high compaction):**
 - **Plot P1:** 38°49'39.06"N, 8°49'49.67"W, elevation 14 m a.s.l.
 - **Plot P2:** 38°49'26.24"N, 8°47'58.42"W, elevation 50 m a.s.l.
 - **Plot P3:** 38°49'39.00"N, 8°49'34.00"W, elevation 24 m a.s.l.
- **Control plots (cork oak forest, no grazing, higher organic input):**
 - **Plot P4:** 38°49'54.84"N, 8°49'33.11"W, elevation 24 m a.s.l.
 - **Plot P5:** 38°49'45.33"N, 8°49'50.45"W, elevation 13 m a.s.l.
 - **Plot P6:** 38°49'37.00"N, 8°49'33.00"W, elevation 13 m a.s.l.

Each plot was 10 m × 10 m, ensuring a representative area for soil respiration monitoring.

At this site, control plots were established beneath the canopy of cork oak trees, following a triangular collar arrangement similar to Monte Morrone (Figure 18-19). In degraded plots, where grazing was continuous, vegetation cover was homogeneous. The site was flat (no slope), ensuring uniformity across treatments and minimizing confounding effects from erosion or runoff.



Figure 18 Control plot at Companhia das Lezírias during summer. The site is characterized by cork oak canopy cover, litter accumulation, and stable microclimatic and soil conditions.



Figure 19 Degraded plot at Companhia das Lezírias during summer. The photo shows low vegetation cover, compacted soil, and drought-exposed conditions due to continuous grazing.

3.5. Soil CO₂ Efflux

The installation of PVC collars was a crucial step in preparing the sites for soil respiration measurements. Each collar measured 10 cm in diameter and was inserted to a depth of approximately 5 cm into the soil, taking care not to disturb the root systems or soil structure, which could affect respiration rates. Collars were installed at least 2 weeks prior to the first SR measurements, allowing the soil to equilibrate and minimize disturbance-induced variability (Dalias et al. 2018; Heinemeyer et al. 2011).

The randomized placement of collars within each plot ensured that spatial variability was captured, allowing for robust comparisons between treatments. This was particularly important in the degraded plots at Companhia das Lezírias, where drought-induced patchiness in soil conditions required careful consideration (Uribe et al. 2020).

Soil CO₂ Efflux (SR) and CO₂ flux were measured using the PP Systems EGM-4 infrared gas analyzer (IRGA), equipped with the SRC-1 closed chamber (Figure 20 -21), a widely used instrument for real-time monitoring of soil CO₂ flux in ecological and biogeochemical studies (López-Serrano et al., 2018; Heinemeyer et al., 2011). To ensure accurate and repeatable measurements, PVC collars (10 cm in diameter, 5 cm deep) were installed at each experimental site at least two weeks prior to the first data collection to minimize soil disturbance and potential alterations in CO₂ emissions (Dalias et al., 2018; Heinemeyer et al., 2011).



Figure 20 PVC collars (10 cm diameter, 5 cm depth) installed at the study sites to measure soil CO₂ Efflux using the closed-chamber method.
Collars were left in place for at least two weeks before data collection to allow soil stabilization.



Figure 21 Portable infrared gas analyzer (EGM-4, PP Systems) connected to SRC-1 closed chamber, used to measure soil CO₂ Efflux at each collar.

Measurements were conducted under closed-chamber conditions, with each acquisition lasting 2 to 3 minutes. During this interval, the chamber was positioned airtight over the collar, preventing gas leaks or overpressure that could alter CO₂ dynamics. In addition to CO₂ concentration, soil temperature (T°C Soil) and soil water content (WC%) were recorded simultaneously using ECH2O EC-5 sensors (Decagon Devices), an instrument designed for continuous monitoring of soil microclimatic parameters.

The CO₂ Eflux (F) was calculated using the equation from Pumpanen et al. (2004):

$$F = \frac{\Delta C}{\Delta t} \times \frac{P}{RT} \times VA^{-1}$$

where:

- F = Soil respiration flux ($\mu\text{mol CO}_2 \text{ m}^{-2} \text{ s}^{-1}$),
- $\Delta C/\Delta t$ = Rate of CO₂ concentration increase in the chamber (ppm s^{-1}),
- P = Atmospheric pressure (Pa),
- R = Universal gas constant ($8.314 \text{ J mol}^{-1} \text{ K}^{-1}$),
- T = Absolute temperature (K),
- V = Chamber volume (m^3),
- A = Soil surface area covered by the chamber (m^2).

Simultaneously, soil temperature and moisture were recorded using ECH2O EC-5 sensors (Decagon Devices), ensuring that variations in environmental conditions were accounted for.

Soil respiration and CO₂ Eflux measurements were conducted twice per month, weather conditions permitting, typically between 10:00 AM and 12:00 PM, including during the summer season. This measurement protocol was consistently applied at both study sites, including the Portuguese site at Companhia das Lezírias (CL).

The Monte Morrone site (Italy) was monitored from January 2022 to September 2024, with a total of 19 sampling campaigns conducted.

The Companhia das Lezírias site (Portugal) was monitored from June 2023 to May 2024, with a total of 15 sampling campaigns.

These campaigns were planned to capture seasonal variations and understand the long-term dynamics of soil respiration under different experimental treatments.

3.5.1. Estimation of Temperature Sensitivity (Q_{10}) and Basal Respiration

The final stage of analysis involved the calculation of the Q_{10} coefficient. To assess the temperature sensitivity of soil respiration (SR), we calculated the Q_{10} coefficient, which quantifies the rate of change in respiration per 10°C increase in soil temperature (Davidson & Janssens, 2006). Given the complexity of microbial respiration responses to temperature, we adopted two distinct approaches—an exponential model and a logarithmic model—to compare different formulations of the temperature dependence of SR and minimize potential biases in Q_{10} estimation.

Soil respiration data were obtained from multiple soil collars installed in the Monte Morrone study site, covering burned plots, unburned control plots, and areas treated with Nature-Based Solutions (NBS). The same methodological framework was applied to the Portuguese site, ensuring comparability across different fire-affected Mediterranean ecosystems.

A single mathematical formulation may not fully capture the complexity of soil respiration responses to temperature due to variations in microbial activity, substrate availability, and post-fire recovery dynamics. The exponential model is widely used in soil ecology and assumes that respiration increases exponentially with temperature (Lloyd & Taylor, 1994). However, in some conditions, this approach can lead to overestimation at higher temperatures (Fang & Moncrieff, 2001). The logarithmic model, in contrast, accounts for potential saturation effects or deviations from an ideal exponential response, which may occur under limited substrate availability or extreme temperature fluctuations. By applying both models, we aimed to reduce the risk of systematic overestimation or underestimation of Q_{10} , ensuring robust comparisons between treatment groups.

Mathematical Formulations for Q_{10} Estimation

The temperature dependence of Soil CO₂ Efflux (SR) was modeled using the following equations:

1. Exponential model:

$$SR = \alpha e^{\beta T_{soil}}$$

where α represents the basal respiration (Soil CO₂ Efflux at $T_{soil} = 0^\circ\text{C}$), and β is the temperature coefficient. The Q_{10} value was computed as:

$$Q_{10} = e^{\beta \cdot 10}$$

This formulation assumes an exponential increase in Soil CO₂ Efflux with temperature, a widely accepted approach in ecological studies (Lloyd & Taylor, 1994).

2. Logarithmic model:

$$\ln(SR) = a + bT_{soil}$$

where a is the intercept, and b represents the temperature dependence of respiration. The Q_{10} value in this case was derived as:

$$Q_{10} = 10^b$$

The logarithmic transformation was applied to test for potential deviations from a strict exponential response, particularly under conditions where microbial respiration might stabilize at higher temperatures (Fang & Moncrieff, 2001).

Each model was fitted separately for every individual collar within each treatment group. The statistical significance of the temperature coefficient (β or b) was assessed via p-values, while model fit was evaluated using R^2 . Non-significant or non-convergent models were excluded from further analysis.

Basal Respiration Calculation

Basal respiration (BR) was estimated at a standard reference temperature of 15°C to facilitate cross-site comparisons of microbial activity (Conant et al., 2011). The BR_{15} values were computed from both models as follows:

- Exponential model:

$$BR_{15} = \alpha e^{\beta \cdot 15}$$

- Logarithmic model:

$$BR_{15} = e^{a + b \cdot 15}$$

After comparing the outcomes, we found that the exponential model yielded more consistent and biologically interpretable values of basal respiration, supporting its adoption for subsequent data analysis.

3.6. Soil Analysis

To complement the Soil CO₂ Efflux measurements, soil samples were collected from each plot for physical-chemical and genetic analyses. Samples were taken at a depth of 0-10 cm, dried, and sieved

for chemical analysis, which included tests for pH, electrical conductivity, organic matter content, and nutrient availability (Dalias et al. 2018; Memoli et al. 2020). These parameters provide essential insights into how soil properties influence Soil CO₂ Efflux rates and ecosystem recovery. Figure 22 shows the soil sampling procedure at the Monte Morrone site during field campaigns conducted after the wildfire events.



Figure 22 Soil sampling at the Monte Morrone site following the 2023 wildfire. The sample was taken at a depth of 0–20 cm to assess the impact of fire on soil physical-chemical properties and microbial composition.

Photo: Author's documentation (2023).

In addition to physical-chemical analyses, microbial DNA was extracted from soil samples to analyze microbial diversity. This was achieved using PCR and next-generation sequencing techniques, focusing on both bacterial and fungal communities. Understanding the microbial composition is crucial in post-fire and drought-affected ecosystems, where microbial shifts can greatly influence nutrient cycling and soil respiration (Certini et al. 2021; Heinemeyer et al. 2011).

3.6.1. Soil Sampling

Soil samples were collected from the study site in the Monte Morrone area, which experienced significant fire events in 2017 and 2023. Samples were taken at one depth (0–20 cm), to assess the impact of fire on soil.

3.6.2. Physical and Chemical Soil Analysis

3.6.2.1. Monte Morrone: Methodologies and Analyses

The soil analyses conducted at Monte Morrone focused on understanding the physical and chemical impacts of fire on this Mediterranean forested ecosystem. Due to logistical and analytical constraints, the study centered on core parameters: soil texture, pH, and bulk density. These analyses aimed to capture the essential dynamics of post-fire soil properties.

Soil Texture The particle size distribution of soil was determined using the sieving and sedimentation method in accordance with **ISO 11277:2009**. This protocol classified soil particles into sand, silt, and clay fractions. Understanding soil texture is critical for interpreting water retention capacity, nutrient availability, and soil behavior under varying conditions. The predominance of silty-clay textures observed aligns with the calcareous substrate typical of the Majella National Park.

Soil pH Soil pH was measured using a potentiometric method, following **ISO 10390:2005**. Soil samples were equilibrated in a 1:2.5 soil-to-deionized-water suspension overnight. This approach provides insights into the soil's acidity, a parameter that influences nutrient solubility and microbial community composition. Post-fire plots exhibited slight alkalization due to ash deposition and volatilization of organic acids.

Bulk Density Bulk density was assessed using the core method outlined by **Blake and Hartge (1986)**. Undisturbed soil samples were collected using stainless steel cylinders, oven-dried at 105°C for 24 hours, and weighed to calculate dry weight. The formula applied for bulk density was:

This metric is essential for understanding soil compaction, which can restrict root growth and microbial activity. Increased bulk density observed in fire-affected plots reflects compaction from aggregate collapse and ash deposition.

3.6.2.2. Companhia das Lezírias: Methodologies and Analyses

The Companhia das Lezírias site, part of a Mediterranean agro-silvo-pastoral system, required a broader range of soil analyses to evaluate the effects of grazing and land management on soil health and respiration. Methods adhered to established protocols commonly employed by cE3c - Centre for Ecology, Evolution, and Environmental Changes and CHANGE - Global Change and Sustainability Institute, with a focus on soil properties that influence microbial activity and carbon cycling.

Soil Moisture Content

Gravimetric soil moisture content was determined according to **ISO 11465:1993**. Fresh soil samples were weighed, oven-dried at 105°C for 24 hours, and reweighed to calculate the moisture percentage:

$$\text{Moisture Content} = \frac{\text{Fresh Weight} - \text{Dry Weight}}{\text{Dry Weight}} \times 100$$

This analysis is critical for understanding the influence of water availability on microbial activity and soil respiration, particularly under Mediterranean climatic conditions.

Soil pH

Soil pH was measured using a potentiometric method following **ISO 10390:2005**. Samples were equilibrated in a 1:2.5 soil-to-deionized-water suspension overnight before measurement with a calibrated pH meter. This method helps evaluate soil acidity, which directly impacts nutrient availability and microbial activity.

Electrical Conductivity (EC)

Electrical conductivity was measured using a conductivity meter, following the standardized soil-to-water ratio of 1:5. EC values indicate the concentration of soluble salts in the soil, which is critical for assessing salinity stress on microbial and root functions.

Carbon Analyses: TOC, OC, and IC

-Total Organic Carbon (TOC): TOC was analyzed using the **Walkley-Black Dichromate Oxidation Method**. The process involves:

1. **Digestion:** Soil samples were treated with potassium dichromate ($\text{K}_2\text{Cr}_2\text{O}_7$) and sulfuric acid (H_2SO_4) to oxidize organic carbon.
2. **Titration:** Excess dichromate was titrated with ferrous ammonium sulfate using a ferroin indicator.
3. **Calculation:** TOC was calculated based on the amount of dichromate reduced during the reaction. This method estimates the labile organic carbon pool.

-Inorganic Carbon (IC): IC was determined using the **Pressure Calcimeter Method**, which measures CO_2 released by treating soil samples with hydrochloric acid (HCl). The process includes:

1. Weighing a dry soil sample.
2. Adding HClHClHCl (1M) to react with carbonates in the soil.
3. Measuring CO₂ gas pressure produced during the reaction.
4. Calculating IC content based on the pressure reading using standard equations. This method provides a precise measure of carbonate content in calcareous soils.

-Organic Carbon (OC): OC was calculated as the difference between TOC and IC:

$$\text{OC (\%)} = \text{TOC (\%)} - \text{IC (\%)}$$

This value represents the fraction of carbon associated with organic matter, excluding carbonates. It provides critical insights into soil fertility and carbon storage potential.

Nitrogen and Carbon Content Analysis

-Total Nitrogen (N%) and Total Carbon (C%): Samples were analyzed for total nitrogen and carbon content using **Continuous Flow Isotope Mass Spectrometry (CF-IRMS)** at the Laboratório de Isótopos Estáveis (LIE), Universidade de Lisboa.

3.7. Microbiological analysis

The environmental quality of an area or territory may be estimated through the use of appropriate indicators. These can be defined as instruments capable of representing, with different levels of approximation, particular conditions (events, processes, overall states of quality or criticality) of the environment. The parameters for the assessment of soil quality can be divided into physical, chemical and biological; in our case study biological parameters were also examined. Microorganisms can be used as indicators of soil quality because they perform key functions in the degradation and recycling of organic matter and nutrients or respond readily to changes in the soil environment. In addition, the microbial activity in soil reflects the sum of all the factors regulating the degradation and transformation of nutrients. However, it is extremely difficult to use the values provided by the microbiological parameters because soil microorganisms react very quickly to seasonal variations and adapt to different environmental needs. At each point, a soil sample (core drilling) of about 20 cm deep, litter-free, was extracted. For this study, specific samples were collected from three categories of plots: burned plots, burned plots treated with Nature-Based Solutions (NBS), and control plots. Each plot was sampled at two depths (5 cm and 10 cm) to assess whether the fire's impact extended beyond the surface layer, as suggested by existing literature. The sample distribution was as follows:

- **Burned plots without NBS interventions:**
 - **Plot 1C-5cm, Plot 1C-10cm**
 - **Plot 2C-5cm, Plot 2C-10cm**
 - **Plot 3C-5cm, Plot 3C-10cm**
- **Burned plots with NBS interventions:**
 - **Plot 1A-5cm, Plot 1A-10cm**
 - **Plot 2A-5cm, Plot 2A-10cm**
 - **Plot 3A-5cm, Plot 3A-10cm**
- **Control plots:**
 - **Plot 4C-5cm, Plot 4C-10cm**
 - **Plot 5C-5cm, Plot 5C-10cm**
 - **Plot 6C-5cm, Plot 6C-10cm**

This dual-depth sampling allowed for a detailed assessment of how fire and management interventions influenced microbial activity across surface and subsurface soil layers. After collection, the samples were stored in the field at ambient temperature for a short time, avoiding deterioration. After the field phase, the samples were stored in a chiller at a temperature of -30 °C. From each soil sample (of the Monte Morrone site) the first extraction of total DNA was performed using 250 mg of soil (taken randomly within each individual sample) according to the manufacturer's instructions (Dneasy PowerSoil Pro Kit, QIAGEN). Later for the rapid and accurate quantification of DNA each DNA sample was diluted. This step was performed using Invitrogen Qubit 4 Fluorometer 4. The Invitrogen Qubit 4 Fluorometer accurately and quickly measures the concentration of DNA, RNA or proteins in a single sample (requires only 1 µl of sample, although highly diluted). It can also be used to evaluate the integrity and quality of RNA. The easy-to-use touchscreen menu makes it easy to select and run the necessary tests, with the results displayed in seconds. Following extraction, DNA samples were refrigerated at 4 °C until they were used for the amplification and sequencing process. For the bacterial community analysis 16S rRNA genes were amplified by PCR using primers for hypervariable regions V2, V3, V4, V6-7, V8 and V9, according to the Ion 16S Metagenomics Kit (Life Technologies, 5781 Van Allen Way Carlsbad, CA, USA 92008). The resulting libraries were amplified by using the Ion Plus Fragment Library kit (Thermo-Fisher Scientific) and quantified by Qubit 3.0 fluorometer (Thermo Fisher Scientific, Carlsbad, CA) with Qubit DNA High Sensitivity Assay Kit (Thermo Fisher Scientific). Finally, they were pooled together with an equimolar concentration of 40 pM and prepared for sequencing in Ion Torrent™, Ion S5™ System using Ion 520™ Chip Kit. The templates were then sequenced on the Ion GeneStudio™ S5. The generated unaligned binary data files (Binary Alignment Map, BAM) have been analysed using the Ion Reporter™ Software 5.20 platform with the workflow of Metagenomics 16S w1.1 v5.14. The Ion

reporter workflow is a custom workflow that uses both MicroSEQ™ ID 16S Reference Library v2013.1 and Greengenes v13.5 as reference libraries. All reads shorter than 150 bp were filtered out after trimming primers. A Minimum Alignment Coverage of 90% and a Read Abundance Filter of 10 were set; threshold value for percentage identity was 99%. 16S rRNA sequence reads were finally clustered into operational taxonomic units (OTUs) using the closed-reference method. For the fungal community analysis ITS1 and ITS2 sequences were amplified by multiplex PCR. Primer specific sequences are listed in Tonge et al. (2014). The amplification reactions were performed using the TaqMan™ Environmental Master Mix 2.0 (Applied Biosystems™). The PCR program has been set up as follows: 94 °C for 3 min; 25 cycles of 94 °C for 15 s, 50 °C for 30 s and 68 °C for 60 s and 72 °C for 7 min. The resulting libraries were amplified and sequenced in the same way as the 16S libraries. Taxonomic identification was conducted using the UNITE database by using a length filter >10 bp. All reads shorter than 150 bp were filtered out after trimming primers. A Minimum Alignment Coverage of 90% and a Read Abundance Filter of 10 were set; threshold value for percentage identity was 99%. ITS sequence reads were finally clustered into operational taxonomic units (OTUs) using the closed-reference method.

3.8. Statistical Analysis

All statistical analyses were conducted using R software (R Core Team, 2023), with the objective of quantifying and comparing soil respiration (SR) dynamics and their environmental controls across two Mediterranean ecosystems: Monte Morrone (Italy), affected by wildfire, and Companhia das Lezírias (Portugal), affected by degradation due to grazing and drought. The analyses aimed to evaluate differences in soil CO₂ efflux, estimate temperature sensitivity (Q_{10}), basal respiration (BR₁₅), and assess the role of soil physicochemical properties and microclimatic factors in driving respiration responses under contrasting disturbance regimes.

Data preprocessing involved outlier removal based on ecologically informed thresholds, excluding extreme soil temperature values (>50°C), the upper 5% of Soil CO₂ Efflux measurements, and the lower and upper 5% of volumetric water content (VWC), to minimize distortions due to anomalous environmental conditions. Data cleaning was initially performed in Excel, followed by normalization and statistical modeling in R.

To evaluate the effects of treatment (burned vs. control, degraded vs. control), differences in Soil CO₂ Efflux were tested using ANOVA when assumptions of normality and homoscedasticity were met, verified through the Shapiro-Wilk and Levene tests. When assumptions were violated, non-parametric alternatives were applied, including the Kruskal-Wallis test, Wilcoxon rank-sum test, and

Brunner-Munzel test (Neubert & Brunner, 2007), allowing for robust inference under heteroscedasticity. Pairwise comparisons were performed using Dunn's test with Bonferroni correction.

The influence of environmental drivers (soil temperature, moisture, and organic matter) on Soil CO₂ Efflux was assessed using multiple approaches. Generalized Linear Mixed Models (GLMMs) and Linear Mixed Models (LMMs) were applied to account for repeated measures and nested data structures (Bates et al., 2015), with fixed effects including temperature, water content, and treatment, and random intercepts for plot and measurement date. To capture potential non-linear and extreme-value effects, quantile regression was also employed, allowing the identification of differential temperature sensitivity across SR quantiles (Koenker & Bassett, 1978), particularly useful in ecosystems exhibiting strong seasonal or microhabitat variability. Bootstrap resampling (Efron & Tibshirani, 1993) was used to estimate the confidence intervals of mean differences and regression coefficients, enhancing the robustness of inferential statistics.

For Q₁₀ and BR estimation, soil respiration measurements from individual collars were modeled using both exponential and logarithmic formulations of temperature dependence (Lloyd & Taylor, 1994; Fang & Moncrieff, 2001; Davidson & Janssens, 2006). Q₁₀ was computed to quantify the increase in respiration per 10°C rise in soil temperature, while basal respiration (BR₁₅) was calculated as the modeled SR value at a standard reference temperature of 15°C to enable cross-site comparisons of microbial activity. Both models were fitted to each collar, and the best-fitting formulation was selected based on R² and biological interpretability.

In addition to functional fluxes, soil physicochemical properties were analyzed to identify their contribution to SR variability. Multicollinearity among predictors (e.g., TOC, OM, TC) was tested using the Variance Inflation Factor (VIF), with variables exceeding the standard threshold excluded to ensure model stability (Dormann et al., 2013). Multiple linear regressions were then performed separately for each site and treatment, with SR as the response variable and selected soil properties (e.g., TOC, WC, pH, temperature) as predictors. This allowed for the identification of site-specific drivers and potential shifts in respiration controls under degradation or post-fire conditions.

Finally, the relationship between microclimatic variables and Soil CO₂ Efflux was explored using both ANOVA and regression models to assess how soil temperature and water content varied across treatments and influenced Soil CO₂ efflux. These analyses were crucial for interpreting how vegetation loss, soil structural changes, and environmental stress interact to modulate carbon dynamics in fire- and drought-prone Mediterranean systems.

This comprehensive statistical framework—combining hypothesis testing, mixed-effects modeling, quantile regression, and functional trait estimation—enabled a nuanced evaluation of soil respiration under multiple disturbance scenarios, supporting ecologically meaningful interpretations of ecosystem functioning and informing targeted restoration strategies.

4 Results

4.1. Microclimate Data Analysis for Monte Morrone Monitoring Site

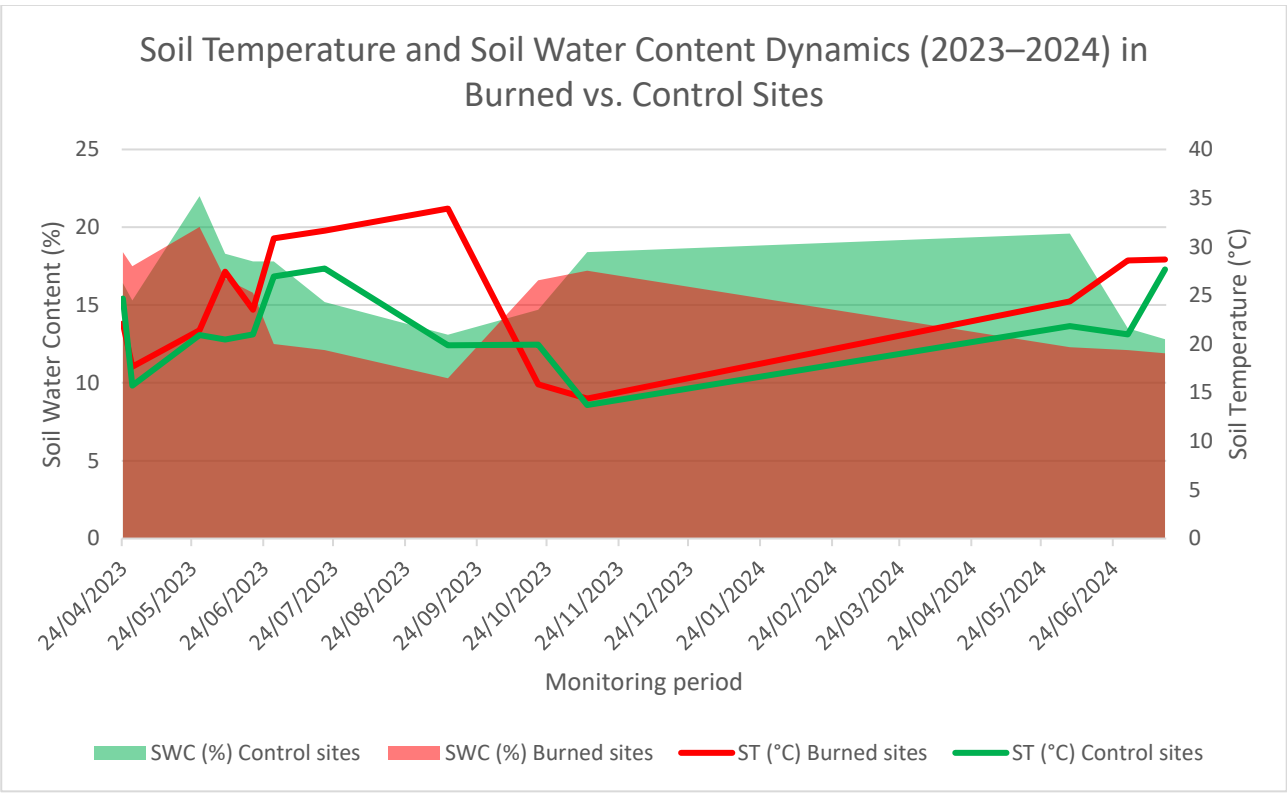


Figure 23 Seasonal variation of soil temperature (°C, lines) and soil water content (SWC, %, shaded areas) Soil temperature is represented with red and green lines for burned and control sites, respectively, and referenced on the left Y-axis. SWC is displayed using shaded areas with the same colour scheme and referenced on the right Y-axis. The figure highlights the contrasting dynamics of soil moisture and temperature between the two site conditions, particularly during summer peaks and dry periods.

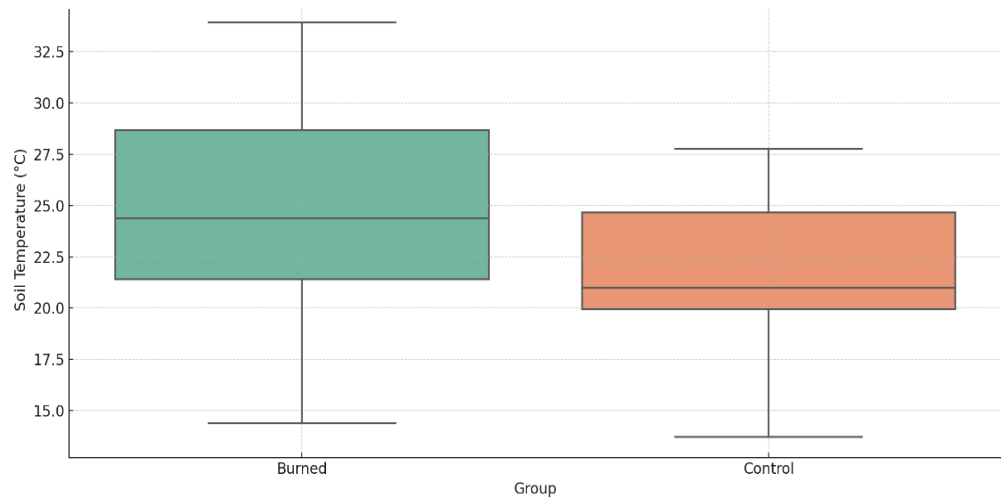


Figure 24 Soil temperature (°C) across control and burned plots at the Monte Morrone site. Boxplots represent the distribution of soil temperature measurements collected between April 2023 and July 2024. Horizontal lines indicate median values, boxes represent interquartile ranges, and whiskers denote the data range excluding outliers. Error bars represent $\pm$ standard error of the mean.

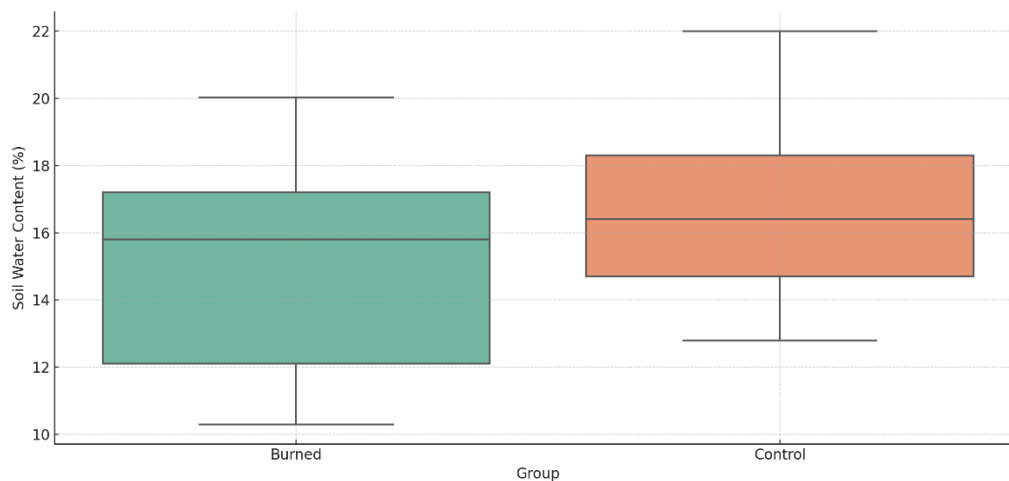


Figure 25 Soil water content (SWC, %) across control and burned plots at the Monte Morrone site. Boxplots show the variation in soil water content recorded during the monitoring period (April 2023 – July 2024). Boxes display interquartile range, medians are shown as horizontal lines, and whiskers represent data variability. Error bars represent $\pm$ standard error of the mean.

Table 3 Mean soil temperature ($^{\circ}\text{C}$) and soil water content (SWC, %) in control and burned plots at Monte Morrone. Values are presented as mean $\pm$ standard error. Letters indicate group differences as determined by Tukey's HSD post-hoc test following one-way ANOVA ($p < 0.05$).

Site	SoilT (Mean $\pm$ SE)	Tukey (SoilT)	ANOVA p (SoilT)	SWC (Mean $\pm$ SE)	Tukey (SWC)	ANOVA p (SWC)
Control	21.66 $\pm$ 1.18	a	0,165	16.53 $\pm$ 0.76	a	0,164
Burned	24.65 $\pm$ 1.72	a	0,165	14.88 $\pm$ 0.86	a	0,164

The analysis of microclimatic dynamics at the Monte Morrone site highlights significant differences in soil temperature and moisture content between burned, NBS-treated, and control plots. Soil temperature (SoilT $^{\circ}\text{C}$) was consistently higher in burned plots throughout the monitoring period, especially during the warmest months, reflecting the absence of vegetation cover and the consequent exposure to direct solar radiation (Figure 23, Figure 24). This effect was slightly mitigated, yet still evident, in NBS-treated plots, suggesting partial thermal buffering from the presence of deadwood structures. In contrast, control plots maintained significantly lower and more stable soil temperatures, benefiting from canopy shading and higher evapotranspiration capacity (Table 3).

Soil water content (SWC) also varied markedly across treatments, with control plots showing significantly higher moisture levels compared to both burned and NBS plots (Figure 23, Figure 25). The reduced SWC in burned areas is likely attributable to a combination of factors: decreased organic matter, impaired soil structure, and lack of vegetative cover, all of which limit water retention and increase surface evaporation. Although NBS plots retained slightly more moisture than fully burned areas, their SWC remained significantly lower than that of the control, indicating that passive restoration alone may be insufficient to restore soil hydrological function (Table 3).

4.2. Spectral Index Processing and Burned Area Mapping

The Monte Morrone region has experienced significant wildfire events in 2017 and 2023, leading to extensive vegetation loss and ecosystem disruption. The analysis of burned area extent and severity provides crucial insights into the ecological impact of these fires, with implications for management and restoration strategies.

According to data from the European Forest Fire Information System (EFFIS), the burned area in the 2017 wildfire covered approximately 2165 hectares, while the 2023 wildfire affected about 432 hectares. These events have led to varying levels of vegetation loss, as captured through remote sensing indices such as dNBR, RdNBR, dLAI, and dkNDVI, which quantify changes in vegetation cover, photosynthetic activity, and burn severity (Key & Benson, 2006; De Santis et al., 2010; Garcia & Caselles, 1991).

4.2.1. Results and Analysis: Monte Morrone Wildfire 2017

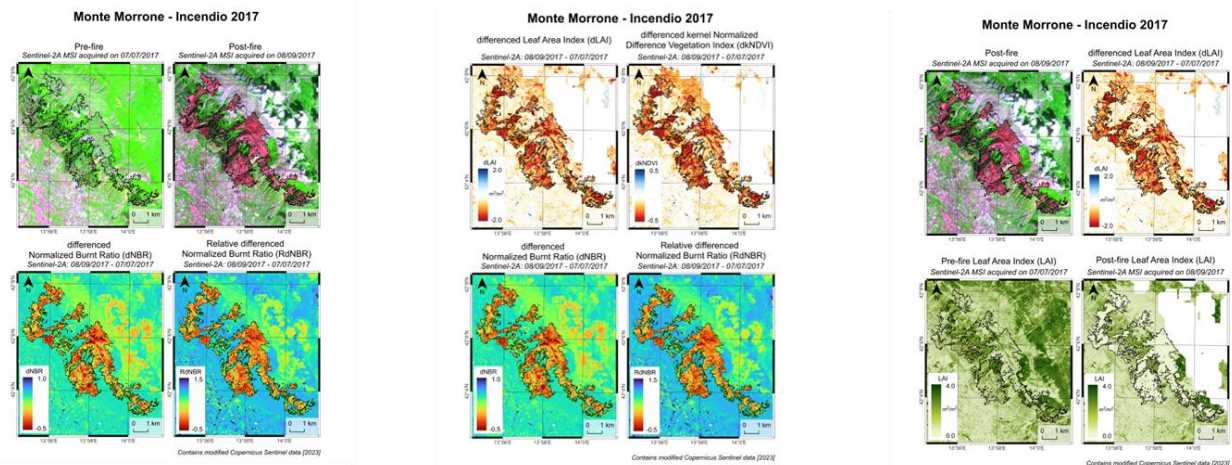


Figure 26 Impact of the 2017 Wildfire on Vegetation Dynamics in the Monte Morrone Area: A Remote Sensing Analysis

Figure 26a shows the pre-fire (07/07/2017) and post-fire (08/09/2017) conditions in the Monte Morrone area. The pre-fire imagery indicates dense, healthy vegetation cover, while the post-fire imagery highlights burn scars and vegetation loss, especially in forested regions. Figure 26b presents the dNBR and RdNBR maps derived from Sentinel-2A data. High values in both indices (shown in red) denote areas of severe vegetation damage, with the most intense burns concentrated in the central and southern zones. Figure 26c illustrates the reduction in Leaf Area Index (dLAI) caused by the fire. Red areas correspond to significant canopy loss, which reflects a decrease in biomass and photosynthetic capacity. Figure 26d shows changes in dkNDVI values, highlighting zones with

altered vegetation health. Negative dkNDVI values, shown in red, indicate areas with strong vegetation decline.

4.2.2. Results and Analysis of the Monte Morrone Wildfire 2023

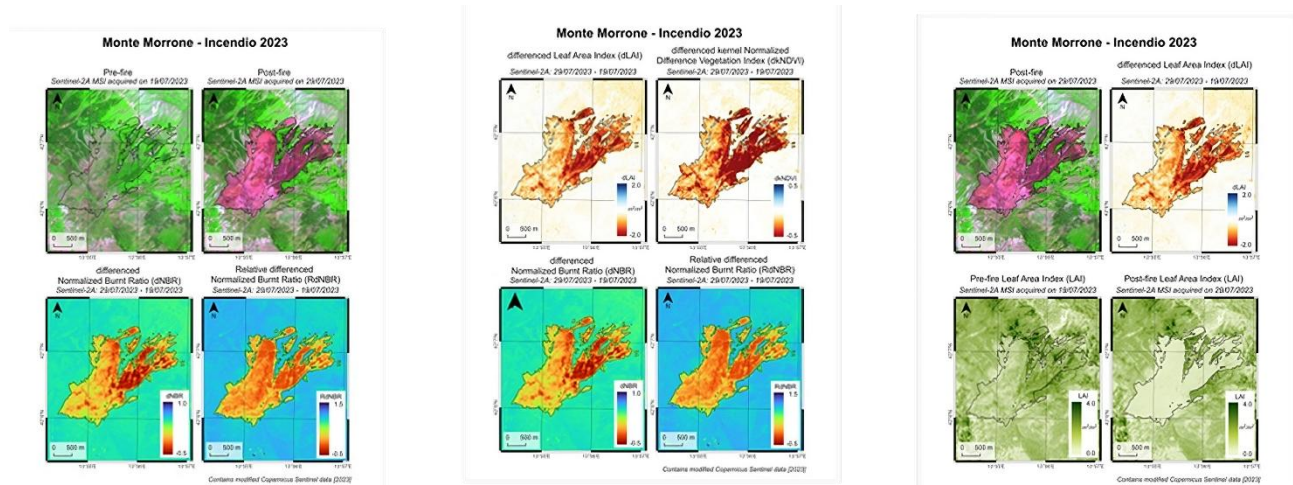


Figure 27 Impact of the 2023 Wildfire on Vegetation Dynamics in the Monte Morrone Area: A Remote Sensing Analysis

Figure 27a displays the 2023 pre-fire (19/07/2023) and post-fire (29/07/2023) imagery. While smaller in extent, this fire event caused notable damage, especially in previously unaffected zones. Figure 27b and 27c present dNBR, RdNBR, dLAI, and dkNDVI maps. High values in dNBR and RdNBR denote severe damage, with lower intensity than the 2017 fire. Red areas in dLAI and dkNDVI maps indicate significant reductions in vegetation cover and photosynthetic activity.

The tables below provide a breakdown of the burn severity distribution across different indices for the 2017 and 2023 wildfires, offering a quantitative representation of fire impact.

Table 4 dkNDVI Burn Severity Comparison (2017 vs. 2023)

<i>Index</i>	<i>Class ID</i>	<i>Fire Damage Class</i>	<i>Coverage Area 2017 (hectares)</i>	<i>Coverage Area 2023 (hectares)</i>
dkNDVI	0	Unburned	324,75	86,4
dkNDVI	1	Low Severity	433	108
dkNDVI	2	Moderate Severity	757,75	129,6
dkNDVI	3	High Severity	433	64,8
dkNDVI	4	Very High Severity	216,5	43,2

Table 5 dLAI Burn Severity Distribution (2017 vs. 2023)

<i>Index</i>	<i>Class ID</i>	<i>Fire Damage Class</i>	<i>Coverage Area 2017 (hectares)</i>	<i>Coverage Area 2023 (hectares)</i>
dLAI	0	Unburned	324,75	64,8
dLAI	1	Low Severity	540	129,6
dLAI	2	Moderate Severity	756	259,2
dLAI	3	High Severity	324,75	126,9
dLAI	4	Very High Severity	216,5	65

Table 6 RdNBR Burn Severity Distribution (2017 vs. 2023)

<i>Index</i>	<i>Class ID</i>	<i>Fire Damage Class</i>	<i>Coverage Area 2017 (hectares)</i>	<i>Coverage Area 2023 (hectares)</i>
RdNBR	1	Low Severity	1220,1	273,1
RdNBR	2	Moderate Severity	424	77
RdNBR	3	High Severity	502,4	73.3
RdNBR	4	Very High Severity	255	8.6

4.2.3. Results and Analysis of Vegetation Recovery Following the 2017 and 2023 Monte Morrone Wildfires

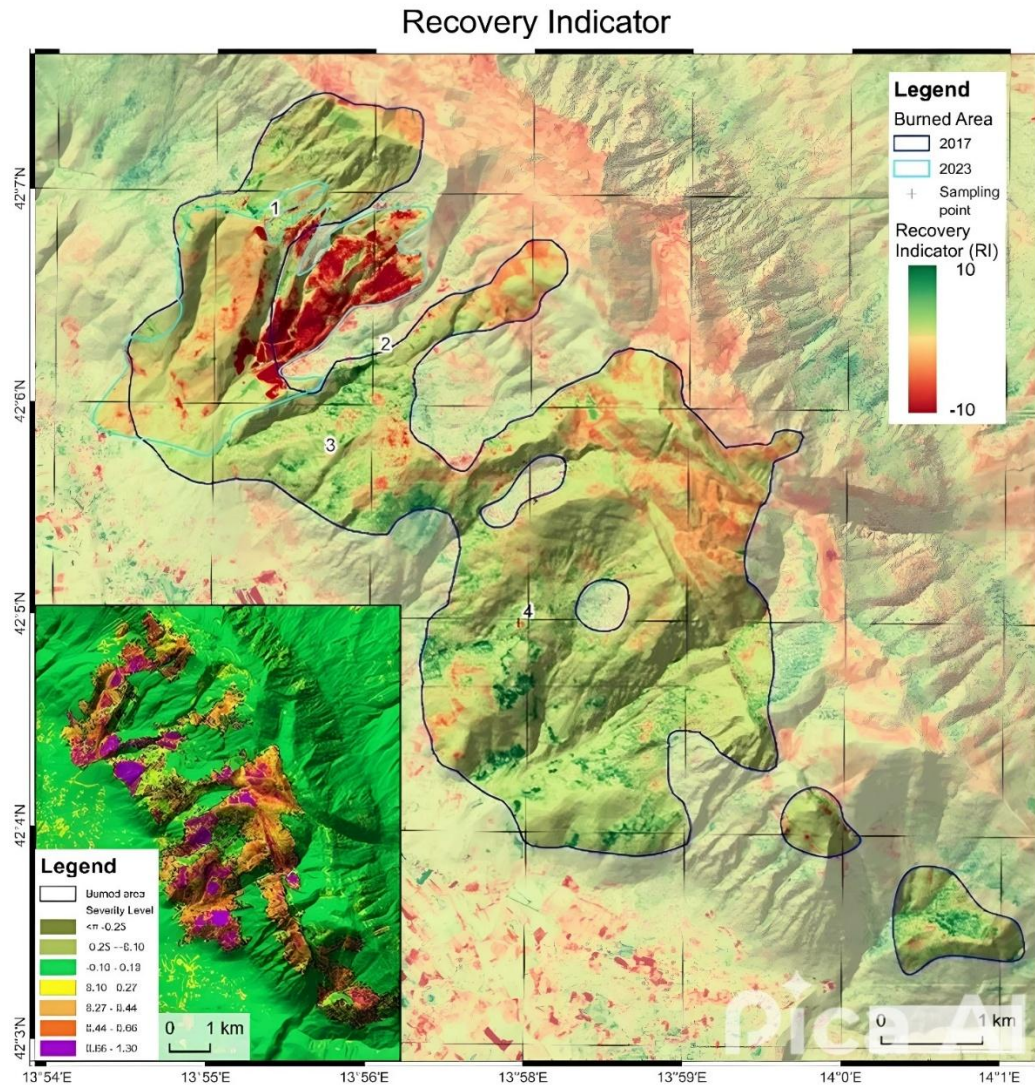


Figure 28 Recovery Indicator Map for Burned Areas in the Monte Morrone Post-Wildfire

Figure 28 illustrates the spatial patterns of vegetation recovery using the Recovery Indicator (RI), with values ranging from -10 (indicating poor recovery) to +10 (indicating strong recovery). The RI map reveals that the areas burned in 2017, delineated in blue, generally exhibit higher recovery values compared to those affected by the 2023 fire. Four distinct recovery zones were defined based on spatial clustering of RI values and overlap with burn scars from 2017 and 2023, in order to capture varying disturbance histories and ecological responses. Zones 1 and 2, located in the northern and central sectors of the 2017 burned area, show prominent recovery, with high RI values illustrated in green. These regions appear to benefit from favorable microclimatic or soil conditions that facilitate regrowth. In contrast, Zone 3 demonstrates limited recovery capacity, which may be attributed to topographic constraints or more severe initial burn conditions. Zone 4, which was affected by both

the 2017 and 2023 wildfires, shows notably poor recovery, suggesting that repeated fire disturbances hinder vegetation regrowth.

Figure 29 shows Leaf Area Index (LAI) trends across three categories—unburned, burned, and regrowth areas—from 2016 to 2023. Unburned areas exhibit stable LAI values over time, reflecting consistent vegetation structure and low disturbance. Burned areas show a marked decline in LAI following fire events, with slow and often incomplete recovery. Regrowth areas, particularly those affected by the 2017 fire, demonstrate an initial increase in LAI, indicating active vegetation recovery; however, this trend stagnates after the 2023 fire event. In this context, the term "pre-regrowth" refers to vegetation cover conditions observed before fire events in areas that later undergo regrowth, while "regrowth" indicates the post-fire increase in vegetation observed in areas previously burned.

Figure 30 presents the temporal evolution of LAI across the four recovery zones previously identified. Zone 1 shows a steady increase in LAI following the 2017 fire, consistent with healthy regrowth. Zone 2 follows a similar trend but is disrupted by the 2023 fire, which leads to a new decline. Zone 3 maintains persistently low LAI values throughout the monitoring period, confirming limited vegetation regeneration. Zone 4 exhibits a fluctuating LAI pattern without clear signs of recovery, underlining the negative effect of repeated fire exposure.

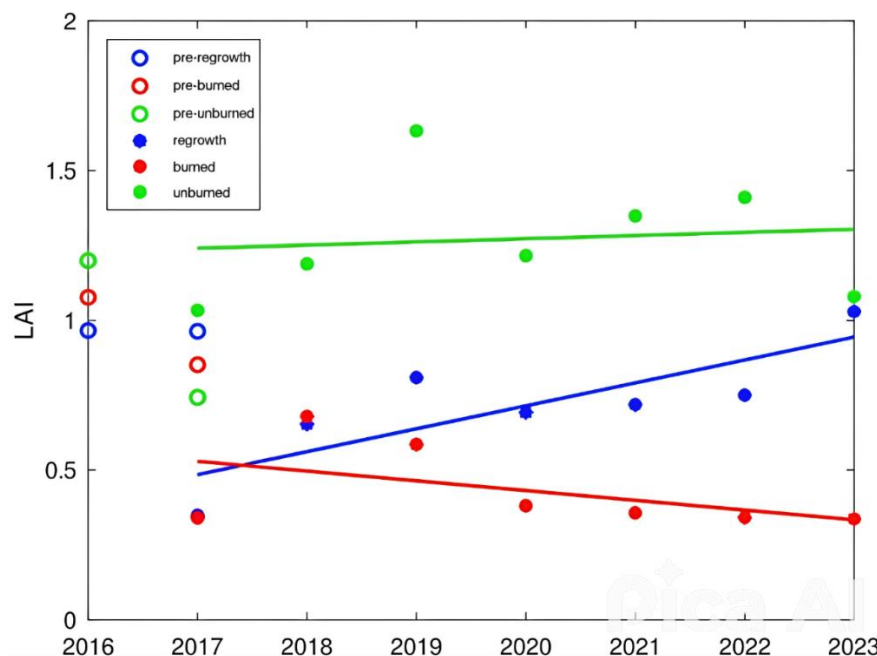


Figure 29 Leaf Area Index (LAI) trends across three categories—unburned, burned, and regrowth areas—from 2016 to 2023

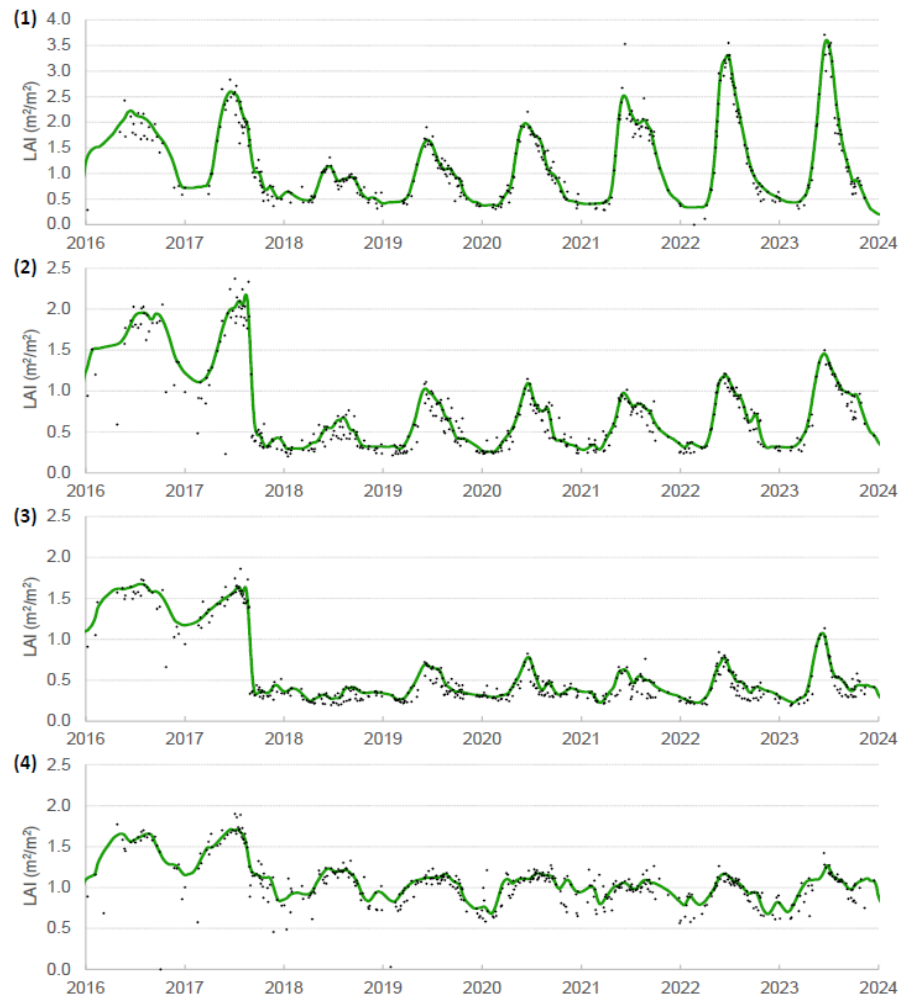


Figure 30 LAI trends from 2016 to 2024 across four zones show post-fire declines in 2017 and 2023, with Zones 1 and 2 displaying recovery, while Zones 3 and 4 indicate limited regrowth, highlighting varied resilience to fire.

4.3. Soil CO₂ Efflux Measurement in Relation to Environmental Factors and Soil Properties: A Comparative Study Across Different Stress Drivers

4.3.1. Introduction

This chapter compares Soil CO₂ Efflux responses under contrasting stressors across two Long-Term Ecological Research (LTER) sites: the Companhia das Lezírias (CL) in Portugal and Monte Morrone in Majella National Park, Italy. CL represents an agro-silvo-pastoral system impacted by drought and grazing, while Monte Morrone exemplifies a forest site affected by wildfire disturbance. The study investigates the relationships between SR and soil properties (temperature, moisture, organic matter), and assesses the role of management interventions such as Nature-Based Solutions (NBS). Multiple statistical approaches—ranging from descriptive and non-parametric tests to linear mixed models (LMM) and quantile regression—are applied to evaluate the magnitude, sensitivity, and variability of Soil CO₂ Efflux across spatial and temporal gradients. This integrative analysis aims to provide a robust understanding of soil carbon fluxes under varying environmental constraints and inform

restoration strategies in fire-prone and drought-affected Mediterranean systems, forming the basis for the site-specific results and discussion presented in the following sections.

4.3.2. Soil CO₂ Efflux in Monte Morrone Site

4.3.2.1. General Trends of Soil CO₂ Efflux in Relation to Temperature Across Fire and Control Sites

The results presented in this section are structured by site and analytical focus. Monte Morrone is analysed first, with sub-sections dedicated to temporal Soil CO₂ Efflux patterns, temperature responses, moisture influences, and functional indicators (Q_{10} , BR), followed by the statistical modelling of the Soil CO₂ Efflux-environment relationship.

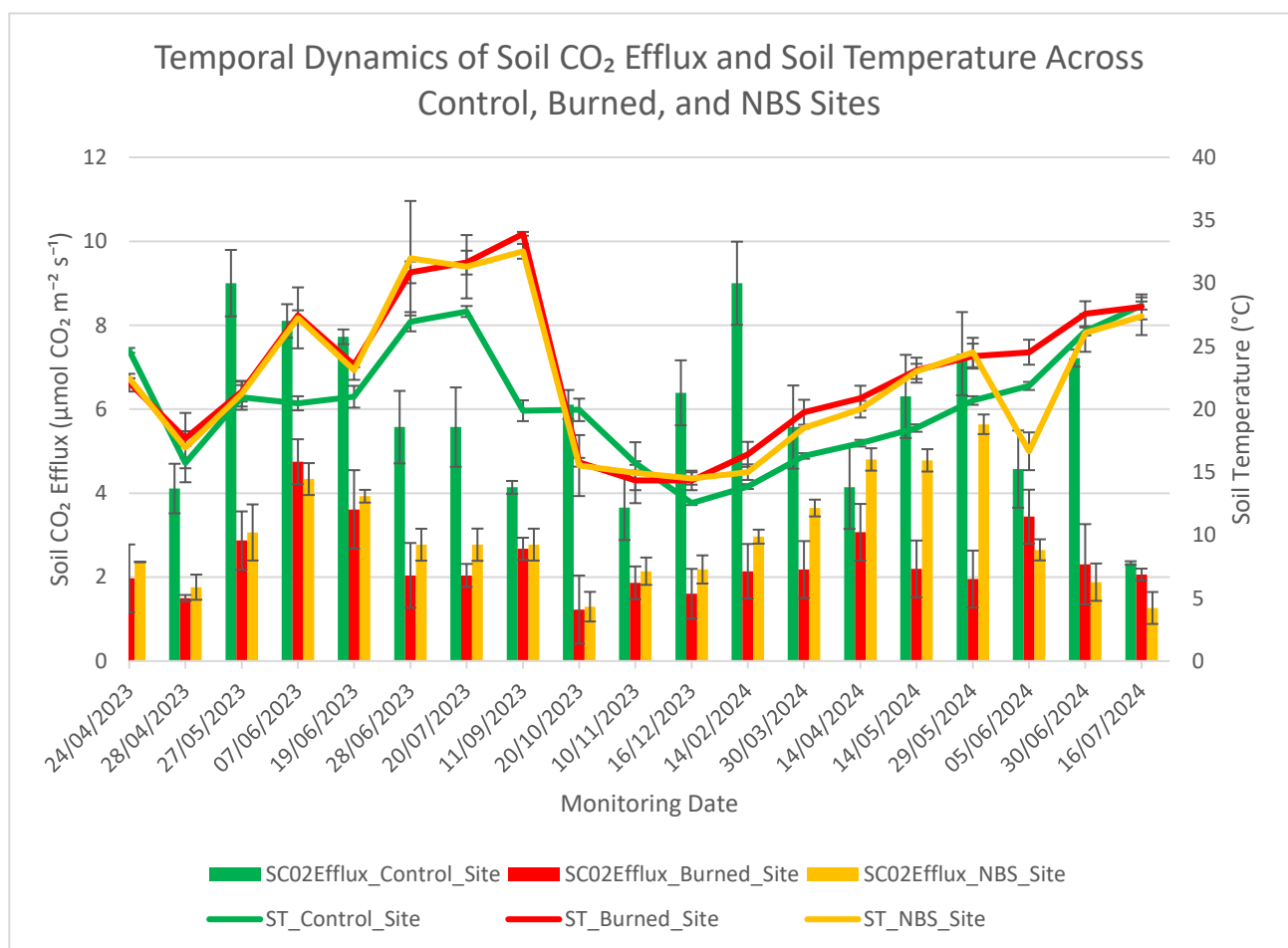


Figure 31 Temporal variation in soil CO₂ efflux (bars) and soil temperature (lines) in burned, control, and NBS plots from April 2023 to July 2024.

CO₂ efflux is reported in $\mu\text{mol CO}_2 \text{ m}^{-2} \text{ s}^{-1}$ (left y-axis), and soil temperature in $^{\circ}\text{C}$ (right y-axis). Bars represent mean values $\pm$ SE. Green, red, and yellow bars indicate Control, Burned, and NBS sites respectively; solid lines represent the corresponding temperature values.

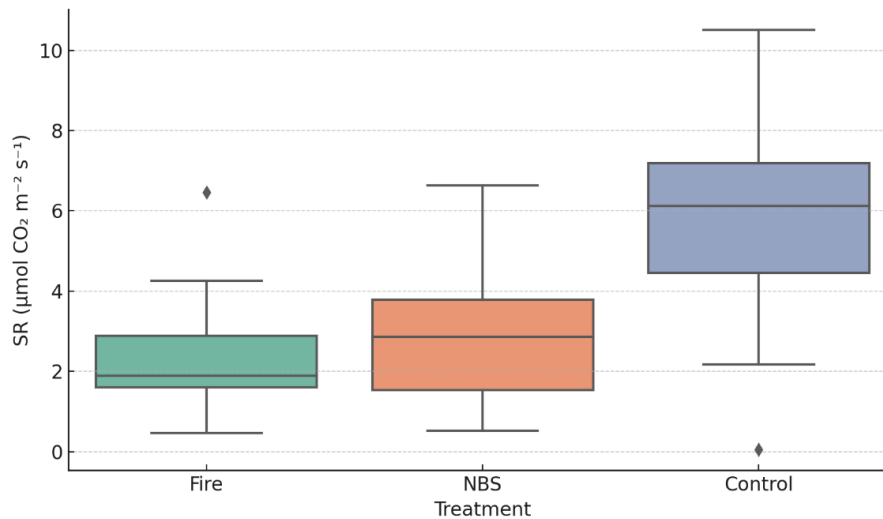


Figure 32 Boxplots represent the distribution of Soil CO₂ Efflux (SR) values across treatments (aggregated across all monitoring periods, n = 72 per group).

Whiskers indicate 1.5×IQR, boxes the interquartile range, and central lines the median.

Table 7 Mean Soil CO₂ Efflux (SR, $\mu\text{mol CO}_2 \text{ m}^{-2} \text{ s}^{-1} \pm \text{standard error}$) across treatments at Monte Morrone (Control, Fire, and Nature-Based Solutions – NBS). Different letters indicate significant differences between treatments according to Tukey’s HSD test ($p < 0.05$). A one-way ANOVA showed highly significant differences among groups ($p < 0.001$).

Thesis	SR (Mean $\pm$ SE, $\mu\text{mol CO}_2 \text{ m}^{-2} \text{ s}^{-1}$)	Tukey	ANOVA p	Significance
Control	5.99 $\pm$ 0.27	a	< 0.001	***
Fire	2.22 $\pm$ 0.12	b		
NBS	2.96 $\pm$ 0.16	c		

Soil CO₂ efflux across burned, control, and NBS-treated plots showed marked seasonal patterns, with higher values during spring and early summer and sharp declines in late summer (Figure 31). Burned plots consistently exhibited significantly lower fluxes (Figure 32), suggesting fire-induced degradation of soil structure and hydrological properties. The control site maintained stable efflux values despite summer temperature peaks, likely due to its higher organic content and porosity (Table 7).

4.3.2.2. Temperature Dependency of SR

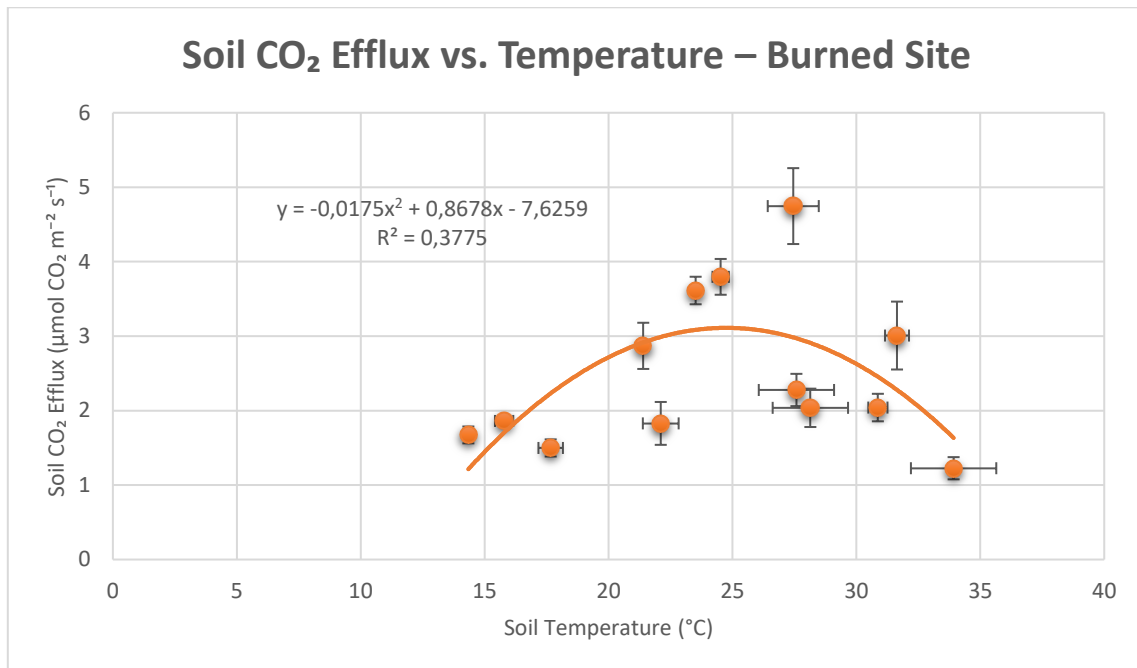


Figure 33 Relationship between soil CO₂ efflux and soil temperature in the burned site. A quadratic curve shows a peak around 25 °C, indicating optimal thermal conditions for microbial and root activity. Error bars represent ± standard error of the mean.

Soil CO₂ efflux–temperature relationships differed between burned and control plots. In the burned site, CO₂ efflux displayed a parabolic response with an optimal peak near 25 °C (Figure 33). Below this threshold, CO₂ efflux increased exponentially with temperature ($R^2 = 0.7087$; Figure 34), while above it, a clear decline was observed ($R^2 = 0.3042$; Figure 35), consistent with microbial thermal stress (Hart et al., 2005; Rey et al., 2011).

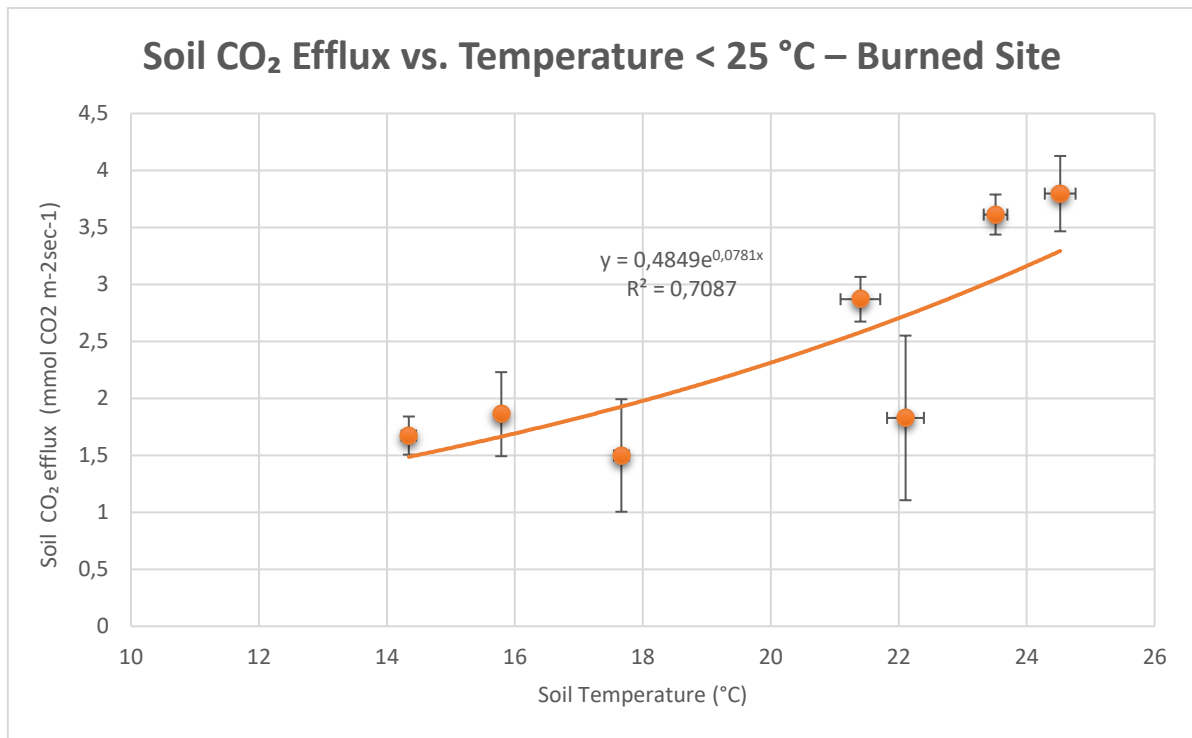


Figure 34 Soil CO₂ efflux in the burned site under temperatures below 25 °C.
An exponential trend indicates high temperature sensitivity in this range ($R^2 = 0.7087$). Error bars represent $\pm$ standard error of the mean.

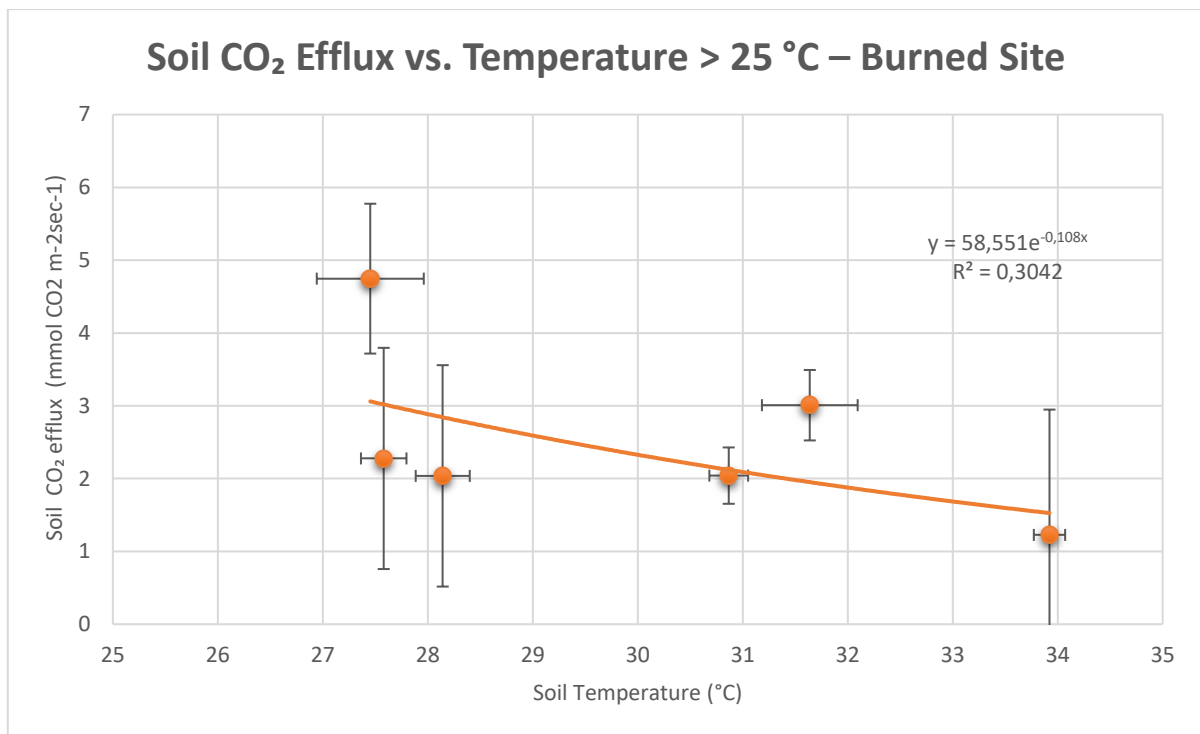


Figure 35 Soil CO₂ efflux above 25 °C in the burned site. The exponential decay suggests microbial inhibition due to heat stress ($R^2 = 0.3042$). Error bars represent $\pm$ standard error of the mean.

In the control site, soil CO₂ efflux peaked at ~20 °C (Figure 36). A strong exponential increase occurred below this value ($R^2 = 0.8197$; Figure 37), while the decline above 20 °C was less

pronounced ($R^2 = 0.4031$; Figure 38), suggesting a greater thermal tolerance due to more favorable soil structure and moisture retention (López-Serrano et al., 2018).

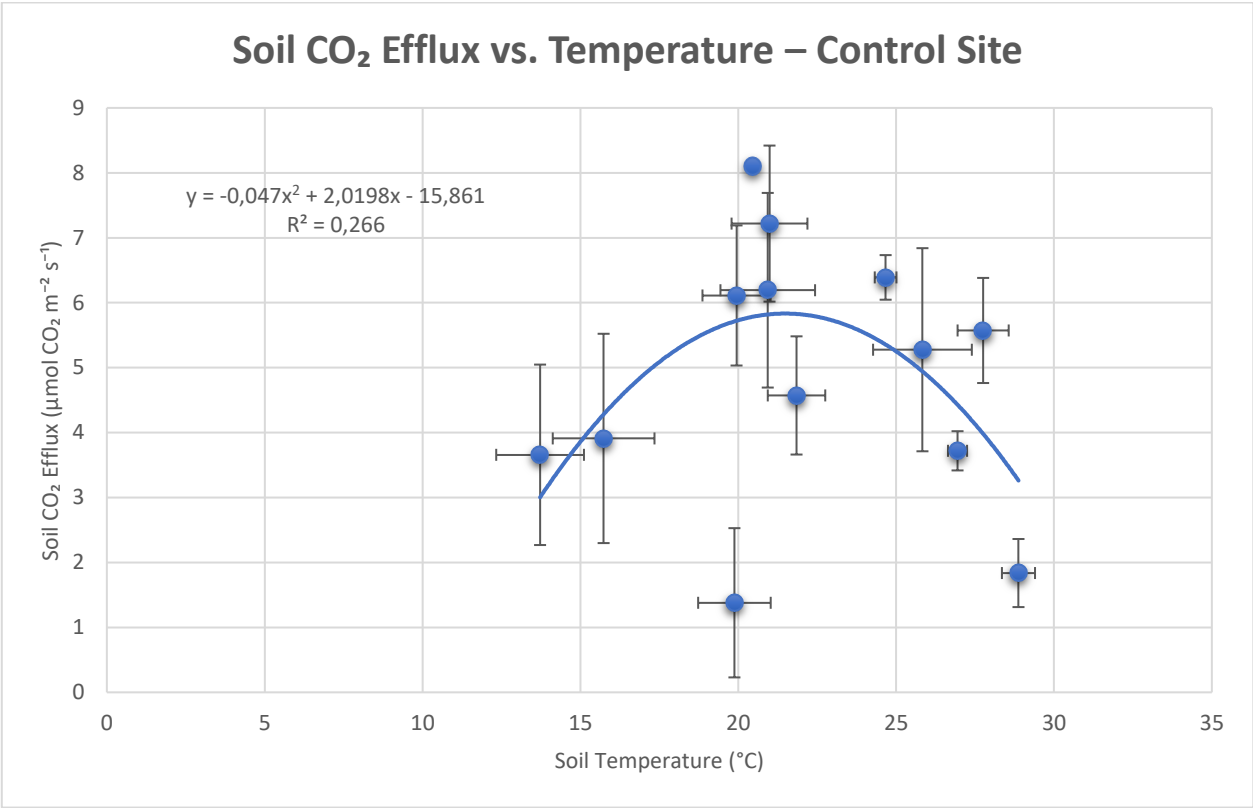


Figure 36 Relationship between soil CO₂ efflux and soil temperature in the control site. A quadratic trend reveals a maximum CO₂ efflux at ~20 °C ($R^2 = 0.266$). Error bars represent ± standard error of the mean.

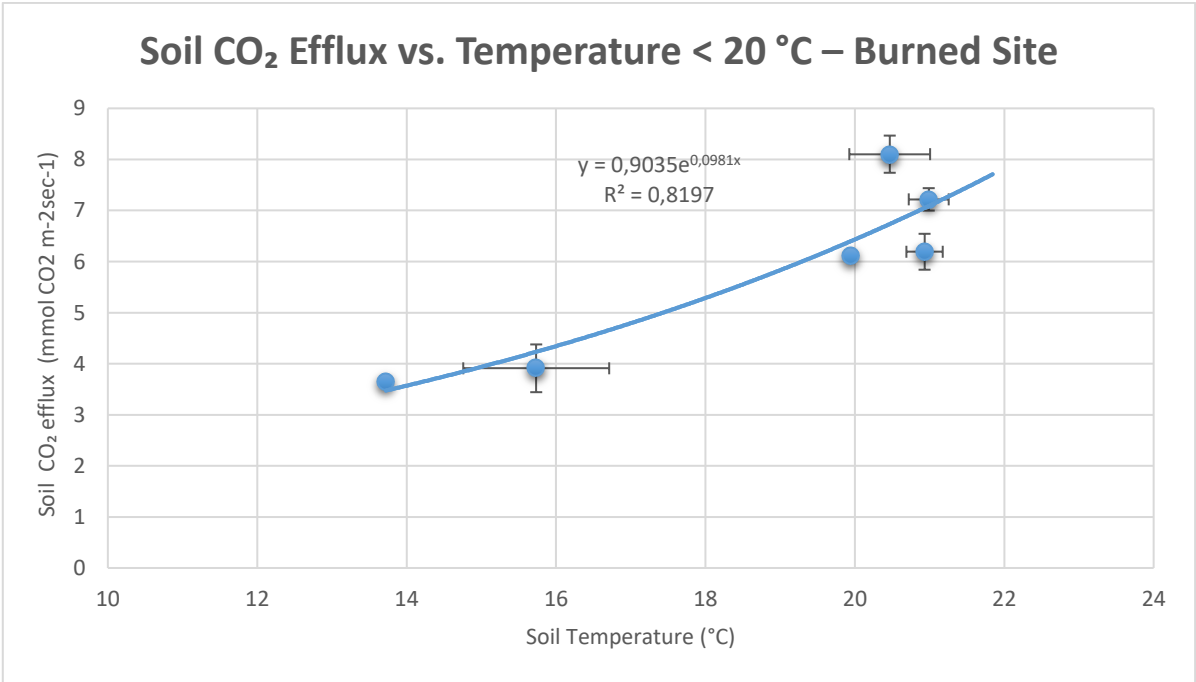


Figure 37 Soil CO₂ efflux in the control site under temperatures below 20 °C. The exponential trend indicates increased microbial and root respiration activity ($R^2 = 0.8197$). Error bars represent ± standard error of the mean.

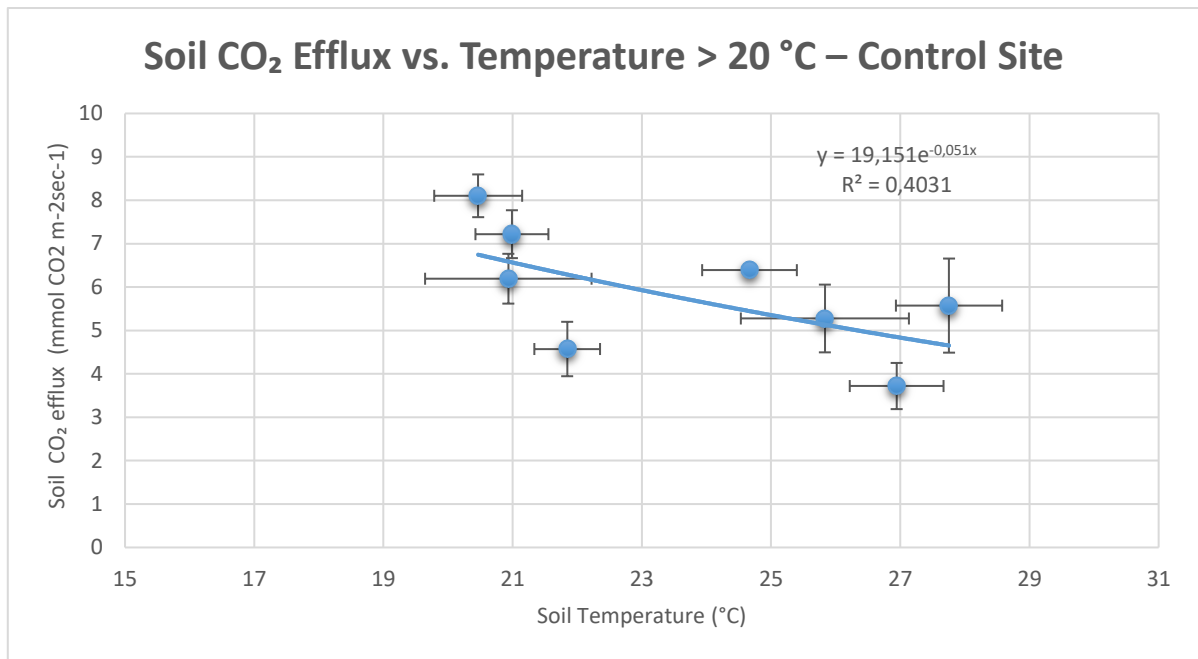


Figure 38 Soil CO₂ efflux above 20 °C in the control site shows a mild exponential decline, suggesting slight thermal stress ($R^2 = 0.4031$). Error bars represent $\pm$ standard error of the mean.

4.3.2.3. Influence of Soil Moisture on Soil CO₂ Efflux in Burned Site

In burned plots, soil CO₂ efflux showed a weak quadratic relationship with soil water content (SWC), peaking around 15% ($R^2 = 0.0268$; Figure 39). Both excessive moisture and dryness appeared to suppress efflux, possibly due to oxygen limitation or microbial dormancy. In contrast, control plots

exhibited a stable response across moisture ranges, reflecting the buffering effect of soil structure and higher organic content ($R^2 = 0.0251$; Figure 40).

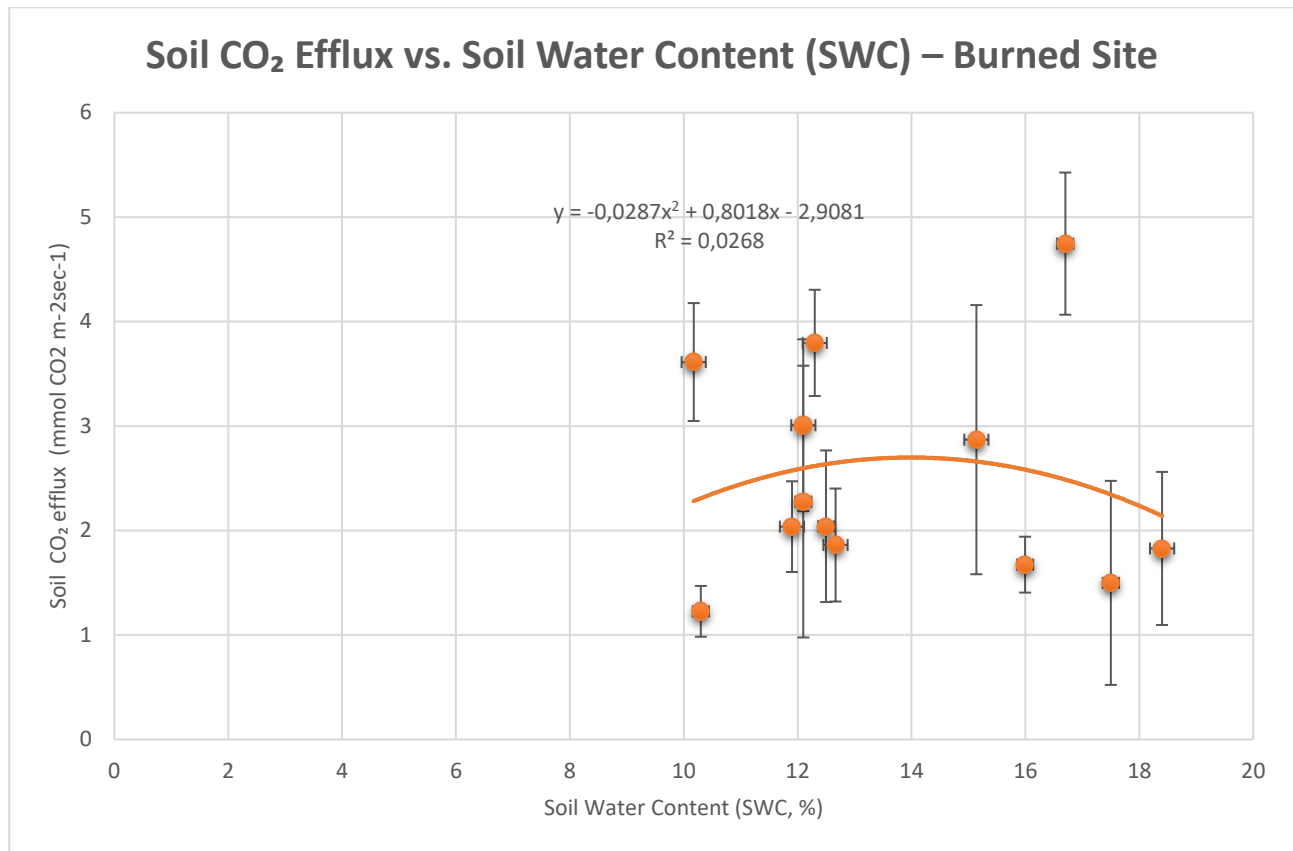


Figure 39 Relationship between soil CO₂ efflux and soil water content (SWC) in the burned site. A weak quadratic trend indicates peak efflux at ~15% SWC ($R^2 = 0.0268$), with declines under drier or wetter conditions. Error bars represent $\pm$ standard error

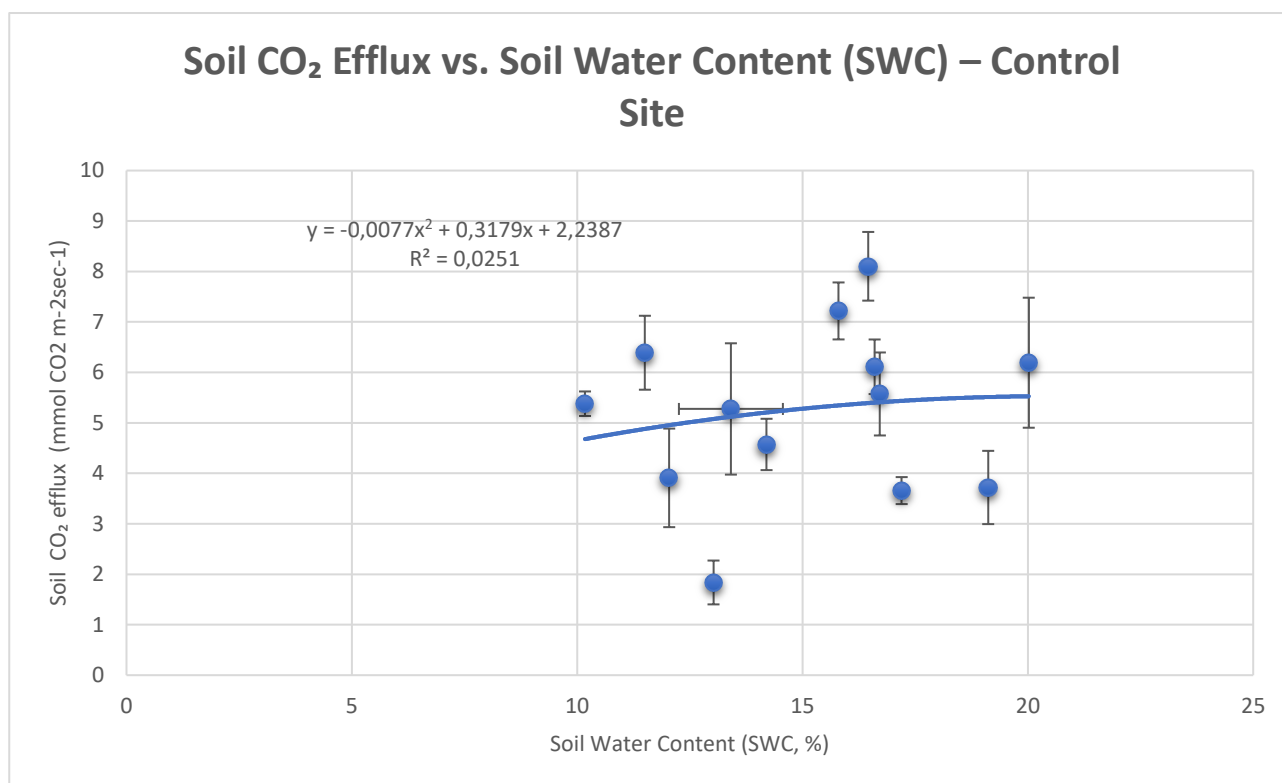


Figure 40 Relationship between soil CO₂ efflux and soil water content (SWC) in the control site. A near-flat trend suggests consistent efflux across a range of moisture levels ($R^2 = 0.0251$), demonstrating soil buffering capacity. Error bars represent $\pm$

4.3.2.4. Influence of Soil Temperature and Water Content Interaction on Soil CO₂ Efflux

Soil CO₂ Efflux vs. Soil Temperature by Site and SWC Class

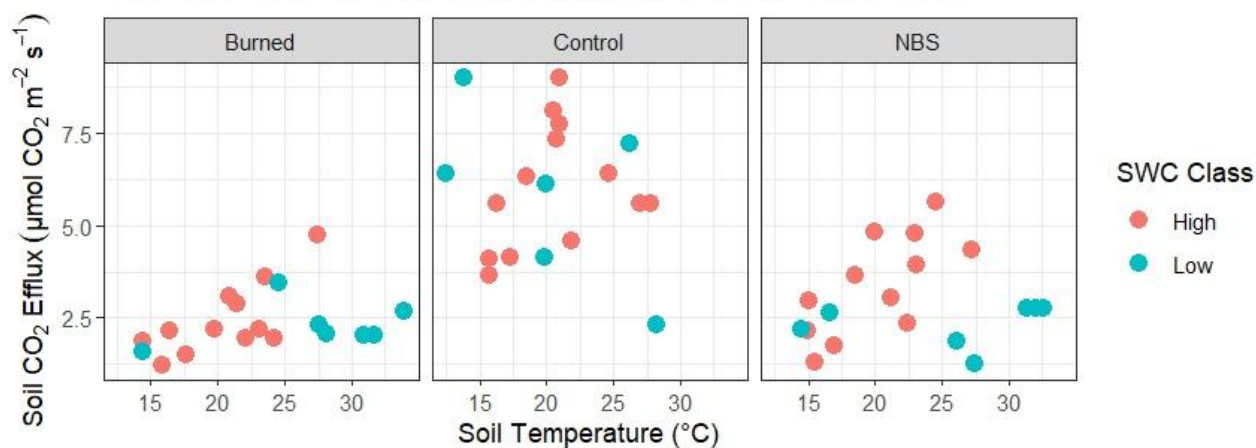


Figure 41 Relationship between soil CO₂ efflux ($\mu\text{mol CO}_2 \text{ m}^{-2} \text{ s}^{-1}$) and soil temperature ($^{\circ}\text{C}$) in burned, NBS and control sites, grouped by soil water content (SWC) class. Each point represents a measurement under high (red) or low (blue) SWC conditions.

Figure 41 shows the relationship between Soil CO₂ Efflux ($\mu\text{mol CO}_2 \text{ m}^{-2} \text{ s}^{-1}$) and Soil Temperature ($^{\circ}\text{C}$), stratified by site (Burned, Control, NBS) and Soil Water Content (SWC) class (High vs. Low).

Each point corresponds to a paired measurement of efflux and temperature under either high or low moisture availability.

The observed patterns indicate that the interaction between soil temperature and SWC class influences Soil CO₂ efflux differently across sites. In burned plots, Soil CO₂ effluxes were generally low and did not show a strong increase with rising temperatures, regardless of moisture class. This suggests that post-fire soil degradation may constrain microbial and root respiration even under favourable thermal or moisture conditions.

In control plots, Soil CO₂ Efflux values were significantly higher overall and showed a clear positive trend with temperature under high SWC conditions. By contrast, under low SWC, fluxes were more scattered and reduced, particularly at higher temperatures. This pattern suggests that adequate soil moisture enhances the temperature sensitivity of microbial and root activity, while drought conditions may limit CO₂ production despite optimal thermal ranges.

NBS plots exhibited intermediate behavior. Fluxes increased moderately with temperature under both moisture classes, though variability was higher compared to the control. This may reflect partial recovery of soil structure and microbial functionality due to the presence of deadwood, which supports microhabitat heterogeneity and moisture retention.

Overall, the combination of temperature and SWC appears to modulate Soil CO₂ Efflux in a site-dependent manner, with the control plots showing the most responsive and stable patterns, and the burned plots showing constrained fluxes under both dry and moist conditions. These findings underline the importance of restoring both thermal and hydrological soil buffering capacity to support post-fire soil carbon dynamics.

4.3.2.5. Q_{10} and Basal Respiration (BR)

To complement the analysis of Soil CO₂ Efflux dynamics and environmental drivers, BR and Q_{10} were derived from exponential temperature–response functions fitted under site-specific thresholds (20°C for Control, 25°C for Fire), following methods from Mediterranean studies (López-Serrano et al., 2018; Rey et al., 2011). Temperature Sensitivity (Q_{10}) (Figure 42) values were not significantly different among treatments (Control = 1.06 ± 0.11 ; Fire = 1.01 ± 0.05 ; Fire_NBS = 1.05 ± 0.05 ; $p = 0.889$) (Table 8).

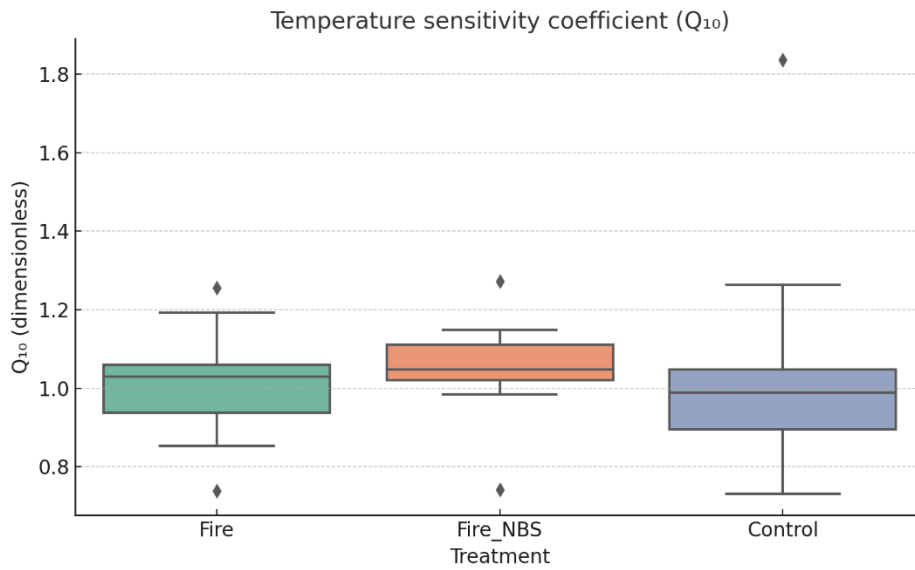


Figure 42 Temperature sensitivity coefficient (Q_{10}) of soil CO₂ efflux across fire-affected (Fire), Nature-Based Solution treated (Fire_NBS), and control plots. Boxplots represent median, interquartile range, and outliers.

Basal Respiration (BR) (Figure 43) was significantly higher in control plots ($5.26 \pm 0.31 \mu\text{mol CO}_2 \text{ m}^{-2} \text{ s}^{-1}$) than in Fire (2.29 ± 0.17) and Fire_NBS (2.52 ± 0.13) (ANOVA, $p < 0.001$; Tukey HSD) (Table 8). This confirms that fire reduces microbial biomass and root respiration, while NBS supports only partial recovery within the observed timeframe (Certini, 2005; Santonja et al., 2017).

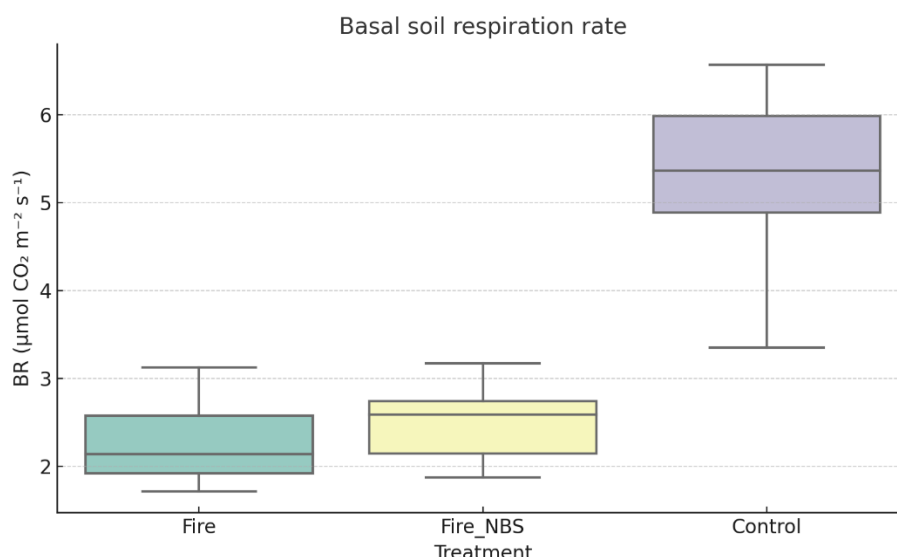


Figure 43 Basal soil respiration rate (BR; $\mu\text{mol CO}_2 \text{ m}^{-2} \text{ s}^{-1}$) across fire-affected (Fire), Nature-Based Solution treated (Fire_NBS), and control plots. Boxplots indicate median, interquartile range, and outliers.

Table 8 Q_{10} values and Basal Respiration (BR_{15} , $\mu\text{mol CO}_2 \text{ m}^{-2} \text{ s}^{-1}$) reported as mean $\pm$ standard error (SE) for degraded and control plots at the Morrone site. Tukey letters indicate group differences.

Treatment	Q_{10} (Mean $\pm$ SE)	Tukey	ANOVA p	BR (Mean $\pm$ SE, $\mu\text{mol CO}_2 \text{ m}^{-2} \text{ s}^{-1}$)	Tukey	ANOVA p
Control	1.06 $\pm$ 0.11	a	0.889	5.26 $\pm$ 0.31	a	< 0.001 *
Fire	1.01 $\pm$ 0.05	a		2.29 $\pm$ 0.17	b	
Fire_NBS	1.05 $\pm$ 0.05	a		2.52 $\pm$ 0.13	b	

4.3.2.6. Statistical Modeling of SR Drivers

The Levene test confirmed heteroscedasticity in soil temperature and SWC ($p < 0.05$), justifying non-parametric analyses. Kruskal-Wallis and Dunn's post-hoc tests indicated significantly higher Soil CO_2 Efflux in control than in Fire or Fire_NBS plots ($p < 0.05$), with no significant difference between Fire and Fire_NBS. Wilcoxon and Brunner-Munzel tests further confirmed lower SR under burned conditions, particularly in dry scenarios.

Linear Mixed Models (LMMs) with repeated measures demonstrated significant effects of temperature, SWC, and treatment interaction ($p < 0.001$). Control plots exhibited stronger Soil CO_2 Efflux responses to temperature, suggesting better functional integrity. Quantile regression analysis revealed that temperature most strongly influenced Soil CO_2 Efflux in lower quantiles ($Q_{0.25}$ – $Q_{0.5}$), while high SR values likely depended on additional factors such as substrate availability or microbial diversity (Wang et al., 2020) (Table 9-10).

Table 9 Summary of statistical tests evaluating Soil CO₂ Efflux (SR) dynamics and environmental drivers at the Monte Morrone site.

<i>Test</i>	<i>Variable(s)</i>	<i>Statistic / Estimate</i>	<i>p-value</i>
Levene Test	T, WC	F = 8.92 (T), F = 5.44 (WC)	< 0.001
Kruskal-Wallis	SR ~ Treatment	$\chi^2 = 73.38$	< 0.001
Dunn Post-hoc	SR ~ Treatment (pairwise)	Z = 8.72 (Ctrl > Fire)	< 0.001
Brunner-Munzel	SR ~ Burned/Unburned	WBM = 7.45	< 0.001
Bootstrap CI	SR difference	CI = [2.90, 4.66]	CI does not include 0
LMM (repeated)	SR ~ T, WC, Treatment	F = 85.32 (T), F = 66.47 (WC), F = 23.10 (T:Trt)	< 0.001
Quantile Regression	SR ~ T (per quantile)	R ² = 0.27 (Q0.25), R ² = 0.31 (Q0.5)	< 0.001

Table 10 Summary Interpretation Table

<i>Test</i>	<i>Variable(s)</i>	<i>Significant Results (p < 0.05)</i>	<i>Notes</i>
Levene Test	T, WC	Yes (heteroscedasticity)	Justifies non-parametric modeling
Kruskal-Wallis	SR ~ Treatment	Yes	Control > Fire, NBS *
Dunn Post-hoc	SR ~ Treatment (pairwise)	Control > Fire*, Control > NBS*	Fire vs NBS not significant
Wilcoxon	SR ~ Burned/Unburned	Fire < Control *	Especially in dry conditions
Brunner-Munzel	SR ~ Burned/Unburned	Confirmed Wilcoxon results	Robust to variance assumptions
Bootstrap CI	SR difference	Control > Fire confirmed	Narrow CI range
LMM (repeated)	SR ~ T, WC, Treatment	T*, WC*, Interaction T:Treatment*	Stronger response in Control
Quantile Regression	SR ~ T (per quantile)	Median and lower quantiles *	Indicates limited functional range in Fire

4.3.3. Soil Properties in Monte Morrone Site

The analysis of soil texture, bulk density, and pH highlights distinct alterations resulting from fire events and provides insight into the physical and chemical status of soils across burned and control plots in Monte Morrone.

4.3.3.1. Soil Properties: Texture, Bulk Density, and pH

The analysis of soil texture, bulk density, and pH reveals distinct patterns that underscore the interplay between fire disturbances and soil characteristics.

Table 11 Changes in Soil Properties Between Control and Burned Sites (2017 and 2017-2023)

<i>Soil Property</i>	<i>Control</i>	<i>Fire 2017</i>	<i>Fire 2017-2023</i>
Texture	Silt: 49.7%; Clay: 42.9%; Sand: 7.4%	Silt: 44.1%; Clay: 41.7%; Sand: 14.2%	Silt: 53.7%; Clay: 38.2%; Sand: 8.1%
Bulk Density (g cm ⁻³)	1.10 ± 0.06	1.12 ± 0.13	1.17 ± 0.04
pH	7.76 ± 0.05 (b)	7.80 ± 0.04 (b)	7.92 ± 0.07 (a)

Soils across all plots are predominantly classified as silty clay according to USDA soil taxonomy (USDA, 2005), a texture that typically ensures good water retention and moderate aeration. The Fire 2017 plot presents a notable increase in sand content (14.2%) compared to the control (7.4%), suggesting post-fire sediment redistribution driven by increased runoff and erosion following vegetation loss (Shakesby & Doerr, 2006; Pereira et al., 2020). In contrast, the Fire 2017-2023 plot exhibits higher silt content, possibly due to progressive deposition of finer materials or differential erosion (Table 11).

Bulk density values are slightly elevated in the fire-affected plots, with the Fire 2017-2023 site showing the highest value (1.17 g cm⁻³), indicative of potential post-fire compaction. Although not statistically significant, this trend is ecologically relevant (Table 11).

A significant increase in soil pH is observed in the Fire 2017-2023 plot (7.92) compared to the control (7.76), attributed to ash deposition and the volatilization of organic acids during combustion (Certini, 2005) (Table 11).

4.3.4. Companhia das Lazirias Site

4.3.4.1. Temporal Variations in Soil Respiration and Temperature

Overall, the measurements—mostly concentrated in winter and spring—do not reveal strong seasonal variability, partly due to limited data collection in the summer and autumn. The June 2023 measurement does not deviate substantially from the spring values.

The degraded site shows lower Soil CO₂ Efflux compared to the control, with greater variability between measurements (Figure 44). The t-test indicated a significant difference ($p < 0.001$) between Soil CO₂ Efflux in degraded and control plots, with lower flux in the degraded site (Figure 45; Table 12). These results suggest that reduced soil organic matter and altered structure in degraded areas may constrain CO₂ emissions.

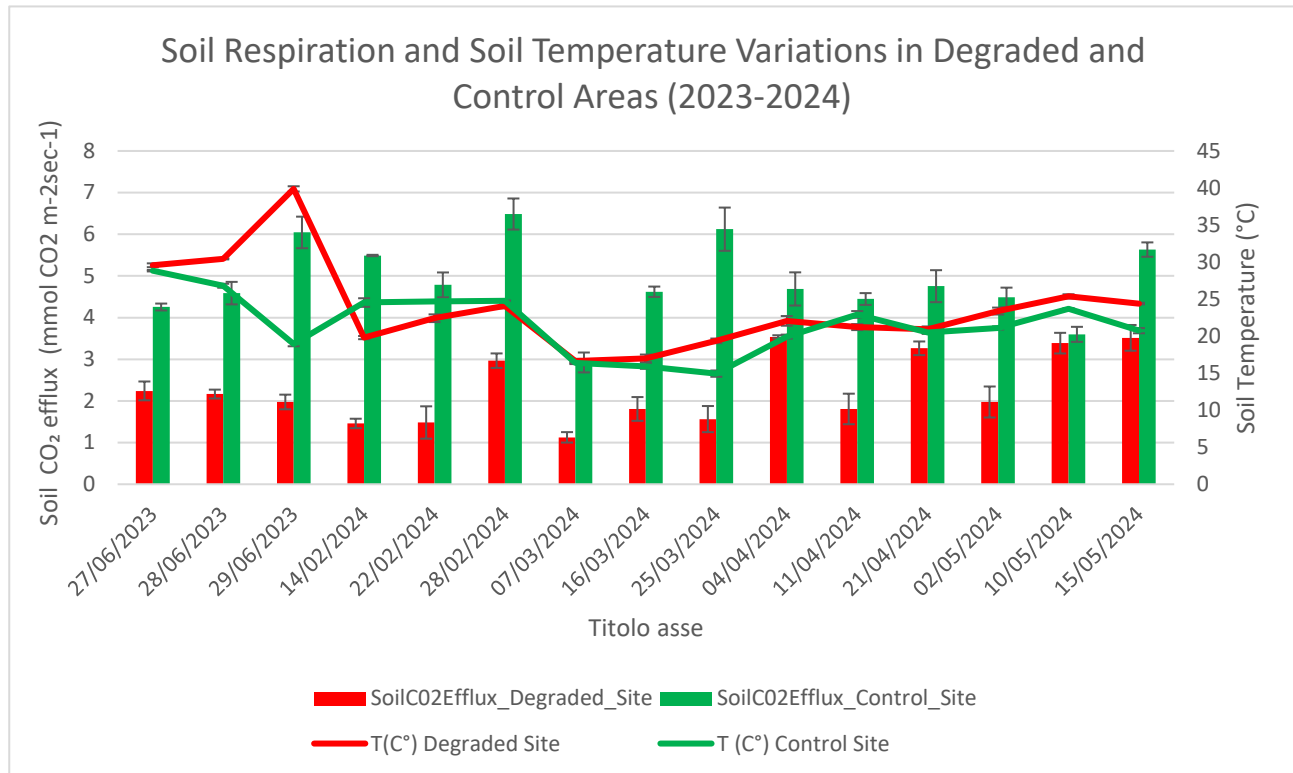


Figure 44 Temporal variation of soil CO₂ efflux (bars, left y-axis) and soil temperature (lines, right y-axis) in degraded (red) and control (green) plots from June 2023 to May 2024. Bars represent mean CO₂ efflux (mmol CO₂ m⁻² s⁻¹). Lines represent mean soil temperature (°C) over the same monitoring periods. Error bars represent $\pm$ standard error of the mean.

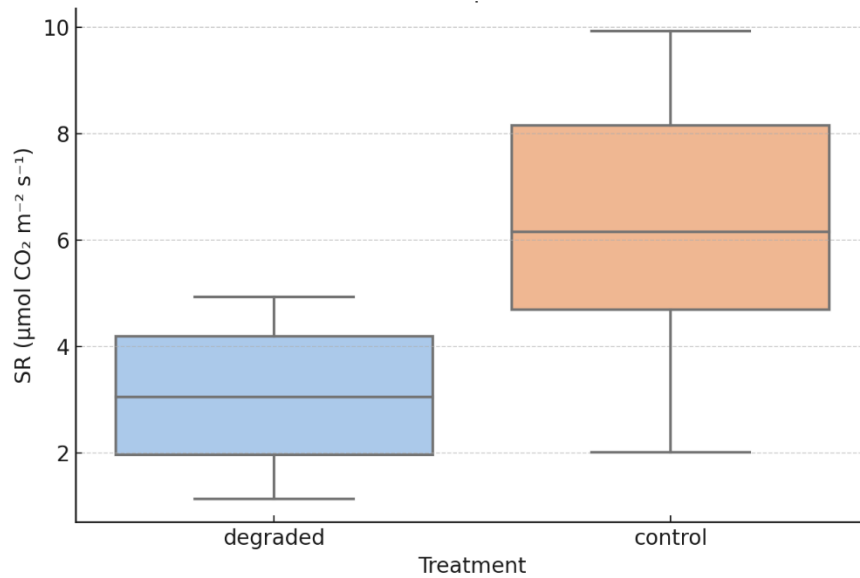


Figure 45 Boxplots and barplots represent soil respiration (SR, $\mu\text{mol CO}_2 \text{ m}^{-2} \text{ s}^{-1}$) measured in degraded and control plots. Bars indicate mean $\pm$ SE. Different letters denote significant differences (one-way ANOVA, $p < 0.001$; Tukey HSD, $p < 0.05$).

Table 12 Mean soil CO₂ efflux (SR, $\mu\text{mol CO}_2 \text{ m}^{-2} \text{ s}^{-1} \pm \text{SE}$) in control, fire-affected, and NBS-treated plots. Different letters indicate statistically significant differences between treatments based on Tukey HSD post-hoc test following one-way ANOVA ($p < 0.001$).

Thesis	SR (Mean $\pm$ SE, $\mu\text{mol CO}_2 \text{ m}^{-2} \text{ s}^{-1}$)	Tukey	ANOVA	Significance
			<i>p</i>	
Control	5.99 $\pm$ 0.27	a	< 0.001	***
Degraded	3.05 $\pm$ 0.25	b		

4.3.4.2. Temperature Dependency of SR

Figure 46 illustrates the relationship between Soil CO₂ Efflux and soil temperature in the degraded site. A clear parabolic trend is observed, with Soil CO₂ Efflux (SR) peaking around 25°C and declining beyond that point ($R^2 = 0.2992$), consistent with thermal optima for microbial activity. This response aligns with previous studies indicating enzyme inhibition and physiological stress at elevated temperatures (Davidson & Janssens, 2006).

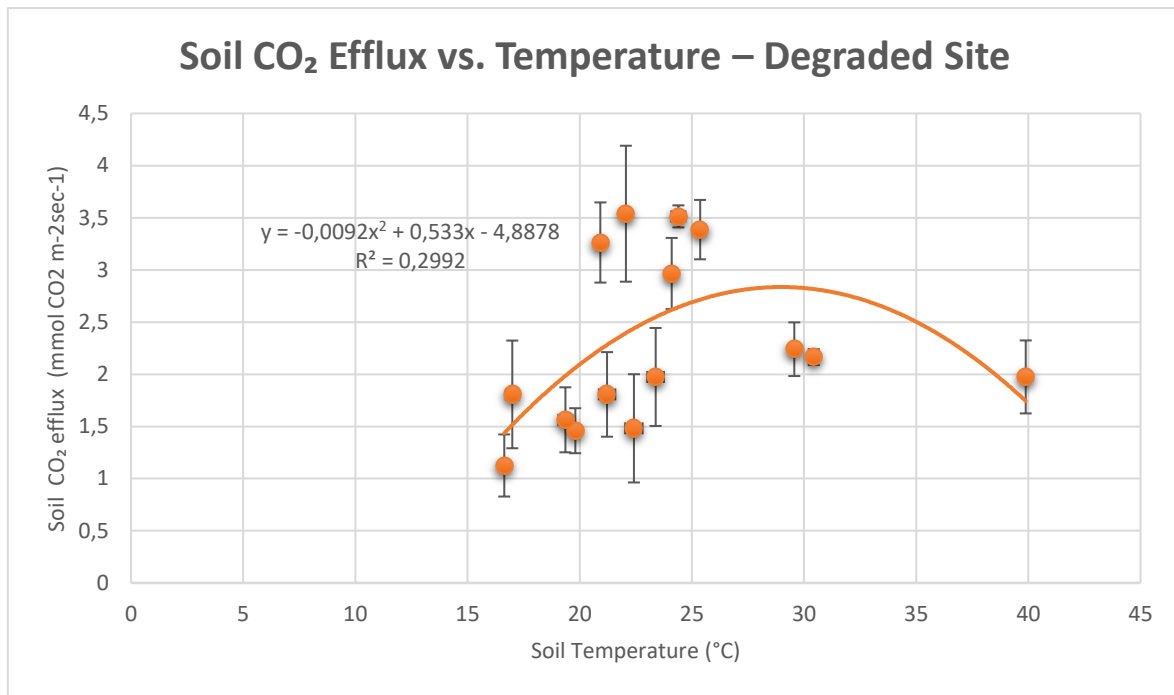


Figure 46 Relationship between soil respiration (SR) and soil temperature (T°C) in the degraded site. Orange points represent mean SR (mmol CO₂ m⁻² s⁻¹) for each sampling date. The second-degree polynomial regression line (R² = 0.2992) indicates a temperature optimum at ~25°C. Error bars represent ± standard error of the mean

When data are partitioned below and above 25°C, two distinct trends emerge. Figure 47 shows an exponential increase in Soil CO₂ Efflux at temperatures <25°C, with moderate sensitivity to warming (R² = 0.388). This suggests optimal microbial and root activity under cooler conditions.

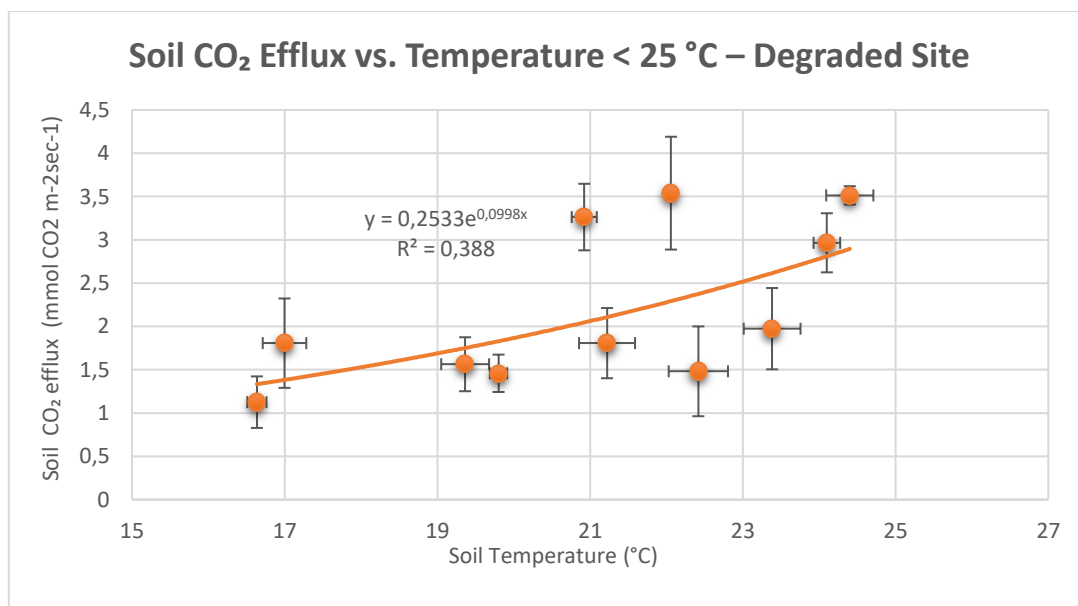


Figure 47 Relationship between soil respiration (SR) and soil temperature (T°C) below 25°C in the degraded site. An exponential function (R² = 0.388) captures the rising SR trend with increasing temperature. Error bars represent ± standard error of the mean.

In contrast, Figure 48 shows a clear exponential decline in Soil CO₂ Efflux above 25°C, with $R^2 = 0.6542$. This sharp drop likely reflects the compounding effects of thermal stress and reduced moisture availability in the degraded plots.

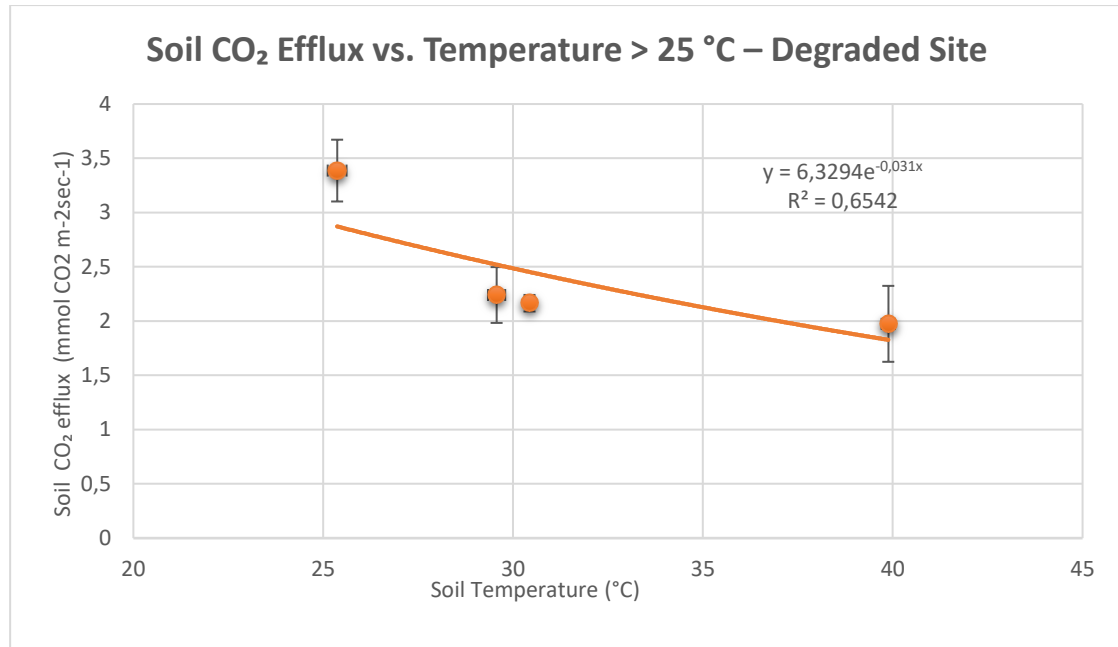


Figure 48 Soil respiration (SR) at temperatures exceeding 25°C in the degraded site. The exponential decay function ($R^2 = 0.6542$) indicates strong sensitivity of SR to heat-induced stress. Error bars represent $\pm$ standard error of the mean.

In the control site, Figure 49 shows a relatively flat relationship between Soil CO₂ Efflux and temperature, with a second-degree polynomial regression explaining only a small fraction of variance ($R^2 = 0.0108$). This suggests high resilience of the microbial community and soil structure to thermal fluctuations.

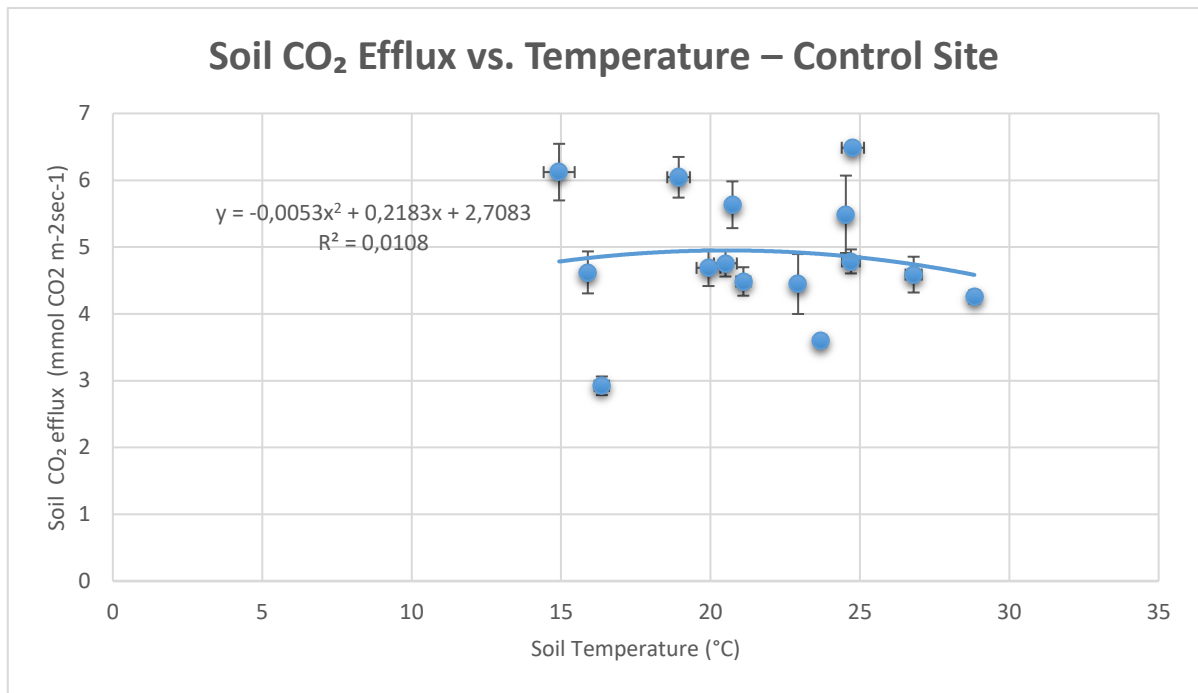


Figure 49 Relationship between soil respiration (SR) and soil temperature (T°C) in the control site. Blue points indicate mean SR values across time points. The second-degree polynomial regression line ($R^2 = 0.0108$) suggests weak temperature dependence. Error bars represent $\pm$ standard error of the mean.

To better understand the stability of Soil CO₂ Efflux under temperature thresholds, Figures 50 and 51 examine responses below and above 20°C in the control site. Figure 50 reveals a nearly flat trend below 20°C ($R^2 = 0.0515$), while Figure 51 shows a slight positive exponential trend ($R^2 = 0.1611$) above 20°C, still indicating low sensitivity.

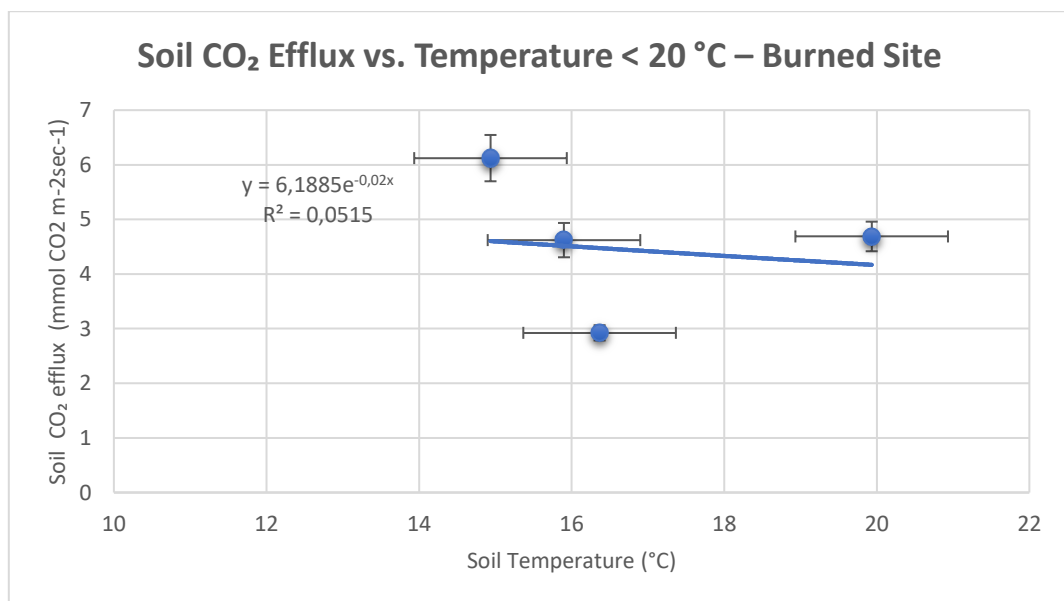


Figure 50 Soil respiration (SR) as a function of soil temperature (T°C) below 20°C in the control site. The exponential regression ($R^2 = 0.0515$) shows minimal change in SR with increasing temperature. Error bars represent $\pm$ standard error of the mean.

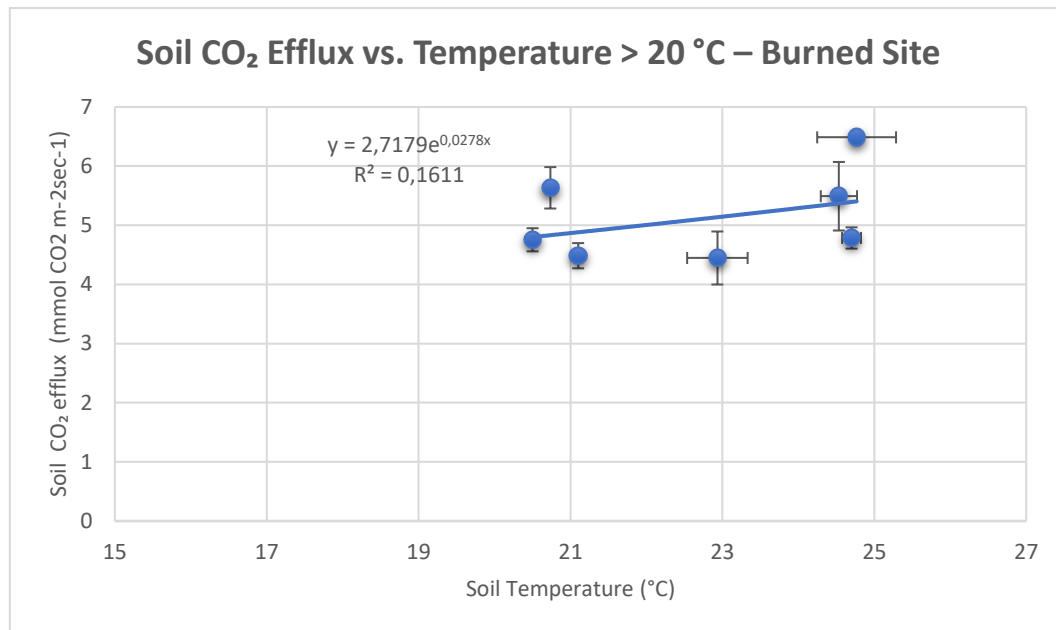


Figure 51 Relationship between soil respiration (SR) and soil temperature (T°C) above 20°C in the control site. A slight exponential increase is observed ($R^2 = 0.1611$), suggesting marginal sensitivity to warming. Error bars represent $\pm$ standard error of the mean.

4.3.4.3. Influence of Soil Moisture on Soil CO₂ Efflux

Figure 52 shows the relationship between Soil CO₂ Efflux and soil water content (SWC) in the degraded site. A second-degree polynomial regression reveals a clear moisture optimum at ~15% SWC ($R^2 = 0.3201$). Beyond this threshold, Soil CO₂ Efflux declines due to potential oxygen limitation under saturated conditions.

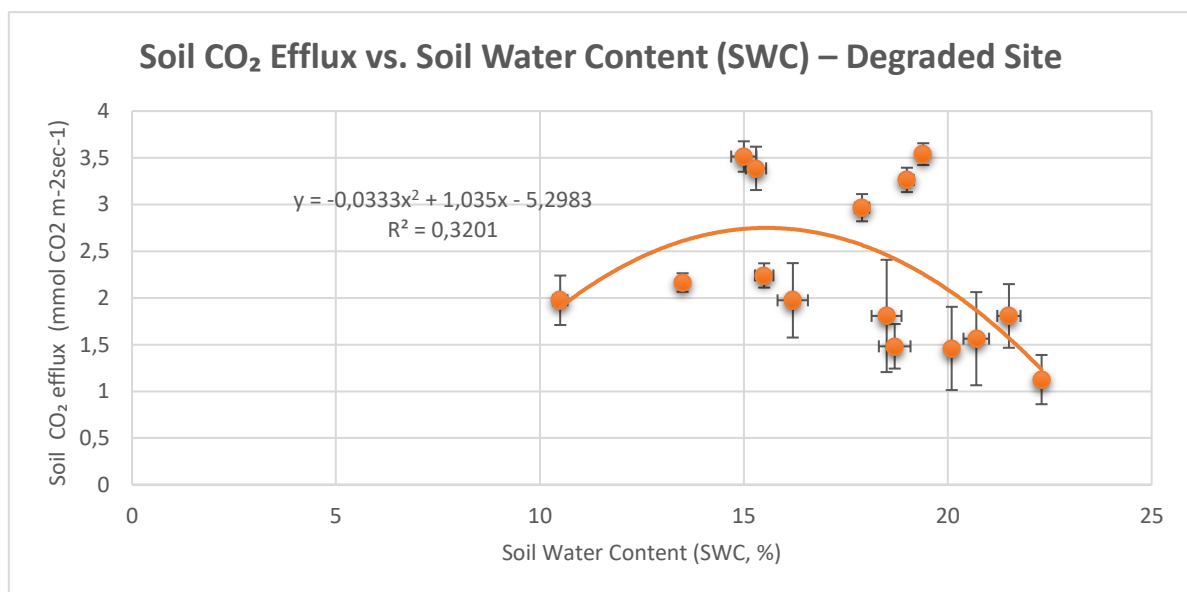


Figure 52 Soil respiration (SR) as a function of soil water content (SWC%) in the degraded site. The quadratic regression model ($R^2 = 0.3201$) highlights a peak in SR around 15% SWC, with a subsequent decline under wetter conditions. Error bars represent $\pm$ standard error of the mean.

In the control site (Figure 53), the quadratic relationship is nearly flat ($R^2 = 0.0544$), indicating the microbial community is well-buffered against moisture fluctuations, likely due to higher organic matter and soil porosity.

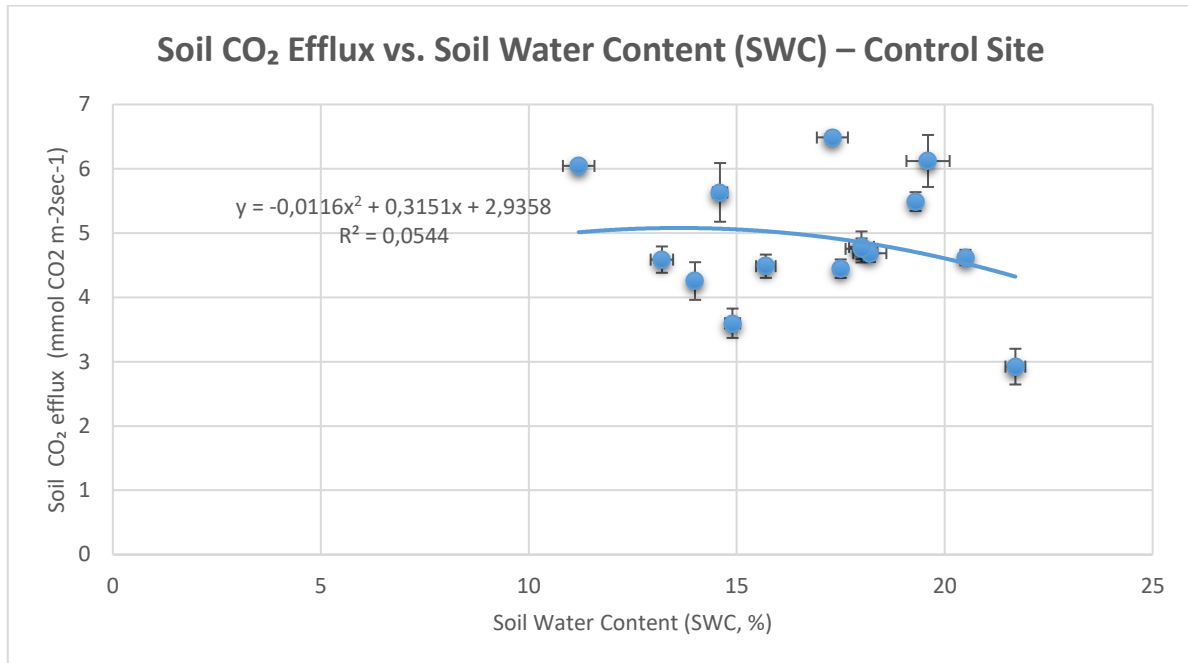


Figure 53 Relationship between Soil CO₂ Efflux (SR) and soil water content (SWC%) in the control site. Blue points represent mean SR values with a quadratic regression line ($R^2 = 0.0544$) showing low sensitivity to changes in moisture. Error bars represent $\pm$ standard error of the mean.

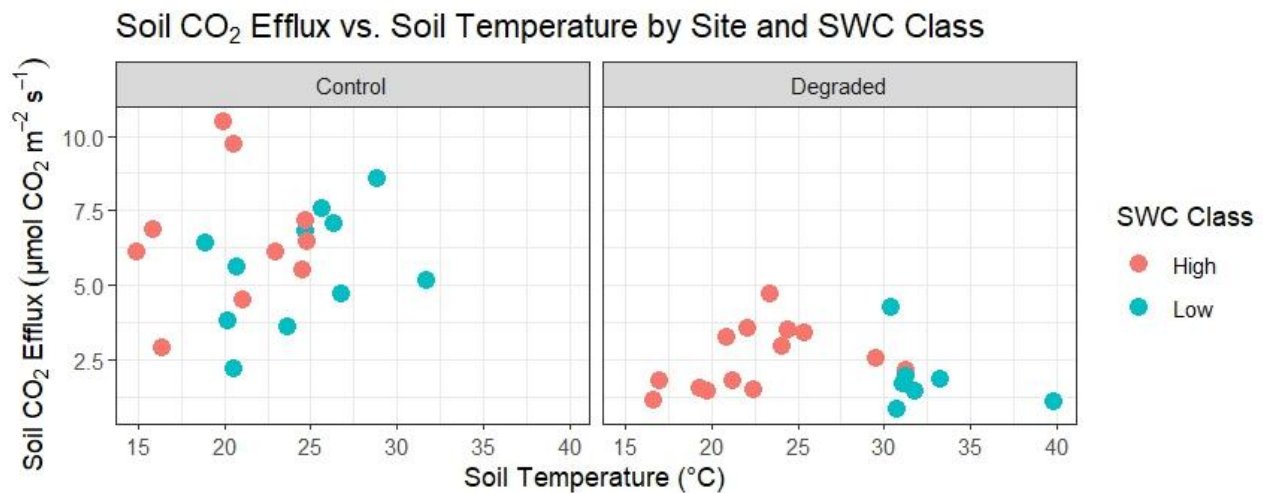


Figure 54 Relationship between soil respiration flux ($\mu\text{mol CO}_2 \text{ m}^{-2} \text{ s}^{-1}$) and soil temperature ($^{\circ}\text{C}$) in control and degraded plots, categorized by soil water content (SWC) class. Each point represents a measurement under high (red) or low (blue) SWC conditions.

Soil water content (SWC) strongly modulated the relationship between soil temperature and soil respiration (Soil CO₂ Efflux) in both the degraded and control sites (Figure 54). In degraded plots,

Soil CO₂ Efflux was generally lower and exhibited reduced responsiveness to temperature under both high and low SWC conditions. The highest flux values occurred around 25°C, particularly under high SWC, with a decline at both cooler and warmer temperatures, suggesting stress from either drought or potential oxygen limitation at higher moisture levels. In control plots, Soil CO₂ Efflux was more responsive to increasing temperatures, especially under high SWC conditions, and maintained elevated flux values even at warmer temperatures, indicating a more robust microbial and root system. The interaction between temperature and moisture class highlights the synergistic effects of thermal and hydrological controls on soil carbon fluxes, with degraded soils showing limited plasticity and functional range compared to the control.

4.3.4.4. Soil Temperature Differences Between Degraded Pastures and Oak Forests in Portugal

A notable microclimatic distinction exists between the two land uses: the oak forest control site, with dense canopy cover, maintains cooler and more stable soil temperatures compared to the open degraded pasture, which experiences greater thermal extremes due to solar exposure. This results in higher soil temperatures and diurnal variability in the degraded site (Ferreira et al., 2010; Correia et al., 2009).

Canopy shading in the control site moderates temperature and enhances soil moisture retention through evapotranspiration and organic matter input (Chaves et al., 2008). In contrast, the degraded site suffers from compaction, lower organic content, and poor moisture retention, limiting its capacity for evaporative cooling (David et al., 2013).

These findings are consistent with broader Mediterranean studies (Pereira et al., 2007), which emphasize the importance of tree cover for buffering against climate stress, sustaining microbial activity, and supporting Soil CO₂ Efflux. The oak forest's microclimatic stability plays a critical role in maintaining soil health and carbon cycling under changing environmental conditions.

4.3.4.5. Temperature Sensitivity (Q₁₀) and Basal Soil Respiration (BR) in Companhia das Lezírias

The Q₁₀ coefficient represents the temperature sensitivity of soil respiration (Soil CO₂ Efflux), indicating the factor by which Soil CO₂ Efflux increases with a 10°C rise in soil temperature. In Companhia das Lezírias, the mean Q₁₀ values were:

- Control: 1.00 ± 0.17 (SD), 0.06 (SE)
- Degraded: 1.00 ± 0.15 (SD), 0.05 (SE)

Statistical analysis using ANOVA showed no significant difference in Q_{10} values between treatments ($F = 0.0011$; $p = 0.975$), suggesting that the temperature responsiveness of microbial communities remains stable across both degraded and control plots (Figure 55; Table 13). These results align with findings by Schindlbacher et al. (2011) and Pietikäinen et al. (2005), which propose that microbial temperature response functions may be conserved even when microbial abundance or diversity is reduced due to environmental stress.

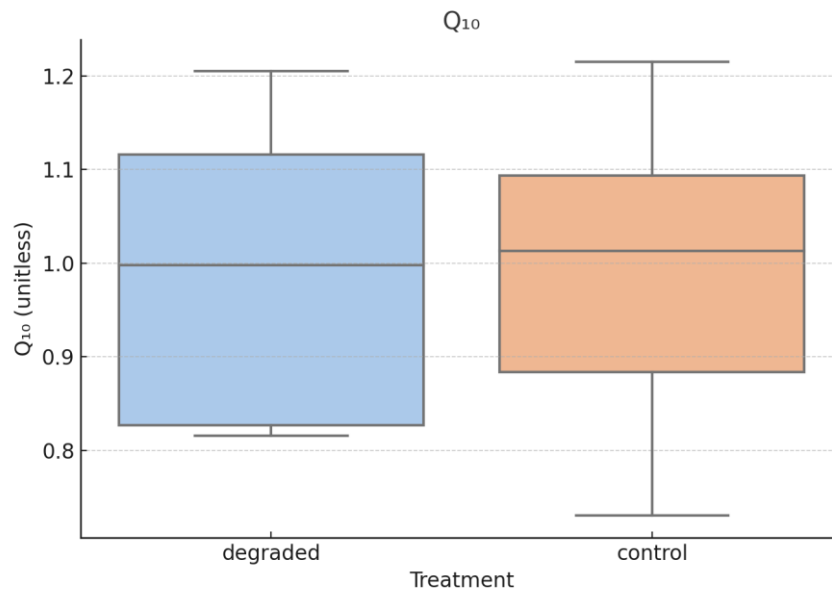


Figure 55 Boxplot of the temperature sensitivity coefficient (Q_{10}) in degraded and control plots at Companhia das Lezírias. Q_{10} reflects the rate of change in soil respiration with a 10 °C temperature increase. No significant differences were observed between the treatments ($p > 0.05$). Error bars represent $\pm$ standard error of the mean.

Basal respiration (BR), calculated at a standardized temperature of 15°C, provides a proxy for microbial and root metabolic activity under non-stress conditions. The results showed marked differences between degraded and control sites:

- Control: $5.91 \pm 0.48 \mu\text{mol CO}_2 \text{ m}^{-2} \text{ s}^{-1}$ (mean $\pm$ SE)
- Degraded: $2.75 \pm 0.17 \mu\text{mol CO}_2 \text{ m}^{-2} \text{ s}^{-1}$ (mean $\pm$ SE)

The ANOVA and Tukey HSD test revealed a statistically significant difference ($F = 15.66$; $p = 0.017$), with BR significantly higher in the control plots (Figure 55; Table 13). These findings suggest that, while temperature sensitivity (Q_{10}) is conserved, the degraded plots exhibit markedly reduced microbial or root activity. This decoupling between BR and Q_{10} highlights that Q_{10} alone is

insufficient to assess degradation, and that BR provides essential insight into ecosystem function (Fang et al., 2005).

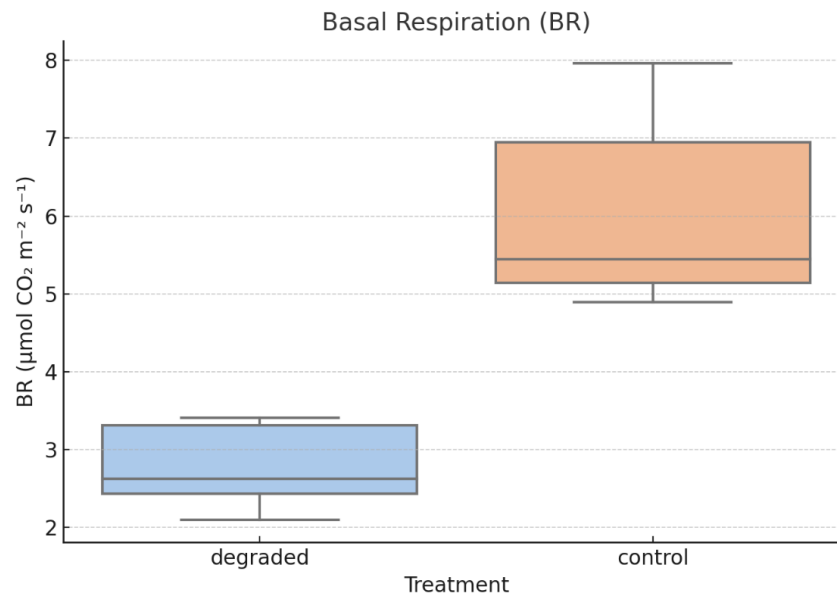


Figure 56 Boxplot of basal soil respiration (BR) at 15 °C in degraded and control plots at Companhia das Lezírias. BR was significantly higher in the control site ($p < 0.05$), indicating greater microbial/root metabolic activity under non-stress temperature. Error bars represent $\pm$ standard error of the mean.

Table 13 Q_{10} values and Basal Respiration (BR_{15} , $\mu\text{mol CO}_2 \text{ m}^{-2} \text{ s}^{-1}$) reported as mean $\pm$ standard error (SE) for degraded and control plots at the Companhia das Lezírias site. Tukey letters indicate group differences.

Treatment	Q_{10} (Mean $\pm$ SE)	Tukey (Q_{10})	ANOVA p (Q_{10})	BR_{15} (Mean $\pm$ SE, $\mu\text{mol CO}_2 \text{ m}^{-2} \text{ s}^{-1}$)	Tukey (BR_{15})	ANOVA p (BR_{15})
Control	1.00 ± 0.06	a	0.996	5.91 ± 0.16	a	< 0.001 ***
Degraded	1.00 ± 0.05	a		2.75 ± 0.17	b	

4.3.4.6. Statistical Modelling and Environmental Controls on Soil Respiration

The statistical analyses confirmed significant differences in Soil CO_2 Efflux (SR) between degraded and control plots at the Companhia das Lezírias site. The Levene test indicated heteroscedasticity for soil temperature (T), water content (WC), and organic matter (OM), justifying the use of non-parametric methods (Table 14). A Kruskal–Wallis test followed by Dunn’s post-hoc revealed significantly higher SR in control plots compared to degraded plots ($p < 0.001$), a pattern further supported by the Brunner–Munzel test ($p = 0.032$), particularly under low water content conditions (Table 14; Table 15).

A linear mixed model (LMM) showed that soil temperature, water content, organic matter, and treatment significantly influenced Soil CO₂ Efflux (SR), with interaction effects also detected (temperature × treatment, $p < 0.001$) (Table 14). Similarly, the generalized linear mixed model (GLMM) confirmed that all fixed effects (T, WC, OM, treatment) were significant predictors of SR ($p < 0.01$), with relevant interactions between treatment and both T and WC (Table 14).

Quantile regression results indicated that, in the control site, soil temperature was a significant predictor of Soil CO₂ Efflux (SR) at lower quantiles (Q0.25, $p = 0.001$; Q0.5, $p = 0.045$). In contrast, no significant pattern was observed in the degraded site, suggesting a weaker environmental control on Soil CO₂ Efflux (SR) (Table 14; Table 15).

Table 14 Summary of statistical tests evaluating Soil CO₂ Efflux (SR) differences and environmental drivers at the Companhia das Lezírias site.

<i>Test</i>	<i>Variable(s)</i>	<i>Statistic / Estimate</i>	<i>p-value</i>
Levene Test	T, WC, OM	F = 6.71 (T), F = 5.29 (WC), F = 4.12 (OM)	< 0.001
Kruskal-Wallis	SR ~ Treatment	$\chi^2 = 48.12$	< 0.001
Dunn Post-hoc	SR ~ Treatment (pairwise)	Z = 6.93 (Control > Degraded)	< 0.001
Brunner-Munzel	SR ~ Degraded/Control	WBM = 2.17	0.032
Bootstrap CI	SR difference	CI = [2.36, 4.02]	< 0.001
LMM (repeated)	SR ~ T, WC, OM, Treatment	F = 33.78 (T), F = 21.63 (WC), F = 10.34 (OM), F = 6.29 (T:Trt)	< 0.001
Quantile Regression	SR ~ T (per quantile)	R ² = 0.24 (Q0.25), R ² = 0.19 (Q0.5)	0.001 / 0.045
GLMM	SR ~ T, WC, OM, Treatment	$p < 0.01$ (all fixed effects); interaction T×Trt, WC×Trt significant	< 0.01

Table 15 Summary Interpretation Table

<i>Test</i>	<i>Variable(s)</i>	<i>Significant Results ($p < 0.05$)</i>	<i>Notes</i>
Levene Test	T, WC, OM	Yes	Justifies non-parametric modeling
Kruskal-Wallis	SR ~ Treatment	Yes ($p < 0.001$)	Control > Degraded

Dunn Post-hoc	SR ~ Treatment (pairwise)	Control > Degraded (p < 0.001)	Confirms pairwise difference
Brunner-Munzel	SR ~ Degraded/Control	Degraded < Control (p = 0.032)	Especially under low WC
Bootstrap CI	SR difference	Difference confirmed (p < 0.001)	Robust confirmation of significant SR difference
LMM (repeated)	SR ~ T, WC, OM, Treatment	T***, WC***, OM***, T: Treatment***	Stronger SR response to predictors in control plots
Quantile Regression	SR ~ T (per quantile)	Control: Q0.25 (p = 0.001), Q0.5 (p = 0.045)	Weak or no pattern in degraded site
GLMM	SR ~ T, WC, OM, Treatment	All fixed effects p < 0.01; interactions significant	Interaction effects: T × Treatment, WC × Treatment

4.3.5. Soil Properties in Companhia das Lezírias Monitoring Site

4.3.5.1. Introduction

Soil respiration (Soil CO₂ Efflux), primarily driven by microbial and root activity, is a key indicator of ecosystem health and carbon cycling in agro-silvo-pastoral systems such as Companhia das Lezírias (CL). It reflects CO₂ emissions resulting from organic matter decomposition and microbial activity (Davidson et al., 2002). Understanding the influence of soil properties on SR is particularly relevant in Mediterranean environments, where temperature and moisture fluctuations shape soil processes.

In this chapter, we present empirical findings on the relationship between Soil CO₂ Efflux and measured soil properties — including nitrogen (N%), carbon (C%), pH, electrical conductivity (EC), water content (WC%), organic matter (OM%), total organic carbon (TOC), inorganic carbon (IC), total carbon (TC), porosity, and soil temperature (T°) — from both degraded and control plots.

4.3.5.2. Soil Property Comparison between Degraded and Control Sites

The analysis (Table 16) revealed significant differences in soil properties between degraded and control sites. Sampling was conducted in February 2024, during the late winter period, which precedes the typical spring flush of microbial and root activity in Mediterranean environments. Table 16 summarizes the mean values and statistical differences observed between sites.

Table 16 Comparison of soil properties between degraded and control sites (n = 9 per site). Sampling was conducted in July 2023. Asterisks indicate significance levels: *p < 0.05, **p < 0.01, *p < 0.001.**

Soil Property	Degraded Mean	Degraded DVS	Degraded Error	Degraded CI	Control Mean	Control DVS	Control Error	Control CI	P-value
N%	0.185	0.062	0.021	0.046	0.08	0.019	0.006	0.013	***
C%	2.627	0.816	0.272	0.615	0.997	0.232	0.077	0.162	***
pH	6.042	0.192	0.064	0.145	7.026	0.222	0.074	0.155	***
EC mS/cm	178.244	36.332	12.111	27.395	131.933	24.362	8.121	16.997	***
WC%	16.86	3.312	1.104	2.497	9.076	2.351	0.784	1.64	***
OM%	6.649	6.873	0.775	1.753	5.79	4.668	1.049	2.196	0.504
TOC mg/L	137.734	25.066	8.355	18.9	74.182	19.805	6.602	13.817	***
IC mg/L	1.097	0.132	0.044	0.1	1.423	0.406	0.135	0.283	***
TC mg/L	138.831	25.105	8.368	18.929	75.605	19.873	6.624	13.865	***
Porosity	0.35	0.01	0.333	0.769	0.5	0.01	0.333	0.769	***
T° Soil (°C)	25,3084	1,09243	0,23838	0,46724	19,630	2,0890	0,4558	0,8935	***
	4444	0013	7773	0034	88889	95498	78012	209	

Empirical results indicate significantly higher water content, TOC, EC, and soil temperature in the degraded site, consistent with fire-induced alterations and canopy loss. Notably, the pH was lower in the degraded site, which may suppress microbial diversity and favor acid-tolerant taxa (Pietikäinen et al., 2005). Porosity was significantly reduced, reflecting compaction likely due to grazing and loss of vegetation.

4.3.5.3. Influence of Soil Properties on Soil Respiration

To explore the relationship between soil properties and SR fluxes, multiple linear regression analyses were performed separately for degraded and control sites. Due to multicollinearity among variables (e.g., TOC, N, pH), highly correlated predictors were excluded based on VIF thresholds, following Dormann et al. (2013) (Table 17-18).

Table 17 Multiple linear regression results for SR in the control site. Asterisk (*) indicates statistical significance (p < 0.05).

Variable	Estimate	Std. Error	P-Value	VIF
(Intercept)	4.85	0.21	0.002	-
T_Control	-0.0012	0.0010	0.624	1.45
WC	0.0036	0.0015	0.048*	1.32
TOC	0.0029	0.0014	0.053	1.47

pH	-0.0048	0.0021	0.040*	1.40
-----------	---------	--------	--------	------

Soil CO₂ Efflux in the control site was positively associated with WC% and marginally with TOC, suggesting favorable aerobic conditions for microbial activity. The negative pH effect indicates reduced respiration under acidified conditions (Table 17).

The degraded site, characterized by reduced vegetation and increased exposure to environmental stressors, exhibited different patterns of Soil CO₂ Efflux drivers compared to the control site. The regression results for the degraded site are shown in Table 18, with statistically significant predictors marked with "**"

Table 18 Multiple Linear Regression Results for Soil Respiration in the Degraded Site

<i>Variable</i>	<i>Estimate</i>	<i>Std. Error</i>	<i>P-Value</i>	<i>VIF</i>
(Intercept)	2.15	0.30	0.001	-
T_Degraded	0.0095	0.0025	0.012*	1.39
WC	-0.0061	0.0021	0.042*	1.45
TOC	-0.0053	0.0023	0.048*	1.50
pH	-0.0038	0.0018	0.046*	1.32

Unlike the control site, Soil CO₂ Efflux in the degraded site was strongly temperature-dependent and negatively affected by excess water, possibly due to anaerobic conditions in compacted soils. The negative impact of TOC may reflect the presence of less labile organic material (Moyano et al., 2012).

4.3.5.4. Ecological Implications and Comparative Insights

These contrasting drivers underscore how degradation shifts the biophysical controls of Soil CO₂ Efflux. While the control site benefits from structural and chemical stability, the degraded site exhibits heightened sensitivity to heat and moisture stress. These vulnerabilities may be exacerbated under climate change, particularly during extreme summer conditions (Table 19).

4.3.5.5. Summary of Soil Property Differences

Table 19 Summary of Changes in Soil Properties Between Degraded and Control Sites

<i>Soil Property</i>	<i>Degraded Site</i>	<i>Control Site</i>	<i>Primary Impact</i>
Nitrogen (N%)	Increased	Lower	Altered microbial cycling
pH	Decreased	Neutral	Suppressed microbial activity
Electrical Conductivity (EC)	Increased	Lower	Salinity stress on microbes

Water Content (WC%)	Higher	Balanced	Anaerobic stress in degraded site
Organic Matter (OM%)	Higher (recalcitrant)	Lower (labile)	Reduced decomposition in degraded
Total Organic Carbon (TOC)	Higher	Lower	Low-quality carbon in degraded
Soil Temperature (T°)	Higher	Lower	Thermal stress in degraded
Porosity	Lower	Higher	Compaction limits aeration

4.4. Soil Microbial Community Composition and Functional Groups in Burned and Control Soils

4.4.1. Bacterial Community Composition

4.4.1.1. Relative Abundance of Bacterial Families

The analysis of 16S rRNA gene sequences revealed clear differences in bacterial family composition between burned, control, and Nature-Based Solutions (NBS) plots. The dataset includes bacterial reads from 5 cm and 10 cm depth layers collected during spring 2022 sampling campaign (Figure 57, Table 20).

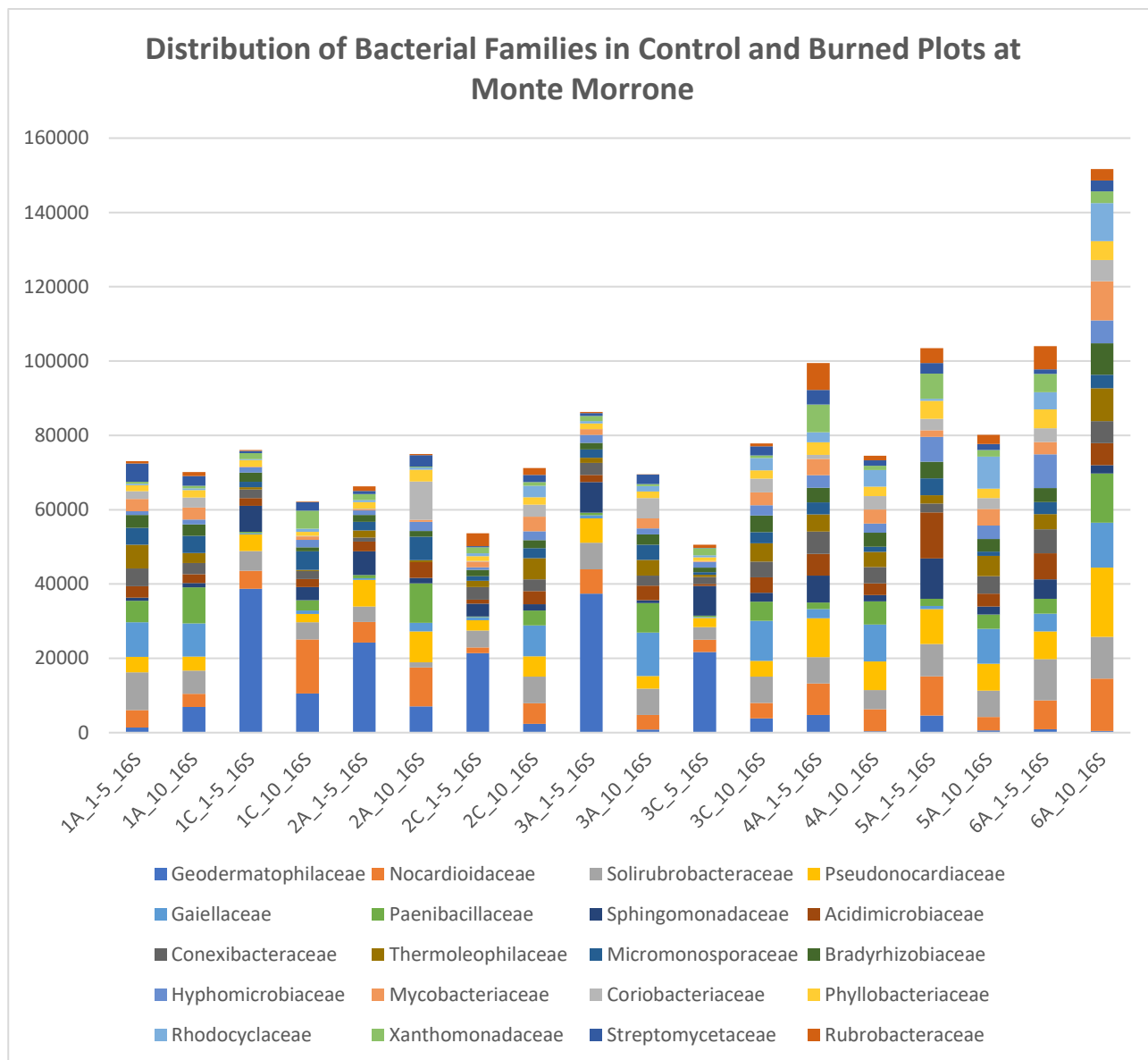


Figure 57 Distribution of the 20 most abundant bacterial families across soil plots. Each bar shows the cumulative relative abundance (%) of dominant families in control, burned, and NBS-treated burned soils. NBS plots include those where burned logs were left in situ.

Table 20 Relative Abundance of the 20 Most Abundant Bacterial Families Across Soil Samples. The table presents the relative abundance of the 20 most abundant bacterial families identified in the soil samples. The values indicate the total RITS (Reads of Interest per Taxonomic Sequence), representing unique sequences associated with each bacterial family in the respective sample. The sample IDs in the header correspond to the plot and depth: "5" denotes 5 cm depth, and "10" denotes 10 cm depth. Burned plots are labeled as "C," burned plots treated with Nature-Based Solutions (NBS) as "A," and control plots as "A."

FAMILY	1A 5	1A 10	1C 5	1C 10	2A 5	2A 10	2C 5	2C 10	3A 5	3A 10	3C 5	3C 10	4A 5	4A 10	5A 5	5C 10	6A 5	6A 10
GEODERMATOP HILACEAE	1400	6920	38722	10543	24221	7045	21374	2344	37382	858	21703	3830	4749	390	4642	596	916	425
NOCARDIOIDAC EAE	4663	3535	4846	14572	5558	10503	1585	5571	6554	3912	3333	4187	8505	5882	10495	3602	7805	14103
SOLIRUBROBAC TERACEAE	10191	6210	5332	4565	4174	1389	4552	7144	7157	7085	3347	7043	7032	5216	8740	7112	11062	11272
PSEUDONOCAR DIACEAE	4145	3771	4441	2285	7149	8341	2736	5478	6568	3356	2456	4240	10521	7695	9337	7243	7471	18594

GAIELLACEAE	928 4	89 79	337	918	490	231 1	824	83 13	866	117 58	279	107 94	246 3	99 41	907	93 50	475 7	121 26
PAENIBACILLA CEAE	578 1	97 15	370	274 2	835	106 25	214	40 08	648	790 4	395	514 6	177 9	61 99	192 2	38 52	403 6	132 50
SPHINGOMONA DACEAE	837	11 52	702 4	357 3	639 8	146 1	344 8	17 06	823 7	777	799 6	236 4	721 8	16 93	108 16	21 91	518 0	221 3
ACIDIMICROBIA CEAE	308 2	23 44	206 4	213 3	257 9	417 4	106 7	35 61	193 7	387 8	489	417 1	582 0	31 29	124 47	34 31	704 4	591 5
CONEXIBACTER ACEAE	476 6	29 87	228 2	217 3	112 3	0	338 3	31 14	322 6	270 6	189 8	424 8	602 5	44 14	238 6	47 54	645 5	597 7
THERMOLEOPHI LACEAE	646 3	27 31	663	310	192 3	591	170 2	57 14	144 6	424 2	570	490 3	462 2	41 03	222 0	54 53	411 2	885 4
MICROMONOSP ORACEAE	447 8	46 15	146 2	502 6	227 3	626 9	120 1	26 81	216 9	408 9	749	304 5	327 4	14 01	455 6	11 55	322 0	355 3
BRADYRHIZOBI ACEAE	349 0	30 54	244 5	104 9	182 4	155 4	168 4	22 05	182 6	278 8	120 0	443 6	388 1	37 38	440 4	33 71	372 6	846 5
HYPHOMICROBI ACEAE	991	13 52	147 3	196 9	126 9	247 5	729	23 52	212 9	163 4	155 0	280 5	339 0	24 59	676 0	36 23	909 5	622 6
MYCOBACTERI ACEAE	330 4	32 35	105	922	284	532	151 6	39 59	152 6	266 3	147	343 8	436 6	37 50	173 1	44 60	337 4	104 96
CORIOBACTERI ACEAE	210 1	26 66	40	150	172	102 99	198	31 76	49	548 2	39	374 4	112 6	36 43	311 8	29 65	362 8	570 1
PHYLLOBACTER IACEAE	161 1	19 91	177 1	109 5	174 6	320 4	125 2	20 44	148 8	180 4	102 3	217 1	335 6	25 45	484 3	24 82	512 3	509 9
RHODOCYCLAC EAE	366	50 1	256	830	556	573	794	30 64	654	139 4	517	333 3	272 0	44 47	525	86 64	458 5	102 83
XANTHOMONA DACEAE	547	64 9	157 2	488 5	164 7	256	163 7	10 00	137 7	597	194 4	708	746 6	11 57	674 3	17 21	498 4	310 8
STREPTOMYCET ACEAE	496 4	26 17	626	228 7	820	307 7	324	19 36	790	248 5	66	245 8	391 1	14 79	291 4	16 70	119 1	296 5
RUBROBACTER ACEAE	624	11 33	268	143	124 9	319	341 2	18 64	283	34	854	769	722 0	12 58	399 8	24 46	626 0	303 4

Sampling codes: depth “5” = 5 cm, “10” = 10 cm; burned plots = "C", NBS plots = "A", control plots = "A".

Control plots showed higher relative abundances of *Geodermatophilaceae*, *Solirubrobacteraceae*, and *Acidimicrobiaceae*, especially at 10 cm depth. Burned plots had increased presence of *Thermoleophilaceae*, *Pseudonocardiaceae*, *Gaiellaceae*, and *Paenibacillaceae*. These differences were evident across both depth layers.

4.4.1.2. Depth-Specific Bacterial Distribution

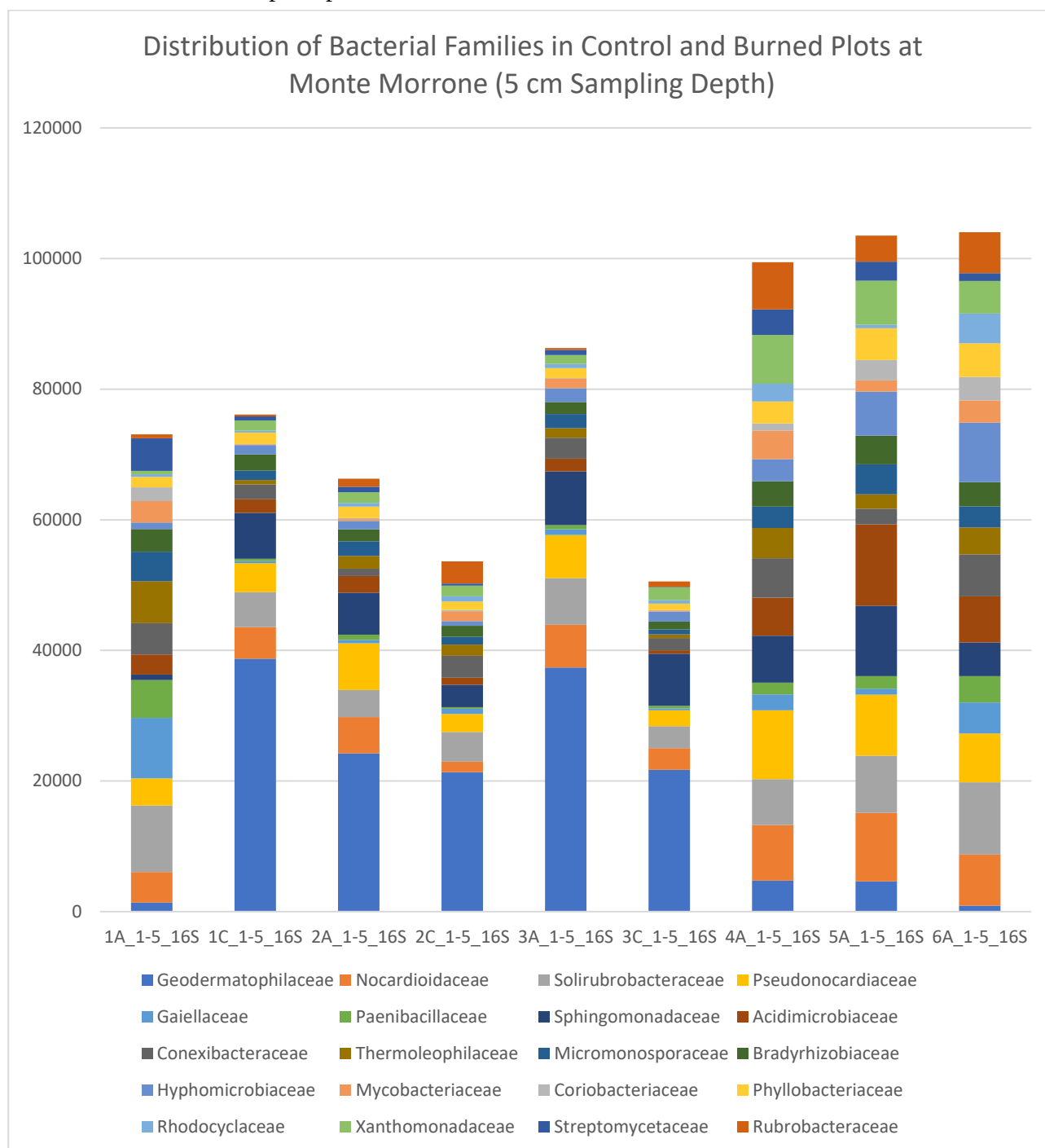


Figure 58 Distribution of bacterial families in control and burned plots at Monte Morrone (5 cm sampling depth). This chart shows the relative abundance of the 20 most abundant bacterial families within the 5 cm soil layer of each sampling plot. Control plots (4A_1-5_16S, 5A_1-5_16S, 6A_1-5_16S) are compared with burned plots (1A_1-5_16S, 1C_1-5_16S, 2A_1-5_16S, 2C_1-5_16S, 3A_1-5_16S, 3C_1-5_16S). Among the burned plots, those incorporating Nature-Based Solutions (NBS: 1A_1-5_16S, 2A_1-5_16S, 3A_1-5_16S) are specifically highlighted to assess the effects of leaving burned logs on the soil to prevent erosion and enhance microbial activity. Each bar segment represents a bacterial family, with its height corresponding to the proportional abundance in the respective plot.

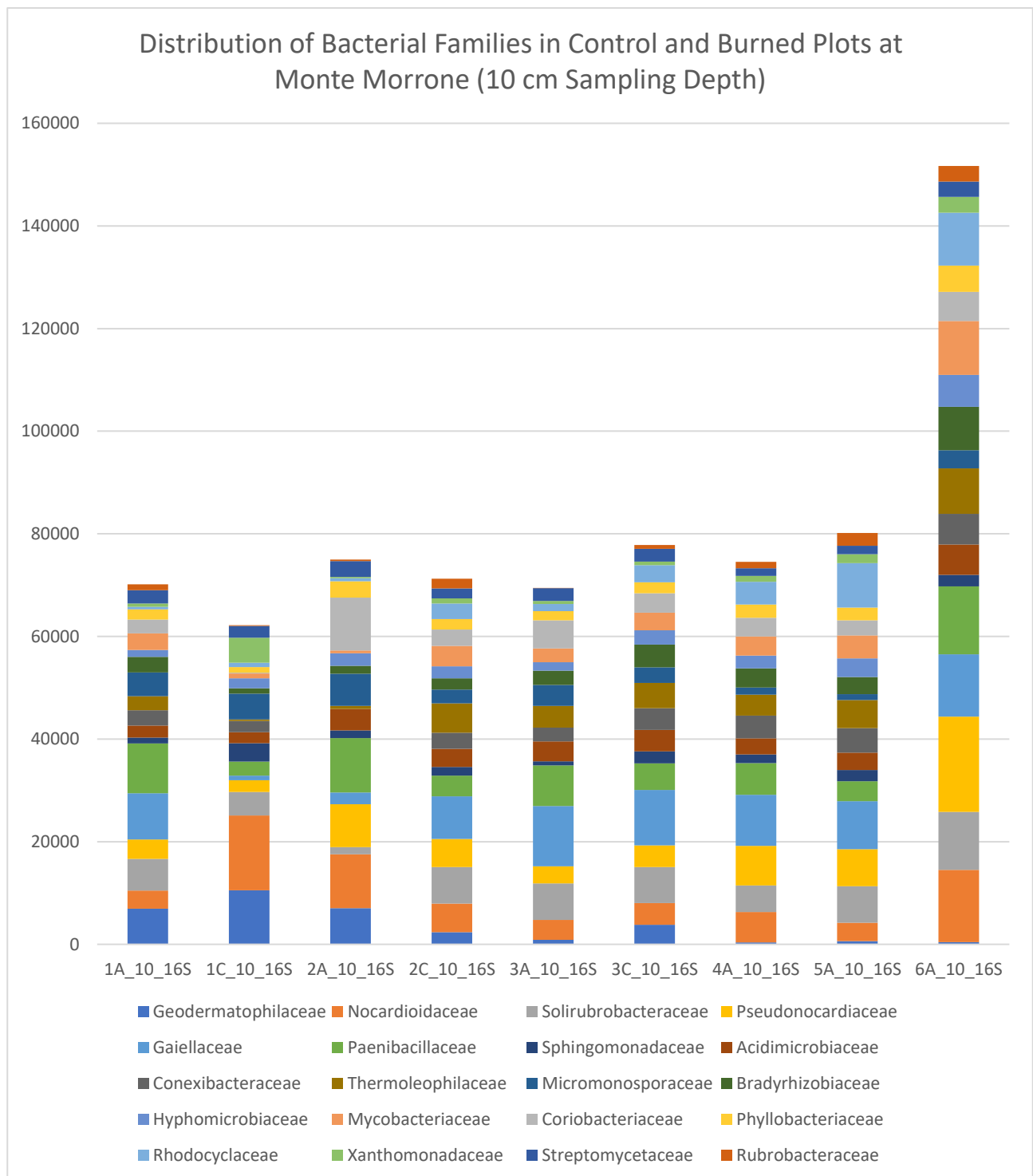


Figure 59 Distribution of bacterial families in control and burned plots at Monte Morrone (10 cm sampling depth). The bar chart illustrates the relative abundance of the 20 most abundant bacterial families across different sampling plots. Control plots (4A_10_16S, 5A_10_16S, 6A_10_16S) are compared with burned plots (1A_10_16S, 1C_10_16S, 2A_10_16S, 2C_10_16S, 3A_10_16S, 3C_10_16S). Plots with Nature-Based Solutions (NBS: 1A_10_16S, 2A_10_16S, 3A_10_16S) are highlighted to examine the impact of burned logs left on-site to reduce erosion and enhance microbial activity. Each bar segment represents the abundance of a bacterial family, with colors denoting distinct taxa.

At 5 cm depth, bacterial composition in burned soils was characterized by elevated abundances of *Thermoleophilaceae* and *Pseudonocardiaceae*. Control plots were dominated by *Geodermatophilaceae* and *Acidimicrobiaceae* (Figure 58). At 10 cm depth, burned soils showed increased abundance of *Nocardiodaceae* and *Paenibacillaceae*, while *Solirubrobacteraceae* and *Rubrobacteraceae* were more represented in control soils (Figure 59).

4.4.1.3. Effect of Nature-Based Solutions on Bacterial Communities

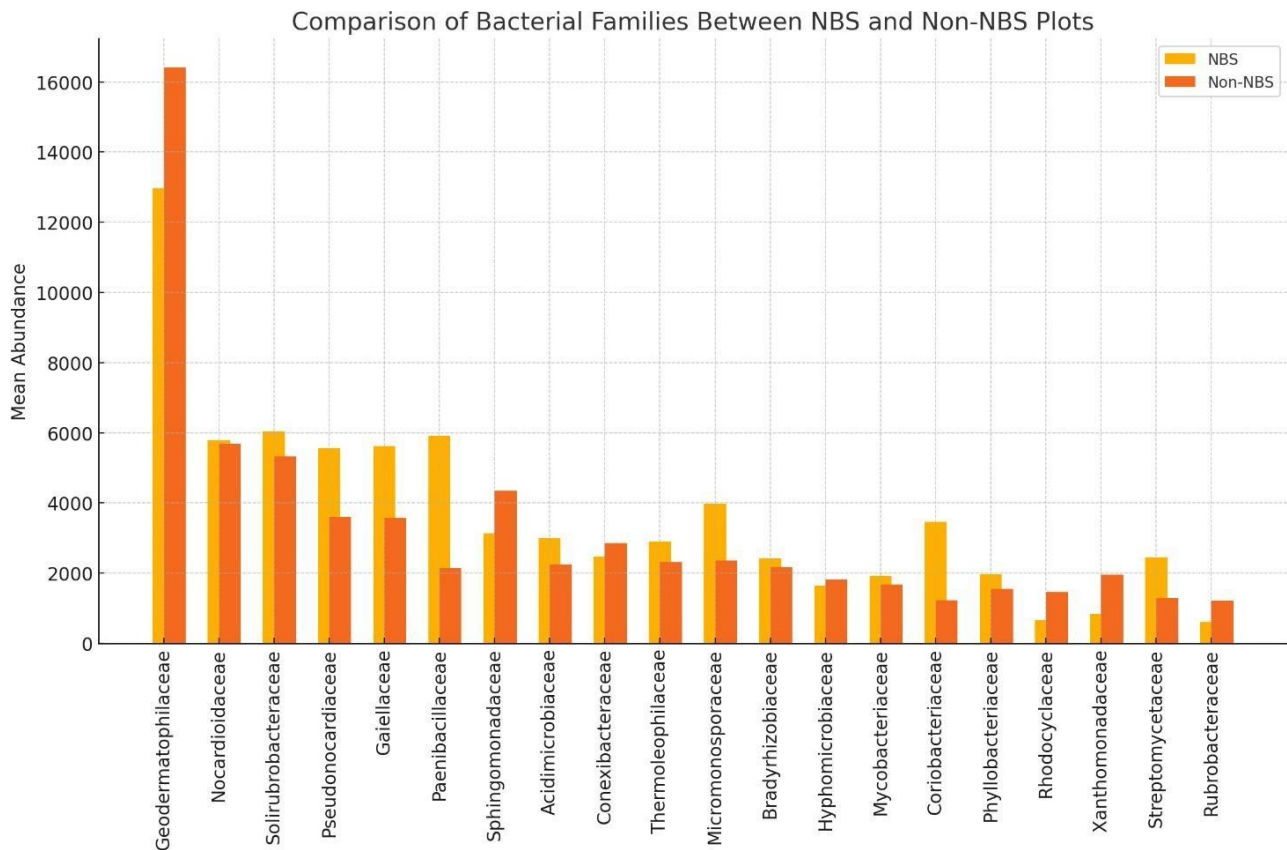


Figure 60 Comparison of bacterial families between NBS and non-NBS plots at Monte Morrone. The chart displays the mean abundance of the 20 most abundant bacterial families across plots with Nature-Based Solutions (NBS: 1A_1-5_16S, 1A_10_16S, 2A_1-5_16S, 2A_10_16S, 3A_1-5_16S, 3A_10_16S) and plots without NBS (non-NBS: 1C_1-5_16S, 1C_10_16S, 2C_1-5_16S, 2C_10_16S, 3C_5_16S, 3C_10_16S). Each bar represents the average relative abundance of bacterial families, categorized by their taxonomic classification. The yellow bars correspond to NBS plots, while the orange bars represent non-NBS plots.

NBS plots displayed higher mean relative abundances of *Paenibacillaceae*, *Thermoleophilaceae*, and *Gaiellaceae*, while non-NBS burned plots exhibited lower bacterial diversity and dominance by a narrower range of taxa (Figure 60).

4.4.2. Fungal Community Composition

4.4.2.1. Relative Abundance of Fungal Families

The ITS sequencing data highlighted distinct fungal profiles in burned and control soils. Burned plots were enriched with families such as *Thelephoraceae*, *Helotiales_fam_Incertae_sedis*, *Aspergillaceae*, and *Pyronemataceae*. In contrast, control plots were dominated by *Sebacinaceae*, *Glomeraceae*, *Clathraceae*, and *Trichocomaceae* (Figure 61, Table 21).

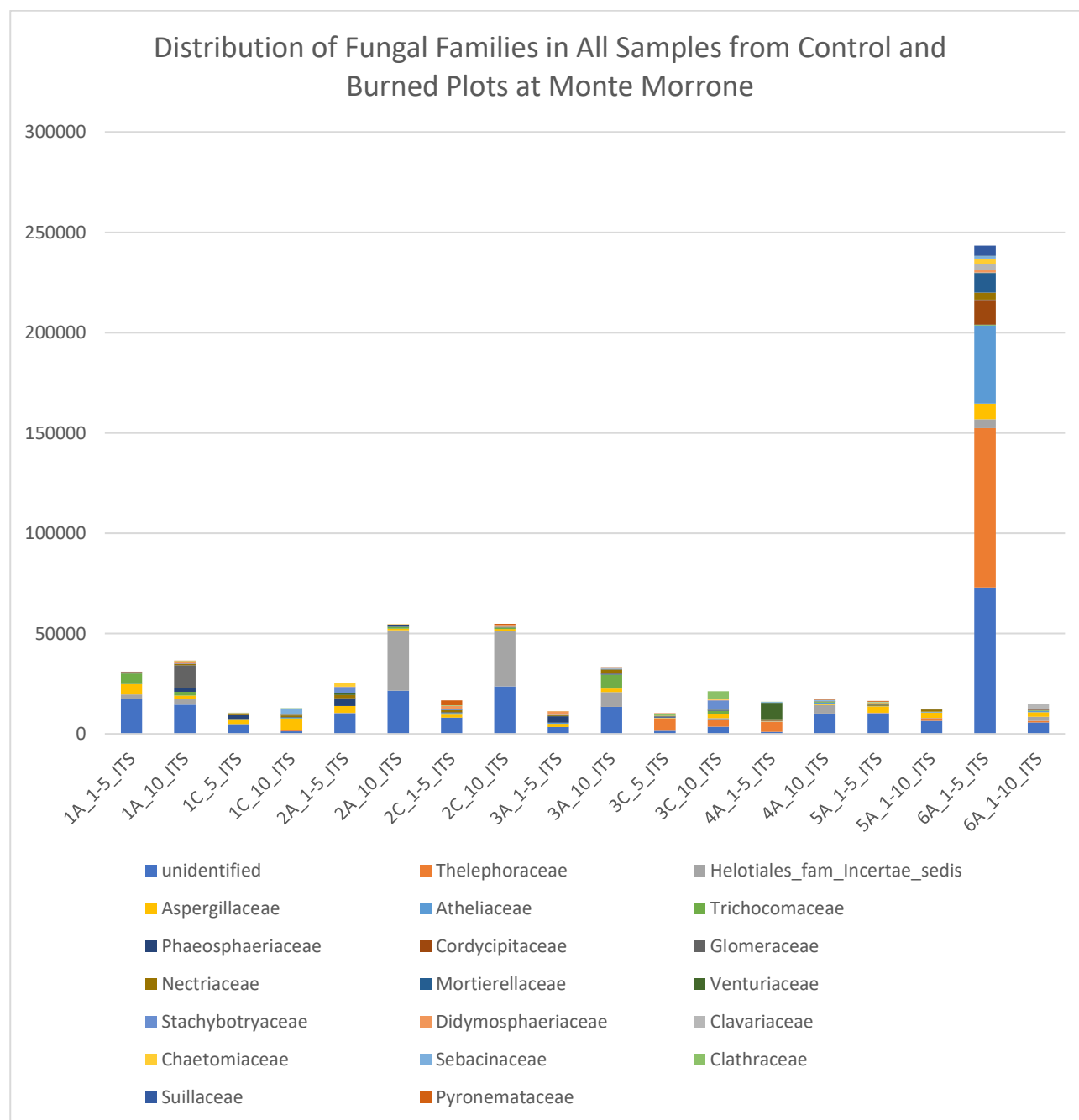


Figure 61 Distribution of the most abundant fungal families across soil plots at Monte Morrone. Each bar represents the total number of ITS reads per sample, with stacked segments indicating the relative abundance of

dominant fungal families in control and burned plots. NBS plots, where burned logs were left in situ, are included among the burned treatments. Samples are grouped by plot ID and ITS region (e.g., 1A_1.5 ITS), showing compositional differences across treatments.

Table 21 Relative Abundance of the 20 Most Abundant Fungi Families Across Soil Samples. The table shows the relative abundance of the 20 most abundant fungal families identified in the soil samples. The values represent the total RITS (Reads of Interest per Taxonomic Sequence), which are unique sequences found for each microbial family in the respective sample. The sample IDs in the header indicate the plot and depth: "5" represents 5 cm depth, and "10" represents 10 cm depth. Burned plots are labeled as "C," burned plots treated with Nature-Based Solutions (NBS) as "A," and control plots as "A."

Family	1A 5	1A 10	1C 5	1C 10	2A 5	2A 10	2C 5	2C 10	3A 5	3A 10	3C 5	3C 10	4A 5	4A 10	5A 5	5C 10	6A 5	6A 10
unidentified	175 21	145 68	48 80	12 96	101 18	214 59	81 01	236 68	34 49	134 85	15 86	35 43	10 78	97 59	102 19	65 02	729 65	56 53
Thelephoraceae	0	0	0	41 8	0	0	0	0	0	10	61 89	32 07	49 54	34 9	226	13 98	794 17	85 8
Helotiales_fam_Inc ertae_sedis	210 0	275 4	0	39 6	436	302 08	13	276 30	48	740 9	77	10 91	20 2	43 15	101	97	440 3	20 77
Aspergillaceae	518 9	180 1	25 34	55 20	336 8	822	15 32	994	15 90	166 0	10 0	21 45	15 8	54 1	323 0	26 52	785 6	22 17
Atheliaceae	48	0	46	20 4	0	0	61 7	83	50 3	0	23 9	0	10 7	67 9	160	35 4	388 89	63 7
Trichocomaceae	530 1	188 0	0	22 5	0	960	10 0	661	0	701 6	0	14 72	10	31 9	24	69	468	30 9
Phaeosphaeriaceae	0	190 0	18 80	28 6	381 8	605	43 6	103	32 43	333	52 5	31 9	0	0	69	0	10	0
Cordycipitaceae	0	0	0	0	0	0	0	0	0	32	0	0	64 0	0	0	0	123 22	27 3
Glomeraceae	744	112 89	0	0	10	31	0	0	0	363	0	0	0	0	0	0	122	18 8
Nectriaceae	0	575	34 3	85 3	168 4	163	10 32	262	36 3	184 5	22	17 9	13 0	35	548	10 20	352 7	0
Mortierellaceae	0	39	13 7	24 3	0	48	0	0	0	59	0	31	0	26	708	17 2	966 7	21 9
Venturiaceae	0	0	21 3	38	933	114	57	0	69	0	57	0	82 78	19 6	14	0	238	0
Stachybotryaceae	0	227	22 9	15 0	302 8	94	37 5	89	23 3	184	12 9	47 53	0	0	0	0	45	0
Didymosphaeriacea e	0	845	11 1	12	157	85	17 66	514	15 25	43	13 6	29 0	0	0	39	0	119 5	0
Clavariaceae	0	35	0	0	0	0	0	0	0	11	0	0	0	19 2	0	21	296 4	25 10
Chaetomiaceae	0	366	58	10 9	177 5	41	0	0	34	0	53	15 9	0	0	335	0	280 0	0
Sebacinaceae	0	0	0	29 36	34	0	0	0	0	285	0	18 8	36 7	52 4	32	28	127 9	12
Clathraceae	0	83	85	62	0	0	19 8	0	0	0	58 7	38 97	81	0	373	71	108	0
Suillaceae	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	511 3	17 8
Pyronemataceae	11	55	0	0	0	0	24 52	933	14 6	54	59 9	0	0	41 3	352	17	0	0

4.4.2.2. Depth-Specific Fungal Distribution

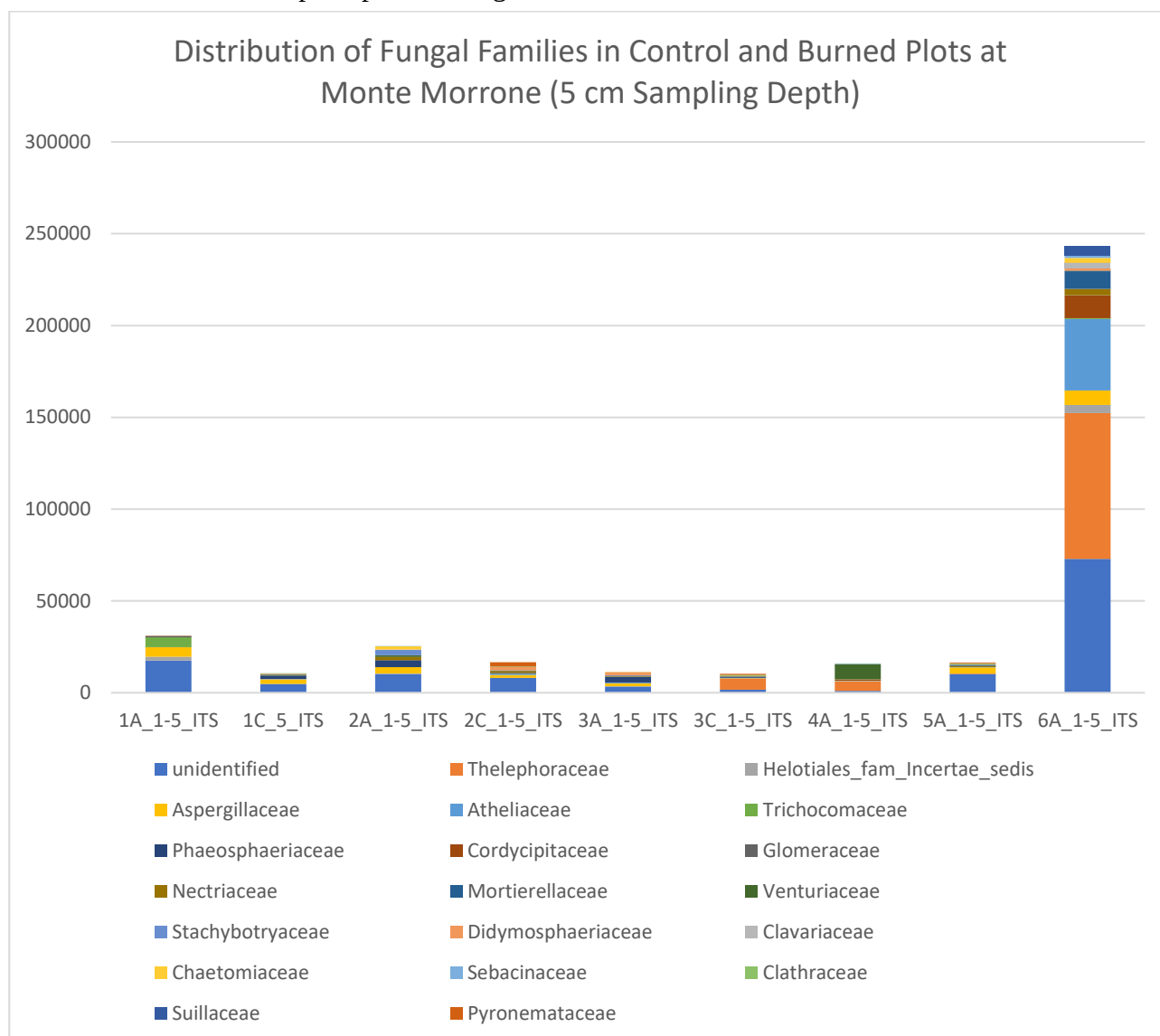


Figure 62 Distribution of fungal families in control and burned plots at Monte Morrone at 5 cm soil depth based on ITS sequencing. Stacked bars represent the total number of reads assigned to each fungal family per sample (1A to 6A). Samples 1A–3C refer to control plots, while 4A–6A represent burned plots. Taxonomic groups are color-coded in the legend, including both identified and unidentified fungal families.

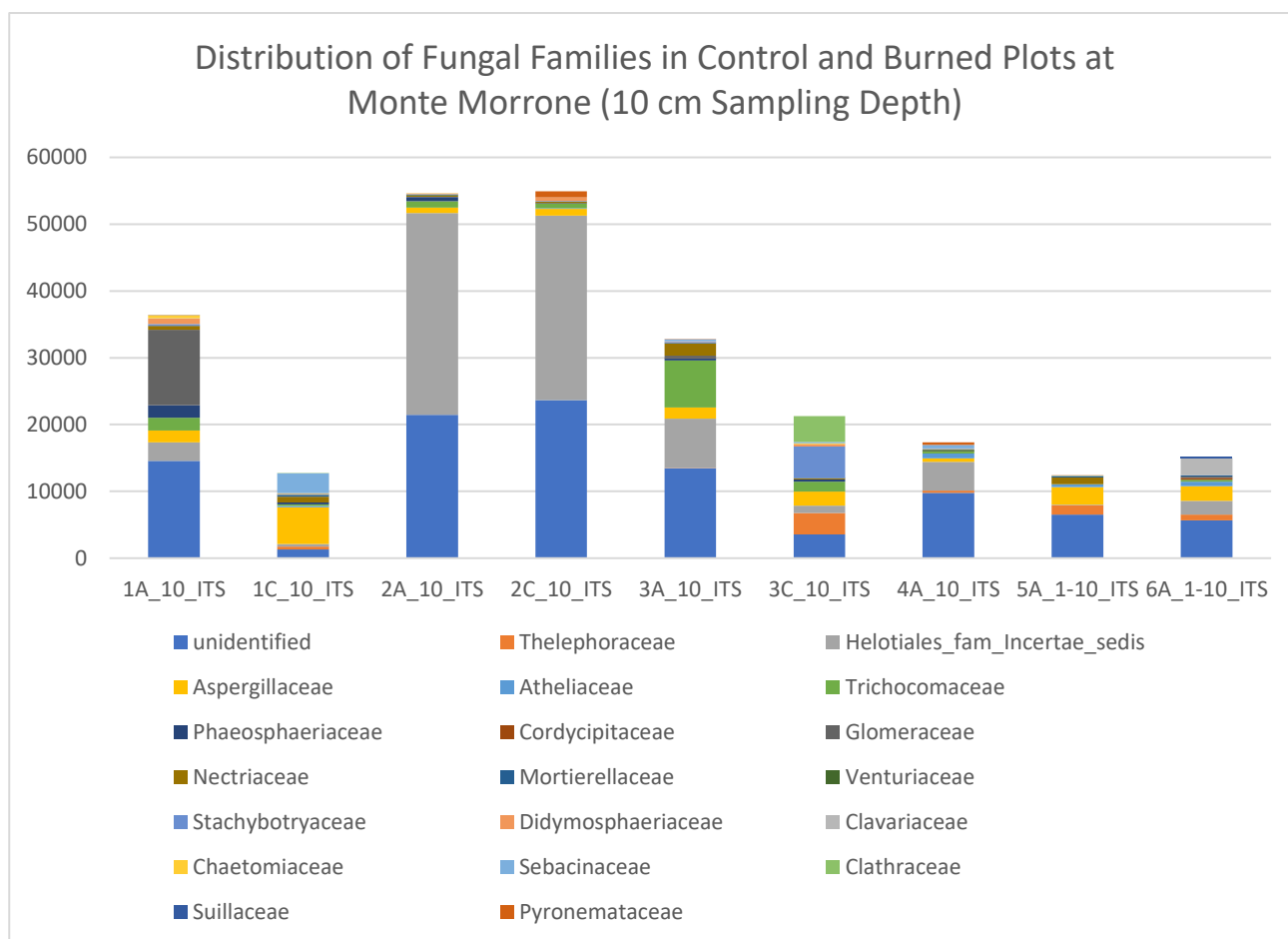


Figure 63 Distribution of fungal families in control and burned plots at Monte Morrone at 10 cm soil depth based on ITS sequencing. Stacked bars represent the total number of reads assigned to each fungal family per sample (1A to 6A). Samples 1A–3C refer to control plots, while 4A–6A represent burned plots. Taxonomic groups are color-coded in the legend, including both identified and unidentified fungal families.

At 5 cm depth, burned soils had elevated abundances of *Aspergillaceae* and *Pyronemataceae*. Control plots at the same depth showed predominance of *Sebacinaceae* and *Clathraceae* (Figure 62). At 10 cm, fungal composition in burned soils shifted toward *Helotiales_fam_Incertae_sedis* and *Thelephoraceae*, while *Glomeraceae* and *Trichocomaceae* remained dominant in control plots (Figure 63).

4.4.2.3. Effect of Nature-Based Solutions on Fungal Communities

Comparison of Fungal Families Between NBS and Non-NBS Plots

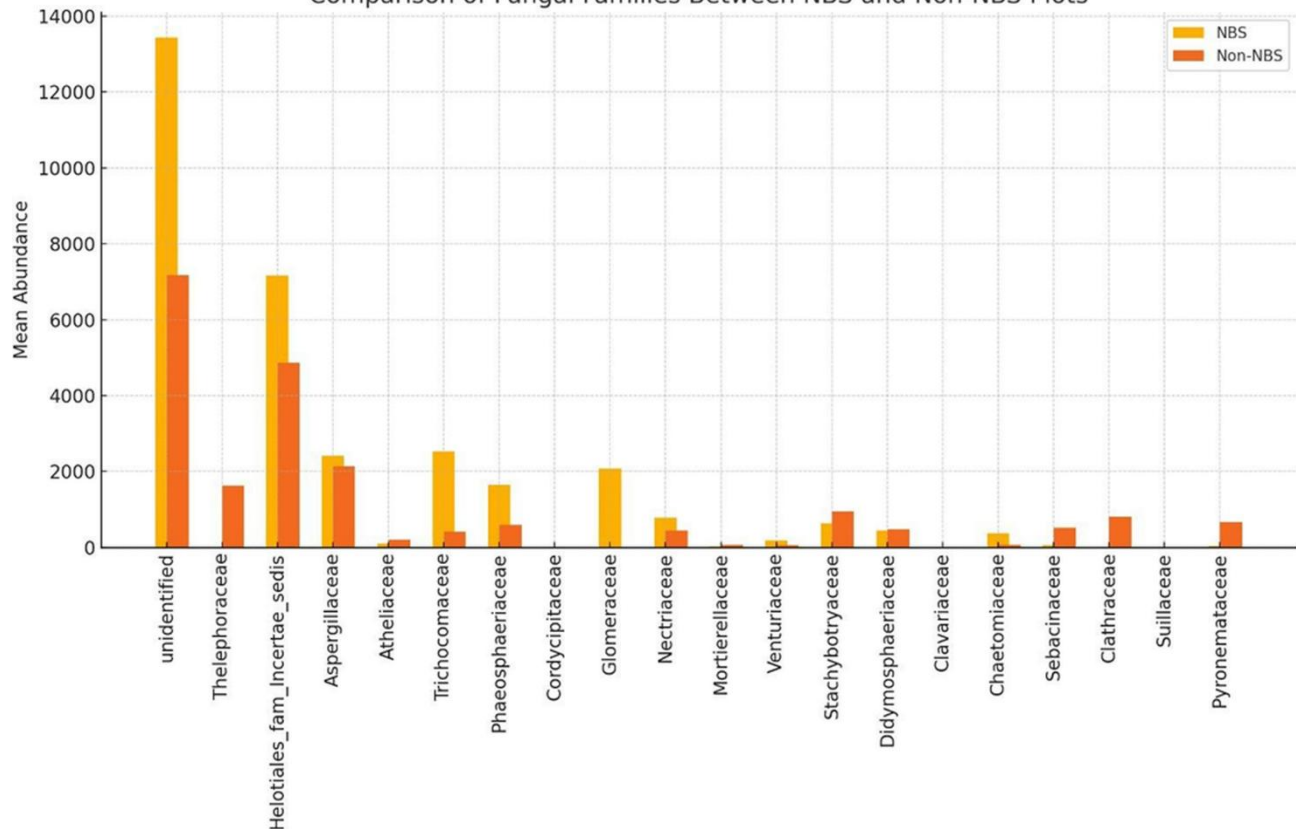


Figure 64 Comparison of fungal families between NBS and non-NBS plots at Monte Morrone.

The bar chart shows the mean abundance of the 20 most abundant fungal families in plots treated with Nature-Based Solutions (NBS: 1A_1-5_ITS, 1A_10_ITS, 2A_1-5_ITS, 2A_10_ITS, 3A_1-5_ITS, 3A_10_ITS) versus untreated burned plots (non-NBS: 1C_1-5_ITS, 1C_10_ITS, 2C_1-5_ITS, 2C_10_ITS, 3C_1-5_ITS, 3C_10_ITS).

Each bar represents the mean relative abundance of the most represented fungal families, aggregated across depths (5 cm and 10 cm). Taxonomic classification was based on ITS rRNA sequencing.

NBS plots exhibited higher mean abundances of *Thelephoraceae*, *Helotiales_fam_Incertae_sedis*, and *Glomeraceae* compared to non-NBS plots. In contrast, non-NBS burned plots showed a greater presence of stress-adapted families such as *Aspergillaceae* and unidentified fungi. These differences suggest that NBS interventions may support the recolonization of more functionally diverse and symbiotic fungal communities following fire disturbance (Pereira et al., 2020; García-Carmona et al., 2023) (Figure 64).

5 Discussion

5.1. Microclimate Dynamics and Their Implications for Post-Fire Soil and Ecosystem Recovery

The microclimatic patterns observed at Monte Morrone reflect a strong divergence between burned and control plots, driven by changes in surface cover, vegetation structure, and post-disturbance hydrological behavior. The elevated and more variable soil temperatures recorded in burned areas—particularly during summer months—are consistent with findings from Mediterranean forest systems affected by wildfire, where vegetation loss increases albedo and solar radiation penetration (López-Serrano et al., 2018; Fernández-García et al., 2021). These thermal shifts have significant implications for soil biota and biogeochemical cycling, accelerating desiccation, altering microbial metabolic rates, and reducing root activity (Marañón-Jiménez et al., 2019; Hart et al., 2005).

Soil moisture dynamics also displayed a marked contrast, with burned plots exhibiting lower and more erratic soil water content (SWC). The destruction of surface litter, loss of root networks, and formation of hydrophobic layers reduce infiltration rates and increase runoff, leading to moisture loss even during precipitation events (Doerr et al., 2000; Lucas-Borja et al., 2020). This pattern was especially evident in the summer dry season, when the absence of vegetative buffering in burned plots exacerbated water stress conditions. By comparison, control sites maintained more stable SWC levels, supported by intact vegetation cover that promotes water retention through shading, root absorption, and litter-induced hydraulic redistribution (López-Serrano et al., 2018; Rey et al., 2011).

These findings have critical implications for ecosystem resilience. Soil temperature and moisture are primary regulators of microbial activity, nutrient cycling, and plant regeneration—processes essential to recovery after fire. In burned plots, the combination of thermal stress and moisture scarcity likely restricts microbial recolonization and root development, contributing to the persistent reduction in soil respiration and biological function observed in subsequent analyses (Santonja et al., 2017; Bárcenas-Moreno et al., 2011). Moreover, temperature–moisture decoupling impairs enzymatic processes related to carbon turnover, particularly in fine-textured soils with low porosity and disrupted structure (Certini, 2005; Marañón-Jiménez et al., 2019).

The irregular rainfall pattern documented during the study period—characterized by short intense events followed by prolonged drought—further challenges post-fire recovery. In Mediterranean regions, such precipitation regimes are increasingly frequent under climate change and are associated with heightened erosion risks, especially in steep slopes like those of Monte Morrone (Shakesby & Doerr, 2006; Guerra et al., 2021). Without sufficient vegetation, these events can trigger topsoil loss,

nutrient depletion, and the breakdown of microbial habitats, reinforcing degradation loops (Pereira et al., 2020; Fernández-García et al., 2021).

Ultimately, the stability of microclimatic conditions in control plots underscores the protective role of vegetation cover in maintaining biophysical equilibrium. These ecosystems exhibit higher resistance and faster recovery due to the sustained regulation of temperature and moisture, highlighting the importance of preserving vegetative structure and promoting natural regeneration in restoration strategies. Integrating microclimatic monitoring into post-fire assessments is thus essential for identifying critical thresholds, predicting recovery trajectories, and designing site-specific interventions that enhance resilience under future climate scenarios (Guénon et al., 2016; Guerra et al., 2021; Rey et al., 2011).

5.2. Remote Sensing Indicators and Post-Fire Vegetation Recovery in Monte Morrone

The spatial and temporal patterns of vegetation damage and regrowth captured by remote sensing indices in Monte Morrone provide valuable insights into the ecological impact of wildfire and the site-specific variability in post-fire resilience. The differential burn severity observed between the 2017 and 2023 wildfires, as revealed by the dNBR, RdNBR, dLAI, and dkNDVI indices, reflects not only the fire intensity and extent but also the legacy effects of previous disturbances and site conditions (Key & Benson, 2006; De Santis et al., 2010).

The 2017 fire, affecting over 2,100 hectares, caused widespread canopy loss and high burn severity, as indicated by strong negative values of dkNDVI and dLAI, and high RdNBR classes. In contrast, the 2023 fire—although smaller in extent—affected both previously undisturbed areas and plots already burned in 2017. The overlap of fire scars is particularly relevant, as repeated fire events are known to significantly reduce ecosystem resilience by altering soil structure, depleting the seed bank, and disrupting microbial and mycorrhizal networks essential for regrowth (Keeley et al., 2011; D’Angelo et al., 2021; Díaz-Raviña et al., 2012).

The analysis of the Recovery Indicator (RI) and LAI trends further supports this interpretation. Areas burned only in 2017 (Zones 1 and 2) displayed relatively higher RI and progressive increases in LAI from 2018 to 2022, reflecting partial natural regeneration under favorable microsite conditions. These zones likely benefit from sufficient recovery time, moderate burn severity, and geomorphological features such as lower slope or shaded exposures. Conversely, Zones 3 and 4, characterized by either repeated fire exposure or harsher topographic constraints, exhibited limited LAI recovery and poor RI scores. The stagnation or decline in vegetation indices in these zones confirms that repeated

disturbances severely constrain ecological recovery and may trigger transitions toward degraded states (Pausas & Keeley, 2009; Pausas et al., 2008; Fernández-García et al., 2018).

This spatial heterogeneity underscores the necessity of site-specific restoration strategies. While passive regeneration may be sufficient in areas with moderate damage and natural seed sources, more severely degraded or repeatedly burned areas require targeted interventions. These may include mulching to control erosion and improve microclimate, seeding or planting of native pioneer species, and deadwood management to promote microbial and faunal recolonization (Castro et al., 2011; Robinne et al., 2021). The monitoring of spectral indices over time, combined with in situ assessments, is essential to evaluate the efficacy of such measures and guide adaptive management. Importantly, these results highlight the role of fire history and temporal spacing between events in shaping recovery potential. In Mediterranean environments, fire-adapted vegetation can withstand single fire events under natural return intervals, but increased fire frequency—amplified by climate change—surpasses ecological thresholds and drives irreversible degradation (Pausas & Keeley, 2019; Moreira et al., 2020). The case of Zone 4 exemplifies this dynamic, where overlapping fires appear to have erased regeneration signals entirely, raising concerns about long-term forest persistence in such hotspots.

In conclusion, the integration of remote sensing with ecological knowledge provides a powerful framework for post-fire landscape assessment. Indices like RI, dLAI, and RdNBR not only quantify burn severity and recovery trends but also support early identification of high-risk areas in need of intervention. As fire regimes intensify across Mediterranean forests, the use of these tools will be crucial for prioritizing restoration actions and improving landscape-scale resilience to future disturbances.

5.3. Soil CO₂ Efflux and Environmental Controls in Post-Fire Ecosystems: Functional Responses and Constraints at Monte Morrone

The analysis of Soil CO₂ Efflux (SR) at the Monte Morrone site provides clear evidence of the long-term impact of wildfire on soil biological activity and carbon cycling. Despite the six years elapsed since the 2017 fire event, Soil CO₂ Efflux in burned plots remained significantly lower than in control plots, indicating persistent disruption in belowground processes. This aligns with previous studies in Mediterranean ecosystems, which highlight the delayed recovery of soil microbial activity following intense fires due to losses in organic matter, microbial biomass, and soil structural integrity (Certini, 2005; Santonja et al., 2017; Guénon et al., 2013).

The contrasting Soil CO₂ Efflux patterns between burned and control plots also indicate a fundamental shift in the regulation of carbon fluxes. In control areas, Soil CO₂ Efflux responded

positively to increases in both soil temperature and moisture, reflecting a well-coupled system where abiotic factors support microbial and root respiration. In burned soils, however, this relationship was weaker and more variable, particularly under summer conditions characterized by elevated soil temperatures and reduced water content. Microclimatic data confirmed that burned plots experienced higher temperature peaks and more intense desiccation during dry periods. These environmental constraints—combined with structural degradation and limited substrate availability—appear to restrict microbial activation despite the presence of moisture, as also suggested by the reduced correlation between Soil CO₂ Efflux and SWC in burned soils (Doerr et al., 2000; García-Cervigón et al., 2021; Conant et al., 2011).

Statistical modelling, including Linear Mixed Models (LMMs) and quantile regression, further supported this interpretation. While control plots exhibited a consistent response to increasing temperature, particularly in the lower quantiles of Soil CO₂ Efflux distribution, burned plots showed no significant temperature dependency in the upper quantiles. This pattern underscores a diminished capacity for carbon flux under favourable conditions, likely attributable to fire-induced loss of labile carbon pools and the disruption of fine-root and fungal networks essential for heterotrophic respiration (Fang et al., 2005; Bárcenas-Moreno et al., 2011).

Basal respiration (BR), measured under standardized conditions (15°C), was significantly lower in burned and NBS-treated plots compared to controls, despite similar Q₁₀ values across treatments. This decoupling suggests that while thermal response mechanisms may remain conserved, the functional microbial biomass capable of sustaining respiration has been reduced in fire-affected soils. Among the models tested to estimate temperature sensitivity, the exponential model provided the best fit in terms of explanatory power (R²) and biological interpretability of the coefficients. This result supports the established ecological understanding that SR increases exponentially with temperature under non-limiting conditions (Lloyd & Taylor, 1994; Davidson & Janssens, 2006). Accordingly, the exponential model was adopted for data interpretation.

The limited recovery observed in Nature-Based Solutions (NBS) plots, where burned logs were left in place to promote soil protection and microhabitat diversity, raises important considerations for passive restoration strategies. Although such interventions may help retain moisture and provide organic input, the low Soil CO₂ Efflux values observed suggest that these benefits do not yet translate into full microbial functional recovery. These findings are consistent with recent literature emphasizing that structural measures alone may be insufficient and require integration with targeted actions—such as mulching, organic amendments, or assisted revegetation—to effectively support microbial processes (Marchetti et al., 2023).

From an ecological perspective, the persistent reduction in Soil CO₂ Efflux across burned and NBS-treated plots points to a long-lasting impairment of carbon cycling, with consequences for nutrient availability, soil aggregation, and vegetation feedback. In Mediterranean fire-prone systems, where drought events are expected to intensify under climate change scenarios, the ability of soils to recover respiratory function is a key determinant of long-term ecosystem resilience (Pausas & Keeley, 2019; de Vries et al., 2013).

In conclusion, soil respiration at Monte Morrone serves as a sensitive indicator of post-fire soil degradation and ecosystem recovery. The integration of basal respiration, Q₁₀ analysis, microclimatic data, and advanced statistical modelling offers a comprehensive framework for evaluating soil functional health. Findings from this study suggest that passive restoration strategies alone may be insufficient to restore soil functionality within a six-year timeframe, highlighting the need for more targeted, site-specific interventions to accelerate microbial resilience and promote carbon flux recovery.

Additionally, soil physical and chemical properties contributed to shaping post-fire Soil CO₂ Efflux dynamics. Although bulk density and texture variations were recorded in burned plots, their ecological implications remain limited due to the low resolution and statistical insignificance of the available data. Nonetheless, a significant increase in soil pH was observed in fire-affected areas, particularly in the Fire_2017-2023 plots. This alkalization is a common post-fire response, typically resulting from ash deposition and volatilization of organic acids during combustion (Certini, 2005). Elevated pH can transiently enhance nutrient availability but may also alter microbial community composition, reducing overall diversity and functional capacity (González-Pérez et al., 2004; Bárcenas-Moreno et al., 2011). These changes, although subtle, may contribute to the persistent reduction in microbial activity and Soil CO₂ Efflux observed in burned soils. Overall, the combination of thermal stress, moisture scarcity, and chemical shifts such as increased alkalinity underscores the complex barriers to microbial and ecosystem recovery in post-fire Mediterranean soils.

5.4. Integrated Responses of Soil CO₂ Efflux and Soil Properties to Degradation in Companhia das Lezírias

The empirical evidence from Companhia das Lezírias underscores the profound biogeochemical divergence between degraded and control soils in this Mediterranean agro-silvo-pastoral landscape. The significantly lower Soil CO₂ Efflux (SR) in degraded plots reflects impaired carbon turnover capacity, with statistical support from Brunner-Munzel tests and bootstrap confidence intervals confirming robust and significant differences ($p < 0.05$). These flux reductions likely arise from

structural degradation, reduced microbial biomass, and lower substrate availability due to past disturbances such as grazing and vegetation loss.

Microbial respiration in the degraded site was positively associated with soil temperature and negatively with soil water content, total organic carbon (TOC), and pH, as confirmed by linear regression models. This contrasts with the control site, where SR was supported by higher water availability and labile TOC, suggesting a more favourable environment for microbial activity. These divergent controls reflect distinct ecological responses shaped by site-specific microclimatic and structural conditions.

Degraded plots experienced higher mean soil temperatures (25.3°C) compared to the forested control site, a consequence of canopy loss and reduced shading—common in Mediterranean post-fire and overgrazed systems (Certini, 2005; Pereira et al., 2020). However, despite higher thermal input, SR peaked around 25°C and declined at higher temperatures, indicating thermal inhibition likely driven by enzymatic stress and moisture limitations (Davidson & Janssens, 2006). The GLMM and LMM models confirmed soil temperature, water content, organic matter, and treatment as significant SR predictors ($p < 0.01$), with notable interaction effects such as SWC $\times$ Treatment, emphasizing the capacity of control soils to utilize water more efficiently likely due to higher porosity and microbial connectivity.

Water availability acted as a major control of Soil CO₂ Efflux. In the degraded site, Soil CO₂ Efflux peaked at ~15% SWC and declined under wetter conditions, consistent with the formation of anaerobic microsites in compacted soils. Conversely, SR in the control site remained stable across moisture levels, demonstrating better structural buffering. This resilience is attributed to intact soil aggregation, better aeration, and higher-quality organic inputs that support aerobic microbial activity (Fierer & Schimel, 2002; Moyano et al., 2012).

The degraded soils' elevated TOC levels paradoxically corresponded to lower Soil CO₂ Efflux, likely due to a shift towards recalcitrant, pyrogenic carbon forms that resist decomposition. This decoupling suggests that TOC alone is not a reliable proxy for microbial function without considering carbon lability and accessibility (Moyano et al., 2012; Czimczik & Masiello, 2007). Similarly, higher EC and lower pH in degraded plots imposed chemical stress on microbial communities. High EC—likely due to ash inputs—can impair microbial osmotic balance, while reduced pH narrows enzymatic functionality and microbial diversity (Rousk et al., 2010; Zhang et al., 2019).

Q₁₀ values did not differ between treatments (both ≈ 1.00), suggesting that temperature sensitivity remains a conserved microbial trait even under degraded conditions (Pietikäinen et al., 2005). However, the basal respiration (BR₁₅) was significantly lower in the degraded site (2.75 $\mu\text{mol CO}_2$

$\text{m}^{-2} \text{ s}^{-1}$ vs 5.91 in control), indicating loss of active microbial biomass or functionality. This decoupling further supports the idea that Q_{10} alone is insufficient to assess soil function, while BR_{15} offers a more reliable indicator of microbial metabolic potential (Fang et al., 2005).

Finally, quantile regression revealed that, in control plots, temperature was a significant predictor of Soil CO_2 Efflux at lower quantiles, highlighting system responsiveness under suboptimal conditions. This pattern was absent in degraded plots, confirming a loss of functional plasticity and reduced environmental control over respiration. Such rigidity in degraded systems has broad implications for carbon cycling resilience under increasing climatic variability.

Altogether, these findings highlight that soil degradation in Mediterranean systems not only suppresses respiration rates but also restricts the dynamic range and environmental responsiveness of Soil CO_2 Efflux. Restoration strategies should thus target both structural recovery (e.g., porosity, aggregation) and organic matter quality to reinstate microbial resilience and carbon flux regulation under future climate scenarios.

The differences in Soil CO_2 Efflux between degraded and control plots at Companhia das Lezírias are closely linked to changes in soil physical and chemical properties. Empirical data revealed significantly higher water content, soil temperature, total organic carbon (TOC), and electrical conductivity (EC) in degraded plots, alongside markedly lower porosity and pH. These variables are known to exert strong control over microbial respiration and soil carbon dynamics (Moyano et al., 2012; Rousk et al., 2010).

The reduced porosity in degraded plots reflects soil compaction, likely caused by grazing pressure and vegetation loss, which limits gas exchange and water infiltration—two critical processes for sustaining microbial metabolism. While the high-water content in these plots might initially suggest favourable conditions, it is more likely indicative of poor drainage and limited evapotranspiration, exacerbated by the absence of canopy cover. This can lead to localized anoxic conditions that suppress aerobic microbial activity and reduce respiration efficiency (Schimel, 2018; Rey et al., 2011).

In chemical terms, the lower pH observed in degraded plots is particularly significant. Acidification can alter enzyme kinetics, limit nutrient availability, and selectively inhibit microbial taxa, thereby reducing overall soil metabolic activity. The concurrent rise in EC, likely due to ash-derived solutes and reduced plant uptake, imposes osmotic stress that further constrains microbial function, especially under high-temperature conditions (Zhang et al., 2019). These factors, taken together, suggest that chemical stressors act synergistically with structural degradation to suppress microbial respiration and reduce soil carbon turnover.

Interestingly, although TOC and organic matter (OM) content were higher in degraded plots, regression analysis and respiration data suggest that this carbon pool may be largely recalcitrant. This supports the hypothesis that fire disturbance and subsequent degradation shift the quality of organic inputs towards more chemically resistant forms, which do not fuel microbial respiration effectively (Czimczik & Masiello, 2007; Ferreira et al., 2016).

In contrast, control plots benefited from more favourable pH levels, lower salinity, and higher porosity, all of which support a more active and resilient microbial community. These conditions help maintain higher basal respiration and more stable CO₂ efflux dynamics, even under fluctuating environmental conditions.

Taken together, the data demonstrate that post-disturbance soil degradation affects not only the quantity but also the quality and functionality of key soil properties. Structural compaction, altered pH, and reduced substrate accessibility converge to limit microbial efficiency, thereby weakening the soil's capacity to sustain carbon cycling and recover ecosystem functionality. Effective restoration in such systems should thus prioritize not only vegetation recovery but also improvements in soil structure and biochemical balance to reinstate belowground ecological processes.

5.5. Soil Microbial and Fungal Community Responses to Wildfire and Environmental Conditions at Monte Morrone (Abruzzo)

Soil microbial communities are key regulators of ecosystem processes such as organic matter decomposition, nutrient cycling, and soil structure stabilization, and their composition is highly responsive to environmental disturbances (Fierer & Jackson, 2006; van der Heijden et al., 2008). At the Monte Morrone site in Abruzzo, preliminary observations of microbial community profiles, based on taxonomic assignment prior to bioinformatic and diversity analyses, suggest notable shifts associated with wildfire disturbance, depth stratification, and passive restoration practices. Soil samples collected in spring 2022, before the onset of summer drought, provide insights into microbial responses in an intermediate post-fire recovery phase.

In burned plots, microbial communities appear to be dominated by taxa commonly associated with stress tolerance and disturbance adaptation. These include bacterial groups capable of metabolizing recalcitrant organic matter (Fierer et al., 2012) and fungal families known for their opportunistic colonization of fire-altered substrates (Tedersoo et al., 2014; Glassman et al., 2016). Such community patterns are frequently reported in Mediterranean fire-affected systems (Barcenas-Moreno et al., 2011; Baldrian, 2016) and reflect both thermal selection and resource limitation following combustion. Although these taxa may initiate early-stage decomposition and nutrient transformation

(Allison & Treseder, 2008), their dominance may also indicate a reduced functional complexity compared to unburned soils.

Conversely, control plots showed greater representation of microbial groups typically associated with stable soil conditions, plant-microbe mutualisms, and efficient biogeochemical cycling (Smith & Read, 2008; Rousk et al., 2010). This suggests that undisturbed soils maintain a higher degree of functional connectivity with vegetation inputs and root activity, supporting a more resilient microbial network across soil horizons. Observed differences in microbial composition between surface and subsurface layers further highlight the impact of fire on vertical microbial distribution, with surface layers in burned soils more strongly affected by thermal disturbance (Rey et al., 2011; Weiss et al., 2011).

Plots treated with Nature-Based Solutions (NBS), where charred woody debris was retained, appeared to support a more diverse set of microbial taxa involved in nutrient cycling and organic matter turnover. These observations, though preliminary, are in line with studies suggesting that structural heterogeneity and microhabitat availability enhance microbial resilience in post-fire environments (Lucas-Borja et al., 2020; Fernández-García et al., 2021).

When considered alongside environmental parameters such as soil temperature, water content, and total organic carbon, these microbial shifts provide ecological context for the observed reduction in Soil CO₂ Efflux in burned plots. The decoupling between organic matter quantity and microbial respiration, particularly in degraded soils with high but recalcitrant TOC, points to the importance of organic matter quality and microbial community functionality in driving ecosystem recovery (Certini, 2005; Moyano et al., 2012). Control plots, characterized by higher porosity, balanced moisture conditions, and stable microbial composition, exhibited more consistent soil CO₂ fluxes, reinforcing the link between microbial integrity and soil functional resilience (Fierer et al., 2009; Conant et al., 2011).

While these initial findings reveal meaningful trends, further bioinformatic analyses—including alpha and beta diversity metrics—will be essential to quantify the extent of microbial disruption and recovery. These upcoming analyses will allow for a deeper understanding of the ecological trajectories of soil microbial communities in fire-affected Mediterranean forest systems and the potential of passive restoration strategies to facilitate their re-establishment.

6 Conclusion

This research has provided an in-depth, multidisciplinary investigation into the effects of wildfire, drought, and land degradation on Mediterranean forest and agro-silvo-pastoral ecosystems, focusing on two contrasting long-term monitoring sites: Monte Morrone in Majella National Park (Italy), affected by severe wildfires in 2017 and 2023, and Companhia das Lezírias (Portugal), impacted by chronic drought and overgrazing. By integrating remote sensing, microclimatic data, soil physical-chemical analyses, soil respiration (Soil CO₂ Efflux) measurements, and environmental DNA sequencing of microbial communities, this study offers a comprehensive understanding of how different degradation drivers disrupt ecosystem processes and how soil-based indicators can support restoration strategies.

At Monte Morrone, the impacts of wildfire were evident at all levels of ecosystem functioning. Remote sensing analyses revealed extensive and long-lasting vegetation loss following the 2017 fire, with areas affected again in 2023 showing particularly poor recovery. Spectral indices such as dLAI and dkNDVI confirmed a significant decline in canopy cover, photosynthetic activity, and regeneration capacity, especially in zones affected by repeated fire. These patterns were directly linked to microclimatic stressors, with burned plots experiencing higher soil temperatures and lower, more erratic moisture availability compared to controls. These altered abiotic conditions were shown to constrain microbial activity and plant regrowth, reinforcing the tight feedback between aboveground vegetation and belowground processes.

Soil respiration analyses further confirmed the degradation of biological activity in fire-affected plots. SR in burned soils was consistently lower than in control areas, particularly under dry conditions and elevated temperatures. Statistical models revealed a decoupling of SR from its usual environmental drivers (temperature and soil water content), suggesting structural constraints, such as compaction and organic matter loss, are limiting biological responses. Basal respiration (BR) was also significantly lower in burned soils, highlighting reduced microbial biomass and activity, while Q₁₀ values indicated a conserved—but limited—temperature sensitivity. Nature-Based Solutions (NBS), including the placement of burned logs, offered only partial improvement, emphasizing that passive interventions may not be sufficient in the short term. These findings align with broader evidence that post-fire soil recovery requires not just time, but also targeted strategies to rebuild structure, porosity, and carbon inputs.

Soil property analyses at Monte Morrone revealed slightly higher bulk density and elevated pH in fire-affected soils compared to control plots. Although these differences were not always statistically significant, they may have relevant ecological implications. Increased compaction can reduce soil

aeration and water infiltration, while pH shifts toward alkalinity may influence microbial community composition, potentially favoring stress-tolerant taxa and reducing functional redundancy. Moreover, localized changes in soil texture—possibly linked to post-fire erosion and ash deposition—could alter water retention capacity and the structure of microbial habitats, further influencing recovery dynamics.

At Companhia das Lezírias, the degraded plots—subjected to prolonged drought and livestock grazing—exhibited similar patterns of stress, including reduced Soil CO₂ Efflux, altered pH and EC values, and diminished organic matter content. However, the drivers differed in nature and intensity, and the absence of fire allowed for some resilience in microbial responses, particularly under moderate temperature and moisture conditions. Control plots at both sites consistently showed greater biological activity, diversity, and functional stability, reinforcing the importance of intact vegetation and soil structure in buffering against environmental stress.

Microbial community analyses provided a unifying lens for interpreting ecosystem function across both sites. At Monte Morrone, burned soils were dominated by bacterial and fungal families adapted to nutrient-poor, high-stress conditions—such as *Thermoleophilaceae*, *Pseudonocardiaceae*, and *Thelephoraceae*—reflecting a shift toward opportunistic taxa capable of degrading recalcitrant substrates and fixing nitrogen. While these microbes contribute to early recovery, their dominance limits community resilience and nutrient cycling efficiency over the long term. Control plots instead harbored diverse, functionally redundant communities—including *Acidimicrobiaceae*, *Glomeraceae*, and *Sebacinaceae*—indicative of mature, stable ecosystems with strong capacity for carbon sequestration and nutrient retention.

By integrating these diverse datasets, this thesis demonstrates the value of Soil CO₂ Efflux and microbial composition as sensitive indicators of ecosystem degradation and recovery. These metrics, when interpreted alongside remote sensing and soil property data, can reveal both the current state of an ecosystem and its potential for restoration. Importantly, the findings emphasize that fire and land degradation do not simply reduce ecosystem function; they alter the very relationships between climate, soil, and biology that sustain Mediterranean landscapes.

In terms of restoration, the research supports the implementation of active, site-specific interventions. At Monte Morrone, areas affected by repeated fire should be prioritized for stabilization and revegetation using native species and organic amendments. The addition of mulch, biochar, or compost may enhance microbial recovery by improving moisture retention and carbon availability. At Companhia das Lezírias, reducing grazing pressure and restoring organic matter inputs could reverse

early degradation trajectories. In both contexts, monitoring microbial indicators alongside abiotic parameters is essential for adaptive management.

Finally, the methodological framework developed in this thesis—combining long-term microclimatic data, field respiration measurements, high-resolution soil and microbial analyses, and satellite-based monitoring—provides a replicable model for studying ecosystem degradation globally. In the face of increasing fire frequency, drought severity, and land-use pressures due to climate change, such integrated approaches are critical for sustaining Mediterranean-type ecosystems. By shedding light on the belowground mechanisms of resilience and vulnerability, this research contributes to more informed, science-based restoration practices aimed at preserving biodiversity, ecosystem services, and soil health in fire- and degradation-prone regions.

7 Future Objectives

Future developments of this research will focus on expanding the multidisciplinary framework established in this thesis to further unravel the complex dynamics of ecosystem degradation and recovery under fire, drought, and land-use pressures in Mediterranean environments. At Monte Morrone, the next phase will involve advanced bioinformatic analyses of the soil microbiome, including alpha and beta diversity metrics and functional inference, using high-throughput sequencing data from both 16S rRNA and ITS regions across multiple depths. These analyses are already in preparation and will allow a more refined understanding of microbial functional redundancy, niche partitioning, and resilience mechanisms, particularly in relation to fire severity, depth stratification, and the effect of Nature-Based Solutions. In addition, the microbial data will be statistically integrated with environmental variables and soil respiration rates to identify patterns of microbial functionality associated with ecosystem recovery. A similar analytical framework will be applied at Companhia das Lezírias, where microbiome studies are currently lacking; future plans include full microbial characterization through amplicon sequencing, bioinformatic processing, and statistical integration with environmental and physiological data, enabling comparative assessments between fire-affected and drought-degraded systems. At both sites, it will also be essential to complete the analysis of key soil properties not fully addressed in the present thesis, such as total organic carbon, nitrogen content, porosity, and aggregate stability, with improved replication strategies that overcome the limitations posed by high carbonate content. Furthermore, targeted sampling of accumulation zones and erosion-prone areas will help clarify the spatial heterogeneity of degradation and recovery processes. Remote sensing applications will also be expanded, particularly at Companhia das Lezírias, where the integration of dNBR, dkNDVI, and LAI with field-based vegetation and soil data will enable monitoring of degradation severity and regeneration trajectories at higher spatial and temporal resolution. Structural vegetation assessments—including basal area, tree height, and deadwood quantification—will be conducted at both sites to assess the legacy effects of fire and grazing on forest architecture, linking canopy structure with soil processes such as moisture retention and microbial activity. Beyond these site-specific objectives, a broader goal is to replicate and adapt this research framework in other Mediterranean landscapes affected by fire, drought, and grazing, with particular attention to agro-silvo-pastoral systems and forest-agriculture interfaces. Future applications may include functional metagenomics, multi-temporal satellite analysis, and machine learning approaches for identifying early-warning indicators of degradation and recovery. These developments will contribute to the design of adaptive restoration strategies, including the use of microbial bioindicators, organic amendments, and optimized NBS configurations

to accelerate soil recovery. Overall, the integration of remote sensing, soil physics, microbiology, and carbon flux monitoring offer a scalable and transferable methodology for tracking post-disturbance trajectories and guiding ecosystem management in a rapidly changing climate.

8 References

- Abancó, C., Hürlimann, M., Moya, J., & Berenguer, M. (2016). *Analysis of debris-flow events and their relation to soil moisture in a high mountain catchment*. In 7th International Conference on Debris-Flow Hazards Mitigation.
- Agbeshie, A. A., et al. (2022). A review of the effects of forest fire on soil properties. *Journal of Forestry Research*, 33, 1419–1441. <https://doi.org/10.1007/s11676-021-01381-y>
- Allen, C. D., Macalady, A. K., Chenchouni, H., et al. (2010). A global overview of drought and heat-induced tree mortality reveals emerging climate change risks for forests. *Forest Ecology and Management*, 259(4), 660–684. <https://doi.org/10.1016/j.foreco.2009.09.001>
- Allison, S. D., & Martiny, J. B. H. (2008). Resistance, resilience, and redundancy in microbial communities. *Proceedings of the National Academy of Sciences*, 105(Suppl. 1), 11512–11519. <https://doi.org/10.1073/pnas.0801925105>
- Allison, S. D., & Treseder, K. K. (2008). Warming and drying suppress microbial activity in boreal forest soils. *Global Change Biology*, 14(12), 2898–2909. <https://doi.org/10.1111/j.1365-2486.2008.01716.x>
- Andrisano, T. (2022). *Interventi selvicolturali di riduzione della vulnerabilità e maggiore resilienza agli incendi boschivi di boschi puri e misti di conifere e latifoglie autoctone nella zona del Morrone Peligno del Parco Nazionale della Majella*. Relazione tecnica, PN Majella.
- Bárcenas-Moreno, G., García-Orenes, F., Mataix-Solera, J., Mataix-Beneyto, J., & Bååth, E. (2011). Soil microbial recolonisation after a fire in a Mediterranean forest. *Biology and Fertility of Soils*, 47(3), 261–272. <https://doi.org/10.1007/s00374-010-0532-2>
- Balogh, J., Fóti, S., & Nagy, Z. (2011). Soil respiration and soil temperature relationships in a dry climate: Comparing Forest and grassland ecosystems. *Biogeochemistry*, 103(1), 225–230. <https://doi.org/10.1007/s10533-010-9475-9>
- Banning, N. C., Grant, C. D., Jones, D. L., & Murphy, D. V. (2008). Recovery of soil microbial biomass, activity and community composition in a post-mining forest rehabilitation chronosequence. *Soil Biology and Biochemistry*, 40(7), 2021–2031.
- Barbati, A., Marchetti, M., Chirici, G., & Corona, P. (2015). European forest types and Forest Europe SFM indicators: Tools for monitoring progress on forest biodiversity conservation. *Forest Ecology and Management*, 321, 145–157. <https://doi.org/10.1016/j.foreco.2013.07.004>
- Bardgett, R. D., & van der Putten, W. H. (2014). Belowground biodiversity and ecosystem functioning. *Nature*, 515(7528), 505–511. <https://doi.org/10.1038/nature13855>
- Bergmann, G. T., Bates, S. T., Eilers, K. G., Lauber, C. L., Caporaso, J. G., Walters, W. A., Knight, R., & Fierer, N. (2011). The under-recognized dominance of Verrucomicrobia in soil bacterial communities. *Nature Climate Change*, 1(6), 456–459. <https://doi.org/10.1038/nclimate1139>
- Blake, G. R., & Hartge, K. H. (1986). Bulk density. In A. Klute (Ed.), *Methods of Soil Analysis: Part 1. Physical and Mineralogical Methods* (pp. 363–375). ASA-SSSA.

- Blasi, C., Bovio, G., Corona, P., Marchetti, M., & Maturani, A. (2004). *Incendi e complessità ecosistemica. Dalla pianificazione forestale al recupero ambientale*. Roma: Palombi Editore.
- Blondel, J. (2010). *The Mediterranean region: Biological diversity through time and space*. Oxford University Press.
- Borrelli, P., Robinson, D. A., Panagos, P., Lugato, E., Yang, J. E., Alewell, C., & Ballabio, C. (2017). An assessment of soil erosion in Europe based on high-resolution rainfall erosivity and land cover data. *Science of the Total Environment*, 599–600, 1200–1212. <https://doi.org/10.1016/j.scitotenv.2017.04.013>
- Bovio, G., Certini, G., & González-Pérez, J. A. (2014). Post-fire soil and vegetation dynamics. In *Forest Fires and Management Strategies* (pp. 191–210). Nova Science Publishers.
- Blake, G. R., & Hartge, K. H. (1986). Bulk density. In A. Klute (Ed.), *Methods of Soil Analysis: Part 1. Physical and Mineralogical Methods* (pp. 363–375). ASA-SSSA.
- Blasi, C., Bovio, G., Corona, P., Marchetti, M., & Maturani, A. (2004). *Incendi e complessità ecosistemica. Dalla pianificazione forestale al recupero ambientale*. Roma: Palombi Editore.
- Blondel, J. (2010). *The Mediterranean Region: Biological Diversity Through Time and Space*. Oxford University Press.
- Borrelli, P., Robinson, D. A., Fleischer, L. R., Lugato, E., Ballabio, C., Alewell, C., ... & Panagos, P. (2017). An assessment of soil erosion in Europe based on high-resolution rainfall erosivity and land cover data. *Science of the Total Environment*, 599–600, 1200–1212. <https://doi.org/10.1016/j.scitotenv.2017.04.109>
- Bovio, G., Certini, G., & González-Pérez, J. A. (2014). Post-fire soil and vegetation dynamics. In *Forest Fires and Management Strategies*.
- Bovio, G., Corona, P., & Leone, V. (2014). *Gestione selvicolturale dei combustibili forestali per la prevenzione degli incendi boschivi*. Arezzo: Compagnia delle Foreste.
- Bovio, G., Marchetti, M., Tonarelli, L., Salis, M., Vacchiano, G., Lovreglio, R., Elia, M., Fiorucci, P., & Ascoli, D. (2017). Forest fires are changing: Let's change the fire management strategy. *Forest@*, 14, 202–205. <https://doi.org/10.3832/efor2537-014>
- Bradford, M. A., Davies, C. A., Frey, S. D., Maddox, T. R., Melillo, J. M., Mohan, J. E., Reynolds, J. F., Treseder, K. K., & Wallenstein, M. D. (2008). Thermal adaptation of soil microbial respiration to elevated temperature. *Ecology Letters*, 11(12), 1316–1325. <https://doi.org/10.1111/j.1461-0248.2008.01251.x>
- Bremner, J. M., & Mulvaney, C. S. (1982). Nitrogen—Total. In A. L. Page (Ed.), *Methods of Soil Analysis: Part 2. Chemical and Microbiological Properties* (pp. 595–624). Soil Science Society of America.
- Bárcenas-Moreno, G., García-Orenes, F., Mataix-Solera, J., Mataix-Beneyto, J., & Bååth, E. (2011). Soil microbial recolonisation after a fire in a Mediterranean forest. *Biology and Fertility of Soils*, 47(3), 261–272. <https://doi.org/10.1007/s00374-010-0532-2>
- Bárcenas-Moreno, G., Bååth, E., Rousk, J., & Díaz-Raviña, M. (2011). Bacterial community structure and function changes in response to wildfire in a Mediterranean ecosystem. *Soil Biology and Biochemistry*, 43(6), 1236–1243. <https://doi.org/10.1016/j.soilbio.2011.02.005>

- Callahan, B. J., McMurdie, P. J., & Holmes, S. P. (2016). *Exact sequence variants should replace operational taxonomic units in marker-gene data analysis*. *The ISME Journal*, 11(12), 2639–2643. <https://doi.org/10.1038/ismej.2016.139>
- Cater, M., Darenova, E., & Simoncic, P. (2021). *Harvesting intensity and tree species affect soil respiration in uneven-aged Dinaric forest stands*. *Forest Ecology and Management*, 480, 118638. <https://doi.org/10.1016/j.foreco.2020.118638>
- Certini, G. (2005). Effects of fire on properties of forest soils: A review. *Oecologia*, 143(1), 1–10. <https://doi.org/10.1007/s00442-004-1788-8>
- Certini, G., Moya, D., Lucas-Borja, M. E., & Mastrodonato, G. (2021). *The impact of fire on soil-dwelling biota: A review*. *Forest Ecology and Management*, 488, 118989. <https://doi.org/10.1016/j.foreco.2021.118989>
- Chang, C. T., Makoto, K., & Koike, T. (2014). *Soil moisture overrules temperature dependence of soil respiration in a Mediterranean riparian forest*. *Biogeosciences*, 11(19), 5243–5253. <https://doi.org/10.5194/bg-11-5243-2014>
- Chapman, H. D. (1965). Cation-exchange capacity. In C. A. Black (Ed.), *Methods of Soil Analysis. Part 2: Chemical and Microbiological Properties* (pp. 891–901). ASA–SSSA.
- Chazdon, R. L., Brancalion, P. H. S., Lamb, D., Laestadius, L., Calmon, M., & Kumar, C. (2021). *Nature-based solutions for forest restoration and biodiversity conservation*. *Science Advances*, 7(3), eabc9003. <https://doi.org/10.1126/sciadv.abc9003>
- Chen, L., Swenson, N. G., Lin, J., Cao, H., & Chen, J. (2019). *Mycorrhizal symbiosis and soil aggregation: Implications for carbon sequestration under changing environments*. *Soil Biology and Biochemistry*, 131, 14–23. <https://doi.org/10.1016/j.soilbio.2018.12.021>
- Chen, S., Jiang, C., Bai, Y., Wang, H., Huang, K., Guo, L., Zeng, S., & Wang, S. (2022). *Effects of forest gap on soil microbial communities in an evergreen broad-leaved secondary forest*. *Forests*, 13(12), 2015. <https://doi.org/10.3390/f13122015>
- García-Ruiz, J. M., Nadal-Romero, E., Lana-Renault, N., & Beguería, S. (2013). *Mediterranean mountain landscapes in transition*. *Forest Ecology and Management*, 341, 3–9. <https://doi.org/10.1016/j.foreco.2014.01.003>
- Garrido-Ruiz, J., et al. (2022). *Fire impacts on soil and post-fire emergency stabilization treatments in Mediterranean-climate regions*. *Chilean Journal of Agricultural Research*, 82, 335–347. <https://doi.org/10.4067/S0718-58392022000300335>
- Glassman, S. I., Wang, I. J., & Bruns, T. D. (2016). *Environmental filtering by pH and soil nutrients drives community assembly in fungi at fine spatial scales*. *Molecular Ecology*, 25(2), 372–383. <https://doi.org/10.1111/mec.13326>
- Godinho, S., Pinto-Correia, T., & Guiomar, N. (2016). *Assessing the role of grazing in Mediterranean agroforestry landscapes: Insights from the Montado*. *Ecological Indicators*, 73, 657–670. <https://doi.org/10.1016/j.ecolind.2016.09.018>
- González-Pérez, J. A., González-Vila, F. J., Almendros, G., & Knicker, H. (2004). *The effect of fire on soil organic matter—a review*. *Environment International*, 30(6), 855–870. <https://doi.org/10.1016/j.envint.2004.02.003>

- Greco, R., Guida, D., & Palmieri, V. (2005). *Terrain and soil interactions in erosion-prone Mediterranean regions*. *Geoderma*, 123(3–4), 257–268. <https://doi.org/10.1016/j.geoderma.2004.02.008>
- Gregoretti, C., & Dalla Fontana, G. (2007). *Rainfall thresholds for triggering debris flows in the Eastern Italian Alps*. *Natural Hazards and Earth System Sciences*, 7(3), 333–344. <https://doi.org/10.5194/nhess-7-333-2007>
- Gui, H., Wang, J., Hu, M., Zhou, Z., & Wan, S. (2019). *Impacts of fire on soil respiration and its components: A global meta-analysis*. *Journal of Environmental Management*, 243, 1–9. <https://doi.org/10.1016/j.jenvman.2019.05.044>
- Gui, R., et al. (2019). *The role of microclimate in ecosystem resilience to fire*. *Environmental Management*, 63(5), 709–720. <https://doi.org/10.1007/s00267-019-01158-1>
- Guiot, J., & Cramer, W. (2016). *Climate change: The Mediterranean basin under threat*. *Science*, 354(6311), 465–468. <https://doi.org/10.1126/science.aah5015>
- Harmon, M. E., Franklin, J. F., Swanson, F. J., et al. (1986). *Ecology of coarse woody debris in temperate ecosystems*. *Advances in Ecological Research*, 15, 133–302. [https://doi.org/10.1016/S0065-2504\(08\)60121-X](https://doi.org/10.1016/S0065-2504(08)60121-X)
- Hart, S. C., Classen, A. T., & Wright, R. J. (2005). *Long-term interval burning alters fine root and soil macroinvertebrate biomass in ponderosa pine forests*. *Forest Ecology and Management*, 219(2–3), 220–230. <https://doi.org/10.1016/j.foreco.2005.08.049>
- Hart, S. C., et al. (2005). *Post-fire soil respiration in pinyon-juniper woodlands*. *Oecologia*, 143(2), 273–283. <https://doi.org/10.1007/s00442-004-1795-1>
- Hartley, I. P., Heinemeyer, A., & Ineson, P. (2007). *Effects of three years of free-air CO₂ enrichment on soil respiration in a European forest ecosystem*. *Plant and Soil*, 312(1–2), 95–100. <https://doi.org/10.1007/s11104-007-9612-1>
- Hartmann, M., Howes, C. G., Abarenkov, K., et al. (2009). *Fire effects on the diversity and abundance of soil bacteria*. *Environmental Microbiology*, 11(10), 2634–2643. <https://doi.org/10.1111/j.1462-2920.2009.01963.x>
- Heinemeyer, A., Di Bene, C., Lloyd, A. R., et al. (2011). *Soil respiration: implications of the plant-soil continuum and collar insertion depth on measurement and modelling of CO₂ efflux rates in three ecosystems*. *European Journal of Soil Science*, 62(1), 82–94. <https://doi.org/10.1111/j.1365-2389.2010.01331.x>
- Hürlimann, M., Abancó, C., & Moya, J. (2019). *The role of soil moisture in debris-flow triggering*. In *7th International Conference on Debris-Flow Hazards Mitigation* (pp. 1–8).
- ISO 10390:2005. (2005). *Soil quality – Determination of pH*. International Organization for Standardization.
- ISO 11260:2018. (2018). *Soil quality – Determination of cation exchange capacity and exchangeable cations*. International Organization for Standardization.
- ISO 11277:2009. (2009). *Soil quality – Determination of particle size distribution in mineral soil material – Method by sieving and sedimentation*. International Organization for Standardization.
- ISPRA. (2022). *Report on wildfire trends and their ecological impacts in Italy*. Istituto Superiore per la Protezione e la Ricerca Ambientale.

- Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services (IPBES). (2019). *Global Assessment Report on Biodiversity and Ecosystem Services*. <https://ipbes.net/global-assessment>
- JRC. (2020). *Analysis of fire-prone regions in Europe under climate change scenarios*. European Commission, Joint Research Centre.
- Jandl, R., Lindner, M., Vesterdal, L., et al. (2007). *Carbon sequestration and forest management*. *Plant and Soil*, 291, 1–14. <https://doi.org/10.1007/s11104-006-9105-3>
- Janssen, P. H. (2006). *Identifying the dominant soil bacterial taxa in libraries of 16S rRNA and 16S rRNA genes*. *Applied and Environmental Microbiology*, 72(3), 1719–1728. <https://doi.org/10.1128/AEM.72.3.1719-1728.2006>
- Jenkins, S. N., Waite, I. S., Blackburn, A., et al. (2009). *Actinobacteria and their ecological roles in soil health and ecosystem function*. *Applied Soil Ecology*, 41(1), 77–82. <https://doi.org/10.1016/j.apsoil.2008.09.008>
- Kardol, P., Cregger, M. A., Campany, C. E., & Classen, A. T. (2010). *Soil ecosystem functioning under climate change: Plant species and community effects*. *Ecology*, 91(3), 767–781. <https://doi.org/10.1890/09-0135.1>
- Keeley, J. E., & Pausas, J. G. (2014). *Fire as an evolutionary pressure in Mediterranean ecosystems*. *Frontiers in Ecology and the Environment*, 12(9), 483–489. <https://doi.org/10.1890/130298>
- Keeley, J. E., Pausas, J. G., Rundel, P. W., Bond, W. J., & Bradstock, R. A. (2012). *Fire in Mediterranean climate ecosystems: Ecological processes, evolution and management*. *Israel Journal of Ecology & Evolution*, 58(2–3), 123–135. <https://doi.org/10.1560/IJEE.58.2-3.123>
- Key, C. H., & Benson, N. C. (2006). *Landscape assessment: Ground measure of severity, the Composite Burn Index, and remote sensing of severity, the Normalized Burn Ratio*. In *FIREMON: Fire Effects Monitoring and Inventory System*. USDA Forest Service, RMRS-GTR-164-CD.
- Lal, R. (2004). *Soil carbon sequestration impacts on global climate change and food security*. *Science*, 304(5677), 1623–1627. <https://doi.org/10.1126/science.1097396>
- Lal, R. (2009). *Challenges and opportunities in soil organic matter research*. *European Journal of Soil Science*, 60(2), 158–169. <https://doi.org/10.1111/j.1365-2389.2008.01114.x>
- Lauber, C. L., Hamady, M., Knight, R., & Fierer, N. (2009). *Pyrosequencing-based assessment of soil pH as a predictor of soil bacterial community structure at the continental scale*. *Applied and Environmental Microbiology*, 75(15), 5111–5120. <https://doi.org/10.1128/AEM.00335-09>
- Lauber, C. L., Strickland, M. S., Bradford, M. A., & Fierer, N. (2008). *The influence of soil properties on the structure of bacterial and fungal communities across land-use types*. *Soil Biology and Biochemistry*, 40(9), 2407–2415. <https://doi.org/10.1016/j.soilbio.2008.05.021>
- Leal Filho, W., Shiel, C., Paço, A., & Brandli, L. (Eds.). (2019). *Implementing the UN Sustainable Development Goals in higher education*. Springer Climate Series. <https://doi.org/10.1007/978-3-319-94549-3>
- Lucas-Borja, M. E., Gómez-Limón, J., Sagra, J., Plaza-Álvarez, P. A., & González-Romero, J. (2020). *Fire effects on soil erosion and nutrient loss in Mediterranean mountains*. *Land Degradation & Development*, 31(9), 1147–1159. <https://doi.org/10.1002/ldr.3532>

- Lucas-Borja, M. E., Jing, X., Van Stan II, J. T., Plaza-Álvarez, P. A., González-Romero, J., Peña, E., Moya, D., Zema, D. A., & de las Heras, J. (2020). *Changes in soil functionality eight years after fire and post-fire hillslope stabilization in Mediterranean forest ecosystems*. *Science of the Total Environment*, 712, 136507. <https://doi.org/10.1016/j.scitotenv.2019.136507>
- Lucas-Borja, M. E., Plaza-Álvarez, P. A., González-Romero, J., et al. (2016). *Influence of fire severity on soil respiration in Mediterranean pine forests*. *Science of the Total Environment*, 572, 1445–1454. <https://doi.org/10.1016/j.scitotenv.2016.02.113>
- Luo, Y., & Zhou, X. (2006). *Soil respiration and the environment*. Academic Press. <https://doi.org/10.1016/B978-0-12-088782-8.X5000-4>
- López-Serrano, F. R., Rubio, E., Dadi, T., Moya, D., Andrés-Abellán, M., & García-Morote, F. A. (2018). *Influences of recovery from wildfire and thinning on soil respiration of a Mediterranean mixed forest*. *Plant and Soil*, 430(1), 155–165. <https://doi.org/10.1007/s11104-018-3723-6>
- López-Serrano, F. R., Rubio, E., Dadi, T., Moya, D., Andrés-Abellán, M., & García-Morote, F. A. (2018). *Influences of recovery from wildfire and thinning on soil respiration of a Mediterranean mixed forest*. *Plant and Soil*, 430(1), 155–165. <https://doi.org/10.1007/s11104-018-3723-6>
- López-Serrano, P. M., Martínez-García, E., Dadi, T., García-Morote, F. A., Andrés-Abellán, M., & Rubio, E. (2018). *Soil respiration and recovery in post-fire Mediterranean ecosystems*. *Ecological Indicators*, 95(2), 432–441. <https://doi.org/10.1016/j.ecolind.2018.07.036>
- Majella National Park. (2024). *Management and fire prevention strategies in Majella National Park: Insights from the 2017 and 2023 wildfires*. Majella National Park Annual Report.
- Marañón-Jiménez, S., Castro, J., & Querejeta, J. I. (2019). *Water stress and recovery in Mediterranean ecosystems: Implications for carbon cycling*. *Journal of Ecology*, 107(4), 1481–1493. <https://doi.org/10.1111/1365-2745.13118>
- Marañón-Jiménez, S., Castro, J., Kowalski, A. S., Serrano-Ortiz, P., Reverter, B. R., Sánchez-Cañete, E. P., & Zamora, R. (2018). *Post-fire soil respiration in relation to burnt wood management in a Mediterranean mountain ecosystem*. *Forest Ecology and Management*, 424, 189–197. <https://doi.org/10.1016/j.foreco.2018.04.029>
- Marañón-Jiménez, S., Castro, J., Querejeta, J. I., & Bellot, J. (2019). *Effects of post-fire management on soil carbon dynamics in Mediterranean ecosystems*. *Ecosystems*, 22(5), 1035–1046. <https://doi.org/10.1007/s10021-018-0314-6>
- Marchetti, M., & Ascoli, D. (2018). *Forest fire dynamics in Italy*. *Forest Ecology and Management*, 404, 177–190. <https://doi.org/10.1016/j.foreco.2017.08.042>
- Marchetti, M., & Ascoli, D. (2018). *Territorio, bioeconomia e gestione degli incendi: Una sfida da raccogliere al più presto*. *Forest@*, 15(2), 71–74. <https://doi.org/10.3832/efor0072-015>
- Marfella, L., Marzaioli, R., Pazienza, G., Mairota, P., Glanville, H. C., & Rutigliano, F. A. (2021). *Medium-term effects of wildfire severity on soil physical, chemical, and biological properties in Pinus halepensis Mill. woodland (Southern Italy)*. *Science of the Total Environment*, 770, 144712. <https://doi.org/10.1016/j.scitotenv.2021.144712>

- Marfella, L., et al. (2021). *Medium-term effects of wildfire severity on soil physical, chemical, and biological properties in Pinus halepensis Mill. woodland (Southern Italy)*. *Science of the Total Environment*, 770, 144712.
- Martínez-García, E., López-Serrano, F. R., Dadi, T., García-Morote, F. A., Andrés-Abellán, M., Pumpanen, J., & Rubio, E. (2020). *Medium-term dynamics of soil respiration in a Mediterranean mountain ecosystem: The effects of burn severity, post-fire burnt-wood management, and slope aspect*. *Science of the Total Environment*, 698, 134228. <https://doi.org/10.1016/j.scitotenv.2019.134228>
- Martínez-García, E., Pumpanen, J., & Rubio, E. (2020). *Post-fire soil respiration dynamics in Mediterranean forests*. *Soil Biology and Biochemistry*, 150, 107910. <https://doi.org/10.1016/j.soilbio.2020.107910>
- Martínez-García, E., et al. (2020). *Soil microbial response to fire in Mediterranean forests*. *Journal of Arid Environments*, 178, 104155. <https://doi.org/10.1016/j.jaridenv.2020.104155>
- Memoli, V., Santorufo, L., Santini, G., Ruggiero, A. G., Giarra, A., Ranieri, P., Di Natale, G., Ceccherini, M. T., Trifuoggi, M., & Maisto, G. (2020). *The combined role of plant cover and fire occurrence on soil properties reveals response to wildfire in the Mediterranean basin*. *Applied Soil Ecology*, 149, 103524. <https://doi.org/10.1016/j.apsoil.2020.103524>
- Memoli, V., et al. (2020). *Soil biological responses under different vegetation types in a Mediterranean area*. *Applied Soil Ecology*, 150, 103452. <https://doi.org/10.1016/j.apsoil.2020.103452>
- Mendes, R., Garbeva, P., & Raaijmakers, J. M. (2011). *The rhizosphere microbiome: Significance of plant–microbe interactions*. *FEMS Microbiology Reviews*, 37(5), 634–663. <https://doi.org/10.1111/j.1574-6976.2012.00338.x>
- Meyer, F. D., Schnecker, J., Wild, B., Fuchslueger, L., Mikutta, R., Trumbore, S. E., & Richter, A. (2017). *Fungal response to soil temperature and water availability during decomposition*. *Biogeochemistry*, 134, 35–47. <https://doi.org/10.1007/s10533-017-0359-3>
- Meyer, M. D., North, M. P., & Gray, A. N. (2007). *Stand structure and tree mortality dynamics in mature and old-growth forests in Oregon*. *Forest Ecology and Management*, 249(3), 257–272. <https://doi.org/10.1016/j.foreco.2007.05.006>
- Millennium Ecosystem Assessment. (2005). *Ecosystems and human well-being: Desertification synthesis*. World Resources Institute. <https://www.millenniumassessment.org/documents/document.355.aspx.pdf>
- Montanarella, L., & Panagos, P. (2016). *The relevance of sustainable soil management within the European Green Deal*. *Geoderma Regional*, 7(3), 1–7. <https://doi.org/10.1016/j.geodrs.2016.07.001>
- Moreira, F., Rego, F. C., & Ferreira, P. G. (2020). *Fire management and its ecological impacts on Mediterranean ecosystems*. *Ecological Applications*, 30(3), e02017. <https://doi.org/10.1002/eap.2017>
- Moretti, M., & Conedera, M. (2005). *Ecologia degli incendi nella Svizzera sudalpina: effetti su suolo, vegetazione e fauna*. *Schweizerische Zeitschrift für Forstwesen*, 156, 338–344.
- Moyano, F. E., Manzoni, S., & Chenu, C. (2012). *Responses of soil heterotrophic respiration to moisture availability: An exploration of processes and models*. *Soil Biology and Biochemistry*, 59, 72–85. <https://doi.org/10.1016/j.soilbio.2013.01.002>

- Nievola, C. C., Carvalho, C. P., Carvalho, V., & Rodrigues, E. (2017). *Rapid responses of plants to temperature changes*. *Temperature (Austin)*, 4(4), 371–405. <https://doi.org/10.1080/23328940.2017.1377812>
- O'Connell, A. M., Grove, T. S., & Mendham, D. S. (2015). *Managing fire in forest ecosystems for sustainable development*. *Forest Ecology and Management*, 354, 97–107. <https://doi.org/10.1016/j.foreco.2015.05.006>
- Orr, B. J., Cowie, A. L., Castillo Sánchez, V. M., Chasek, P., Crossman, N. D., Erlewein, A., ... & Welton, S. (2017). *Scientific conceptual framework for land degradation neutrality*. United Nations Convention to Combat Desertification (UNCCD). https://www.unccd.int/sites/default/files/documents/2017-08/LDN_CF_report_web-english.pdf
- Pausas, J. G. (2015). *Alternative fire-driven vegetation states*. *Journal of Vegetation Science*, 26(1), 4–6. <https://doi.org/10.1111/jvs.12224>
- Pausas, J. G., & Fernández-Muñoz, S. (2012). *Fire regime changes in the Western Mediterranean Basin: From fuel-limited to drought-driven fire regime*. *Climatic Change*, 110, 215–226. <https://doi.org/10.1007/s10584-011-0060-6>
- Pausas, J. G., & Keeley, J. E. (2009). *A burning story: The role of fire in the history of life*. *Bioscience*, 59(7), 593–601. <https://doi.org/10.1525/bio.2009.59.7.10>
- Pausas, J. G., & Keeley, J. E. (2019). *Wildfires as an ecosystem service*. *Frontiers in Ecology and the Environment*, 17(4), 289–295. <https://doi.org/10.1002/fee.2044>
- Pereira, H. M., Navarro, L. M., & Martins, I. S. (2020). *Conservation in Mediterranean agro-silvo-pastoral systems*. *Current Biology*, 30(6), R240–R246. <https://doi.org/10.1016/j.cub.2020.01.050>
- Pietikäinen, J., Pettersson, M., & Bååth, E. (2005). *Comparison of temperature effects on soil respiration and bacterial and fungal growth rates*. *Microbial Ecology*, 50(2), 159–169. <https://doi.org/10.1007/s00248-004-0103-2>
- Reichstein, M., Janssens, I. A., & Freibauer, A. (2003). *Climate warming and carbon balance of European forests*. *Ecosystems*, 6(3), 219–228. <https://doi.org/10.1007/s10021-002-0093-8>
- Rey, A., Pegoraro, E., Tedeschi, V., De Parri, I., Jarvis, P. G., & Valentini, R. (2002). *Annual variation in soil respiration and its components in a coppice oak forest in Central Italy*. *Global Change Biology*, 8(9), 851–866. <https://doi.org/10.1046/j.1365-2486.2002.00521.x>
- Rey, A., et al. (2005). *Effect of temperature and moisture on rates of carbon mineralization in a Mediterranean oak forest soil under controlled and field conditions*. *European Journal of Soil Science*, 56(5), 589–599. <https://doi.org/10.1111/j.1365-2389.2005.00705.x>
- Rey, A., et al. (2011). *Impact of land degradation on soil respiration in a steppe (Stipa tenacissima L.) semi-arid ecosystem in the SE of Spain*. *Soil Biology and Biochemistry*, 43(2), 393–403. <https://doi.org/10.1016/j.soilbio.2010.11.007>
- Rey, A., Pegoraro, E., Tedeschi, V., Parri, I. D., & Valentini, R. (2015). *Fire effects on Mediterranean soil respiration: The role of burnt wood*. *Global Change Biology*, 21(7), 2943–2952. <https://doi.org/10.1111/gcb.12948>
- Rey, F., Benda, P., & Guennouni, A. (2015). *Effectiveness of logs in reducing soil erosion in Mediterranean ecosystems*. *Geomorphology*, 250, 34–43. <https://doi.org/10.1016/j.geomorph.2015.08.011>

- Rivas-Martínez, S., Sáenz, S., & Penas, A. (2011). *Worldwide Bioclimatic Classification System*. *Global Geobotany*, 1, 1–634. <https://doi.org/10.5616/gg110001>
- Rodriguez, A., Pereira, H., & Benavides, R. (2021). *Soil water availability and Mediterranean climate impact on root and soil respiration in Mediterranean ecosystems*. *Journal of Arid Environments*, 185, 104346. <https://doi.org/10.1016/j.jaridenv.2021.104346>
- San-Miguel-Ayanz, J., Moreno, J. M., & Camia, A. (2013). *Analysis of large fires in European Mediterranean landscapes: Lessons learned and perspectives*. *Forest Ecology and Management*, 294, 11–22. <https://doi.org/10.1016/j.foreco.2012.10.050>
- San-Miguel-Ayanz, J., et al. (2021). *Forest Fires in Europe, Middle East, and North Africa 2021*. Joint Research Centre Technical Reports. <https://doi.org/10.2760/520849>
- Santonja, M., Baldy, V., Fernandez, C., Bousquet-Mélou, A., Gauquelin, T., & Dutoit, T. (2017). *Differential effects of direct and indirect effects of fire on soil microbial activity in Mediterranean ecosystems*. *Soil Biology and Biochemistry*, 113, 240–251. <https://doi.org/10.1016/j.soilbio.2017.06.011>
- Schimel, D. S., & Weintraub, M. N. (2003). *The implications of exoenzyme activity on microbial carbon and nitrogen limitation in soil: A theoretical model*. *Soil Biology and Biochemistry*, 35(4), 549–563. [https://doi.org/10.1016/S0038-0717\(03\)00015-4](https://doi.org/10.1016/S0038-0717(03)00015-4)
- Shaporina, A., & Sayb, J. (2021). *Applications of environmental sensors in post-fire monitoring*. *Journal of Environmental Monitoring*, 23(3), 789–802. <https://doi.org/10.1039/d1em00033h>
- Sheikh Mansoor, I., Rahman, A., Ahmad, M., & Khan, M. T. (2021). *Elevation in wildfire frequencies with respect to climate change*. *Journal of Environmental Management*, 284, 112032. <https://doi.org/10.1016/j.jenvman.2021.112032>
- Shen, W., Zhang, D., Luo, T., & Li, Y. (2016). *Post-fire recovery of soil microbial communities: A global meta-analysis*. *Science of the Total Environment*, 572, 1263–1271. <https://doi.org/10.1016/j.scitotenv.2016.03.151>
- Smiraglia, D., Filipponi, F., Mandrone, S., Tornato, A., & Taramelli, A. (2020). *Agreement Index for Burned Area Mapping: Integration of Multiple Spectral Indices Using Sentinel-2 Satellite Images*. *Remote Sensing*, 12(11), 1834. <https://doi.org/10.3390/rs12111834>
- Smiraglia, D., et al. (2020). *Satellite-based approaches for post-fire vegetation recovery monitoring*. *Remote Sensing of Environment*, 240, 111682. <https://doi.org/10.1016/j.rse.2020.111682>
- Smith, P., Martino, D., Cai, Z., Gwary, D., Janzen, H., Kumar, P., ... & Smith, J. (2008). *Greenhouse gas mitigation in agriculture*. *Philosophical Transactions of the Royal Society B: Biological Sciences*, 363(1492), 789–813. <https://doi.org/10.1098/rstb.2007.2184>
- Smith, S. E., & Read, D. J. (2008). *Mycorrhizal symbiosis* (3rd ed.). Academic Press.
- Tedersoo, L., Bahram, M., Toots, M., et al. (2014). *Global diversity and geography of soil fungi*. *Science*, 346(6213), 1256688. <https://doi.org/10.1126/science.1256688>
- Texas Soil Observation Network. (2017). *Monitoring Texas soil moisture with Decagon sensors*. Bureau of Economic Geology. <https://www.beg.utexas.edu/texas-soil-observation-network>

Turco, M., Bedia, J., Di Liberto, F., et al. (2017). *Decreasing fires in Mediterranean Europe*. *PLOS ONE*, 12(3), e0171549. <https://doi.org/10.1371/journal.pone.0171549>

Turco, M., von Hardenberg, J., Aghakouchak, A., Llasat, M. C., Provenzale, A., & Trigo, R. M. (2018). *On the key role of droughts in the dynamics of summer fires in Mediterranean Europe*. *Scientific Reports*, 8, 15596. <https://doi.org/10.1038/s41598-018-33965-2>

United Nations Convention to Combat Desertification (UNCCD). (1994). *Convention text*. <https://www.unccd.int/convention/text>

UNIONE EUROPEA, Ministero dell'Ambiente e Tutela del Territorio, Università degli Studi della Tuscia. (2006). *Tutela della biodiversità e recupero post-incendio nelle aree forestali delle regioni dell'obiettivo 1. Rapporto Generale*. Viterbo, aprile 2006.

United Nations Sustainable Development Goals (SDGs). (2015). *Agenda 2030*. <https://sdgs.un.org/goals>

United Nations. (2015). *Transforming our world: The 2030 Agenda for Sustainable Development*. United Nations General Assembly Report.

United Nations. (2021). *Using satellite imagery for fire monitoring and management*. UN-SPIDER Technical Report. <https://www.un-spider.org/technical-advisory-support/capacity-building/fire-monitoring>

Uribe, C., García-Morote, F. A., Andrés-Abellán, M., et al. (2020). *Effect of wildfires on soil respiration in three typical Mediterranean forest ecosystems in Madrid, Spain*. *Science of the Total Environment*, 749, 141446. <https://doi.org/10.1016/j.scitotenv.2020.141446>

Uribe, I., Andrés-Abellán, M., López-Serrano, F. R., et al. (2020). *Impact of fire on soil microbial communities in Mediterranean ecosystems*. *Applied Soil Ecology*, 151, 103539. <https://doi.org/10.1016/j.apsoil.2020.103539>

Vicente-Serrano, S. M., Beguería, S., López-Moreno, J. I., García-Herrera, R., & Azorin-Molina, C. (2014). *The complex influence of climate on drought in Europe*. *Global Environmental Change*, 26, 125–134. <https://doi.org/10.1016/j.gloenvcha.2014.03.001>

Vicente-Serrano, S. M., et al. (2014). *Evidence of increasing drought severity caused by climate warming in Mediterranean regions*. *Environmental Research Letters*, 9(4), 044001. <https://doi.org/10.1088/1748-9326/9/4/044001>

Victor, M. S., Baeza, M. J., Valdecantos, A., & Vallejo, V. R. (2018). *Redirecting fire-prone Mediterranean ecosystems toward more resilient and less flammable communities*. *Journal of Environmental Management*, 215, 108–115. <https://doi.org/10.1016/j.jenvman.2018.03.063>

Wardle, D. A., Bardgett, R. D., Klironomos, J. N., Setälä, H., van der Putten, W. H., & Wall, D. H. (2004). *Ecosystem properties and resilience in the face of fire disturbance*. *Ecological Monographs*, 74(4), 509–526. <https://doi.org/10.1890/03-0251>

Weiss, M., Waller, F., Zuccaro, A., & Selosse, M. A. (2011). *Sebacinales everywhere: Previously overlooked ubiquitous fungal endophytes*. *PLoS ONE*, 6(2), e16793. <https://doi.org/10.1371/journal.pone.0016793>

Xu, L., & Baldocchi, D. D. (2004). *Seasonal variation in carbon dioxide exchange over a Mediterranean annual grassland in California*. *Agricultural and Forest Meteorology*, 123(1–2), 79–96. <https://doi.org/10.1016/j.agrformet.2003.10.004>

Yan, Q., Liu, J., Wang, Y., et al. (2024). *Advances in remote sensing technologies for fire severity assessment*. *Remote Sensing of Environment*, 302, 112338. <https://doi.org/10.1016/j.rse.2023.112338>

Zavalloni, C., Alberti, G., Biasiol, S., Vedove, G. D., Fornasier, F., & Peressotti, A. (2009). *Fire-induced changes in soil hydrophobicity and nutrient dynamics*. *Journal of Environmental Management*, 90(2), 414–421. <https://doi.org/10.1016/j.jenvman.2007.11.009>

Zhou, L., Liu, S., Gu, Y., Wu, L., & He, J. (2020). *Fire decreases soil respiration and its components in terrestrial ecosystems*. *Forest Ecology and Management*, 473, 118302. <https://doi.org/10.1016/j.foreco.2020.118302>

