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Evaluating ecological restoration success by mapping regulating ecosystem services based on field and remote sensing approaches: a case study in Arrábida Natural Park

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Na noite calma,
a poesia da Serra adormecida
vem recolher-se em mim.
E o combate magnífico da Cor,
que eu vi de dia;
e o casamento do cheiro a maresia
com o perfume agreste do alecrim;
e os gritos mudos das rochas sequiosas que o Sol castiga
— passam a dar-se em mim.

E todo eu me alevanto e todo eu ardo.
Chego a julgar a Arrábida por Mãe,
quando não serei mais que seu bastardo.

Sebastião Gama in *"Serra-Mãe"* (1943)

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Abstract

Regulating ecosystem services (ES) are important for sustaining life on Earth, as they maintain the integrity of ecosystems and benefit human well-being. However, anthropogenic activities like quarrying put these services under great pressure by reducing biodiversity and extensively damaging the soil. Thus, restoration actions are necessary to recover these highly degraded areas and the ES they provide, and are one of the main focuses of the current "Decade on Ecosystem Restoration (2021-2023)" declared by the United Nations. However, the success of most restoration interventions remains unknown or is typically assessed solely through spatially-limited local approaches e.g. field sampling. In contrast, remote sensing (RS) repeatedly records and assesses changes in ecosystem characteristics at different spatial (and temporal) scales and can be combined with field data to produce valuable models and maps for large areas that can support adaptive management decisions.

This dissertation aimed to evaluate the success of the ecological restoration of a quarry located within the Arrábida Natural Park, a protected area in Portugal with a high ecological value that served as this study's reference ecosystem. For this, field and RS data were combined to assess key ecosystem attributes which are closely linked with several ES. Furthermore, field and RS data were used to analyse the trajectory over time (based on a chronosequence) and evaluate the restoration success through statistical analysis, modelling, and mapping of the ecosystem attributes. The restoration intervention mode applied in each site was also taken into consideration.

Overall, ecosystem recovery in the SECIL-Outão quarry was incomplete. Most studied attributes and associated ES did not resemble or had values below those of the reference ecosystem (e.g. soil carbon sequestration, soil fertility, habitat quality, resilience to fire and drought). Moreover, the hypothesis that older restored quarry areas would be closer than recent ones to providing the same regulating ES as the reference ecosystem was only observed for some attributes. Additionally, the recovery trajectory differed between ecosystem attributes. While some variables showed positive signs of improvement with restoration age (e.g. seedling density), the majority showed a stabilisation 15-20 years after restoration (e.g. productivity, soil decomposition, species similarity with the reference ecosystem). Furthermore, there was a decrease over time in several ecosystem functioning indicators, which might negatively affect the restored ecosystem's functioning and resilience.

In the older plantations at the limestone benches, pine cover seems to contribute to slow decomposition rates and decreased functional diversity, which might negatively affect ecosystem functioning and the quality of ES. This indicates limitations in the restored ecosystem, possibly due to the high competition and exhaustion of limited resources caused by the high Aleppo pine cover in the older restored areas. Furthermore, ecosystem resilience variables were negatively affected by pine cover, such as the proportion of resprouters (resilient to fire) or sclerophyll vegetation (resilient to drought). This tendency is particularly worrying in a climate change scenario, where higher fire frequency and severe droughts are expected. The pollination service estimated through flowering duration and entomophily was also negatively affected by pine cover.

Thus, older restored sites might be in a vulnerable state due to their decreasing resilience and pollination services, which could jeopardise restoration efforts in case of future disturbances, calling the need for adaptive management actions. Since the high pine cover became an obstacle to the progression of the plant community in the direction of the reference system, pine thinning is highly recommended to promote key functional traits and ecosystem functioning, resilience, and biotic fluxes.

On the other hand, shrub density in the restored sites proved to be very important in increasing species similarity with the reference ecosystem, taxonomic and functional diversity, and seedling density. Therefore, these results emphasise the importance of using native shrub species in the recovery of the quarry locations. Additionally, interventions that promote higher native shrub densities are highly suggested to improve the functioning, resilience, and self-maintenance of the restored areas, especially in the hydroseeded slopes where herbaceous cover is dominant and similarity to the reference is low.

Furthermore, the produced maps provided estimates for the spatial distribution of several ecological variables and ES, indicating which areas are more in need of adaptive management actions. Their estimates were overall comparable with the statistical analysis of the different intervention modes and the recovery success index calculated for each ecosystem attribute (which considered the different intervention modes and zone types). Therefore, the multiple maps created by statistical models supported the hypothesis that RS data, together with field data, allow upscaling information and the creation of valuable models that can be used to produce spatial information at the landscape scale.

In conclusion, this study found that the restored ecosystem differs from its reference model and the ES it provides, even after 30 years of restoration. Furthermore, these findings endorse the idea that ecological restoration should not be viewed as a substitute for conservation, nor should the promise of restoration be used to justify further destruction or unsustainable use, as restoration may not succeed in re-establishing the full assemblage of native species or the full extent of the original ecosystem's structure and function.

Keywords: Quarry ecological restoration; regulating ecosystem services; remote sensing; functional traits; Arrábida Natural Park

Resumo

Os serviços de ecossistema (SE) de regulação são importantes não só para sustentar a vida na Terra e manter a integridade dos ecossistemas, como também para beneficiar o bem-estar humano. No entanto, atividades antrópicas como a exploração de pedreiras colocam estes serviços sob grande pressão, reduzindo a biodiversidade e danificando extensivamente o solo. Deste modo, são necessárias ações de restauro para recuperar as áreas altamente degradadas e os SE que elas fornecem, o que constitui um dos objetivos principais da "Década para a Recuperação dos Ecossistemas" (2021-2030) declarada pela Assembleia Geral das Nações Unidas.

Avaliar o sucesso das ações de restauro ecológico é essencial para confirmar se o restauro segue a direção desejada no sentido de alcançar os objetivos do mesmo, e para determinar a eventual necessidade de estratégias de gestão adaptativa para garantir a eficácia dos esforços de restauro. Neste sentido, a monitorização do ecossistema em recuperação através de medidas locais no campo é necessária. Contudo, a recuperação da funcionalidade e dos serviços do ecossistema ocorre à escala da paisagem. Assim sendo, complementar os dados de campo com dados de deteção remota (DR) poderá permitir extrapolar as mudanças no ecossistema para diversas escalas espaciais (e temporais), visto que a DR regista e avalia repetidamente as mudanças nas características do ecossistema e na saúde da vegetação a diferentes escalas espaciais. Assim, complementar dados de campo com dados de DR poderá permitir a criação de modelos espaciais e mapas informativos para áreas maiores, que poderão apoiar nas decisões de gestão adaptativa.

Esta dissertação teve como objetivo avaliar o sucesso do restauro ecológico de uma pedreira inserida no Parque Natural da Arrábida (PNA), uma área protegida em Portugal com elevado valor ecológico que serviu de ecossistema de referência neste estudo. Para isso, foram combinados dados de campo e DR para avaliar atributos-chave do ecossistema (que estão intimamente ligados a vários SE). Além disso, foram usados dados de campo e DR para analisar a trajetória dos atributos ao longo do tempo (com base numa cronossequência) e avaliar o sucesso do restauro através de análise estatística (tendo em consideração o modo de intervenção em cada local restaurado) e produção de mapas dos atributos. Os atributos-chave considerados foram: 1) Produtividade e Sequestro de Carbono; 2) Fertilidade e Decomposição do Solo; 3) Qualidade do Habitat; 4) Similaridade com a Vegetação Natural; 5) Diversidade (Taxonómica e Funcional) e 6) Funcionamento do Ecossistema (Polinização, Resiliência e Adaptação ao Fogo, Resiliência e Adaptação à Seca, e Capacidade de Dispersão, Interação com a Fauna e Regeneração).

No geral, a recuperação do ecossistema na pedreira SECIL-Outão foi considerada incompleta, uma vez que a recuperação total é apenas alcançada quando todos os principais atributos do ecossistema se assemelham fortemente aos do modelo de referência. Tal não se observa nas áreas restauradas em relação à maioria dos atributos estudados e seus SE associados, pois tiveram valores abaixo daqueles do ecossistema de referência (ex. sequestro de carbono no solo, fertilidade do solo, qualidade do habitat, resiliência ao fogo e seca). Além disso, a hipótese de que as áreas de pedreira restauradas há mais tempo estariam mais próximas do que as recentes de fornecer os mesmos SE de regulação que o ecossistema de referência foi apoiada apenas para alguns dos atributos.

A análise da trajetória dos atributos do ecossistema ao longo do tempo apoiou a hipótese proposta de que a trajetória de recuperação seria diferente para cada atributo, uma vez que diferentes atributos recuperam a velocidades distintas devido às suas dinâmicas naturais inerentes e outros fatores, como o modo de intervenção aplicado. Enquanto algumas variáveis recuperaram rapidamente e continuaram a aumentar com a idade do restauro (ex. densidade de plântulas), mostrando sinais positivos de melhoria,

outras variáveis mostraram uma recuperação mais lenta (ex. carbono orgânico no solo). Alguns atributos do ecossistema mostraram sinais de estabilização 15-20 anos após as ações de restauro terem ocorrido (ex. produtividade, decomposição, semelhança com a referência), indicando limitações no ecossistema, possivelmente devido à alta competição e exaustão de recursos limitados causados pela alta abundância de pinheiro de Alepo. Além disso, a diminuição observada ao longo da cronossequência nas variáveis de funcionamento do ecossistema, como a proporção de resprouters (resilientes ao fogo) ou vegetação esclerófila (resiliente à seca), pode ter implicações negativas na resiliência do ecossistema restaurado. Esta tendência é particularmente preocupante num cenário de alterações climáticas, onde se preveem secas mais prolongadas e severas e maior frequência e intensidade de incêndios, o que poderá colocar os locais restaurados há mais tempo num estado vulnerável e comprometer os esforços de restauro, reforçando a necessidade de ações de gestão adaptativa.

De um modo geral, não foi possível determinar qual o modo de intervenção (plantações, hidrossementeira ou misto) mais adequado para um restauro bem-sucedido, pois cada um apresentou vantagens e desvantagens quando comparado ao ecossistema de referência, havendo também *trade-offs* entre os atributos estudados (ex. prestação de serviço de polinização e similaridade do ecossistema restaurado com o ecossistema de referência). Os resultados dos diferentes modos de intervenção divergem do ecossistema de referência devido à dominância do pinheiro de Alepo nas plantações dos patamares de calcário e da dominância do estrato herbáceo nos taludes de calcário e de marga, onde foram realizadas hidrossementeiras. Deste modo, as áreas restauradas apresentaram baixos níveis de recuperação em relação à cobertura arbustiva, dominante no ecossistema de referência. Esses fatores contribuíram para diferenciar as áreas restauradas do ecossistema de referência em termos de composição de espécies, com baixos índices de similaridade, e da qualidade do habitat.

Outros atributos associados aos SE, como o sequestro de carbono do solo, foram significativamente maiores nos solos das áreas naturais não perturbadas. Os solos das áreas naturais armazenaram cerca de três vezes e até 10 vezes mais carbono do que os solos das áreas restauradas, revelando limitações das áreas restauradas em contribuir para o sequestro de carbono na mesma medida que as áreas naturais envolventes à pedreira. Além da baixa fertilidade do solo, é possível que as agulhas de pinheiro recalcitrantes estejam a contribuir para as baixas taxas de decomposição, limitando a produtividade primária e a taxa de crescimento nos locais restaurados mais antigos. Da mesma forma, nas plantações dos patamares de calcário verificou-se que a diversidade funcional, a vegetação esclerófila e os resprouters na comunidade lenhosa estão abaixo dos níveis do ecossistema de referência. Por outro lado, locais com hidrossementeira tiveram menor densidade de plântulas do que as plantações nos patamares, possivelmente devido à falta de dispersores e recursos alimentares como frutos carnudos de arbustos nativos, o que pode diminuir as interações bióticas e constituir um risco para a autossustentabilidade e regeneração das áreas restauradas mais recentemente.

Adicionalmente, a modelação de 28 variáveis permitiu identificar fatores promotores ou limitantes da recuperação. A cobertura de pinheiros contribuiu para menores taxas de decomposição no solo e diminuiu a diversidade funcional nas áreas restauradas, o que pode ter implicações negativas para o funcionamento do ecossistema e a qualidade dos SE fornecidos pelas áreas restauradas. A cobertura de pinheiro também diminuiu significativamente a resiliência à seca (vegetação esclerófila) e ao fogo (resprouters), além de influenciar negativamente o serviço de polinização (duração da floração e entomofilia). Uma vez que a alta cobertura de pinheiros se tornou num obstáculo à progressão da comunidade vegetal na direção do sistema de referência, intervenções de desbaste de pinheiros são altamente recomendadas. Espera-se que tais intervenções promovam características-chave funcionais e, a médio prazo, contribuam para melhorar o ciclo de nutrientes no solo, a resiliência do ecossistema e fluxos bióticos. Por outro lado, a densidade de arbustos nos locais restaurados provou ser muito importante para aumentar a diversidade funcional e taxonómica, a densidade de plântulas e a semelhança de espécies com o ecossistema de referência. Deste modo, intervenções que promovam uma maior densidade de arbustos nativos são altamente recomendadas para melhorar a recuperação, funcionamento e autossustentabilidade das áreas restauradas.

Por fim, vários mapas foram produzidos com sucesso através de modelos espaciais baseados em DR e dados de campo. As suas estimativas concordaram com os padrões observados na análise estatística e com os índices de sucesso da recuperação calculados (considerando os modos de intervenção e as zonas-tipo). Assim, os mapas produzidos apoiaram a hipótese de que os dados de DR podem ser validados por dados de campo e permitir a extrapolação (*upscaling*) da informação, fornecendo modelos úteis que podem ser usados para produzir mapas à escala da paisagem. Desta forma, foi possível produzir mapas que fornecem a distribuição espacial de indicadores ecológicos relevantes, o que contribuiu para a gestão do restauro ao permitir identificar quais as áreas mais necessárias de futuras ações de gestão adaptativa.

Este estudo mostra que o ecossistema restaurado difere de seu modelo de referência e dos SE que este fornece, mesmo após 30 anos de restauro. Assim sendo, o restauro ecológico não deve ser visto como um substituto para a conservação, nem a promessa de restauro deve ser utilizada para justificar mais destruição ou uso insustentável, pois o restauro pode não conseguir restabelecer o conjunto total de espécies nativas ou a extensão total da estrutura e função do ecossistema original, conforme concluído por este estudo.

Palavras-chave: Restauro ecológico de pedreiras; serviços de ecossistema de regulação; deteção remota; atributos funcionais; Parque Natural da Arrábida

List of Publications

Oral communications:

Mendes C., Serafim J., Oliveira, A., Nunes, A. & Branquinho, C. (2021, December 1-4). *Evaluation of the degree of recovery of restored areas in a quarry within the protected area of the Arrábida Natural Park*. 20th National Ecology Meeting - Knowledge at the service of classified areas (SPECO), Ponte de Lima, Portugal.

Mendes C., Serafim J., Oliveira, A., Nunes, A. & Branquinho, C. (2021, September 7-10). *Evaluating quarry ecological restoration success with field and remote sensing approaches (Arrábida Natural Park, Portugal)*. 12th European Conference on Ecological Restoration (SERE2021), Online event.

Mendes C., Serafim J., Oliveira, A., Nunes, A. & Branquinho, C. (2020, December 10-11). *Evaluation of the ecological restoration success based on field data and remote sensing: a case study in the Arrábida Natural Park)*. 19th National Ecology Meeting (SPECO), Online event.

Scientific papers (in preparation):

Mendes *et al.* Evaluating ecological restoration success based on field and remote sensing approaches: a case study in Arrábida Natural Park, Portugal (*In prep*)

This provisional title corresponds to a publication in preparation to be submitted to the international peer-reviewed journal "Restoration Ecology".

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List of Abbreviations

ANP	Arrábida Natural Park
CBD	Convention on Biological Diversity
CICES	Common International Classification of Ecosystem Services
CO_2	Carbon dioxide
CW	Crown width
CWM	Community Weighted Mean
DBH	Diameter at breast height
EBV	Essential Biodiversity Variables
EEA	European Environment Agency
ES	Ecosystem services
FDis	Functional dispersion
FRic	Functional richness
GHGs	Greenhouse gases
IPBES	Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services
LOI	Loss on Ignition
LULC	Land use/Land cover
n.s.	statistically non-significant
N	Nitrogen in soil
NDVI	Normalized Difference Vegetation Index
NV	Natural vegetation (existent in the ANP –reference ecosystem)
P	Phosphorous in soil
PSR	Potential solar radiation
RS	Remote sensing
SER	Society for Ecological Restoration
SLA	Specific leaf area
SNAP	Sentinel Application Platform
SOC	Soil organic carbon
SOM	Soil organic matter
TEEB	The Economics of Ecosystems and Biodiversity
UN	United Nations

CHAPTER 1 | INTRODUCTION

1.1. The Biodiversity Crisis

Biodiversity - the variety of all life on Earth - plays a fundamental role in our lives. Approximately 40% of the global economy and 80% of human needs are derived from biological resources, making the efficient use of biodiversity a necessary condition for sustainable development (CBD, 1994). However, global biodiversity is currently undergoing a major crisis, resulting in the degradation of ecosystems, their functions in the living system, and thus their services to humanity (Hooper et al., 2012).

Ecosystems, defined as a dynamic complex of plant, animal and microorganism communities and their non-living environment interacting as a functional unit, provide a wide range of services that vary in quantity and quality depending on the ecosystem’s type and status (CBD, 1992). Ecosystem services (ES) are essential for sustaining life on Earth and maintaining the integrity of an ecosystem while also benefiting human well-being (Millennium Ecosystem Assessment, 2005). They include provisioning services such as food and water, regulating services like carbon sequestration and pollination, and cultural services that include spiritual, recreational and educational values (Haines-Young & Potschin, 2012). Such services are generated by ecosystem functions supported by biophysical structures and processes. When a good or service provides benefits (e.g. health, air quality, pleasure), it can be valued in economic and monetary terms (De Groot et al., 2010). Nevertheless, ES are currently under pressure, not only due to anthropogenic activities but also due to climate change (Mooney et al., 2009).

Among the human activities that negatively impact ES are land use and land cover (LULC) change and intensity, typically driven by unsustainable agricultural activities, built-up areas, mining and quarrying. These not only contribute to climate change by affecting carbon storage but also weaken ecosystems’ resilience via changes in biodiversity and landscape heterogeneity (De Groot et al., 2010). LULC change primarily acts as a slow variable disturbing planetary processes which provide the underlying resilience of the Earth System, like the functioning of sinks and sources of carbon and the regulation of water, nutrient, and mineral fluxes (Rockström et al., 2009). These dynamics underpin the resilience that enables planet Earth to stay within a state conducive to human development.

Consequently, LULC change is likely to be the driver of biodiversity change that will have the most significant effect on terrestrial ecosystems in the current century (Sala et al., 2000). Furthermore, the constant acceleration of urbanization and industrialization worldwide, along with the accompanying demand for mineral resources, has led to the development of new quarries that have largely damaged many natural ecosystems (Zhang et al., 2018). The large scale of this loss has led to significant changes in vegetation and forests worldwide (Sala et al., 2000).

For instance, quarrying activities decimate flora and fauna by the removal of vegetation cover and habitat destruction thereby reducing biodiversity and disrupting fundamental ecological relationships (Lameed & Ayodele, 2010; Meira-Neto et al., 2011). Additionally, drainage and the physical-chemical erosion of the substrate cause a scarcity in the nutritional and hydric status, which hinders the natural germination and establishment of young plants (Juan Puigdefábregas & Mendizabal, 1998). Quarrying activities also extensively damage the soil by modifying the original site topography and depleting soil microbial communities (Zhang et al., 2018). Besides this, in Mediterranean areas in particular, usual environmental constraints imposed on vegetation by the Mediterranean climate, such as summer drought and interannual rainfall variability, are further enhanced at the quarry by the lack of soil and the extent of barren surfaces, increasing local thermal amplitudes and erosion (Nunes et al., 2014).

Because quarrying operations are usually carried out on a large scale, and the quarry sites are often abandoned after extraction when resources have become depleted, quarrying results in significant ecological and extremely visual impacts (Vitousek et al., 1997). In the past, the recovery of abandoned quarries was left to slow natural processes. However, the natural colonisation of limestone quarries is very slow due to unfavourable conditions, taking decades or even hundreds of years to reach new woodland communities with satisfactory cover (similar to a pre-degraded ecosystem state) (Correia et al., 2001; Meira-Neto et al., 2011). The factors responsible for the slow recovery in limestone quarries are (i) low input or retention of propagules; (ii) climatic and edaphic limitations on plant establishment and survival; and (iii) biotic checks, including disturbance, grazing, invertebrate attack and competition (Correia et al., 2001). Because the time scales involved in creating new communities are not considered acceptable for reclamation or restoration, active quarry restoration instead of passive restoration is necessary to guarantee a quicker recovery of ES, at least in an initial phase (Moreno-Mateos et al., 2020).

Fortunately, the awareness concerning the biodiversity crisis is increasing at the centre of international policy-making, under multilateral agreements such as the Convention on Biological Diversity (CBD), which sought to achieve the Aichi Biodiversity Targets, namely “enhancing the benefits to all from biodiversity and ES” (O’Connor et al., 2015). Furthermore, restoring biodiversity and ES can provide part of the solution to many of the United Nations (UN) Sustainable Development Goals and others, such as the Bonn Challenge (Alleaume et al., 2018). Additionally, the UN General Assembly declared 2021 – 2030 as the UN Decade on Ecosystem Restoration, which intends to massively scale up the restoration of degraded and destroyed ecosystems (Gann et al., 2019). In this way, the restoration of abandoned quarry sites must be an objective to achieve.

1.2. Ecological Restoration

1.2.1. Ecological restoration theory and main goals

A restorative activity is one that directly or indirectly supports or achieves the recovery of ecosystem attributes that have been lost or degraded (Gann et al., 2019). In particular, ecological restoration is “the process of assisting the recovery of an ecosystem that has been degraded, damaged, or destroyed”, and aims to restore it to its original or to a reference state, with similar species composition and functional characteristics (SER, 2004). It is distinct from other forms of environmental repair seeking to assist recovery of native ecosystems and ecosystem integrity, such as remediation (i.e., of polluted and contaminated sites), rehabilitation (which only aims to make ecosystems healthy and useful after the disturbance) or mitigation (compensates for environmental damage). Some projects may cover more than one category, particularly those carried out within larger frameworks, such as nature-based solutions (i.e., green infrastructure). Thus, ecological restoration is one of a family of restorative activities that can be conceived of as a continuum of allied activities in repairing the world’s ecosystems (Gann et al., 2019). As such, ecological restoration complements other conservation activities and nature-based solutions and vice versa.

The practice of ecological restoration requires a high degree of ecological knowledge which can be drawn from many different sources (e.g. practitioner experience, traditional ecological knowledge, local ecological knowledge, and scientific research) resulting from observation, experimentation, and trial and error. Hence, the best available knowledge should inform the design and implementation of ecological restoration, as well as monitor restoration results over time to indicate the need to modify management approaches and, in this way, contribute to adaptive management (Gann et al., 2019).

Furthermore, the concept of a reference system is a bedrock principle in ecological restoration, serving as a template to guide restoration and identify when recovery is complete. Most importantly, it provides ecological information about the system of interest, suggesting where that system would currently be had it not been disrupted, and enlightens important dynamics that might be lacking. In this way, a reference ecosystem "usually represents a nondegraded version of the ecosystem complete with its flora, fauna, abiotic elements, functions, processes and successional states that might have existed on the restoration site had degradation not occurred" (Gann et al., 2019). Nevertheless, ecological restoration has never been limited to the literal use of historical conditions as a target for restoration (i.e., "returning to the past"), and maintaining or restoring resilience or adaptive capacity may be as or more important than literal historical authenticity (Palmer et al., 2016).

Thus, restoration activities focus on reinstating components and conditions suitable for natural processes to recommence and support the recovery of ecosystem attributes, including capacity for self-organization and ecosystem resilience to future stresses (Gann et al., 2019). The resilience of an ecosystem, that is, its ability to structurally and functionally recover from damage resulting from stress or disturbances, largely arises from the resilience of the different plant species existing on the site. More specifically, it is the diversity of functional response mechanisms to environmental variation among species in an ecosystem that maintains resilience to disturbances (Rockström et al., 2009). Additionally, impacts on ecosystems originated or aggravated by natural causes (e.g. fires) may compromise the restoration of their original properties (SER, 2004). In areas prone to fires, for example, the conservation of water and soil and the promotion of species resilient to this disturbance must be considered to allow the revegetated ecosystem to recover (Meira-Neto et al., 2011).

The Society for Ecological Restoration (SER) has formulated nine attributes for determining when the restoration has been accomplished (SER, 2004). Although the full expression of the attributes is not essential to demonstrate restoration, these should demonstrate an appropriate trajectory of ecosystem development towards the intended goals or reference. The restored ecosystem should:

1. Present a species composition similar to the reference ecosystem.
2. Have the greatest possible number of native species.
3. Contain the functional groups necessary for the continued development and/or stability of the restored ecosystem or, colonization by individuals of the missing functional groups from adjacent natural environments is possible.
4. Support reproductive populations of the species necessary for its stability and development along the desired trajectory.
5. Function normally for its ecological state of development, and signs of dysfunction are absent.
6. Be integrated into a larger ecological matrix or landscape, with which it interacts through abiotic and biotic flows and exchanges.
7. Not present potential threats to the integrity of the ecosystem, such as invasive species, which, if present, occur in small numbers.
8. Be sufficiently resilient to withstand normal periodic stress and/or disturbance events.
9. Be self-sustainable to the same degree as its reference ecosystem and have the potential to persist indefinitely under existing environmental conditions.

In practice, most restoration studies assess ecosystem condition using indicators that can be grouped into three ecosystem attributes: vegetation structure, species diversity and abundance, and ecological processes (Ruiz-Jaen & Aide, 2005). Vegetation structure includes measures of plant growth such as height, cover and biomass of the strata (tree, shrub, herb). Diversity and abundance include flora and fauna species, as well as functional diversity. Ecosystem processes often include measures of primary productivity, reproductive success or dispersal, nutrient cycling, soil development, pollination, and other biological interactions (Wortley et al., 2013). The measurement of these attributes and their comparison between the restored and reference sites can reflect the recovery trajectory and the self-sustainability of the restored ecosystem, in addition to qualifying restoration success.

More recently, SER has defined that the assessment of progress towards the ecological target should include indicators for each of six key ecosystem attributes of the reference ecosystem: absence of threats, physical conditions, species composition, structural diversity, ecosystem functional and external exchanges. Full recovery can then be identified as the state or condition where, following restoration, all key ecosystem attributes closely resemble those of the reference model (Figure 1.1) (Gann et al., 2019).

a)

Attribute	*Description*
Absence of threats	Direct threats to the ecosystem such as overutilisation, contamination, or invasive species are absent
Physical conditions	Environmental conditions (including the physical and chemical conditions of soil and water, and topography) required to sustain the target ecosystem are present
Species composition	Native species characteristic of the appropriate reference ecosystem are present, whereas undesirable species are absent
Structural diversity	Appropriate diversity of key structural components, including demographic stages, trophic levels, vegetation strata and spatial habitat diversity are present
Ecosystem function	Appropriate levels of growth and productivity, nutrient cycling, decomposition, species interactions, and rates of disturbance
External exchanges	The ecosystem is appropriately integrated into its larger landscape or aquatic context through abiotic and biotic flows and exchanges

b)

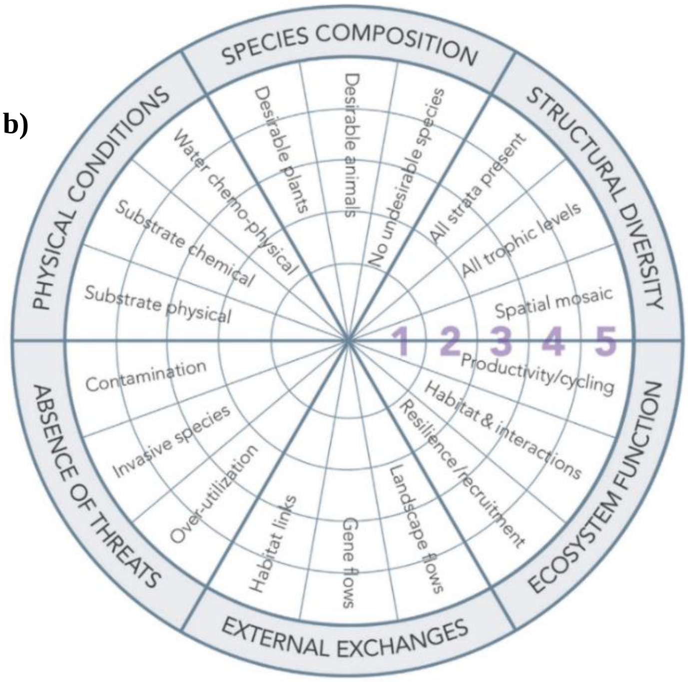


Figure 1.1. a) Description of the key ecosystem attributes used to characterize the restored and reference ecosystem. b) Ecological Recovery Wheel used to monitor the degree of recovery at a restoration site. From (Gann et al., 2019).

1.2.2. Relevance of ES assessment in ecological restoration

Recovery, whether full or partial, is a slow process and does not produce immediate results, as different ecosystem components recover at distinct speeds. While some variables are slow to recover (e.g. soil organic matter or nitrogen availability), fast variables (e.g. annual plant productivity) respond quickly to restoration efforts (Chapin et al., 2002; Walker et al., 2012). Thus, components with diverse turnover times establish the tempo of ecosystem dynamics (Carpenter & Turner, 2000). Furthermore, recovery gains from scientific evaluation and medium/long-term monitoring, considering the application of adaptive management and continuous learning. Ideally, to evaluate an ecological restoration, it would be necessary to study all the attributes listed by SER. However, it may be time-consuming and exceed most projects' financial resources. For this reason, in many cases, it can be helpful to study ecological restoration using a framework covering crucial aspects of an ecosystem's functioning and health, such as the ES concept.

As mentioned briefly in *section 1.1*, ES are ecosystems' direct and indirect contributions to human wellbeing. The CBD, the Millennium Development Goals (MDGs), and the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services (IPBES), explicitly link the conservation of biodiversity with the provision of ES to support the sustainable development of social-ecological systems, protecting the functioning and resilience of ecosystems, which are essential for human health and wellbeing (Bullock et al., 2011; Sachs et al., 2009). ES are derived from ecosystem functions supported by basic ecosystem processes maintained by biodiversity, which plays a vital role in the structural setup of ecosystems, as depicted in Figure 1.2.

Healthy native ecosystems assure the flow of ES, defined as the transmission of services from ecosystems to people (Bagstad et al., 2013). In other words, ES flow can be conceptualized as the movements of materials, energy, value, or information between systems, transferred due to actions taken by agents (Liu et al., 2013). However, ES are not isolated independent units but rather depend on each other and interact, causing trade-offs and synergies. Some of these interactions can be determined by stakeholders' use and management of ES. Thus, the flow of ES is shaped through the social system (i.e., stakeholders' interactions, roles, and preferences) by several types of complex interactions among multiple stakeholders (Felipe-Lucia et al., 2015).

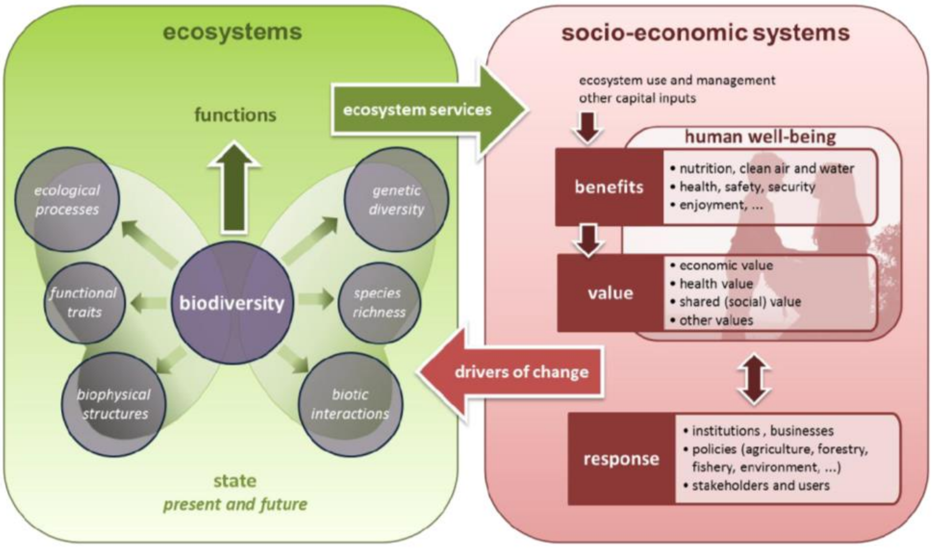


Figure 1.2. Conceptual framework for European Union wide ecosystem assessments. From Maes et al. (2013).

The classification of ES is a conceptually and technically challenging task because there is no single, fully accurate and accepted definition of the term capable of capturing the full range of ways in which ecosystems support human life and wellbeing, and because there is a wide range of purposes or applications with different requirements in terms of spatial and thematic resolution levels (De Groot et al., 2010). Nonetheless, several classification systems of ES currently coexist, and the main internationally adopted ones are the Millennium Ecosystem Assessment (MEA), The Economics of Ecosystems and Biodiversity (TEEB) and Common International Classification of ES (CICES).

In 2005, MEA adopted a classification system based on four groups: provisioning, regulation, cultural and supporting services (Table 1.1). This ES classification was considered quite operational, accessible and easily understood by decision-makers. However, it presents some weaknesses regarding ES categories since it does not distinguish between intermediate ecosystem processes and services that are directly used or consumed by people, which may lead to overlapping estimates of supporting services (Fisher & Turner, 2008).

Table 1.1. Ecosystem services in the MEA classification system (Millennium Ecosystem Assessment, 2005).

PROVISIONING SERVICES	REGULATING SERVICES	CULTURAL SERVICES
The "products" obtained from ecosystems	*Benefits obtained from the regulation of ecosystem processes*	*Nonmaterial benefits obtained from ecosystems*
Foods	Carbon sequestration	Education
Fibers	Flood prevention	Recreational
Ornamentals	Erosion control	Sense of place
Medicines	Air and water purification	Spiritual
Biofuels	Pest and disease control	Cognitive development
Fresh water	Pollination	Stress relief
Genetic resources	Seed dispersal	Gardening
SUPPORTING SERVICES		
Services necessary for the production of all other ecosystem services		
Nutrient and water cycling Primary productivity Soil formation		

These weaknesses have led to the emergence of new classifications such as the one proposed by TEEB, which had the main objective to recognize the value of ecosystems and biodiversity (regardless of their market value), demonstrate this value in economic terms, and ultimately help to capture this value in decision-making processes (TEEB, 2010). In addition, TEEB created the designation of "habitat services" (e.g. habitat for species and maintenance of genetic diversity), which is not included in any of the original categories of ES proposed by MEA, and eliminated supporting services (considered intermediate services), as it focuses on services that have economic value, that is, in the final services (Haines-Young & Potschin, 2012).

Then, in 2013, based on the environmental accounting work of the European Environment Agency (EEA), the first complete version of CICES emerged to facilitate the comparison of assessments with different systems, contributing to the standardization of classification and making ES accounting clearer (Haines-Young & Potschin, 2018). This classification organizes the different types of ES into three major sections: Provisioning, Regulating & Maintenance and Cultural Services. CICES considers habitat services part of a broader 'Regulating & Maintenance' section because they participate in everything that underlies ecosystems (structures, processes and functions), and are indirectly consumed or used, contributing to many final goods and benefits (Haines-Young et al., 2012). In this way, CICES regards habitat services as part of the "Regulation of the biotic environment" in the following categories: lifecycle maintenance (pollination and seed dispersal), habitat and gene pool protection (maintaining nursery populations and habitats) (Table 1.2).

Table 1.2. CICES basic structure and relationship of classes to TEEB classification, which aims to classify ecosystem services. Adapted from Haines-Young et al. (2012).

CICES Section	**CICES Division**	**TEEB Categories**			
Regulating & Maintenance	**Regulation of bio-physical environment**	Air purification	Waste treatment (esp. water purification)		
	Flow regulation	Disturbance prevention or moderation	Regulation of water flows	Erosion prevention	
	Regulation of physicochemical environment	Climate Regulation (incl. Carbon sequestration)	Maintaining soil fertility, decomposition		
	Regulation of biotic environment	Gene pool protection	Lifecycle maintenance and habitat quality	Pollination	Biological control

The CICES classification was used in this research as it is the most operational-oriented ES classification and allows for a high level of detail. More specifically, Regulating & Maintenance ES in a broader sense were studied, as it was considered the most relevant section to focus on in quarry ecological restoration monitoring, and particularly in the context of this case study, located in a Natural Park with high conservation value.

Since quarrying causes the degradation of ecosystems, it is necessary to restore them and recover ES supply. Additionally, healthy ecosystems can reduce the impacts of climate change, given that vegetation functions as a climate regulator by capturing carbon dioxide (CO_2) from the atmosphere. Climate-induced events such as increasing floods, droughts, and pest outbreaks can be mitigated by ES, such as water and erosion regulation, natural hazard mitigation, and pest control (Pereira et al., 2010).

1.2.2.1. Carbon storage and sequestration

One of the most widely recognized Regulating & Maintenance ES is terrestrial-based carbon storage and sequestration (Sharp et al., 2014). Carbon storage measures the responsiveness of productive capacity and ecological resilience to changes in terrestrial ecosystems. It is especially important to consider in cases where land-use change under human activities significantly impacts the dynamic balance of carbon storage due to built-up land growth and vegetation degradation, which in turn may threaten the delivery of multiple ES (Liang et al., 2017). Ecosystems regulate Earth's climate by adding and removing greenhouse gases (GHGs) like CO_2. Forests, grasslands, peat swamps, and other terrestrial ecosystems collectively store far more carbon than the atmosphere alone (Lal, 2004). Thus, ecosystems keep out the atmospheric CO_2 that would further contribute to climate change. Many systems do more than merely store carbon; they also continue to accumulate it in plants and soil over time, "sequestering" more each year. Large volumes of CO_2 can be released when these systems are disturbed by fire, illness, or vegetation change (e.g. LULC conversion). However, changes in management, such as forest restoration or alternative farming practices, might result in huge volumes of CO_2 being stored (Sharp et al., 2014).

Therefore, information about how much and where carbon is stored (or sequestered over time), and how LULC changes affect these amounts is needed to manage landscapes for carbon storage and sequestration. Carbon storage and sequestration maps are ideal for assisting decisions influencing these ES since land managers must select between locations for conservation, harvest, or development. Governments, NGOs, and companies can use such maps to support a variety of decisions, knowing which areas of a landscape store the most carbon and which ones have sequestered or lost it over time (Sharp et al., 2014). Increasing the carbon sequestration rate through boosting plant biomass production (including biomass in soils) is urgently needed to mitigate the effects of climate change.

Restoration goals may specifically refer to the reinstatement of particular ES or ameliorating the quality and flow of one or more services (Gann et al., 2019). Restoration projects can effectively enhance both biodiversity and ES; however, conflicts can arise, especially if single services are targeted in isolation. Criticism regarding restoration actions focusing solely on increasing forest carbon stocks (often to earn REDD credits for carbon payments) has been made, as there is a possibility that other services or biodiversity may be neglected, which could lead to negative impacts on the provision of other ES and adversely affect social issues (Bullock et al., 2011; Putz & Redford, 2009). Evidence also suggests that there is a trade-off between protecting biodiversity and reducing carbon emissions; thus, this needs to be carefully considered in restoration projects to ensure both objectives are met (Venter et al., 2009).

Projects that focus on a limited number of factors, such as single ES, may have limited potential to restore overall ecosystem complexity. On the other hand, projects that incorporate numerous aspects into their reference models and goals may have a better chance of restoring ecosystems that conserve biodiversity, deliver ecological resilience, and provide higher rates of ES over the long term (Gann et al., 2019).

Ecological processes function at landscape and regional scales (e.g. gene flow, colonization, predation, ecological disturbances). Large-scale degradation can overwhelm smaller-scale restoration efforts (i.e., projects may not be able to accommodate species with high minimum habitat needs or that demand greater trophic complexity). Thus, operating at the landscape scale is essential to achieve vital environmental and ecological benefits. This justifies why some ecological restoration projects must be carried out on a large scale (e.g. hundreds or thousands of hectares). Additionally, as part of integrated landscape planning initiatives, planning and prioritizing site-level actions are required (Gann et al., 2019). Furthermore, large-scale monitoring of biodiversity and ES is essential in ecological restoration, as their recovery can be slow and incomplete (Bullock et al., 2011).

1.2.3. Relevance of functional diversity, ecosystem functioning and ecosystem resilience in ecological restoration

Biodiversity includes not only species diversity but also functional diversity. Changes in components of biodiversity, such as functional diversity, influence ecosystem functioning and are, therefore, likely to influence ES. In fact, studies have shown that as biodiversity is reduced, so is the quality of the ES provided by ecosystems (Balvanera et al., 2014; Garland et al., 2021; Wilson, 1999). Functional diversity, defined as the value, range, and relative abundance of the functional traits of biological communities in a given ecosystem, has proved to be a more universal and reliable indicator of ecosystem vulnerability than only species diversity, which does not reflect the uneven role played by species in the maintenance of ecosystem processes (Díaz et al., 2007; Nunes et al., 2014).

Rather than accounting for species, functional diversity measures the diversity of ecological roles required for the ecosystem to function (Science for Environment Policy, 2015). The presence of a few species that perform diversified functions may be more important than the presence of many species with "redundant" functions, especially if species responsible for essential functions are absent. However, the redundancy of functions may be advantageous, as it gives the system greater resilience in the face of disturbances that lead to the disappearance of one or another species (i.e., the function of the disappeared species continues to be performed by another that still remains in the system). Still, the reduction in the number of species in many functional groups may indicate a loss of biotic integrity (Pellant, 2005). The FDis, which is a multi-functional diversity index, reflects the degree of functional dissimilarity within a community (i.e., functional attributes), simultaneously considering all the attributes selected for the study in question. Theories predict that multi-trait dissimilarity could allow species coexistence by decreasing competition for similar resources and improving ecosystem multifunctionality. In this way, multi-functional diversity indices reflect complementary resource use within a community (de Bello et al., 2021; Laliberté & Legendre, 2010; Villéger et al., 2008).

Furthermore, the multiple threats posed by climate and land-use change, such as more frequent droughts, megafires, and loss of biodiversity have put a clear priority on the importance of ecosystem stability and resilience (Bellard et al., 2012). Ecosystem stability is defined as the ability of an ecosystem to maintain its properties in relation to reference conditions through time. Stability's key elements include inter-annual constancy in ecosystem properties, but also resistance and recovery from environmental change and perturbation (de Bello et al., 2021). Stability is maintained by populations, communities, and ecosystems that can buffer the effects of environmental variation, while retaining ecosystem functions (e.g. productivity, pollination or seed dispersal).

Ecosystem multifunctionality is defined as the overall functioning of an ecosystem or the provision of multiple ecosystem functions, which contribute to ES either directly or indirectly (Garland et al., 2021; Manning et al., 2018). Understanding that ecosystem functioning depending on species' functional traits, rather than only on species diversity per se, is becoming a dominant paradigm (de Bello et al., 2021). Approaches based on functional attributes of species, together with measures of functional diversity, make it possible to link species composition to ecosystem processes and make predictions about their response to disturbance (Lavorel & Garnier, 2002; Suding & Goldstein, 2008). This two-fold approach is central to the assessment of the ecological resilience of ecosystems (de Bello et al., 2021). In this sense, the maximization of functional redundancy - measured as the number of species that contribute in a similar way to an ecosystem function - helps to increase the ecological resilience of ecosystems when species loss occurs because of disturbances (Gann et al., 2019; Loreau, 2004; Suding, 2011). Ultimately, the temporal and spatial components of disturbance are important to determine resilience, because the sensitivity of species, the functions they perform, and their responses to the environment depend on scale (Elmqvist et al., 2003; Nash et al., 2014; Timpane-Padgham et al., 2017). For example, an ecosystem with high functional redundancy and broad connectivity at multiple scales is likely to be much more resilient to moderate disturbance and habitat fragmentation than an ecosystem with low functional redundancy and limited connectivity at different scales.

1.2.4. Complementing and upscaling field data with RS for ecological restoration monitoring

Ecosystem monitoring is essential to assess the outcomes of restoration actions. However, the efficacy and effects of most restoration practices remain largely unknown, limiting the ability to determine the overall impact of current investments in restoration (Gatica-Saavedra et al., 2017). Absent or inappropriate monitoring may prevent understanding restoration trajectories, precluding adaptive management strategies, often crucial to create functional ecosystems able to provide ES (Nunes et al., 2016). Therefore, there is a critical need to assess the outcomes of restoration efforts in the short and long term and establish standards for evaluating restoration success.

Additionally, there is a disconnection between two scale-dependent views on Earth’s biota, with one view based on *in-situ* spatially discontinuous field sampling of species composition and the other view based on spatially continuous remote sensing (RS) observations (Asner et al., 2017). While *in-situ* approaches are spatially and temporally limited when it comes to an understanding of ecological processes, species responses to stress, and estimating services, RS repeatedly records and assesses the status and changes of ecosystem functions, disruptions in ecosystem processes, vegetation stresses and spatial-temporal shifts in plant phenology, over the short to long term and in local and global vegetation monitoring (Lausch et al., 2018). Furthermore, RS observations are particularly relevant to automatically quantify habitat loss, degradation, and fragmentation whilst also helping to optimize field data collection by focusing on relevant sites, dramatically improving cost-effectiveness ratios (Alleaume et al., 2018).

Since 2015, Sentinel 2's twin satellites (S-2A and S-2B) have been acquiring reduced-spectrum multispectral data on the Earth's surface, at different wavelengths and with different spatial resolutions, reaching 10m resolution for the red, green, blue and near-infrared, currently acquiring data with a frequency of 2 to 5 days (Houborg & Skidmore, 2015). In addition to these spectral bands, Sentinel 2 also collects reflectance in other bands of the visible and shortwave infrared spectrum (<3000nm) with variable spatial resolutions of 10m, 20m or 60m (E.S.A., 2015). The information acquired is made available free of charge, allowing the monitoring of the state and function of vegetation at a global level with high frequency and spatial resolution (Houborg & Skidmore, 2015).

Light reflectance in vegetation varies with each species' chemical and morphological characteristics, such as tissue water content, nutrient concentration, leaf surface or amount of chlorophyll (Ollinger, 2011; Xue & Su, 2017). The combination of reflectance in different electromagnetic spectrum bands, with an emphasis on near-infrared and red, is used to determine vegetation indices (Xue & Su, 2017). Such indices capture specific plant reflectance signals in the electromagnetic spectrum determined by leaf chemistry, chlorophyll, and water content, as well as canopy and plant-level structure, including leaf and trunk density and proportion (Ollinger, 2011). Thus, indices can reflect the structure and composition of vegetation, landscape elements, and various relevant biophysical processes such as photosynthesis. For example, the NDVI (Normalized Difference Vegetation Index) has become one of the most used vegetative indices, offering strong empirical correlations with vegetation biomass and productivity on a global scale. On a local scale, its use is highly dependent on local factors (e.g. topography, geology, existing plants) (Houborg & Skidmore, 2015).

In addition, monitoring of restoration and management actions has primarily focused on field-sampled structural and compositional features of ecosystems, despite growing evidence that their influence on ecosystem processes depends on species' functional traits (e.g. growth form, specific leaf area or post-fire regeneration strategy) (Pettorelli et al., 2018). Thus, a functional trait approach is more informative and advisable since traits are the key mechanism through which diversity affects ecosystem functioning and generates services, influencing short-term resource dynamics and long-term ecosystem stability and resilience (Cadotte et al., 2011; de Bello et al., 2010).

The lack of perception of adequate spatial and field data to map species functional traits and quantify ES is now easier to perceive with the new concept of Essential Biodiversity Variables (EBVs), which are a set of stipulated critical variables required to study, report and manage biodiversity change (GEO BON, 2015; Pereira et al., 2013). Because changes in biodiversity occur at a variety of spatial and temporal scales, RS (airborne and satellite) can effectively provide many biodiversity-relevant measures of change in ecosystem structure and function, community composition, species traits, and species populations (Figure 1.3). Current and emerging next-generation satellite RS data acquisition and processing is an ideal tool for continuously detecting changes in biodiversity from local to global levels, bridging data gaps in the spatial and temporal coverage of *in situ* observations. However, there has been little research into how ecologists (who study the efficacy of using EBVs for biodiversity monitoring) and RS specialists (who address technologies deriving RS products related to EBVs) might collaborate (Skidmore et al., 2021).

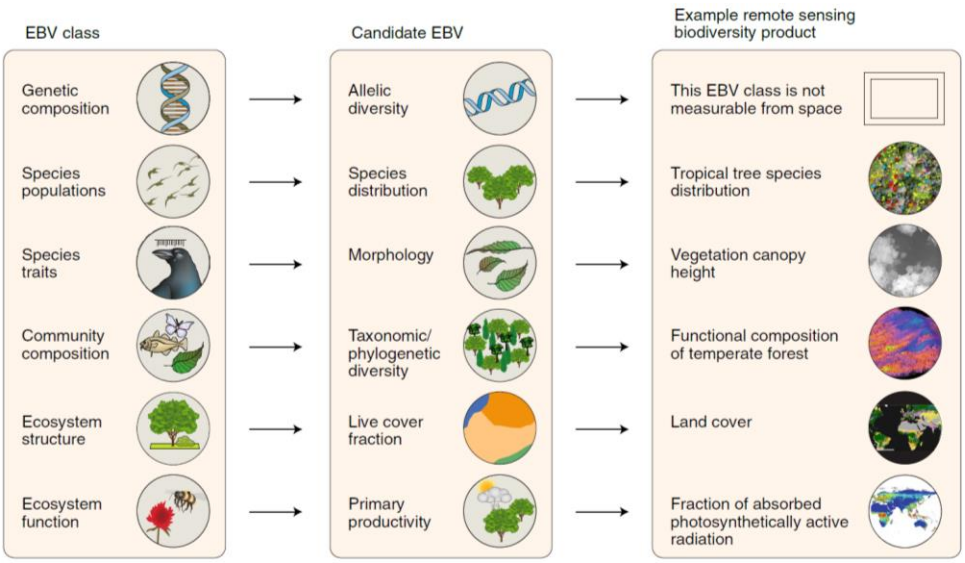


Figure 1.3. Remote sensing products can be used to estimate Essential Biodiversity Variables. Adapted from Skidmore et al. (2021).

Since ecosystem recovery occurs at the landscape scale, complementing field data with RS data for modelling purposes of key variables relevant to ecological restoration can allow the detection of changes in the ecosystem at various scales, improving the monitoring of vegetation and ecological indicators. Because no single approach is sufficient to monitor the complexity and multidimensionality of ecosystems over the short to long term and on local to global scales, *in-situ* and RS data should be combined in a functional trait approach to inform conservation planning and support the assessment of biodiversity, ES, ecosystem functioning, resilience and ecological restoration success (Lausch et al., 2018).

1.3. Restoration project at the SECIL-Outão quarry

1.3.1. Quarry history in the Arrábida Natural Park and the importance of quarry ecological restoration

Regions with a Mediterranean climate have a long history of intensive human occupation that has been accompanied by frequent changes in land use and replacement of the original vegetation (Meira-Neto et al., 2011). The Arrábida Natural Park (ANP) in Portugal is a protected area and an example of well-preserved Mediterranean vegetation where disturbances such as natural fires and quarrying activities are present, but also interventions like revegetation and adaptive management strategies (e.g. SECIL-Outão quarry where ecological restoration is undergoing) (Nunes et al., 2014).

This karst ecosystem has a long tradition of limestone exploration. The industrial complex owned by SECIL – Companhia Geral de Cal e Cimento, S.A. in Outão was established in 1904 and has a total area of 482.7 ha, corresponding to about 4% of the ANP area (Figure 1.4) (Nunes, 2010). About 99 ha are used to extract limestone and marl, which are then processed in the cement factory located next to the quarries (Correia, 2000).

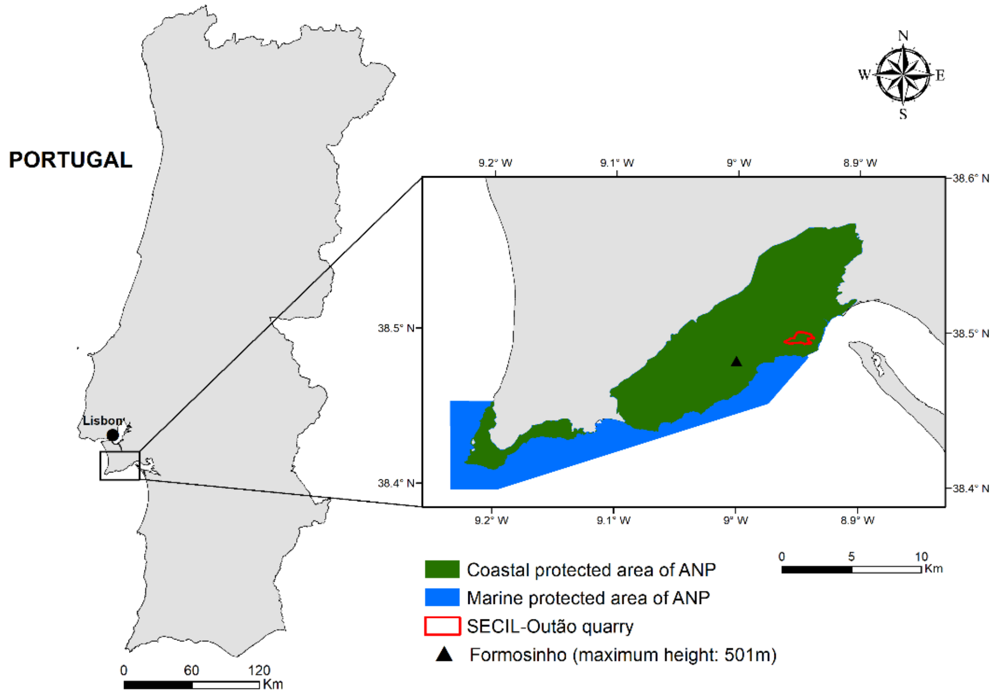


Figure 1.4. Arrábida Natural Park (ANP) map (Portugal). Adapted from ICNF (2022) and Lopes & Videira (2016).

Although quarrying is a necessity that provides much of the construction materials needed to build houses, roads, and buildings, it has long-lasting impacts on local ecosystems (Lameed & Ayodele, 2010). Quarrying adds to the biodiversity crisis by inevitably damaging ecosystems, resulting in biodiversity loss (caused by habitat destruction and other issues described in *section 1.1*) and consequent depletion of ES (Salgueiro et al., 2020).

Restoration, either spontaneous (passive) or assisted (active), stands as a solution to reverse quarrying impacts, thus contributing to improving the environment and, ultimately, human health (Palmer et al., 2010). However, limestone quarries are a good example of "irreversibility", as by completely removing the soil, they produce areas of bare rock which can lead to rocky desertification, taking hundreds of years to obtain a situation similar to the one that existed before – which is considered too long for passive restoration (Correia et al., 2001; Werner et al., 2001). Thus, active restoration in quarries can be seen as an opportunity to increase biodiversity and ES delivery by creating new habitats or restoring existing ones (Salgueiro et al., 2020).

Since the creation of the ANP in 1976, SECIL has planned and carried out a vast plan for the landscape restoration of the quarry, minimizing the environmental and visual impacts it has on the ANP.

This landscape restoration has been done by revegetating areas while simultaneously exploiting aggregates, avoiding the usual practice of only recovering quarries at the end of the exploration or when they have been abandoned. In the context of ecological restoration goals and assessment, described in *section 1.2*¸ a native reference ecosystem is fundamental to serve as a model for the ecosystem being restored. In this study, the natural non-degraded areas of the ANP were considered as the reference ecosystem, and the primary goal of SECIL-Outão quarry ecological restoration is to achieve the highest level of recovery possible relative to this reference (e.g. in species composition, structural diversity, ecosystem function). In this work, the reference ecosystem is interchangeably referred to as the natural vegetation (NV) present in the ANP.

1.3.2. ANP as a reference ecosystem in the ecological restoration of SECIL-Outão quarry

The ANP is situated in the Arrábida Chain (Serra da Arrábida), an elongated mountain located on the southern border of the Setúbal Peninsula, about 50km south of Lisbon, which reaches its maximum height at Formosinho (501m) and boasts several tops over 300 m (Meira-Neto et al., 2011). Here, slopes can be very steep, with a significant area with slopes between 50% to 70% inclination (Figure 1.5) (Catarino et al., 1982). The Arrábida Chain occupies a limestone massif, and due to its steepness, few sites have true soil profiles, mainly corresponding to zones of developed forest ("climax") (Catarino et al., 1982). Soils are generally skeletal, and among the rocky outcrops of limestone and dolomite, red soils ("terra rossa"), yellow Mediterranean soils and dark red limestone soils predominate accumulating in fissures and holes of the parent rock (Catarino et al., 1982). The abundance of inert materials of interest for construction, such as dolomitic, siliceous, and compact white and grey limestones, are characteristic of this area (Correia, 2000).

Figure 1.5. Mediterranean maquis on the hillslopes of the Arrábida Chain, in Arrábida Natural Park.

Although the ANP encloses a relatively limited geographical area, it comprises a great diversity of habitats and, correspondingly, plant and animal species, making it a unique region in Portugal. This area's high ecological and landscape value started to be recognized in the 1940s when the first attempts for its protection were made. Nonetheless, the insufficient protection afforded by the preventive measures decreed for the zone prompted the creation of the ANP in 1976 (Decreto-Lei nº 622/76, de 28 de julho), which was later reclassified by Decreto Regulamentar nº23/98 with an alteration of its limits and inclusion of a Marine Park area (ICNF, n.d.-b). This classification aimed to protect the natural patrimony of the area, more specifically the local geological, floristic, faunal and landscape values, along with material testimonies of cultural and historical nature. Besides its unique scientific value and state of conservation at a national level, the ANP is also of international interest, being part of the European Network of Biogenetic Reserves (area currently integrated into the “Arrábida-Espichel” site from the Natura 2000 Network) (ICNF, n.d.-b).

This protected area is characterized by having a Mediterranean dry sub-humid climate with an oceanic influence (Nunes et al., 2014). Its strong Mediterranean characteristics result in two extreme seasons: the hot dry summer reaching temperatures close to the ones of tropical regions, with prolonged periods of drought that can last for several months (typically, June until August); and the cold wet winter (ICNF, n.d.-b). These are interspersed with two intermediate seasons: autumn and spring. The proximity of the sea (the Atlantic Ocean) is a climate factor of great importance, giving the region higher humidity and, consequently, more amenity in temperatures throughout the year. For this reason, there is an Atlantic influence on the Mediterranean typicality that exerts essentially thermal amplitude reduction and atmospheric humidity increase, which occurs from mid-autumn to mid-spring.

In conjunction with what was previously said, the orientation and consequent exposure of the terrain also influence the amenity of the region’s type of climate. The intensity of solar radiation is affected by slope and aspect (slope orientation); hence north-facing slopes receive less direct sunlight and are cooler than south-facing slopes due to the absence of direct sunlight throughout the day, whether in winter or summer. Due to the low angle of the sun during the winter months, areas of north-facing hills may remain shadowed throughout the day. Thus, the face a slope presents to the sun – north, south, east or west – plays a role in the local climate, driving local variations in temperature and moisture. This "microclimate" influences the plant community types that occupy specific areas and their species diversity, phenology and productivity (Warren, 2008). Finally, regarding climatic factors and based on data obtained from the Setúbal meteorological station, it is important to mention that insolation in this region has two phases: one with increasing luminosity (January to August) and the other with decreasing luminosity (September to December). As can be expected, these greatly influence vegetation (ICNF, n.d.-b).

Due to the complex topography, different microclimates and soil thickness generate favourable conditions for the development of various vegetation types. The morphology of Arrábida causes differentiation of microhabitats with distinct characteristics (Mediterranean/Atlantic), allowing the existence of evergreen, deciduous and semi-deciduous plants. Three floristic elements converge in this territory, including (ICNF, n.d.-a): 1) the Euro-Atlantic vegetation, dominant on north-facing slopes in cooler, humid and shaded areas; 2) the Mediterranean vegetation, dominant on south-facing slopes in warmer, drier, and luminous areas; and 3) the Macaronesian vegetation, occurring under special conditions, occupying the steep maritime cliffs.

The vegetation and flora in this region are of extreme importance, as they are one of the last vestiges of Mediterranean vegetation in its various stages (Pedro, 1998). A total of 1450 taxa have been inventoried in the Arrábida Limestone Massif, representing about 40% of all flora in mainland Portugal, making this area the biggest deposit of national phytodiversity (ICNF, 2021; Pedro, 1997). More than 100 of these taxa are endemic and rare, with many only found in this area (ICNF, 2021). Of the inventoried taxa, 90 were classified with high value as genetic patrimony, and 18 taxa, mostly Portuguese endemisms, are present in the Habitat Directive (ICNF, n.d.-a). Examples of these are *Iberis procumbens* subsp. *microcarpa* and *Arabis sadina*, protected endemic species with occurrence in the Central West Coast of Portugal (included in the Anexos II and IV of the Directive 92/43/CEE) (ICNF, n.d.-a). Additionally, Portuguese limestone regions are one of the richest places in orchids, and the ANP is no exception, hosting around 30 taxa of the Orchidaceae family, which corresponds to almost 50% of all Portuguese wild orchid species concentrated in this small geographical area (Frazão, 2020). Most of these species are associated with semi-natural perennial grasslands (with *Brachypodium phoenicoides* dominance), herbaceous habitats with no or few shrubs, which occupy extensive areas in the interior of Arrábida.

Despite having many similarities with other limestone mountain ranges which develop further north, the vegetation here presents unique aspects such as the arboreal kermes (*Quercus coccifera*) and gorse (*Ulex densus*). According to Gomes Pedro (1942), there are five distinct physiognomic vegetation types in the ANP: rupestrian formations, heathland, scrubland/thickets, machial and woodland. The rupestrian formations in the poorer and rocky soils include lichens, small ferns, mosses and other xerophytes (Pedro, 1998). In the limestone areas, species such as *Quercus coccifera*, *Smilax aspera*, *Rosmarinus officinalis*, *Pistacia lentiscus*, *Phillyrea angustifolia*, *Phillyrea latifolia*, *Olea europea* var. *sylvestris*, *Ceratonia siliqua*, *Cistus* sp., *Rhamnus* sp. and *Daphne gnidium* are present (Pedro, 1998). The heathlands are characterized by having low vegetation, grassland and sparse shrubs. In contrast, the Mediterranean maquis is a habitat comprising closed vegetation, often reaching 100% plant cover, predominantly composed of evergreen sclerophyll shrubs that can reach arboreal size. The presence of semi-deciduous/deciduous shrubs, a few annuals and some geophytes can also be observed in this type of habitat (Thompson, 2020). In ecological succession, the machial precedes the woods, making the transition between these and scrubland/thickets, which consist of a bi-stratified formation of wild olive and carob trees (Mediterranean-characteristic evergreen formations) (Pedro, 1998). In general, semi-deciduous shrubs and evergreen sclerophylls dominate in the initial and end phases of natural succession, respectively (Catarino et al., 1982; A S Clemente, 2002). The last physiognomic type, the woodland forests, consists of carob, olive, oak, and mixed pine trees.

Indeed, the ANP is a rare example of very ancient Mediterranean vegetation that, over time, has been shaped by natural and anthropogenic factors (Guerreiro, 2008). Apart from its exuberant and surprising appearance, the extremely high plant diversity and endemisms make the ANP an area of national relevance. Since the ANP's creation, anthropogenic pressures such as deforestation and grazing were diminished considerably; however, some disturbances have remained until the present, such as the exploitation of open-pit limestone quarries. Current environmental legislation obliges companies to revegetate the mined lands to accelerate the recovery of the degraded areas and favour their reintegration into the natural landscape. Thus, the conservation and restoration of this protected and ecologically important regional area is fundamental (Clemente et al., 1996; Correia et al., 2001).

1.3.3. Intervention history in the ecological restoration of SECIL-Outão quarry

The SECIL-Outão quarry divides into two areas or quarries depending on the mineral extracted (limestone or marl). It was explored downwards from the highest elevation point, near the Arremula Ridge, resulting in a stair-like sequence of narrow benches with a height difference of 20m and maximum width of 20m in the limestone area. Quarry excavation of the hill face resulted first in benches and escarpments between each bench (in the limestone area) and later steep slopes (in the limestone and marl area) (Correia, 2000). As required by law, a Plan of Landscape Recovery was approved in 1981 to reduce the visual impact resultant from the exploration and restore the original existing vegetation cover in that area (SECIL, 2017). As the exploration progressed and exploitation sites were abandoned, revegetation started to be performed, creating restored benches and slopes with different restoration ages and distinct revegetation techniques or intervention modes (Figure 1.6, Figure S2.5).

The revegetation process began in 1983 with intervals of about three years between each bench (roughly the time it takes to be explored), thus giving rise to plant communities of different ages and covers (Correia et al., 2001). Therefore, the SECIL-Outão quarry not only contains heterogeneity among sites related to the local topography but also to temporal and historical differences in the quarry exploitation, resulting in adjacent areas being at different stages of regeneration, according to time after restoration.

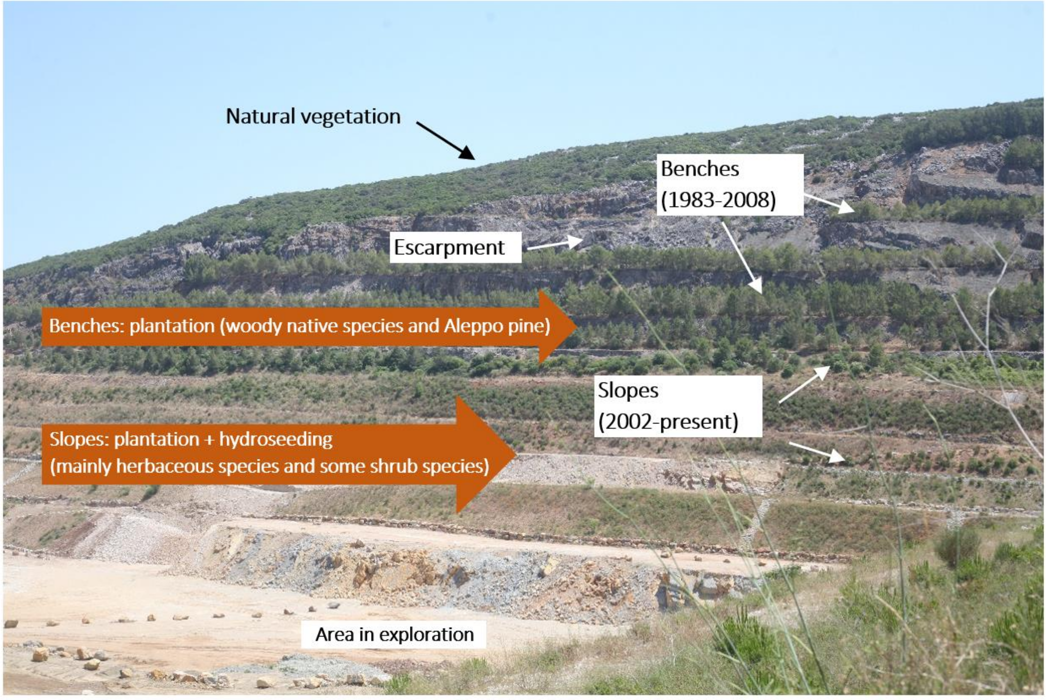


Figure 1.6. Topography and different types of interventions in SECIL-Outão quarry (image of the "Vale de Mós B" limestone quarry).

For the revegetation, a 1m layer of desegregated marl with high clay content was added to each bench's bare horizontal rock surface. Then, woody sclerophyllous Mediterranean species (native) and fast-growing species (introduced) were planted (Table 1.3) (Correia et al., 2001). The plants used in the revegetation were produced in SECIL's plant nursery and placed, with an age of 2 years, in 50 cm holes filled with organic soil (Correia et al., 2001). The seeds of the various species were collected in the natural areas surrounding the exploration and transported to open-air nurseries or greenhouses.

Table 1.3. List of species used in the revegetation process in the SECIL-Outão quarry. Nomenclature according to Franco & Rocha-Afonso (1984).

Native species			**Introduced species**		
Arbutus unedo L.	Strawberry tree	Medronheiro	*Pinus halepensis* Miller	Aleppo pine	Pinheiro de Alepo
Ceratonia siliqua L.	Carob	Alfarrobeira	*Pinus pinea* L.	Stone pine	Pinheiro manso
Juniperus phoenicea L.	Phoenician juniper	Zimbro	*Retama monosperma* (L.) Heywoord	Bridal broom	Piorno
Myrtus communis L.	Myrtle	Murta	*Spartium junceum* L.	Spanish broom	Giesta
Olea europaea var. *sylvestris* Brot	Wild olive tree	Zambujeiro			

Native species			Introduced species
Phillyrea angustifolia L.	Narrow-leaved mock privet	Lentisco	
Phillyrea latifolia L.	Mock privet	Aderno	
Pistacia lentiscus L.	Mastic tree	Aroeira	
Quercus coccifera L.	Kermes oak	Carrasco	
Quercus faginea Lam.	Portuguese oak	Carvalho-cerquinho	
Rosmarinus officinalis	Rosemary	Alecrim	
Viburnum tinus	Laurustinus	Folhado-comum	

In the last two decades, hydroseeding started to be performed in slopes, producing a more effective plant cover of the soil by creating a favourable environment for germination in adverse weather conditions. Hydroseeding has been used since 2005 in rehabilitating limestone and marl slopes to overcome various revegetation difficulties related to the nature of the soil, high slopes and vegetation establishment. This technique involves projecting a homogeneous slurry of seeds, fertilizer, binder, mulch and other components (e.g. moisture retention agents, plant-growth promoters) over large or inaccessible areas (Clemente et al., 2016). This way, it builds an ecological wall with variable slopes up to 80º and no height limitations (keeping the front of the wall fully vegetated and integrated into the landscape).

Initially, commercial seeds were used, but since 2007 allochthonous species were no longer allowed to be utilised in the quarry revegetation. Hence, native species started to be exclusively used. The use of native species instead of fast-growing commercial species was proposed because the latter strongly depends on irrigation, which is incompatible with sustainable water use in arid and semi-arid environments and can outcompete native species and constrain vegetation dynamics in the long term (García-Palacios et al., 2010; Matesanz & Valladares, 2007). In 2010, hydroseeding was performed with a new mixture of 22 herbaceous and shrub species, of which 15 were autochthonous species that occur naturally in the ANP, representing different families and biological types.

This restoration action is a unique case of intervention in the territory that has been ongoing for more than 35 years, resulting in a recovered area of 44 ha, which corresponds to 45% of the total quarry areas (SECIL, 2019). In the limestone quarry 9.5 ha have been recovered (17.6% of the limestone area), and in the marl quarry 32.27 ha have been recovered (72.1% of the marl area) (SECIL, 2017). In the specific case of the restoration of this quarry, the introduction of allochthonous species, especially *P. halepensis*, in the limestone benches, contributes to the distance between the restored areas and the NV (Figure 1.7). Initially, Aleppo pine was planted as a pioneer species and was expected to have a facilitative effect on late-successional hardwoods and gradually generate a diverse ecosystem. However, previous studies evaluating the restoration success in the SECIL-Outão quarry demonstrated that pines might actually have a negative effect on later-successional species and failed to improve ecosystem functioning (Correia et al., 2001; Nunes et al., 2014). Additionally, the vegetation of the restored benches differs from the NV, mainly due to the presence of the allochthonous species planted such as *P. halepensis*, *S.* junceum, *R. monosperma*, and *C. siliqua*, which occur elsewhere in the ANP in lower densities but are not found in areas adjacent to the quarries (Correia et al., 2001; Nunes et al., 2009).

Figure 1.7. Aleppo pine (*P. halepensis*) community in the restored benches of SECIL-Outão's quarry.

Nevertheless, the proximity of well-preserved NV facilitates the entry of native species into the quarry areas. The NV adjacent to the quarry is of the maquis type, composed of shrubs, small trees and sclerophyllous species up to 5 m in height (Nunes et al., 2009; Pedro, 1998). The natural limestone vegetation area exhibits several rocky outcrops with vegetation characteristic of this type of habitat (Figure 1.8) (Pedro, 1998).

Figure 1.8. Mediterranean Maquis vegetation on the rocky outcrops of ANP areas surrounding the SECIL-Outão quarry.

The progressive cessation of exploration in different locations of the quarry was accompanied by the revegetation of those areas resulting in a variety of degradation and regeneration phases that confer the landscape a distinct mosaic of habitats (Correia et al., 2001). Thus, adjacent revegetated areas in different stages of recovery, with different restoration ages, create a chronosequence that allows for valid observations of a series of sites of different ages simultaneously, enabling the study of ecological succession in a single period of time (Correia et al., 2001; L. R. Walker & Moral, 2003). These circumstances make these places very suitable for ecological studies and constitute a precious source of knowledge for adjusting recovery strategies.

The different surface slope (benches vs. slopes) and effects of distinct revegetation techniques play an important role in the restoration outcomes. These should be differentiated from other biophysical factors. Previous studies of the SECIL-Outão quarry restoration have focused on a limited number of restored sites within a small range of topographical, biological and intervention characteristics. The same applies to previously sampled plots in the reference areas (NV), usually only sampled in the quarry's surroundings, which may misrepresent the reference ecosystem. Furthermore, the studied variables focused on a limited number of attributes such as species composition, structure and biomass based on allometric equations. To evaluate restoration success, it is relevant to assess the key variables stated by SER (*section 1.2.1*). Since ecosystem recovery occurs at the landscape scale, large-scale monitoring of the restoration and the ES provided is needed to make more informed adaptive management decisions. Therefore, combining field sampling and RS data will contribute to a better understanding of the restoration trajectory, large-scale calculation of the ES provided by restored areas, and evaluation of ecological restoration success.

1.4. Objectives and Hypotheses

This work aimed to assess the success of the ecological restoration undergoing at the SECIL-Outão limestone quarry, located within the protected area of the ANP, by using field and RS data to model and map several ES and their ecological indicators. Multiple ecosystem attributes were studied, given that they are key ecosystem attributes used to evaluate ecological restoration and can be associated with several ES. In this research, the following ecosystem attributes were analysed: 1) Productivity and Carbon Sequestration; 2) Soil Fertility and Decomposition; 3) Habitat Quality; 4) Similarity to Natural Vegetation; 5) Diversity (Taxonomic and Functional) and 6) Ecosystem Functioning (Pollination, Resilience and Adaptation to Fire, Resilience and Adaptation to Drought and Dispersal capacity, Interaction with Fauna and Regeneration).

More specifically, the main research objectives, which were applied to each of the ecosystem attributes mentioned above, were to:

1. Analyse the trajectory of each ecosystem attribute with restoration age (based on the chronosequence).
2. Determine which predictors best estimate each ecological indicator to identify limiting factors or promoters of recovery and suggest recommendations for adaptive ecosystem management.
3. Identify RS indices associated with vegetation characteristics, and model their relationship with field data, to extrapolate and map regulating ES at the landscape scale.
4. Compare the absolute values of the ecosystem attributes between the different restoration intervention modes and the reference ecosystem.
5. Determine the level of ecosystem recovery in the different restoration intervention modes and zone types through an ecological restoration success index.

Furthermore, three hypotheses were suggested: 1) the recovery trajectory over time is different for each ecosystem attribute due to the inherent dynamics of its indicators; 2) older restored quarry areas are closer than recently restored areas to providing the same regulating ES as the reference ecosystem, and 3) RS data can be validated by field data and allow to upscale field information, providing useful models that can be used to produce maps at the landscape scale. To validate these hypotheses, in-situ and RS data were combined to spatially assess restoration, following the methods described in the next section.

CHAPTER 2 | METHODS

2.1. Study area

The study area of this project is one of the largest cement factories in ANP, the SECIL-Outão quarry (Figure 1.4), located within the Natural Park in Outão (Setúbal). The climate here is considered to be Mediterranean dry-humid, with an average yearly temperature of 16.4ºC and average annual precipitation of 650 mm (Nunes et al., 2014). The ANP area, where the company’s manufacturing facilities are located, includes the 347 m Arremula Ridge and is delimited by the Rasca Valley that descends towards Outão. The elevation within the quarry varies from 120 to 340 m (Correia et al., 2001).

The SECIL-Outão quarry is divided into two quarries depending on the rock type extracted, with “Vale de Mós A” being the marl quarry and “Vale de Mós B” the limestone quarry (Figure 2.1). The different exploration processes in the marl and limestone quarry gave rise to areas with different characteristics in terms of elevation, slope and soil depth and led to the formation of benches or slopes. Several revegetation interventions have been applied in both quarries at different points in time, resulting in a large spatial and temporal heterogeneity. Therefore, the rate of natural or assisted colonization by plant species varies according to these characteristics.

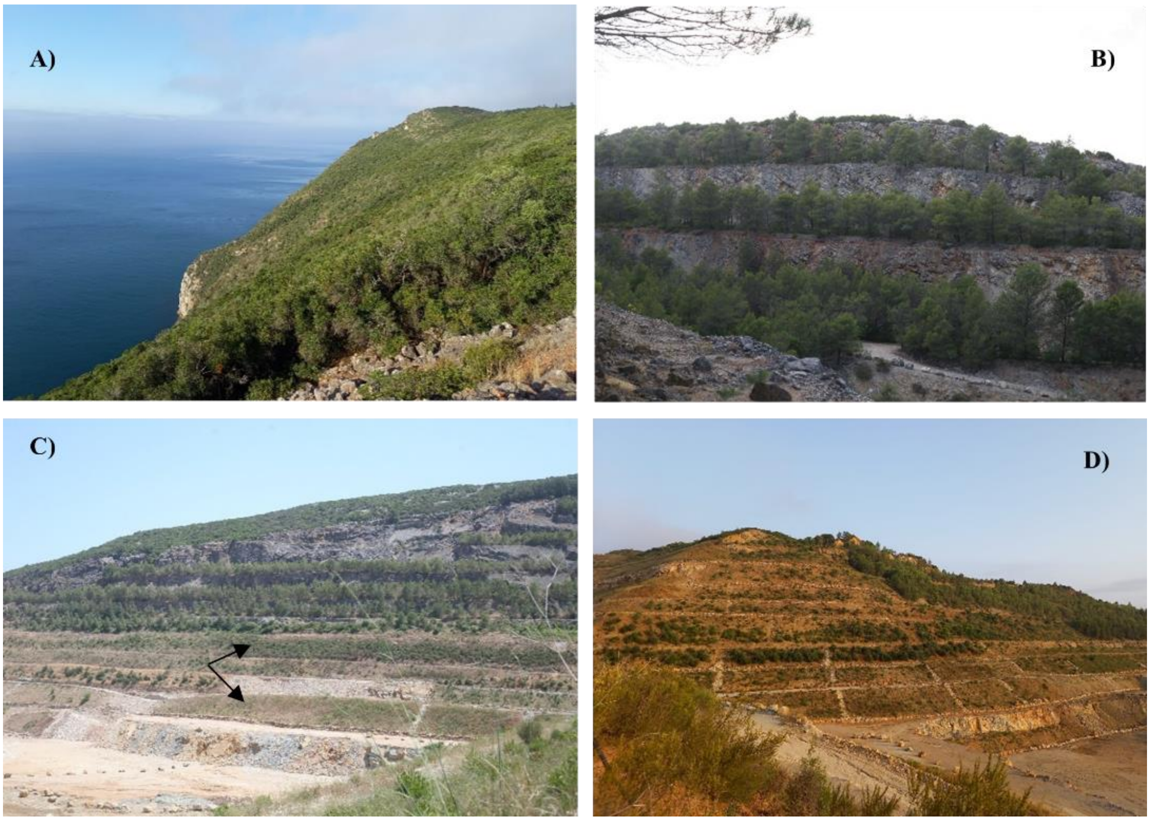


Figure 2.1. The different types of studied sites in the study area. A) Natural vegetation (Maquis) areas – reference ecosystem. B) Revegetated benches in the limestone quarry “Vale de Mós B”. C) Revegetated slopes in the limestone quarry “Vale de Mós B” indicated by the black arrows. D) Revegetated slopes in the marl quarry “Vale de Mós A”.

Different types of revegetated areas are present in SECIL's limestone and marl quarries (Figure 2.2). The limestone quarry is characterized by having a North-facing or Northeast exposure and restored benches (low slope) with plantations of mostly Aleppo pine and shrub species at higher elevations. At lower elevations, limestone slopes were restored with hydroseeding and plantations of mostly herbaceous and shrub species. In the marl quarry, horizontal benches are absent, and slopes dominate the revegetated areas, with some being very steep. Marl slopes were revegetated with hydroseeding, or a mix of hydroseeding and plantation, of mostly herbaceous and some shrub species. However, the presence of Aleppo pine patches can be observed in some of the oldest restored marl slopes. Marl areas have a Southern to Eastern exposure.

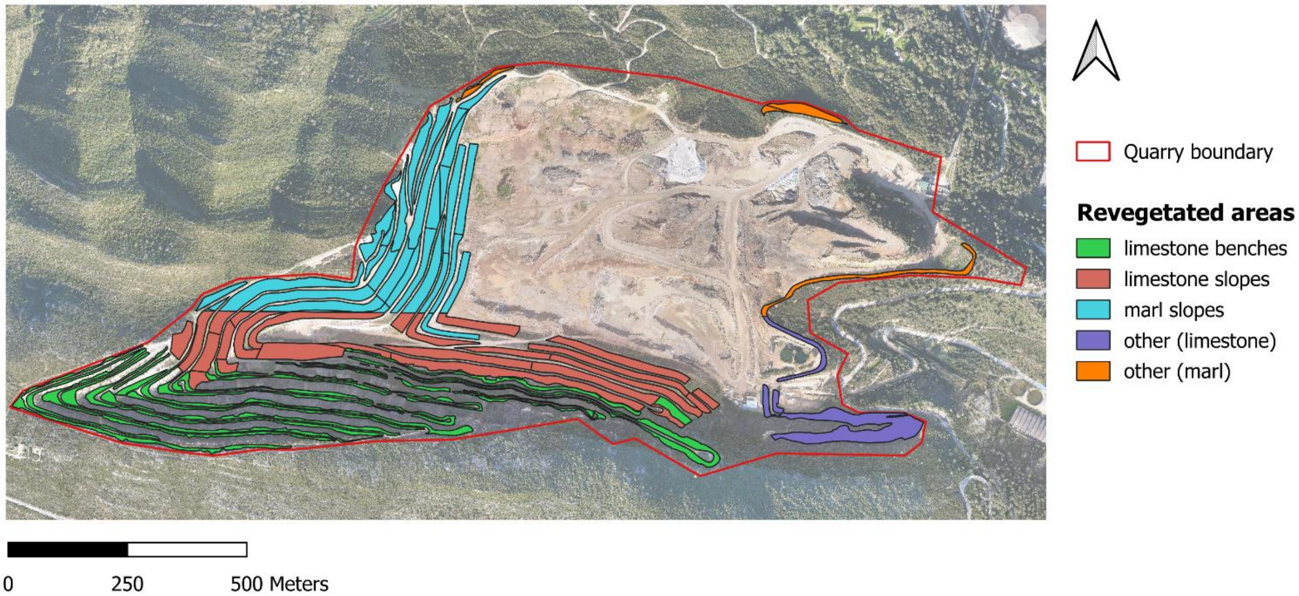


Figure 2.2. Types of revegetated areas regarding the different excavation methods done in the quarry, which resulted in narrow benches or high vertical slopes, in limestone or marl substrates. "Other" refers to revegetated areas in the quarry that are not benches or slopes. NV surrounds the quarry boundary.

To study the success of the revegetation interventions on the quarry, natural non-degraded areas within the ANP were considered as the reference ecosystem in this study. The reference ecosystem areas studied in this research are characterized by having a Mediterranean maquis vegetation type, as described in *section 1.3.2*, and were sampled in the NV areas surrounding the quarry and in remote ANP areas with well-conserved Maquis ecosystems.

2.2. Experimental design

2.2.1. Sampling sites selection

The area of the limestone and marl quarries is very variable in terms of topography and exposure. To control the effect of these factors, it was necessary to select areas with different characteristics, both in the restored quarry areas and in natural vegetation areas. For such, a stratified sampling was done based on Normalized Difference Vegetation Index (NDVI), Potential Solar Radiation (PSR), and elevation above mean sea level w(Figure 2.3). This method not only allows to obtain a representative sampling in the quarry by improving the amount of data distributed along a gradient of different variables, but it also helps to optimize field data collection by focusing on relevant sites instead of focusing on replication in sites that may be similar, improving the cost-effectiveness ratio.

NDVI is a vegetation index indicative of different vegetation types, plant biomass, productivity, and vegetative health. PSR is often used as a proxy of microclimatic conditions as it quantifies incoming solar radiation (Príncipe et al., 2014; Turvey & Mclaurin, 2012). Elevation controls many environmental factors (e.g. temperature, radiation, humidity) in natural ecosystems (Körner, 2007). It also reflects the type of exploration and revegetation technique performed in the quarry (e.g. higher elevations in the limestone quarry correspond mostly to benches, where pine trees are dominant, while lower elevations usually correspond to slopes revegetated by hydroseeding), thus revealing topographical and vegetation characteristics (Figure S2.2 and Figure 2.5). Additionally, in the quarry, higher elevations generally correspond to older restored sites (Figure S2.2 and Figure S2.5). The variation of slope, elevation, PSR, aspect, restoration age and NDVI in the quarry can be observed in Figure S2.1, Figure S2.2, Figure S2.3, Figure S2.4, Figure S2.5 and Figure S2.6, respectively.

Firstly, a Sentinel-2A multispectral image (10 x 10 m resolution) containing the study area from 10-03-2020 was acquired from open-access satellite RS products. The image was then processed in the Sentinel Application Platform (SNAP, version 7.0) and cropped to the subset study area. Next, the image was analysed in the geographic information system application ArcGIS (ESRI® ArcMapTM version 10.7.1), where NDVI values were determined, using the RED and NIR (near-infrared) bands (as described in *section 2*.8). The elevation and PSR were obtained from SECIL's Digital Terrain Model (DTM) from 2019, with a resolution of 20 cm, that was resampled to 10 m to be perfectly overlapped with the Sentinel-2A multispectral image. PSR was calculated in ArcGIS using the "Area Solar Radiation" Spatial Analyst tool (for the whole year). The output incoming solar radiation derived from this tool takes into consideration the absolute position of the area (latitude and longitude) to determine solar declination and solar position used to compute local direct and diffuse radiation, and topographical variables obtained from the DTM, such as slope and aspect (orientation of the slope), which influence the intensity of solar radiation at a given location (ESRI, n.d.).

As a result, spatially explicit gridded information with 10 m resolution was created for NDVI, PSR and elevation covering the study area, showing a perfect spatial overlap between pixels. This information was then transformed into tabular format, each column corresponding to a variable (NDVI, PSR and elevation) and each line corresponding to a pixel. Then, a cluster analysis using the table created was done using the R software for statistical computing, with the selection of 80 clusters or classes created based on how closely associated pixels were regarding the variables, resulting in potential sampling pixels in the same group being more similar to each other than to those in other groups (R Core Team, 2013).

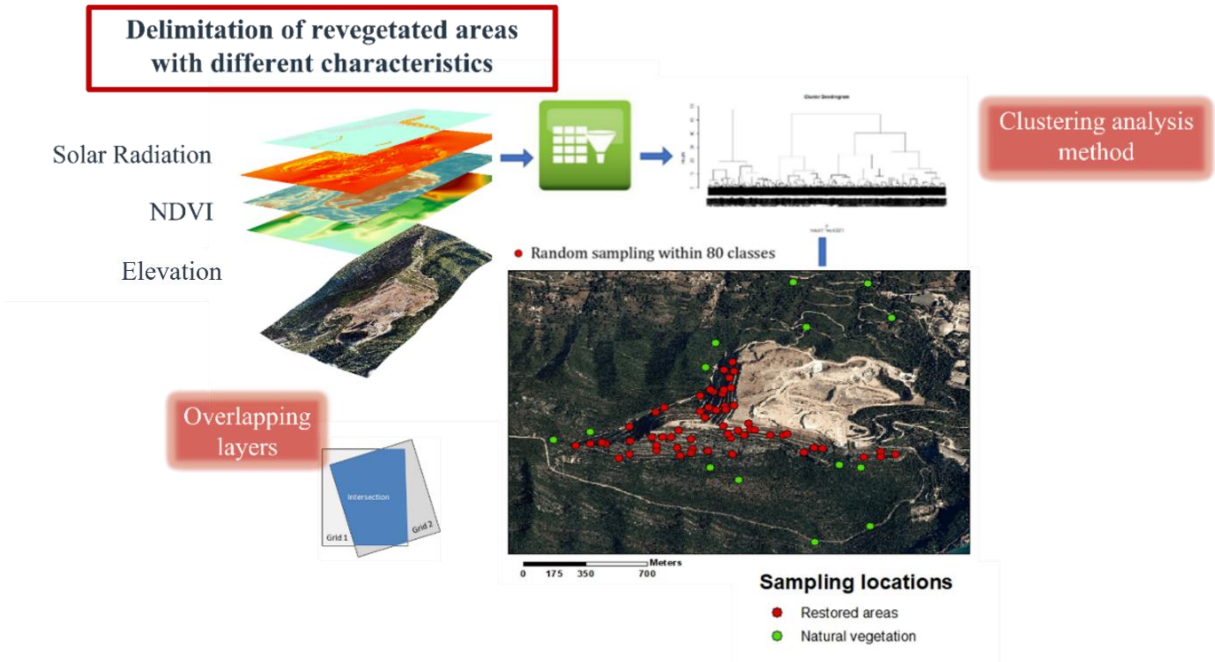


Figure 2.3. Methodological approach used for sites selection in the first sampling phase.

Since each pixel is a 10 x 10m pixel, only classes with more than 20 pixels were considered sufficiently representative of the quarry, resulting in 56 classes. The sampling locations were chosen by selecting the sites with 2 or 3 contiguous pixels of the same class so that no class overlap would occur in field sampling. Additionally, sites were selected according to their accessibility so that a 10 x 10 m parcel could fit. Primacy was given to areas with higher vegetation strip width. Given the inaccessibility and sampling difficulty in some slopes, geographically close classes of pixels were condensed, resulting in a total of 50 classes (or sampling plots) sampled in the first sampling phase. Due to some restoration ages being poorly replicated or represented, six plots from benches and slopes were additionally sampled in the second phase after identifying what restoration ages were missing representation in the data from the first sampling phase (ages between 10 and 20 years).

For the NV, a stratified sampling design based on PSR and elevation was applied, resulting in five classes. Seven plots were sampled in the first sampling phase (one plot in each class and a replica in two classes). For the NV, the elevation and PSR were obtained from ASTER's GDEM v3 with a resolution of 30 m (Abrams et al., 2020), because SECIL's high resolution DTM only covered the quarry area. The NDVI was not used as a variable in the stratified sampling of the NV because the Maquis vegetation type present in the natural areas is similar, contrary to the revegetated areas in the quarry where pine trees contrast with shrubby sclerophyllous or herbaceous dominant areas, and where NDVI is useful in differentiating the vegetation type of those areas.

After the first sampling phase, further analysis of NV's sampled sites showed that these were insufficient to represent the elevation, PSR, and slope variations observed in the quarry. Because the steep slope values within the quarry are not easily found in the NV areas, the reference ecosystem's sampling area was widened to guarantee the same range in elevation, PSR and slope. Then, additional NV plots were identified and sampled. These plots were defined by analysing the range of elevation, PSR, and slope values within the quarry, identifying which range of values was not represented in the first sampling phase plots, and lastly, selecting six classes in the NV that represented the missing elevation, PSR and slope values (Table 2.1). In this way, six additional plots were sampled in the NV's areas in the second sampling phase (all NV plots are represented in Figure 2.4). Although some of these plots are 5 km far from the quarry's study area, the reference ecosystem sampling area continues to be enclosed in the ANP area, and the range of variables considered to affect the vegetation is the same.

Table 2.1. Classes of elevation (m), PSR (WH/m2) and slope (º) defined for the field sampling of NV (reference ecosystem) areas, which serve as a control (reference ecosystem) for the quarry limestone benches, limestone slopes and marl slopes. These classes were sampled in the second sampling phase.

Control for:	Elevation (m)	PSR (WH/m^2)	Slope (º)
Limestone benches	High (≥ 269.34)	Medium (987949-1913864)	Low (≤ 14.67)
Limestone slopes	Low (≤ 201.67)	Low (≤ 987949)	High (≥ 27.3)
Marl slopes	Medium (201.67 – 269.34)	High (≥ 1913864)	High (≥ 27.3)
	Low (≤ 201.67)	High (≥ 1913864)	High (≥ 27.3)
	Medium (201.67 – 269.34)	High (≥ 1913864)	Medium (14.67 – 27.3)
	Low (≤ 201.67)	High (≥ 1913864)	Medium (14.67 – 27.3)

Additionally, the shrub layer data from 9 plots sampled in a previous project between SECIL, the University of Lisbon and the University of Évora (from 27-05-2019 until 04-06-2019) was used in this research to increase the sampling sites number and amount of data studied. These sampling plots (three plots in the limestone quarry, three in the natural areas, and three in a natural forest with planted pines) were revisited and sampled in the second sampling phase for the remaining field variables of interest in the herbaceous and tree layer. However, the natural pine forest plots were excluded from the analysis (except for *section 3.4* in which they are included in one of the similarity graphs) as they represent an "artificial reference" of a planted pine forest and because, effectively, the accurate reference in which the objectives of ecological restoration should be guided towards is the Mediterranean Maquis.

In total, 78 sites were sampled (59 sites in the quarry restored areas and 19 sites in the NV). The sampling locations are shown in Figure 2.4.

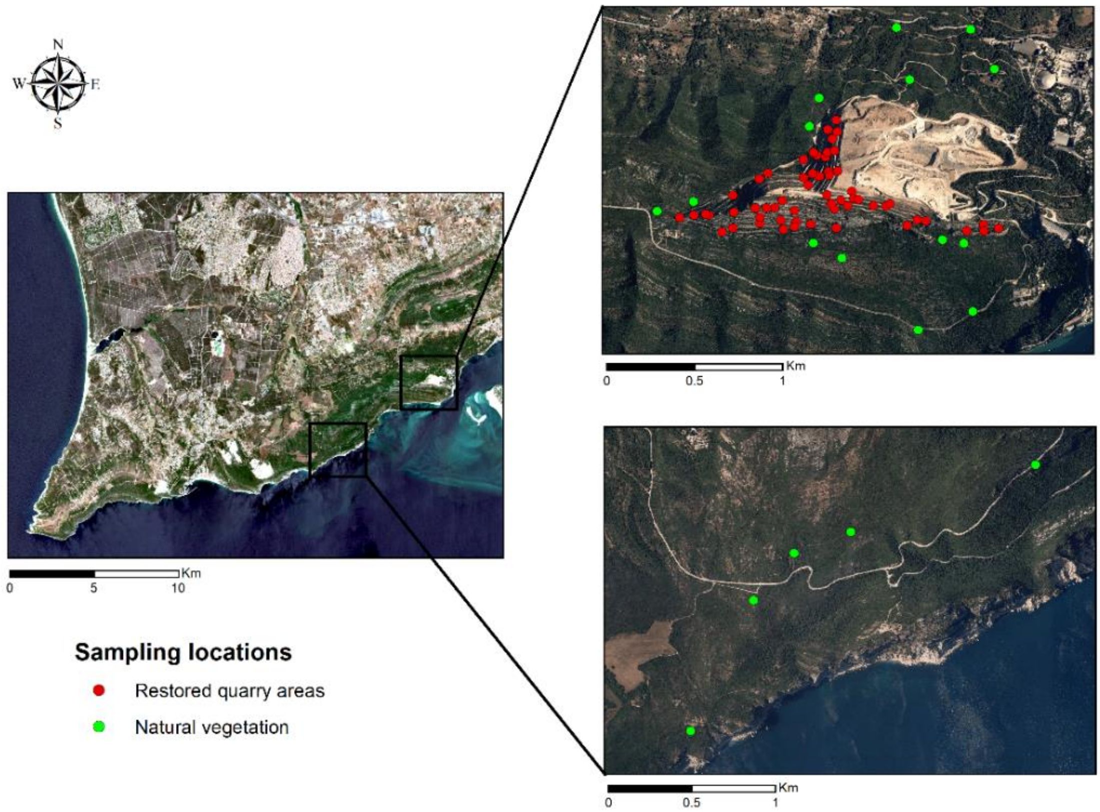


Figure 2.4. Sampling locations in the restored quarry areas and in the natural vegetation (reference ecosystem).

2.2.2. Classification of restoration intervention modes

The type of restoration intervention applied in the quarry's different revegetated areas (e.g. plantation of shrubs and Aleppo pine vs. hydroseeding with different compositions) varied over time and in space, especially in agreement with the best knowledge and plant material available at the time of application and depending on the characteristics of the area to be recovered (benches or slopes). Since the mode of intervention plays a crucial role in the current vegetation composition, a new categorical variable named "intervention mode" was created to study the effects of the different revegetation interventions in the quarry restored areas, to understand which intervention modes have the most similar values to the reference ecosystem, and to include as a variable in the statistical modelling described in the following sections. Additionally, this variable was used in the ecological restoration success index (*section 2.12*)

The plots sampled in the quarry were classified into different modes of intervention, considering the nature of the revegetation technique (plantations, hydroseeding, both or none/unknown) and the application of the recommendations or pilot tests of the FCUL team in previous researc protocols (e.g. application of compost, fertilizers, different seed mixtures with the goal of increasing native species density, watering, etc.), resulting in 6 classes (Figure 2.5). Characteristics of each intervention mode, such as the number of sampled plots and average restoration age, slope, and elevation, are presented in Table 2.2. The variation of shrub, pine and herbaceous cover in each intervention mode category can be seen in Figure S2.23, Figure S2.24 and Figure S2.25, respectively. Information about the intervention modes in the different restored areas was obtained from databases created in previous FCUL protocols with SECIL.

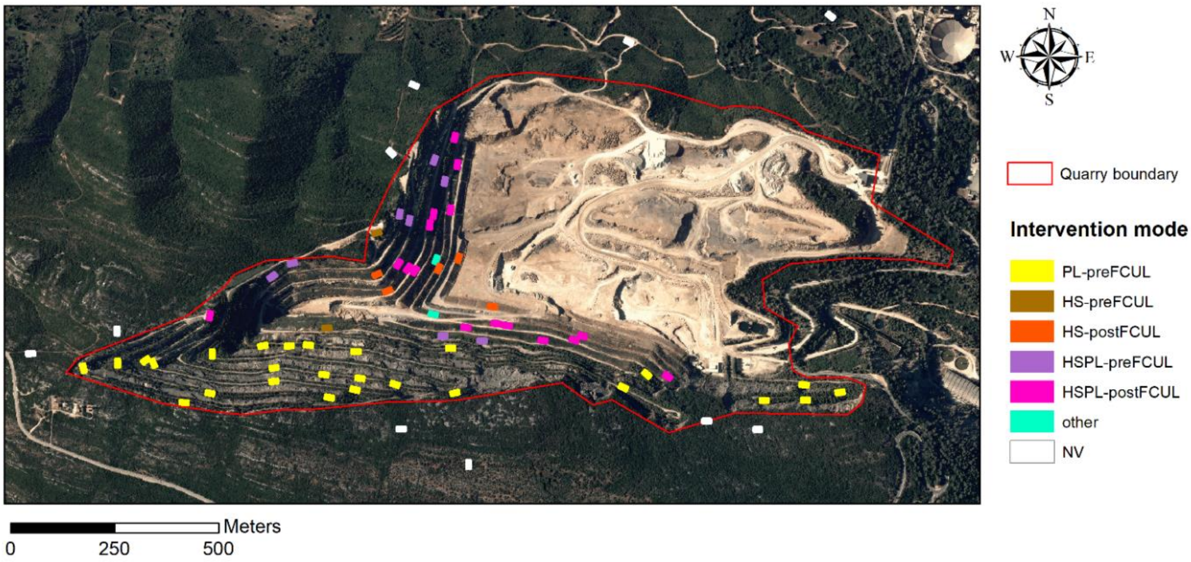


Figure 2.5. Intervention modes of the quarry revegetated study sites, considering the nature of the intervention (planting, hydroseeding, both or none/unknown) and the application of the recommendations (or pilot tests) of the FCUL team in previous protocols. HS- hydroseeding, PL- plantation, HSPL- hydroseeding and plantation, other- none or unknown intervention, NV- NV (Maquis).

Table 2.2. Intervention mode classification defined according to the nature of the intervention (planting, hydroseeding, both or none/unknown) and the application of the recommendations (or pilot tests) of the FCUL team in previous protocols. The number of sampled plots, average restoration age, average slope and average elevation in each category are shown. Standard deviation (± SD) is also presented. The colour code is equivalent to Figure 2.5.

Intervention mode	**Intervention**	**FCUL Support**	**Number of sampled plots**	**Restoration age (years)**	**Slope (°)**	**Elevation (m)**
PL-preFCUL	only plantation	No	26	25.0 ± 4.5	11.3 ± 6.8	251.3 ± 65.7
HS-preFCUL	only hydroseeding	No	2	19.5 ± 14.9	29.1 ± 7.6	233.9 ± 25.5
HS-postFCUL	only hydroseeding	Yes	5	6.2 ± 2.1	26.8 ± 8.9	177.6 ± 34.6
HSPL-preFCUL	hydroseeding + plantation	No	8	17.0 ± 7.2	31.2 ± 6.6	209.9 ± 32.0
HSPL-postFCUL	hydroseeding + plantation	Yes	16	8.0 ± 2.5	30.9 ± 6.5	182.8 ± 34.7
other	none / unknown	No	2	7.0 ± 0	32.8 ± 0.8	166.5 ± 4.0
NV	without any intervention (NV in non-degraded areas)	-	19	-	17.8 ± 10.5	239.1 ± 68.8

2.2.3. Classification of Zone Type according to different surface exposure and slope classes

Since distinct topographical characteristics in its area characterize the quarry, all sampling plots (including NV plots) were classified into five zone types based on different categories of surface exposure (northern, eastern and southern) and slope (low slope: < 21°, high slope > 21°) found in the quarry, as shown in Figure 2.6. The northern-facing benches (low slope) and slopes (high slopes) are part of the limestone quarry, while the southern and eastern-facing slopes are located in the marl quarry.

In the similarity analysis (*section 2.6*) and the ecological restoration success index (*section 2.12*), the restored sites were compared to the controls in the NV that adequately served as their reference regarding these topographical factors. Hence, revegetated sites were compared with reference sites in the same Zone Type, with similar surface exposure and slope characteristics. These two variables were obtained using ArcGIS's Aspect and Slope tools from SECIL's Digital Terrain Model (DTM). Characteristics of each Zone Type regarding average restoration age, slope, elevation, and number of sampled plots, in both quarry and NV, are presented in Table 2.3.

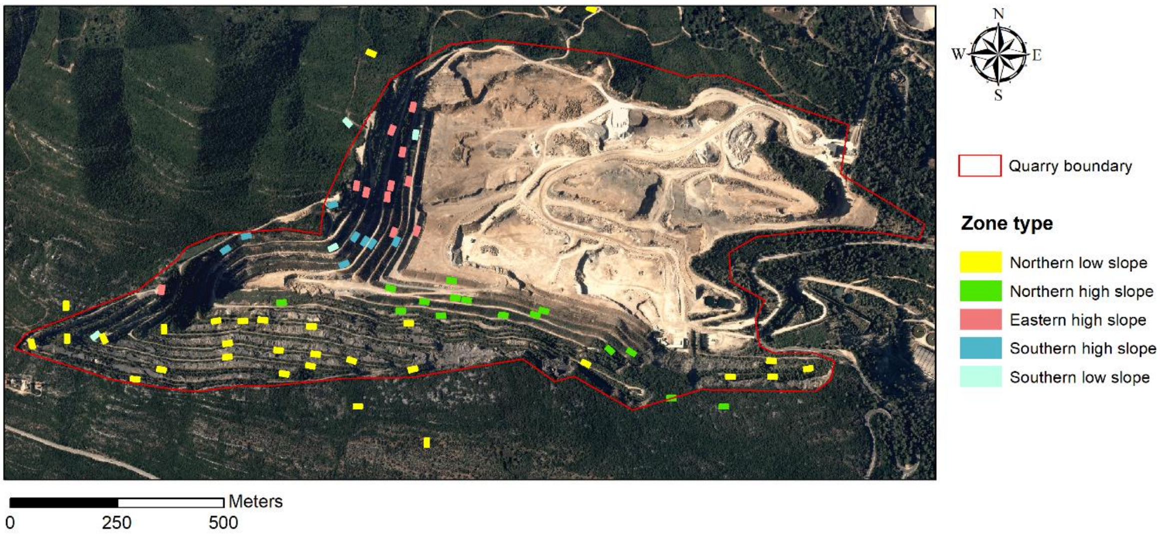


Figure 2.6. Zone Type defined according to five different classes of surface exposure and slope found in the quarry area. NV plots were also classified accordingly. Low slope: < 21°, High slope > 21°.

Table 2.3. Zone Type classification defined according to different classes of surface exposure and slope in the quarry area. The average restoration age, average slope average elevation, and the number of sampled plots in each category are shown for the quarry restored plots and NV (NV) plots. Standard deviation (± SD) is also presented. The colour code is equivalent to Figure 2.6.

Zone type	Description	Restoration age (years)	Slope (°)		Elevation (m)		Number of sampled plots	
			Quarry	NV	Quarry	NV	Quarry	NV
Northern low slope	Limestone benches facing North, with low slope	25.5 ± 3.6	10.9 ± 6.0	11.9 ± 6.1	252.6 ± 65.6	265.5 ± 95.8	24	6
Northern high slope	Limestone slopes facing North, with high slope	8.2 ± 2.8	31.0 ± 4.0	25.4 ± 1.1	171.6 ± 16.8	193.5 ± 5.6	13	2
Eastern high slope	Marl slopes facing East, with high slope	13.0 ± 8.5	30.9 ± 5.1	31.1 ± 5.0	193.3 ± 41.9	226.3 ± 46.2	11	4
Southern high slope	Marl slopes facing South, with high slope	12.1 ± 7.6	32.9 ± 5.1	31.1 ± 5.0	214.6 ± 34.2	226.3 ± 46.2	8	4
Southern low slope	Marl slopes facing South, with low slope	13.3 ± 11.9	8.1 ± 0.3	9.5 ± 5.2	247.85 ± 46.9	235.1 ± 56.8	3	4

The difference in the number of control plots in the NV between the various Zone Types was due to safe access limitations in the natural areas. It should also be noted that the class "Eastern high slope" did not have controls in the NV because sites that could be safely sampled were inexistent. Therefore, controls of the most "similar" and geographically closest Zone Type in the quarry (the Southern high slope sites) were assigned to this class.

2.3. Field sampling

As previously mentioned, field sampling occurred in two phases: the first sampling phase, from 18-06-2020 until 03-07-2020, and the second sampling phase, from 19-04-2021 until 30-04-2021.

Following a sampling unit equal to the pixel size in Sentinel-2 images (10 x 10 m resolution), the centre of a parcel of 10 x 10 m was identified in the field using a permanent marker using stakes. Its geographical coordinates were recorded using a cell phone GPS, aided by high-resolution satellite imagery available via the Google Earth application and, whenever possible, using notable points visible both in the field and in the satellite imagery (e.g. larger trees, changes in the rock scarp, etc).

For the shrub layer, sampling was done using the line-intercept transect method with two transects of 10 m, one in the middle of the parcel and another in one of the extremes (total of 20 m of transect by parcel) (Elzinga & Salzer, 1998). A 10 x 20 m "extended" parcel was sampled for the tree layer to record a higher number of trees. Three subplots of 0.5 x 0.5 m (subdivided in 100 squares of 0.05 x 0.05 m) were sampled for the herbaceous layer, and the presence and number of newly emerged seedlings of woody species were recorded and analysed separately (Figure 2.7). The herbaceous subplots were placed in areas with the highest herbaceous cover to obtain the "maximum potential", and the number of squares occupied by individuals of each species was recorded.

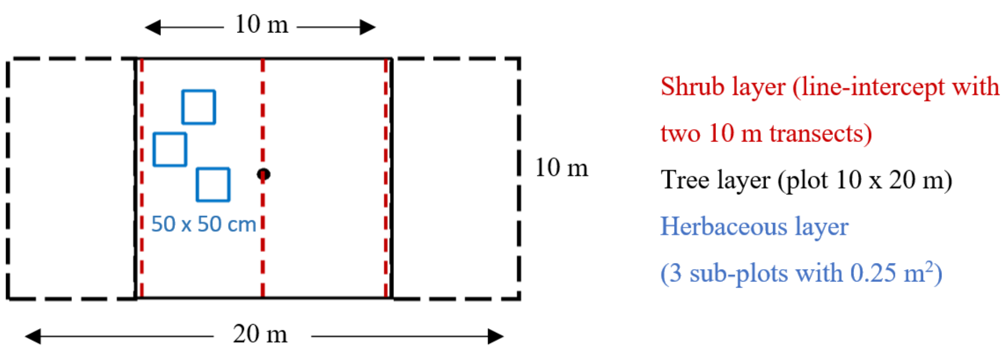


Figure 2.7. Sampling scheme for the shrub, tree and herbaceous layer.

In order to assess vegetation composition, the cover of plant species was recorded for the shrub layer and the number of tree species for the tree layer. The percentage of naked soil cover in the shrub layer was calculated using the sum of the soil length without shrub interception divided by the transect length. Shrub cover was determined as being the complementary measure of naked soil cover. The herbaceous cover was estimated by the sum of the average abundance (from 3 sub-plots) of each herbaceous species in each plot, resulting in a cumulative herbaceous absolute cover.

To analyse vertical ecosystem structure, tree height and circumference were determined using visual estimation and measuring tape, respectively. The diameter at breast height (DBH) was then estimated based on the circumference (c), or perimeter, of the tree using Equation 2.1. Only Aleppo pines with a height larger than 1.80 m and minimum DBH of 5 cm were considered for the tree layer. Other species higher than 4 m (such as *C. siliqua* or *Q. faginea*) were also considered for the tree layer. Furthermore, the height of shrub species was also recorded with a measuring tape.

$$c\ (perimeter) = 2\pi \times r$$
$$DBH = perimeter/\pi$$

Equation 2.1. Equation for diameter calculation at breast height (DBH) (cm). Perimeter corresponds to the circumference (girth) of the tree.

Absolute cover of shrub species was calculated (%) by dividing the sum of the length measured along the transects of a species in a plot by the total transect length, excluding the lengths of rock and road that sometimes intercepted the sampling area. Relative cover of shrub species was calculated by dividing the abundance of each species in a plot (considering the two transects done in each plot) by the sum of the abundances of all species in that plot.

Pine cover was determined firstly by estimating the crown width (CW) of each individual of *Pinus* using Equation 2.2 from Condés & Sterba (2005). Then, the crown area of each individual was calculated with Equation 2.3. The pine projected cover was determined as the percentage of the sum of all crown areas divided by the total plot area (200 m^2). Pine projected cover was then used as the measure of the abundance of *Pinus halepensis* in the abundance matrix of shrub species, and *P. halepensis* relative abundance was calculated by dividing the pine projected cover by the sum of abundances of all woody (shrub and tree) species in each plot (in the same way shrub species' relative cover was calculated).

$$lnCW = a0 + a1\ ln\ d + a2\ ln\ h$$

Equation 2.2. Crown width (CW) (m) equation, where d is the DBH (cm), h the tree height and a0 to a2 are the coefficients. For P. halepensis, a0 = -0.765 , a1 = 0.800 and a2 = -0.111 (Condés & Sterba, 2005).

$$Crown\ area = (\frac{CW}{2})^2 \times \pi$$

Equation 2.3. Crown area (m^2) equation.

Shrub and tree density were calculated using Equation 2.4. Shrub density area was estimated by assuming 1 m for each side of shrub crown (2 m) in two line-intersept 10 m transects (2*10 m*2 m = 40 m^2). Tree density area was 200 m^2 (20 m*10 m).

$$Density\ = \frac{number\ of\ shrubs}{area}$$

Equation 2.4. Density (n/m^2). Area for shrub density: 40 m^2. Area for tree density: 200 m^2.

These methods were used for all plots located in the quarry revegetated areas and in the NV areas. In some of the NV sampling plots, with larger clearings and less dense vegetation (plot reference 5.1VN and 6VN), abundance data were collected twice, first in July 2020 and later in April 2021. In this work and for these specific sampling plots, only the sampled data from April 2021 was used to avoid the non-detection of herbaceous species in July 2020 due to a typically increased dryness during summer periods.

2.4. Soil sampling and characterization

2.4.1. Soil sampling

For soil fertility and decomposition assessment, the soil organic matter (SOM), nutrient content and pH were determined for each plot, with the objective of analysing soil recovery along environmental gradients and restoration age.

Soil sampling was performed in all quarry and NV sampled plots. It occurred on various dates, from 14-06-2021 until 17-06-2021, on 05-07-2021, and from 20-07-2021 until 22-07-2021. First, the litter was removed, and eight sub-samples per plot were collected using 9 cm diameter cores from 0 – 5 cm depth (100 g each) (Figure 2.8). The sub-samples were then added to a plastic bag and mixed, resulting in one composite sample per plot. An effort was made to collect the samples in the inter-patch area (at least 20 cm away from tree or shrub trunks).

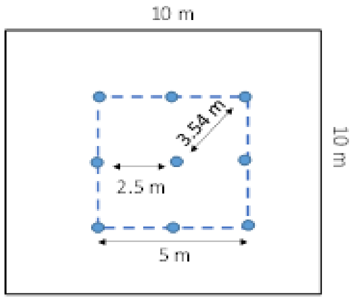


Figure 2.8. Soil sampling scheme in the regular plot of 10 x 10 m.

2.4.2. Soil characterization

2.4.2.1. Organic matter analysis

Soil samples were sieved using a 2 mm mesh and oven-dried for 48h at 60ºC before further processing. Then, SOM was estimated using the weight loss on ignition method (LOI) in a muffle furnace (L3/11, Nabertherm, Germany) (Heiri et al., 2001). This method is based on measuring the weight loss of a dry soil sample (DW) when exposed to high temperatures (dry weight after ignition at 600 ºC, DW600º) for six hours and calculated using Equation 2.5. For each soil sample, three replicates were made using a subsample of circa 10 g of dry soil per crucible. However, the SOM values used for analysis were the ones determined by an external laboratory, as explained in *section 2.4.2.2*.

$$LOI\ (\%) = \frac{DW - DW600ºC}{DW} * 100\%$$

Equation 2.5. LOI equation.

2.4.2.2. Nutrient analysis

Soil analysis of total organic carbon (%) (also referred as soil organic carbon – SOC), SOM (%), available phosphorous (P) using the Olsen method (mg/kg), nitrogen (N) using the Dumas method (mg/kg) and pH was conducted by an external laboratory (AGQ Labs). In order to calculate the carbon-to-nitrogen ratio (C:N), nitrogen values (mg/kg) were converted to a percentage by dividing them by 10000 mg (1 kg = 1000000 mg), and then the ratio was calculated by dividing the carbon percentage by the nitrogen percentage.

The SOM analysis done by the external laboratory was based on the total organic carbon determined using a higher precision method (Leco Carbon Analyzer) than the one described in section 2.4.2.1 (Wang & Anderson, 1998). For that reason, the external laboratory SOM values were used for the analysis. Nonetheless, the SOM values determined by the LOI method had a very strong correlation ($r= 0.74$, $p<0.001$) with the ones determined by the Leco Carbon Analyzer, demonstrating that the LOI method is adequate for SOM analysis of quarry soils (and less expensive).

2.5. Taxonomic diversity

For the taxonomic diversity analysis, the following variables were evaluated for each plot: species richness, species evenness and the Shannon-Wiener diversity index (H'). These were calculated by the diversity function of the “vegan” package version 2.6-2 in R using Equation 2.6 and Equation 2.7 (Oksanen et al., 2017; R Core Team, 2013).

Species richness expresses the total number of species in a site (Jost, 2010). Species evenness is a description of the distribution of abundance across the species in a community, attaining its highest (one) when all species in a sample have the same abundance and approaching zero as relative abundances vary (Alatalo, 1981). Thus, a site with low evenness indicates that one or few species dominate the site (Morris et al., 2014). The Shannon-Wiener diversity index measures the diversity of species in a community, considering the number of species living in a habitat (species richness) and their relative abundance (species evenness) and is strongly influenced by rare species (Shannon, 1948). It is typically found between 1.5 to 3.5, rarely reaching 4.5 (Gaines et al., 1999). Its minimum value is zero, which would tell us that there is no diversity – only one species is found at that site. There is no upper limit to the index since the maximum value occurs when all species have the same abundance (maximum evenness). Thus, a low diversity index value suggests a site with few potential niches where only a few species dominate (low richness and evenness).

$$H' = -\sum_i p_i \log_b p_i$$

Equation 2.6. Shannon-diversity index (H’) formula, where p_i is the proportional abundance of species i and b is the base of the logarithm.

$$J' = \frac{H'}{\log\left(Species\ Richness\right)}$$

Equation 2.7. Evenness (J’) formula, where pi is the proportional abundance of species i.

2.6. Similarity analysis

Pairwise dissimilarity analysis was performed to understand further how similar the restored sites were to the reference ecosystem (NV). The similarity was assessed using the Sørensen-Dice (S_S) and Bray-Curtis (S_{BC}) dissimilarity indices, which quantify the dissimilarity in the species composition between two sites based on the presence/absence (S_S) and abundance (S_{BC}) of species.

The Sørensen–Dice distance was used to compare different sites regarding the presence of species. However, Bray–Curtis distance has been shown to be one of the most effective measures of species dissimilarities. It is recommended for community data since it quantifies dissimilarity between two sites considering species abundances (McCune et al., 2002). Both indices take values from zero to

1, with zero being the most similar to NV and one being the biggest dissimilarity to NV. For visualization purposes, these values were inverted so that zero corresponded to maximum dissimilarity and one to maximum similarity.

The indices were calculated with the beta.pair (S_s) and beta.pair.abund (S_{BC}) functions of the "betapart" package version 1.5.6 in R, using Equation 2.8 (Baselga & Orme, 2012; Team, 2013). The similarity was calculated individually for communities - shrub species; woody (shrub and pine) species and herbaceous species - and for all plant community (woody and herbaceous species).

$$BC_{ii'} = \frac{\sum_{j=1}^{J} |n_{ij} - n_{i'j'}|}{n_i + n_{i'}}$$

Equation 2.8. Bray-Curtis similarity index, where nij= abundance of species j at site i, ni'j'= abundance of species j at site i', ni= number of species at site i and ni'= number of species at site i'. The Bray-Curtis index becomes the Sørensen index when presence/absence data is used instead of abundance data.

Once the similarity between every pair of sites was calculated, the average similarity of each site to the NV was estimated by averaging the similarity values of that site with all its controls.

To understand the pattern of similarity along the chronosequence (with restoration age), three types of analysis and controls were used when considering all plant community,: a) the first analysis was performed by comparing each revegetated site with all the plots in NV altogether, disregarding the Zone type classification, to account for the natural variability that exists in the reference ecosystem(s) (which have a multitude of environmental, topographical and biological characteristics that are variable in space); b) the second analysis was made by comparing each revegetated site with its corresponding Zone Type plots in NV, to understand what if the similarity changed when comparing each site with the NV plots of the same surface exposure and slope classes (as described in section 2.2.2) and c) the third analysis compared each revegetated site with its corresponding Zone Type plots in NV, with the exception of "Northern low slope" (plantations of pine and shrub trees in the limestone benches) which were compared with 3 plots in a natural planted pine forest (considered as an "artificial reference"), to understand if the similarity of these restored sites is higher when compared to a natural planted pine forest or to the reference ecosystem.

Except for the chronosequence analysis, where three types of analysis and controls were used, the similarity values used for the other similarity analysis (figures, models and maps) were obtained by comparing each revegetated site with its corresponding Zone Type plots in NV - analysis of point b). This was done for simplification purposes.

To compare the species composition of the studied sites, an ordering analysis Nonmetric Multidimensional Scaling (NMDS) was made. With the input of an abundance matrix of all species present in all sites, the metaMDS function and the Bray-Curtis coefficient were used to calculate the distance (or similarity) between plots. For these calculations, the "vegan" package in R was used (Oksanen et al., 2017). To determine the proportion of variance explained by the axes, the original data matrices and those resulting from the ordination were correlated using the cor and distance functions of the "ecodist" package (Goslee & Urban, 2007). This was performed by analysing communities individually - shrub species, woody (shrub and pine) species and herbaceous species - and analysing all plant community (woody and herbaceous species).

Vegetation composition and environmental factors were then related using the envfit function from the "vegan" package (Oksanen et al., 2017). Twenty-eight environmental variables were considered and characterized at the plot level, namely: restoration and topographical variables (restoration age, elevation, slope, PSR and the x-coordinate as a proxy for horizontal distance to mean sea level/humidity), variables that characterize the soil (SOM, carbon, nitrogen, phosphorous, C/N ratio as a decomposition proxy and pH) and variables that characterize vegetation structure (shrub cover, pine absolute and relative cover, herbaceous cover, shrub density, tree density, seedling density, shrub height, tree height, and the 10^{th} percentile -PR10- and the 90^{th} percentile -PR90- of shrub and tree height, tree DBH, sum of tree DBH per plot, PR10 and PR90 tree DBH).

2.7. Functional diversity

In order to calculate the structure and functional diversity in each studied site, the cover values of all species found in the field (abundance matrix) and a set of characteristics (functional attributes) considered relevant were used to characterize the recovery process of the degraded areas of SECIL-Outão (Nunes et al., 2014; Perez-Harguindeguy et al., 2013).

Species were classified according to 11 functional traits considered relevant concerning the main limitations to plant establishment and growth and to ecosystem functional recovery in the study area, namely: (i) water shortage, due to climatic constraints and absence of irrigation, (ii) high competition, low soil nutrient availability and limited soil propagules, due to the use of poor quality substrate in plantations after complete soil removal, (iii) lack of dispersers, due to high local fragmentation resulting from the exploitation process and to a low-intensity continuous disturbance from remnant quarry activities, iv) reduced biotic interactions (e.g. with fauna) due to structural (height and cover) homogenization, and v) need for a strong disturbance tolerance, such as high fire resilience, since wildfires are a common disturbance in these Mediterranean dry sub-humid areas (Alday et al., 2012; Clemente et al., 2004; Correia et al., 2001; Šálek, 2012). Thus, the functional traits considered were leaf life cycle, Raunkiaer life form, specific leaf area (SLA), seed mass, height, flowering onset, flowering duration, pollination type, fruit type, dispersal mode and post-fire regeneration strategy. The categories, functional relevance in ecological restoration and units of these functional traits can be seen in Table 2.4 (Cornelissen et al., 2003; Lavorel & Garnier, 2002; Meira-Neto et al., 2011; Nunes et al., 2014; Weiher et al., 1999).

Table 2.4. Functional traits used to characterize the state of the ecosystems in general and, in particular, the recovery process of the degraded areas in the SECIL-Outão quarry.

Trait	Categories	Functional relevance in an ecological restoration	Units
Leaf life cycle	Evergreen sclerophylls / Evergreen conifers / Semi-deciduous / Herbaceous	Drought strategy and adaptation, nutrient cycling, fire resilience, response to disturbances and CO_2, longevity and persistence	
Raunkiaer life form	Chamaephyte / Phanerophyte / Geophyte / Hemicryptophyte / Therophyte	Response to disturbances, edaphic resources, competitiveness, effects in biogeochemical cycles	
Specific leaf area (SLA)	-	Growth rate, photosynthetic rate, nutrient conservation, leaf nitrogen, longevity	mm^2/mg
Seed mass	-	Plant fitness, seed production, competitiveness, tolerance to environmental stress factors, predation, dispersal distance, dormancy, longevity	mg

Trait	Categories	Functional relevance in an ecological restoration	Units
Height	Maximum height measured in the field	Vertical structure and complexity, competitiveness	m for shrubs and trees, cm for herbaceous species
Flowering onset	Number of the month in which flowering starts	Stress avoidance, disturbance avoidance, dispersal strategy	
Flowering duration	Number of flowering months	Response to disturbances, reproductive and dispersal strategy, interaction with other taxa	months
Pollination type	Anemophily (pollination by the wind) / Entomophily (pollination by insects)	Interaction with other taxa, reproductive strategy	
Fruit type	Dry / Fleshy	Interaction with other taxa	
Dispersal mode	Anemochory (seed dispersal by wind) / Barochory (seed dispersal by gravity) / Zoochory (dispersal by animals)	Dispersal ability in a locally fragmented area, lacking soil propagules and dispersers, dispersal distance, response to disturbances, reproductive strategy, interaction with other taxa	
Post-fire regeneration strategy	Resprouter / Seeder / Both / None	Response to fire disturbances and reproductive strategy	

All species were characterized in terms of functional characteristics of interest using open access databases, deepening and extending previous diversity studies in this region. Trait values were taken primarily from BROT 2.0, a reference plant trait database for Mediterranean Basin species (Tavşanoğlu & Pausas, 2018). Missing data was then completed by using other databases such as the TRY database (a global database of plant traits) and the LEDA Traitbase (a database of life-history traits of the Northwest European flora) (Kattge et al., 2011; Kleyer et al., 2008). For the sake of simplicity, I refer to the reproduction period of *P. halepensis* as the flowering duration, even though gymnosperms have no real flowers.

2.7.1. Calculation of functional structure and functional diversity indices

Based on the abundance of the various species and their functional traits (Table 2.4), indexes of functional structure and diversity were calculated for each sampled plot, considering all attributes (functional traits) together (multi-trait analysis). These were then related to the functioning and resilience variables of the plant communities in the recovered and natural areas.

The functional structure index used was the "weighted average of the community" (Community Weighted Mean – CWM), defined as "the mean trait value of species weighted by the species abundances". For categorical traits, the CWM represents the relative level of dominance of each of the considered categories of each trait. It was calculated for each functional trait using the function functcomp of the "FD" package in R (Botta-Dukát, 2005).

Functional diversity was assessed by calculating functional richness (FRic) and functional dispersion (FDis). Both FRic and FDis measure the degree of functional dissimilarity within a community, reflecting the complementarity in the use of resources by the species that compose it (Laliberté & Legendre, 2010; Villéger et al., 2008). FRic is defined as "the convex hull volume of the individual species in multidimensional trait space" and does not consider species abundance. For a single continuous trait (e.g. plant height), it is the trait range (difference between the maximum and minimum value in each community). For a categorical trait (e.g. Raunkiaer life form), it is the number of different functional categories in each community. FDis, on the other hand, considers species abundance and is defined as “the weighted mean distance in multidimensional traitspace of individual species to the centroid of all species”. FDis, as FRic, is a measure of the dispersion of species in a functional space. However, FDis is independent of species richness, ensuring that the number of species does not influence the response of functional diversity to environmental gradients (Almeida et al., 2018). Thus, FDis was used as the primary indicator of functional diversity because it is an intuitive measure of functional diversity that accounts for the volume of occupied functional space and the distribution of species within it. In modelling, in cases where there was no significant relationship between FDis and predictor variables, FRic was used to see if significant relationships could be found.

The dbFD function of the “FD” package (Laliberté et al. 2014) in R was used for calculating FRic and FDis after standardising all numerical traits by applying log(x+2). Standardising was done so that each trait had the same weight in functional diversity estimation, and the units used to measure traits had no influence (Villéger et al., 2008). Analyses were done separately for the shrub community, woody community (shrubs and pines) and herbaceous community. The functional trait “Leaf life cycle” was excluded from the herbaceous community multi-trait analysis because it only has one category (Herbaceous) in this community.

2.8. Extraction and processing of remote sensing indices

Multiple spectral indices were used to spatially analyse the different revegetated and NV areas and later map relevant ecological variables using models based on field and RS data. In total, 31 indices were selected based on their relevance, resulting in the extraction of 21 vegetation radiometric indices, five biophysical indices, one water radiometric index and four soil radiometric indices, listed and defined in Table S1.1.

The indices were extracted using RS data, namely multispectral images from the Sentinel-2 satellite. Its multispectral sensor acquires information in 13 bands of the electromagnetic spectrum, from visible to far infrared, and its images are available with high spatial (10-20m) and temporal (3-5 days) resolution (E.S.A., 2015). Data was extracted from Sentinel open-access hub (https://scihub.copernicus.eu/) and processed with the software Sentinel Application Platform (SNAP) version 8.0.

Three Sentinel-2B satellite images from 2020-05-24, 2020-06-23 and 2020-07-03 were acquired. These images are Level-2A products with bottom of reflectance (BOA) values and therefore are atmospherically corrected (ESA, n.d.). The three dates were chosen due to their proximity to the sampling dates and compared to select the date in which indices had higher correlations with field data. The correlations were performed in the R software, as described in *section 2.10.* The selected date that showed the higher correlations with field data was 2020-07-03.

The NDVI was the most used of all the RS indices analysed. This index detects the sign of green vegetation, even in areas with low cover, but reveals some saturation in places with higher vegetation density or with high brightness from the ground (Pettorelli et al., 2018). Several vegetation indices have emerged to solve this problem; however, these indices have other limitations, such as the underestimation of the actual reflectance values, by reducing ground noise at the cost of decreasing the dynamic range of the indices (Xue & Su, 2017). Such indices, such as SAVI, TSAVI, and MSAVI, are slightly less sensitive to changes in vegetation cover than NDVI, especially at low cover levels, and more sensitive to atmospheric variations than NDVI (Qi et al., 1994). In the quarry, the strongest and most evident relationships with field variables were mainly obtained using the NDVI, while indices less sensitive to the reflectance signal were less effective in predicting, modelling, and mapping variables measured in the field. Other vegetation indices, such as the ARVI, with a range of values similar to the NDVI (and four times more resistant to atmospheric effects), also had significantly high correlations with the key ecological attributes studied (Kaufman & Tanre, 1992).

Furthermore, several biophysical indices (FAPAR, LAI, CWC, etc.) were calculated based on the reflectance at the top of the canopy, obtained from a set of multispectral images with different spatial and spectral resolutions, which include visible and infrared (near, medium and far) bandwidths. These indices were calculated using SNAP software machine learning algorithms (Weiss et al., 2016). FAPAR (Fraction of Photosynthetically Active Radiation) is a biophysical index directly related to primary productivity and photosynthesis. It corresponds to the fraction of photosynthetically active radiation (400-700 nm) absorbed by the forest canopy's green leaves and, therefore, reflects the ability of green vegetation to absorb energy. It depends on vegetation structure, optical properties and lighting conditions and includes only the green parts of plants with high chlorophyll content (Weiss et al., 2016). In addition to being a key variable in models of primary productivity and CO_2 assimilation in vegetation, such as gross and net primary production, to which it is directly proportional (Monteith, 1977; Ruimy et al., 1994), FAPAR also serves as an indicator of the evolution of vegetation cover and its health status (Bojinski et al., 2014; Fensholt et al., 2004). It is recognized as an Essential Climate Variable (ECV) by the Global Climate Observing System (GCOS) and considered one of the most fundamental variables for studying the Earth system in the context of global changes. Here, maximum FAPAR values were used as a proxy for the maximum potential of primary productivity.

Soil indices CI and BI2, calculated based on reflectance in the green, red and infrared bandwidth, also proved helpful due to their inverse relationship with vegetation cover.

2.9. Cartography and mapping carbon sequestration

The method used to determine the spatial distribution and approximate magnitude of carbon storage and sequestration was the InVEST 3.10.2 (Integrated Valuation of Environmental Services and Tradeoffs) model, developed by the Natural Capital Project (Nelson et al., 2009; Tallis et al., 2010). The InVEST model is a spatially explicit modelling tool that can evaluate the impact of different land uses on multiple ES (Polasky et al., 2011). This program was designed to support decision-making in the management of natural resources, making it possible to examine the various services as a whole in order to find effective solutions in an integrated manner and allow to decide where and how to invest in natural capital to ensure the preservation of protected areas (Sharp et al., 2014).

Carbon storage largely depends on the sizes of four carbon pools: aboveground biomass, belowground biomass, soil, and dead organic matter. Aboveground biomass comprises all living plant material above the soil (e.g. bark, trunks, branches, leaves). Belowground biomass encompasses the living root systems of aboveground biomass. SOM is the organic component of soil and represents the largest terrestrial carbon pool. Dead organic matter includes litter and lying and standing dead wood (Fahey et al., 2010). Using maps of LULC classes and a table with the amount of carbon stored in carbon pools of each LULC class, InVEST estimates the net amount of carbon stored in a land parcel at a specific period but also over time when given a second past or future map of LULC classes (Sharp et al., 2014).

Limitations of the model include an oversimplified carbon cycle and an assumed linear change in carbon sequestration over time. The model infers carbon sequestration based on the LULC type, and in this way, LULC changes over time can be used to infer carbon sequestration rates. The model assumes that none of the LULC types in the landscape is gaining or losing carbon over time. Instead, it is assumed that all LULC types are at some fixed storage level equal to the average of measured storage levels within that LULC type. Under this assumption, the only changes in carbon storage over time are due to changes from one LULC type to another. Therefore, any pixel that does not change its LULC type will have a sequestration value of 0 over time, when in reality, many areas are recovering from past land use or are undergoing natural succession. This problem can be addressed by dividing LULC types into age classes (essentially adding more LULC types), such as different forest ages. Then, parcels can move from one age class to another in scenarios and change their carbon storage values. However, no carbon values were found for the LULC types present in the quarry at different restoration ages. Therefore, this limitation could not be overcome. Additionally, although carbon storage differs among LULC types (e.g. conifer forest vs. Maquis/sclerophyll vegetation), there can often be significant variation within a LULC type since factors such as temperature, elevation, rainfall, and the number of years since a major disturbance affect carbon storage which is not considered in this model (Sharp et al., 2014). Biophysical conditions important for carbon sequestration, such as photosynthesis rates and the presence of active soil organisms, are also not included in the model (Sharp et al., 2014).

Since the model relies on carbon storage estimates for each LULC type, the results are only as detailed and reliable as the LULC classification used and carbon pool values supplied, and for this reason, only carbon pool values from similar climate and vegetation found in the literature were considered.

To produce the LULC maps for the model data inputs, manual cartography of the quarry was performed in ArcGIS based on an orthophoto (resolution of 12 cm/px) from SECIL and field knowledge. Due to the difficulty in finding studies in a Mediterranean climate with carbon values for mixed classes (such as coniferous and shrub vegetation or herbaceous and shrub vegetation), the land cover type assigned to each polygon in the new LULC map was based on the dominant land cover type, which resulted in four main land cover types: 1) Coniferous vegetation; 2) Herbaceous vegetation; 3) Sclerophyll vegetation and 4) Natural material surfaces, roads and escarpments.

Land cover maps of the quarry from 1980 and 2020 were produced to estimate carbon sequestration since revegetation started (resulting map is shown in Figure 3.6). The maps in raster format were obtained by transforming the LULC polygons from vector files (shapefiles) to raster, or matrix format, using ArcGIS tools. The final raster cell size chosen was 5m x 5m. The map for 1980 had all the quarry areas classified as “Natural material surfaces, roads and escarpments” since ecological restoration by revegetation interventions only began in 1983.

Carbon pool values were obtained through bibliographic research and are presented in Table 2.5. They correspond to land cover types and vegetation similar to those in the study area. It is relevant to mention that there are no specific studies on the values of carbon reservoirs for the vegetation of Serra da Arrábida. Therefore, carbon pool values were obtained from existing studies with the most similar characteristics to the Mediterranean climate. No carbon density values for dead matter were found in the literature for these vegetation types.

Table 2.5. Bibliographical review of carbon pool data for the vegetation types present in Arrábida Natural Park (Mg/ha). Notes: [a]*Values 0-15 cm of soil,* [b] *Values of 0-30 cm of soil,* [c] *Values 0-100 cm of soil.*

Authors	**Localization**	**Vegetation type or species**	**Carbon in aboveground biomass**	**Carbon in belowground biomass**	**Carbon in soil**	**Carbon in dead organic matter**
			Mg/ha	Mg/ha	Mg/ha	Mg/ha
Almagro et al., 2010	Spain (Mediterranean ecosystem)	Coniferous vegetation	32.938 (20.726)	3.054 (0.429)	51,893 (1.549)[a]	
		Typical mediterranean scrubland	3.513 (0.4964)	2.529 (0.429)	30.6 (4.41)[a]	
		Olive groves	8.303 (0.521)	1.622 (0.438)		
Canaveira et al., 2013	Portugal	Pastures	0.53	0.94		
		Pinus pinaster	26.74	3.14		
		Pinus pinea	18.79	1.46		
		Other conifers	14.51	1.76		
		Scrubland	8.78	4.94		
		Olive groves	7.85	1.15		
		Quercus spp.	15.87	4.69		
Nunes et al., 2013	Vila Real, Portugal	Pines (2008)	78.4 (11.4)			
		Pines (2009)	83.4 (11.9)			
Eaton et al., 2008	Ireland	Herbaceous vegetation			160[c]	
Muñoz-Rojas et al., 2011	Anadaluzia	Herbaceous vegetation	3.4			
Simões et al., 2012	Évora, Portugal (Mediterranean scrublands)	*Cistus salviifolius*	9.1 (0.63)	9.73 (1.97)		
		Cistus ladanifer	15.74 (0.85)	13.56 (0.23)		
Ruiz-Peinado et al., 2013	Western Spain (Iberian dehesas)	*Cistus ladanifer*	8.551			
Nieto et al.,	Spain	Scrubland native vegetation			111.6 (40.8)[b]	
Davies et al., 2011	England	Herbaceous vegetation	1.4 (0.1)			

As for the Herbaceous vegetation class, aboveground carbon values were determined by averaging the values of herbaceous vegetation and pastures (Canaveira et al., 2013; Davies et al., 2011), and carbon belowground was calculated using Equation 2.9 (Canaveira et al., 2013; Muñoz-Rojas et al., 2011). The value for soil carbon of herbaceous plants was estimated by Eaton et al. (2008).

$$\text{Carbon belowground}_{\text{Herbaceous vegetation}} = \frac{\text{Carbon belowground}_{\text{Canaveira et al.,2013}} + (\text{Carbon aboveground and belowground}_{\text{Muñoz-Rojas et al,2011}} - \text{Carbon aboveground}_{\text{Herbaceous vegetation}})}{2}$$

Equation 2.9. Carbon belowground formula for herbaceous vegetation.

Since the delimited forests are predominantly coniferous (planted pine forests), the average values associated with these species are assumed for the Coniferous vegetation class due to absence of Aleppo pine values found in the literature (Almagro et al., 2010; Canaveira et al., 2013; Nunes et al., 2013).

For the sclerophyllous vegetation, the calculation was performed by the average values of scrubland, olive groves, *Quercus* spp. and *Cistus* spp. (Almagro et al., 2010; Canaveira et al., 2013;

Ruiz-Peinado et al., 2013; Simões et al., 2012). The values used for carbon in soil were calculated according to Nieto et al. (2013).

Finally, for areas where there is no vegetation (natural material surfaces, roads and escarpments), the authors are unanimous in assigning zero to the value of sequestered carbon because vegetation captures carbon through the photosynthesis process and because in these areas there is no vegetation, the capture is null.

The final carbon pool table based on species-specific carbon pool data (Table 2.5) is presented in Table 2.6.

Table 2.6. Carbon pool (carbon density) values for each land cover type (Mg/ha).

Land Cover Type	**Carbon aboveground**	**Carbon belowground**	**Carbon in soil**	**Carbon in dead matter**	**Total Carbon**
Coniferous vegetation	42.46	5.29	250	0	297.75
Herbaceous vegetation	0.97	1.69	160	0	162.66
Scrubland and sclerophyllous vegetation	9.71	5.46	111.6	0	126.77
Natural material surfaces, roads and escarpments	0	0	0	0	0

Total carbon sequestration in the quarry revegetated areas was estimated using the Zonal Statistics tool that calculated the sum of the total carbon sequestration values of all the pixels inside the quarry area. Furthermore, this tool was also used for general statistics regarding areas, land cover types and carbon values. It is important to mention that carbon values were underestimated due to the absence of values for carbon density in dead matter and because only large revegetated areas or polygons were defined and characterized, which disregards the existence of vegetation in escarpments and tree crowns that intersect the roads in the quarry. Nonetheless, some average carbon values in the quarry areas are overestimated due to the dominant land cover type assigned in the cartography, especially in the quarry's limestone benches, where pine trees dominate, which show the highest carbon values. Since scrubland and sclerophyllous vegetation are also present, actual average carbon values should be lower.

2.10. Data analysis

Given the complexity of the multiple factors affecting the recovery of the restored areas, the analysis and statistical modelling of the main variables of interest (ecological indicators in Table 2.7) took into account four main objectives: 1) the modelling of the variables of interest after restoration, to observe their recovery trajectory in time based on the chronosequence (section 2.10.1); 2) comparison of the values of the variables of interest in the quarry restored areas and the NV's reference areas (section 2.10.1); 3) the identification of the main predictors of each variable of interest, to identify limiting factors to consider in the recommendations for the adaptive management of the areas in recovery (section 2.10.2); 4) spatialization and mapping of the variables of interest to the entire SECIL quarry's area (section 2.10.3), through models based on continuous variables in space obtained by remote sensing, to support large-scale adaptive management in the restoration of the quarry areas. To analyse the relationship between the multiple variables studied, generalized linear models were built for all response variables of interest. The complete list of the variables and predictors used in modelling and their description, units and sources can be seen in Table S1.2. Predictors used in the modelling of the variables of interest are: restoration age (years after the end of the last revegetation intervention); biological data (cover and height of the tree, shrub and herbaceous strata); edaphic variables (SOM, nitrogen and phosphorous in soil, pH, CN ratio); environmental and topographical variables (slope, elevation, PSR, humidity, longitude, latitude); categorical variables "intervention mode" and "zone type" (described in *sections 2.2 and 2.3*, respectively), and the RS variables (vegetation, soil and biophysical indices) obtained through satellite data and the ratios between those indices in different months (Table S1.3). The complete list of the final predictors selected in the models is shown in Table S1.4.

Firstly, to identify the predictors with the strongest correlations with the variables of interest, all variables and potential predictors were correlated using the Spearman's rank correlation in R (function cor from the "stats" package). Then, those variables were used in general linear models, and a final explanatory model was selected based on the statistical significance of the predictor variables and the parsimony principle (a model achieves a desired level of goodness of fit using as few explanatory variables as possible) (Coelho et al., 2019). This was done using the glm function from the "stats" package, which fits generalized linear models, and the family used was "gaussian" (R Core Team, 2013). The predictor variables were chosen by manual backward selection, which begins with a model containing all variables under consideration and then removes the least significant variables one after the other, until all the remaining variables are statistically significant, and the best explanatory and simpler model is found. For this, the anova function from the "stats" package was used to test whether the model terms were significant and then remove the variables with the higher (less significant) p-values, one at a time.

All analyses were performed in R 4.1.3 (R Core Team, 2013). Unless otherwise stated, all tests were run with a significance level of $P \leq 0.05$. For every model, the assumptions of multiple linear regression (linearity, no/little multicollinearity, independence, homoscedasticity and normality) were inspected through graphical analysis and using the Shapiro-Wilk Normality Test (function shapiro.test from "stats" package) (Garson, 2012). Whenever necessary, the dependent and occasionally independent variables were transformed to fulfil the assumptions of multiple linear regression. To evaluate the predictive power of each model, pseudo R^2 was used as a measure of goodness of fit and calculated as the difference between null deviance and residual deviance divided by the null deviance (Zuur et al., 2009). The levels of statistical significance mentioned in the description of the results for each predictor refer to the p-value of the predictors' coefficients in the models in case they are numerical or the anova function test if predictor variables are categorical. Due to a large number of functional diversity and functional trait variables, only the variables with higher correlations with spatial variables (e.g. RS indices and slope) were chosen to be modelled, with the main goal of producing statistically relevant models and create maps.

<u>Limitations of the models and maps</u>

Since models were produced with data from the quarry revegetated plots, maps are only shown for the quarry, excluding NV areas. Such is because the topography, variables, and revegetation interventions in the quarry differ considerably from the natural areas, with the latter having no interventions nor significant human impact. Thus, producing spatial models that efficiently estimate values in both quarry revegetated areas and the natural ecosystem would be problematic since these are two different ecosystems with different vegetation types (planted species in quarry vs. native species in natural areas). Ultimately, the main objective of this research was to study quarry restoration success.

Therefore, NV plots only served as reference ecosystem plots to compare the values of key ecological attributes concerning restoration success (primary productivity, carbon sequestration, soil fertility, and decomposition, habitat quality, similarity to NV, biodiversity, and ecosystem functional traits) between the quarry and its natural reference.

Hence, the models and maps produced are only valid for the revegetated areas, as those were the only quarry areas sampled, and therefore, only revegetated areas inside the quarry can be estimated by maps through interpolation (predicting values inside of a range of data points), instead of extrapolation (which refers to predicting values that are outside of a range of data points). Likewise, areas occupied by natural mineral surfaces (limestone and marl), roads, and escarpments were omitted from the maps to remove noise for the reader's sake. The shapefile corresponding to “natural mineral surfaces, roads, and escarpments” was obtained from the cartography of the quarry (explained in *section 2.9*).

2.10.1. Models of succession with restoration age

The modelling of the succession after restoration aimed to evaluate how and how much the restored ecosystem recovered. This was accomplished by analysing the ecosystem’s trajectory in time based on the chronosequence, creating models with restoration age as the only predictor, and producing graphs using the “ggplot2” package (Wickham, 2016). Furthermore, to compare the values between the different intervention modes in the restored areas and the reference ecosystem (NV), boxplots were produced using the “ggplot2” package. To see if these values were statistically different between the different intervention modes and the NV, a one-way analysis of variance (ANOVA) test was performed using the aov function from the “stats” package. When statistical differences were observed, a post-hoc Tukey HSD (Honestly Significant Difference) test was used (TukeyHSD function from the “stats” package) to compare group means and identify which groups are statistically different from each other ($p \leq 0.05$) (R Core Team, 2013). Letters were added to the boxplots based on the Tukey test. Means sharing the same letter are not significantly different, and the highest mean receives the letter “a”.

2.10.2. Models based on all predictors

The modelling with all the predictors aimed at selecting the most relevant variables to explain each variable of interest, identify the limiting factors or promoters of recovery and take them into account in future recommendations for the adaptive management of areas in recovery.

For the construction of models with restoration age, biological, environmental, and topographical variables, standardized predictors were used (ranging from -1 to 1). This was done to ensure that differences in the order of magnitude of the units in which each variable is measured (in different scales) did not influence the determination of its importance as a predictor of a given response variable when analysed in sets with the other predictors.

In models for which a significant interaction between two or more independent variables was detected, the nature of the interaction was described in the text (positive or negative). The coefficients of interactions between numerical variables were included in the model formulas of Table S1.5.

2.10.3. Models for upscaling and mapping at a landscape scale

As a tool to support recovery management, models were built to upscale variables and provide maps of various variables of interest spatial variation (in the SECIL quarry area). The mapping was done through models and based on spatially continuous variables such as RS indices, topographical variables (slope, elevation) and latitude, that estimated field-measured variables (usually field-intensive and time-consuming to sample). This involved extensive testing of 31 indices to find the best RS predictors.

Indices NDVI, CI and TSAVI were chosen due to being RS variables with higher correlations with field-measured variables than other indices. Additionally, ratios of NDVI, CI, and TSAVI between different months were calculated to identify potential differences in biomass peaks that could be better indicators than indices from one date alone. NDVI ratios showed the highest correlation with the field variables and were the only ratios used in the models.

In Mediterranean ecosystems, ephemeral herbaceous species appear as an understory layer in forests shortly after the beginning of the rainy season (October–November), reaching a biomass peak at the end of winter (February) and drying out in spring (April), while the evergreen woody vegetation (shrubs and trees) become most active from early spring (March), developing new leaves towards the dry season (June–August). Since herbaceous vegetation is absent or completely dry in the dry season in Mediterranean forests, the mean NDVI in the dry season can be attributed to the woody vegetation. In contrast, in the wet season (September-April), changes in NDVI can be attributed to the development of ephemeral herbaceous vegetation on the forest floor and its maximum value to the peak green cover (Helman et al., 2015).

Thus, NDVI ratios were produced to capture the absence/presence of the different vegetation layers, since NDVI ratios exhibit distinct patterns in relation to ecological variables. In areas with higher pine or shrub cover, the ratio tends to be small due to minimal changes in woody vegetation over the months, contrasting with the noticeable seasonal shifts in herbaceous vegetation. For example, smaller ratios between NDVI February and NDVI July are associated with higher pine cover, as pine canopy masks the signal of the herbaceous layer, resulting in a lower difference in NDVI between February and July. Conversely, areas with lower pine cover exhibit higher NDVI ratios due to greater detectability of herbaceous vegetation (drier in July) and its changing reflectance. In this case, a positive relationship between herbaceous cover and the $\frac{NDVI\ February}{NDVI\ July}$ ratio is observed.

The months used for the ratios were January, February, May, June, July, August, September, October, November and December, and the Sentinel-2 images for these months, in the year of 2020, were downloaded and processed as described *in section 2.8.*

To model and map ecological variables, the RS data (vegetation, soil and biophysical indices, and ratios of indices in different months) produced from Sentinel-2 images was extracted to the specific area of the field sampled plots, using ArcGIS. To do this, a shapefile of the sampling plots (polygons with 10 x 20 m) was obtained from the overlap of sampling plots coordinates and SECIL orthophoto (coordinate system ETRS_1989_Portugal_TM06). Since the coordinate system in Sentinel-2 images is WGS_1984_UTM_Zone_29N, the ETRS_1989 sampling plots shapefile was projected into WGS_1984. Then, zonal statistics was applied to each plot to extract the mean, minimum, maximum, standard deviation and range values of each RS index, by considering the pixels overlapping each polygon of the field sampled plots.

The correlation between these values and the parameters measured in the field was tested. Then, RS and topographical data were used to create generalized linear models able to estimate and map the field-measured variables of interest. All the NDVI ratios used to produce the models were highly correlated. The NDVI ratio variable in each spatially distributed model was chosen based on the ratio that resulted in the model with the highest R^2.

For these models, no standardization was done so that the produced maps would reflect the actual values of variation of each predictor and, in this way, obtain y-values from the dependent variable, which are within its normal range. Maps were only produced based on models with an $R^2 > 0.30$ because it was considered that models with lower R^2 values would not be sufficient to explain a dependent variable or have the desired confidence to do so (due to high variability around the regression line).

Finally, ArcGIS was used to produce maps using the Raster Calculator tool by applying the expressions determined from the multivariate statistical models. The quarry boundary shown in the maps with a red line is the official quarry boundary provided by SECIL. Some vegetated quarry areas in the maps were not sampled because they are not restored areas under ecological restoration. They are represented in the maps because they fall within the quarry's official boundary; however, they are not subject to analysis or interpretation in this study. These areas are located in patches surrounding the natural mineral surface areas in the Southeastern and Northern/Northeastern parts of the quarry.

2.11. Summary of the ecosystem services, key attributes and ecological indicators studied

Two tables are shown below summarising the ecological attributes evaluated in this study, which were selected and classified according to bibliographic information and the specific objectives, conditions and challenges of the SECIL-Outão quarry's recovery. Table 2.7 classifies the ecological attributes and their indicators into ES, while Table 2.8 classifies them according to the key ecosystem attributes established by SER for ecological restoration assessment.

Table 2.7. Ecosystem services according to CICES V5.1 classification system and their indicators, used to evaluate ecological restoration in this study (Haines-Young & Potschin, 2018).

<table>
<tr><th>Ecosystem service division in CICES</th><th>Ecosystem service group in CICES</th><th>Ecological attribute in the “Results” section</th><th>Ecological indicators studied</th><th></th></tr>
<tr><td rowspan="2">Regulation of physicochemical environment</td><td>Climate regulation</td><td>Productivity and Carbon Sequestration</td><td>• Primary productivity: FAPAR
• Carbon sequestration:
- Carbon in soil
- Maps InVEST: total carbon (aboveground, belowground and soil)</td><td></td></tr>
<tr><td>Soil formation and composition</td><td>Soil Fertility and Decomposition</td><td>• Fertility:
- Organic matter
- Nitrogen
- Phosphorous
- pH
• Decomposition: C:N ratio</td><td></td></tr>
<tr><td rowspan="5">Regulation of biotic environment</td><td rowspan="5">Lifecycle maintenance, habitat and gene pool protection</td><td>Habitat Quality</td><td>• Pine, shrub and herbaceous cover
• Tree, woody (CWM) and herbaceous (CWM) height</td><td></td></tr>
<tr><td>Pollination</td><td>• Entomophilous pollination by the shrubs and herbaceous community
• Flower duration in the shrub community</td><td rowspan="4">Ecosystem functioning (CWM of functional traits)</td></tr>
<tr><td>Dispersal Capacity, Interaction with Fauna and Regeneration</td><td>• Dispersal capacity: different types of seed dispersal
• Interaction with fauna: fruit type
• Regeneration: seedling density</td></tr>
<tr><td>Resilience and Adaptation to Fire</td><td>• Resprouters in the woody community
• Resprouters in the shrub community</td></tr>
<tr><td>Resilience and Adaptation to Drought</td><td>• SLA in the woody community
• Sclerophyllous vegetation in the woody community</td></tr>
<tr><td colspan="2" rowspan="2">Not recognized as ecosystem services but are essential attributes to study and compare to the reference ecosystem.

Diversity (taxonomic or functional) is necessary for the production of all other services</td><td>Similarity to NV</td><td>• Similarity indexes of species presence (Sørensen–Dice) and abundances (Bray-Curtis) for all plant community, and the shrub community individually
• NMDS graphs</td><td></td></tr>
<tr><td>Diversity</td><td>• Taxonomic diversity:
- Shannon index in the shrub, woody, herbaceous and all plant community
• Functional diversity:
- Functional dispersion (FDis) (multi-trait) in the shrub, woody, and herbaceous community
- Functional richness (FRic) (multi-trait) in the woody community</td><td></td></tr>
</table>

Table 2.8. SER key ecosystem attributes and their indicators, used to evaluate ecological restoration in this study (Gann et al., 2019).

SER key ecosystem attributes	**Ecological attribute in the "Results" section**	**Ecological indicators studied**	
Ecosystem function	Productivity and Carbon Sequestration	• Primary productivity: FAPAR • Carbon sequestration: - Carbon in soil - Maps InVEST: total carbon (aboveground, belowground and soil)	
	Pollination	• Entomophilous pollination by the shrubs and herbaceous community • Flower duration in the shrub community	Ecosystem functioning (CWM of functional traits)
	Dispersal Capacity, Interaction with Fauna and Regeneration	• Dispersal capacity: different types of seed dispersal • Interaction with fauna: fruit type • Regeneration: seedling density	
	Resilience and Adaptation to Fire	• Resprouters in the woody community • Resprouters in the shrub community	
	Resilience and Adaptation to Drought	• SLA in the woody community • Sclerophyllous vegetation in the woody community	
	Diversity (Functional)	• Functional dispersion (FDis) (multi-trait) in the shrub, woody and herbaceous community • Functional richness (FRic) (multi-trait) in the woody community	
	Soil Decomposition	• C:N ratio	
Physical conditions	Soil Fertility	• Fertility: - Organic matter - Nitrogen - Phosphorous - pH	
Structural diversity	Habitat Quality	• Pine, shrub, and herbaceous cover • Tree, woody (CWM) and herbaceous (CWM) height	
	Diversity (Taxonomic)	• Taxonomic diversity: - Shannon index in the shrub, woody, herbaceous and all plant community	
Species composition	Similarity to NV	• Similarity indexes of species presence (Sørensen–Dice) and abundances (Bray-Curtis) for all plant community, and the shrub community individually • NMDS graphs	

2.12. Ecological Restoration Success Index

The degree of recovery success (or ecological restoration success) was assessed by directly comparing the quarry restored sites with the reference ecosystem. The recovery completeness, which is defined as the degree to which a metric type measured in the restored site reaches the reference level (Meli et al., 2017), was determined considering the main ecological attribute groups considered in the results: 1) Productivity and Carbon Sequestration, 2) Soil Fertility and Decomposition, 3) Habitat Quality, 4) Similarity to NV, 5) Diversity, 6) Pollination, 7) Resilience and Adaptation to Fire, 8) Resilience and Adaptation to Drought and 9) Dispersal capacity, Interaction with Fauna and Regeneration.

As such, the evaluation of the overall recovery of each ecological attribute group was performed by selecting relevant ecological indicators of each group and then using transformed response ratios (RR) as the standardized mean effect size to estimate recovery completeness, a methodology generally used in the context of ecological restoration (Benayas et al., 2009; Crouzeilles et al., 2016; Meli et al., 2017).

As shown in Equation 2.10, RR values tend toward zero as $X_{restored}$ increases to a value approaching $X_{reference}$. Thus, values close to zero indicate a high degree or full recovery for the indicator under analysis, positive values indicate higher values in the restored sites than in the reference ecosystem, while negative RR values indicate incomplete recovery (restored sites with values lower than the reference ecosystem) (Meli et al., 2017). Whereas increases in most response variables indicate improvement, increases in others indicate degradation. For example, an increase in the abundance or richness of non-native species such as the Aleppo pine implies reductions in biodiversity. For interpretation purposes, the RR is referred to as “recovery completeness” in the results section.

$$RR = \ln[(X_{restored} + 0.001) \;/\; (X_{reference} + 0.001)]$$

Equation 2.10. Response ratio (RR) equation where $X_{restored}$ represents the mean values of restored quarry sites (current condition) and $X_{reference}$ represents the mean values of the reference ecosystem sites (ideal and undisturbed condition) (Meli et al., 2017).

The recovery success analysis was done assuming two typologies: (i) intervention mode, to study the effects of the different revegetation interventions in the quarry restored areas when compared to the NV reference, and (ii) zone type, to understand as a whole how different exploration methods led to different results and to analyse in a geographical perspective which areas of the quarry (based on surface exposure and slope) are more similar to NV.

CHAPTER 3 | RESULTS

Overall, 28 variables were modelled through three types of models: 1) models based on restoration age, 2) models based on all predictors (topographical, environmental, and biological variables) except RS indices, and 3) models based on spatially distributed predictors (topographical variables and RS indices) to produce maps. The models' formulas and respective R^2 are shown in Table S1.5. Variables were mapped using statistically significant models with an $R^2 > 0.3$. Below this threshold, models were not considered sufficiently strong to predict data and, therefore, maps were not produced. In general, the spatially distributed predictors that most successfully improved the models used for mapping were: slope, soil indices BI and BI2, and a wide range of ratios of NDVI values in different months, which allow capturing the presence or absence of the different vegetation layers. Statistically significant models could not be generated for two variables (Bray-Curtis similarity in the shrub community and Resprouters in the shrub community), in none of the three types of models.

3.1. Primary Productivity and Carbon Sequestration

3.1.1. Primary Productivity

The primary productivity indicator, FAPAR, significantly increased along the chronosequence, with a tendency towards a productivity peak around 15 years after restoration that stabilised in the next 15 years (Figure 3.1).

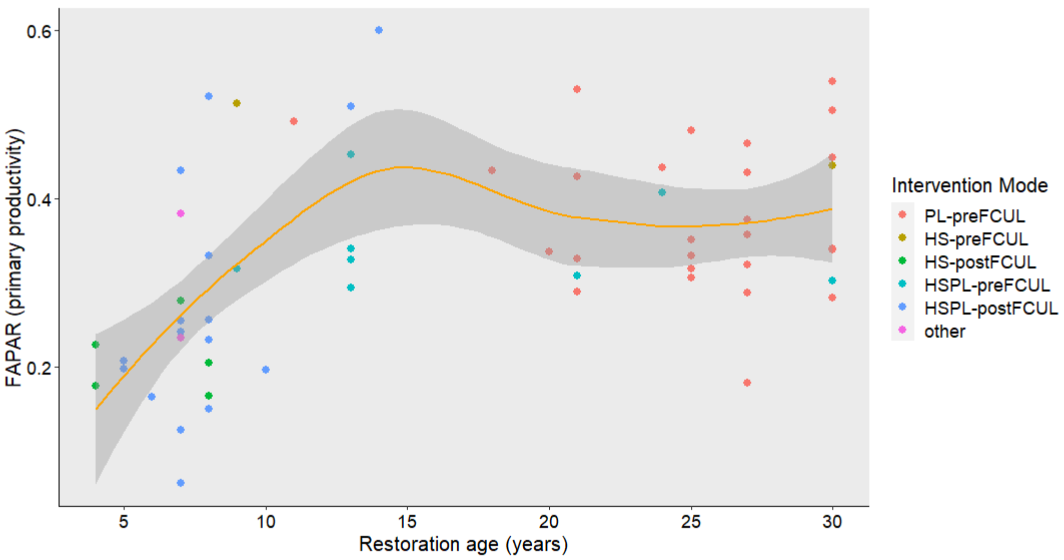


Figure 3.1. Variation of the biophysical variable FAPAR, used as the primary productivity indicator, with restoration age in quarry restored plots (R^2= 0.32, p<0.01).

FAPAR values differed in the intervention modes, with plantations (PL-preFCUL) having significantly higher values than hydroseeded sites (HS-postFCUL and HSPL-postFCUL) (Figure 3.2). Yet, the reference ecosystem (NV) had significantly higher values than the intervention modes, indicating higher productivity in the NV than in the restored areas.

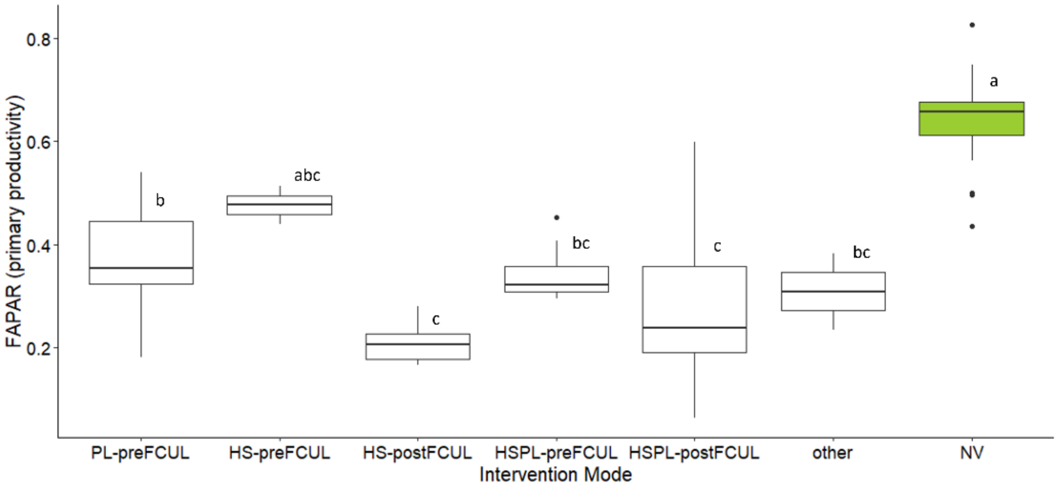


Figure 3.2. Variation of the biophysical variable FAPAR, used as the primary productivity indicator, in the different intervention modes in the restored quarry areas and the NV (reference ecosystem).

Considering all environmental, topographical and biological variables, the most important predictors for estimating the primary productivity indicator FAPAR were pine cover ($p<0.01$), shrub cover ($p<0.05$) and intervention mode ($p<0.05$) ($R^2= 0.45$) (Table S1.5). Pine and shrub cover positively correlate with the FAPAR index, indicating greater primary productivity and aboveground carbon sequestration in places with more shrub and pine cover.

The map in Figure 3.3 represents the FAPAR variation in the quarry and surrounding NV areas (outside the quarry boundary). The higher FAPAR values are found in the NV areas, between 0.6-0.9. Despite North-facing limestone benches and some of the East-facing marl slopes (where pine plantations exist) are the revegetated areas closer to the productivity values of the reference ecosystem, their values are mostly between 0.3-0.6. Limestone slopes have the lowest FAPAR values in the revegetated areas, between 0.1-0.4.

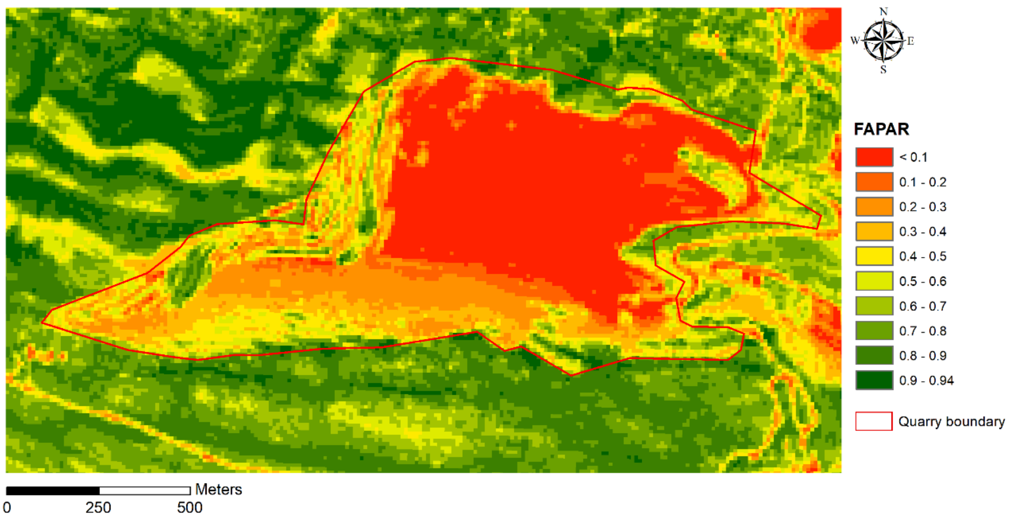


Figure 3.3. Map of the biophysical variable FAPAR used as the primary productivity indicator and calculated with satellite data with a resolution of 10m. Copernicus Sentinel image used in the determination of FAPAR from 05-12-2020.

3.1.2. Carbon Sequestration

3.1.2.1. Carbon sequestration in soil (chemical analysis)

Total organic carbon in soil, or SOC, assessed trough chemical analysis, shows a slight increasing trend with restoration age (Figure 3.4). The increase rate is higher in the first 10 years after restoration, after which there is a stabilisation in the SOC levels. Nevertheless, SOC levels did not differ significantly between different intervention modes in the quarry (Figure 3.5). Still, carbon was significantly higher in the soils of the undisturbed natural areas, which stored up to 10 times more carbon than the restored areas' soils.

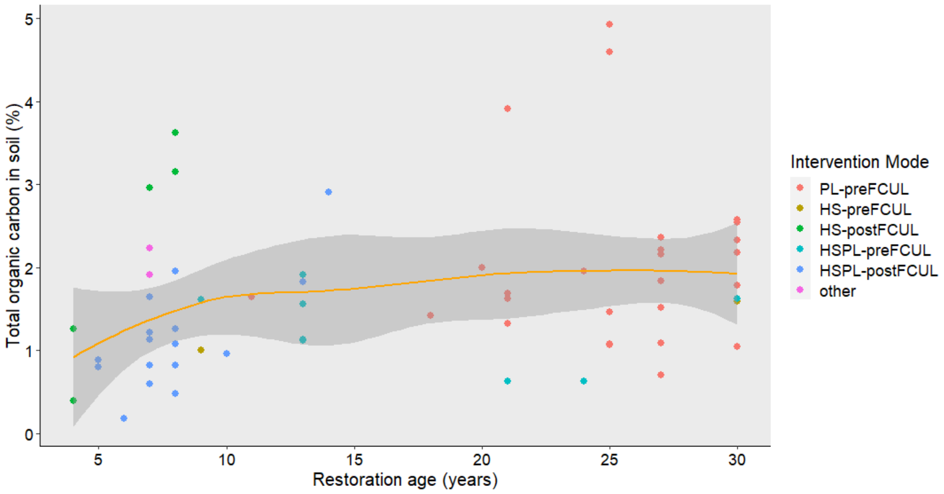


Figure 3.4. Variation of total carbon in the soil with restoration age in quarry restored plots (R^2= 0.10, p<0.05).

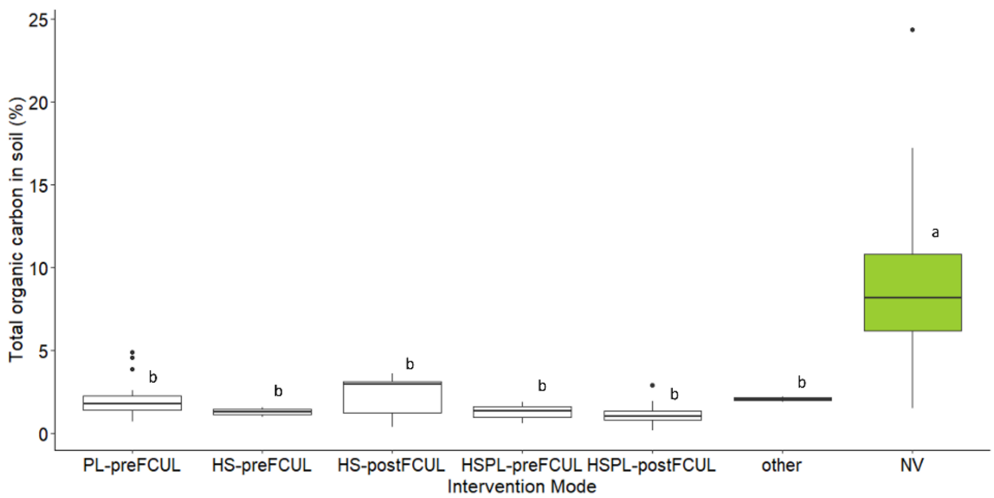


Figure 3.5. Variation of total carbon in the soil in the different intervention modes in the restored quarry areas and the NV (reference ecosystem).

Considering all environmental, topographical and biological variables, the most important predictors for estimating SOC were pine cover (p<0.001) and intervention mode (p<0.05) (R^2= 0.46) (Table S1.5). Quarry areas with higher levels of SOC were associated with increasing pine cover. However, the interaction between pine cover and intervention mode in the model suggests that the effect of pine cover on SOC varies depending on the intervention mode. This way, pine cover is only associated with higher SOC values in plantations where pines are highly abundant (in the limestone benches), as this association is absent in other intervention modes where pines are also present. Further analysis regarding soil fertility is made on *section 3.2.1*.

3.1.2.2. Carbon sequestration (InVEST analysis)

3.1.2.2.1. Land cover dominance

The quarry's cartography made for the InVEST analysis can be observed in Figure 3.6. More than 23 ha have been recovered by revegetation in the quarry, corresponding to 25% of the quarry's area. The most dominant land cover type was natural material surfaces, roads, and escarpments (69 ha). Coniferous vegetation occupies the largest of the revegetated areas (12.1 ha), followed by herbaceous vegetation (8.8 ha) and, lastly, sclerophyll vegetation (2.4 ha) (Figure S2.7).

Coniferous vegetation is dominant in limestone benches and some of the oldest restored marl slopes where Aleppo pine trees were planted. Herbaceous vegetation, on the other hand, dominates most limestone and marl slopes where hydroseeding and plantation of herbaceous species (and few shrub species) occurred. While scrubland and sclerophyll vegetation dominate the entire areas of NV in the reference ecosystem (Maquis), these are only dominant in small areas inside the quarry. Despite being present on both benches and slopes, they only dominate areas where pine trees were not planted or hydroseeding with commercial herbaceous species was not applied. Natural material surfaces, roads and escarpments are extensively present in the quarry where extractive activities are still in progress or where revegetation and natural colonization did not occur.

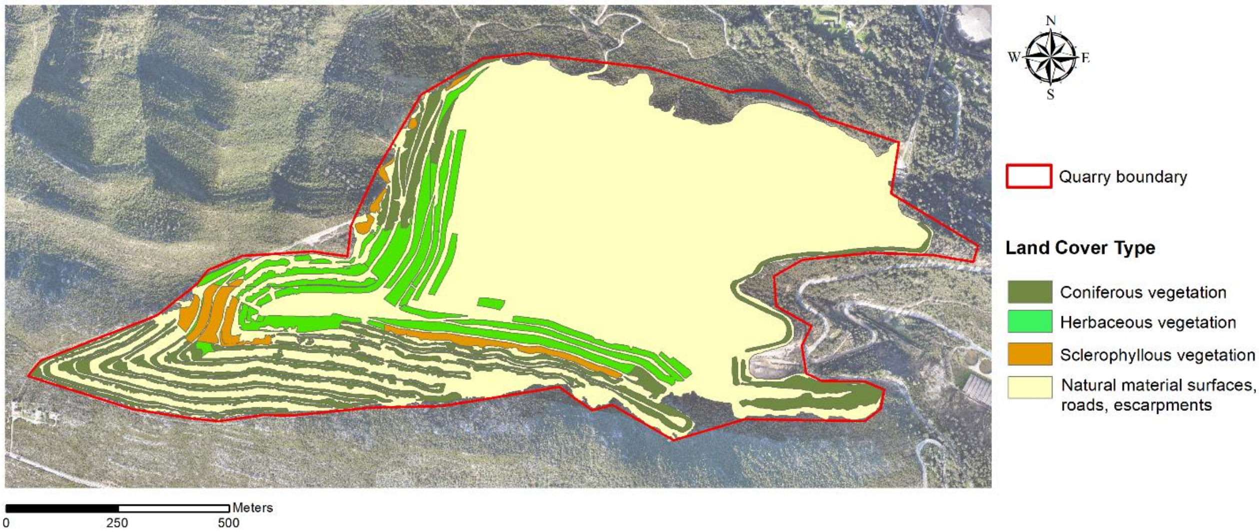


Figure 3.6. Map of the land cover types in the SECIL-Outão quarry.

3.1.2.2.2. Carbon sequestration in the quarry

Since restoration began in 1983, an estimated 5313 Mg of carbon has been sequestered from the atmosphere. This value is equivalent to the annual CO_2 emissions of 311 cars, assuming that the typical passenger vehicle emits about 4.6 metric tons of CO_2 per year (EPA, 2021). Maps for total carbon sequestration by the vegetation in the quarry are shown in Figure 3.7, and contributions from carbon stored aboveground, belowground and in soil are shown in Figure S2.8, Figure S2.9 and Figure S2.10, respectively.

The carbon sequestered by the revegetated areas depends on the type of vegetation present and the known carbon pool values available in the literature (see table 2.5). Thus, coniferous vegetation contributes the most to carbon storage in the quarry, followed by herbaceous and sclerophyllous vegetation (Figure S2.11). Therefore, revegetated areas with pine (limestone benches) store higher values of carbon than areas with dominant herbaceous or sclerophyllous cover (marl and limestone slopes) (Figure 3.7). According to the estimated carbon pool values, conifers contribute more to soil carbon sequestration than even NV (composed of sclerophyllous vegetation) due to their ability to store a large amount of carbon, mainly in the soil but also in their aerial and belowground parts. However, these results are opposite to the soil carbon values determined by the chemical analysis in *section 3.1.2.1*, where the NV had significantly higher carbon storage levels than the restored limestone benches. Herbaceous vegetation, dominant on slopes, mainly contribute to carbon storage in the soil and its belowground biomass (root system). Likewise, sclerophyllous vegetation essentially stores carbon in the soil and belowground. It can store more carbon in its root system than herbaceous or coniferous vegetation (Figure S2.9). In the revegetated areas, it stores more carbon aboveground than herbaceous vegetation (Figure S2.12).

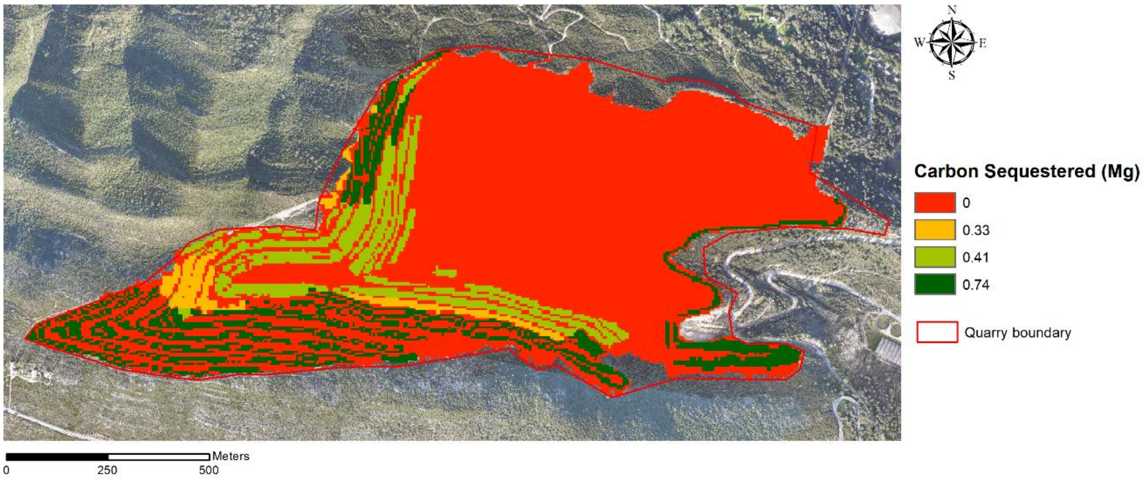


Figure 3.7. Map of the total carbon sequestration provided by the quarry's revegetated areas. Natural material surfaces (rock), roads and escarpments have a value of 0. The Maquis vegetation surrounding the quarry (reference ecosystem), mainly composed of sclerophyllous vegetation, has a value of 0.33 Mg/pixel.

3.1.3. Ecological Restoration Success Index

The response ratio, or the ratio of mean outcome in the restored plots to that of the reference ecosystem control (NV), showed that the restored quarry plots have not recovered to the desired values of the reference ecosystem concerning productivity and carbon sequestration, regardless of the intervention mode or zone type area in the quarry (Figure 3.8). In the quarry, the intervention modes with the closest values to the productivity of the reference ecosystem are the oldest restored sites, where plantations (PL-preFCUL) and hydroseeding (HS-preFCUL) interventions occurred. Soil carbon values in the quarry restored sites are considerably below the values found in NV. Nonetheless, plantations (PL-preFCUL) and hydroseeding after FCUL's recommendations (HS and HSPL-postFCUL) are the intervention modes closest to the reference. Regarding the zone types, a similar pattern is observed, with the Northern low slope (limestone benches) having the closest values of productivity and soil carbon to the reference ecosystem, and the limestone slopes (Northern high slope) having the lower values.

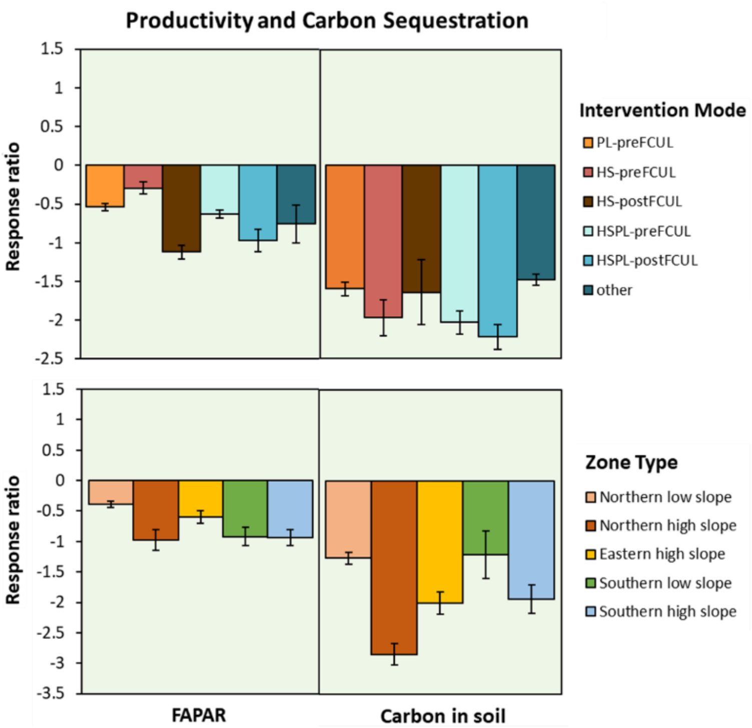


Figure 3.8. Response ratio (restoration success) for FAPAR (primary productivity proxy) and total carbon in soil for the different intervention modes (top image) and zone types (bottom image), compared with the reference ecosystem. Positive values of the indicator under analysis indicate higher values in the restored sites than in the reference ecosystem, values close to zero indicate a high degree or full recovery, and the opposite (incomplete recovery) holds for negative values. Error bars correspond to the standard error of the mean.

3.2. Soil Fertility and Decomposition

SOM, SOC, soil pH and macronutrients such as N and P were the indicators used for the assessment of soil fertility. Results for SOC are presented in *section 3.1.2.1*.

3.2.1. Soil Fertility

SOC and SOM were the only soil fertility variables showing a significant relationship with restoration age. These two variables have the same increasing trend with restoration age since they are proportional to each other (Figure 3.4 and Figure S2.13). Despite increasing with restoration age, the maximum SOM in the quarry soils is around 8%, which is still far from the reference ecosystem values which have a median of 15% but can reach values as high as 40% SOM (Figure S2.14).

Likewise, N content in the soil was significantly higher in the reference ecosystem, reaching values as high as 11190 mg/kg, more than the triple of the maximum N found in the restored plots (Figure S2.15). On the other hand, available phosphorous in the soil is up to almost six times higher in hydroseeded plots with FCUL recommendations (HS-postFCUL) when compared with the NV, far exceeding reference ecosystem levels (Figure S2.16). The only intervention mode where phosphorous levels were significantly below the reference ecosystem was the plantations (PL-preFCUL). Additionally, soil analysis revealed a significantly higher pH in the restored soils, around 8.1, compared to the more acidic soils found in the reference ecosystem, around 7.7, revealing a potential excess of alkalinity (Figure S2.17).

The analysis of which predictors best estimate soil fertility variables indicates that SOM significantly increased with shrub height (p<0.05) and pine cover (p<0.001), and depended on the intervention mode (p<0.05) (Table S1.5). Together, these predictors explained 51% of the SOM variability in the sampled sites in the quarry (R^2= 0.51). Although slope is not a significant predictor in this model, there is a significant negative relationship between slope and SOM (and carbon) in soil, indicating that areas with high slopes have more difficulty accumulating organic matter (Figure S2.18). Furthermore, woody (shrub and pine) cover contribute to increasing SOM and nitrogen in the restored quarry areas, bringing the restored sites closer to reference ecosystem levels. Shrub cover is especially important in the marl and limestone slopes (HS and HSPL intervention modes), where it appears to be a determinant factor for higher SOM values and in decreasing soil pH (Figure S2.19 and Figure S2.20, respectively). In the limestone bench plantations (PL-preFCUL), shrub height (PR90) is significantly associated with increasing nitrogen concentration in the soil (Figure S2.21) and tree height is significantly associated with more acidic soils (Figure S2.21).

3.2.2. Decomposition

The C:N ratio in the restored sites increased with restoration age, with values lower than 15 at more recently restored plots and close to or higher than 20 at older restored sites where succession has happened for a longer period of time (Figure 3.9). The ratio values were significantly above those in NV in the plantations (PL-preFCUL) and were slightly below in hydroseeded sites (HS-postFCUL) (Figure 3.10).

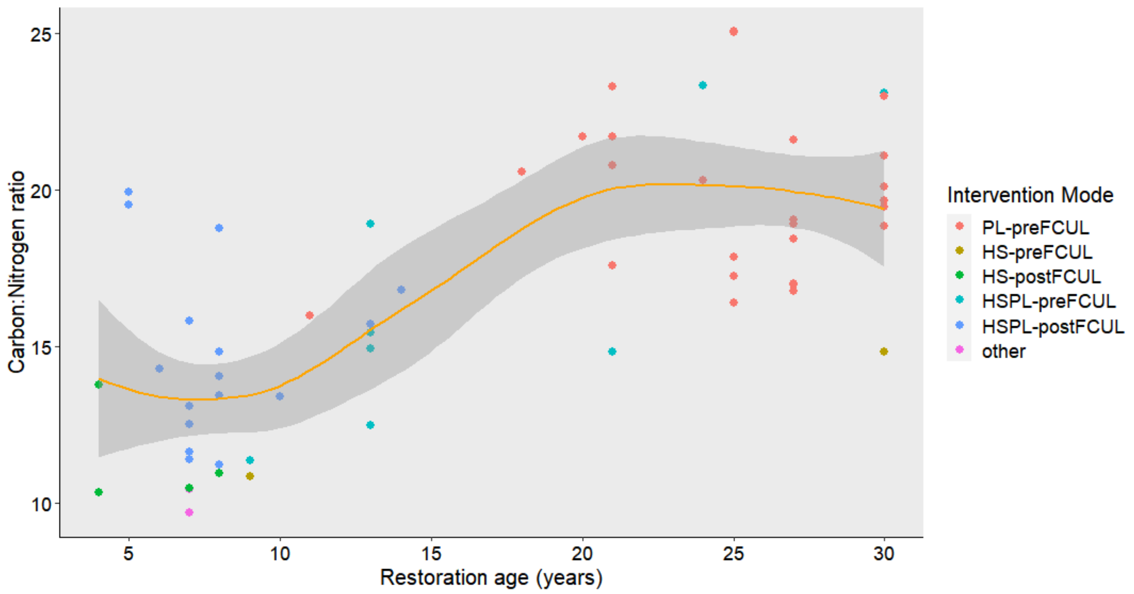


Figure 3.9. Variation of the carbon-to-nitrogen ratio in the soil with restoration age in quarry restored plots (R^2= 0.54, p<0.05).

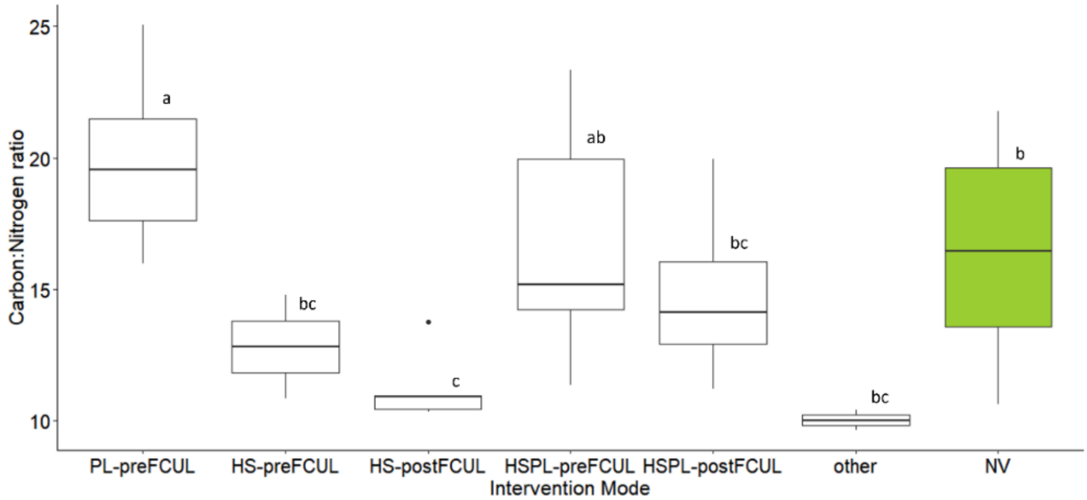


Figure 3.10. Variation of carbon-to-nitrogen ratio in the different intervention modes in the restored quarry areas and the NV (reference ecosystem).

The most significant predictors for estimating the C:N ratio were restoration age ($p<0.05$) and tree density ($p<0.05$) (both positively associated with the ratio) and intervention mode ($p<0.05$) (R^2 = 0.64) (Table S1.5). Therefore, higher C:N ratios can be found in older restored areas with high tree density.

The map in Figure 3.11 represents the C:N ratio variation in the quarry revegetated areas and was produced using a model with predictors BI2 ($p<0.001$) and slope ($p<0.01$), which explained 50% of the C:N variation (R^2= 0.50) (Table S1.5). Both predictors were negatively associated with the ratio, indicating that steeper sites with less vegetation and more soil cover (BI2 is a soil index) have a lower C:N ratio. The map shows higher C:N values in limestone benches (older restored sites) where plantations of pines and shrub species predominate and the lowest values in the marl slopes where hydroseeding (with and without plantations) has been applied.

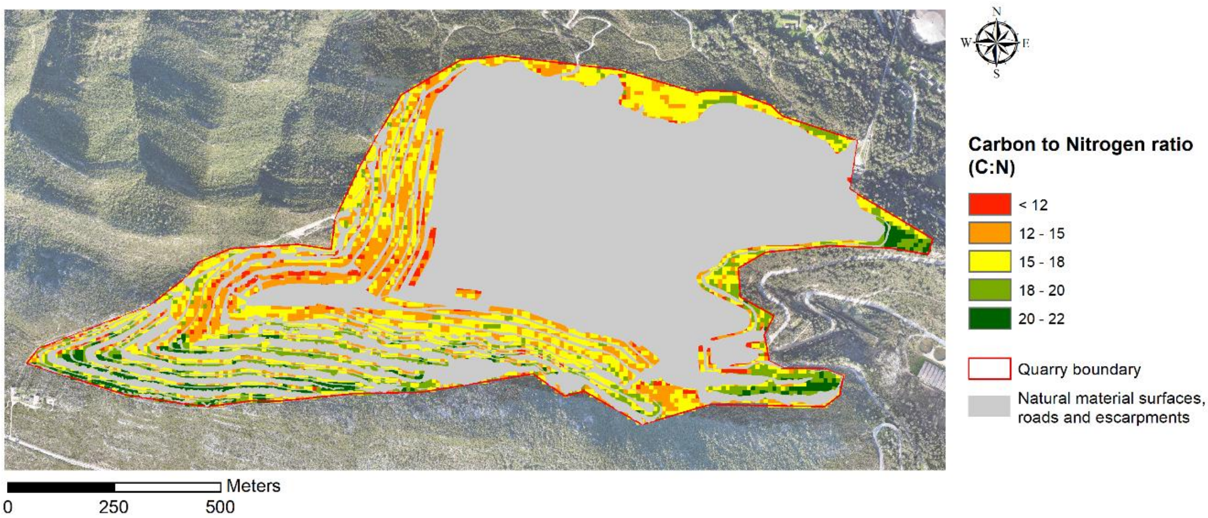


Figure 3.11. Carbon-to-nitrogen ratio map, determined from the BI2 and slope (R^2 = 0.50). Orthophoto from SECIL (2020) with 5 cm resolution.

3.2.3. Ecological Restoration Success Index

Generally, soil fertility in the restored plots is still far from the reference levels (Figure 3.12). SOM and nitrogen have not recovered to reference levels in any intervention modes or zone types. Phosphorous is only higher in hydroseeded sites (HS-posFCUL) and sites classified as having "other" intervention (2 plots). Additionally, soil pH is higher (more alkaline) in the quarry restored plots than in the reference ecosystem. Regarding soil decomposition, the C:N ratio is only higher than the reference ecosystem in the limestone benches (Northern low slope) where plantations occurred, indicating lower decomposition rates than the reference ecosystem. The most similar decomposition values to the reference ecosystem could be found in sites where hydroseeding and plantations (HSPL-preFCUL) were applied, located in the zone type "Eastern high slope". The "Northern high slope" (limestone slopes) zone type is the one with the most distant values to the reference regarding all soil fertility and decomposition values (except for pH, where limestone benches – Northern low slope – have the most alkaline soils).

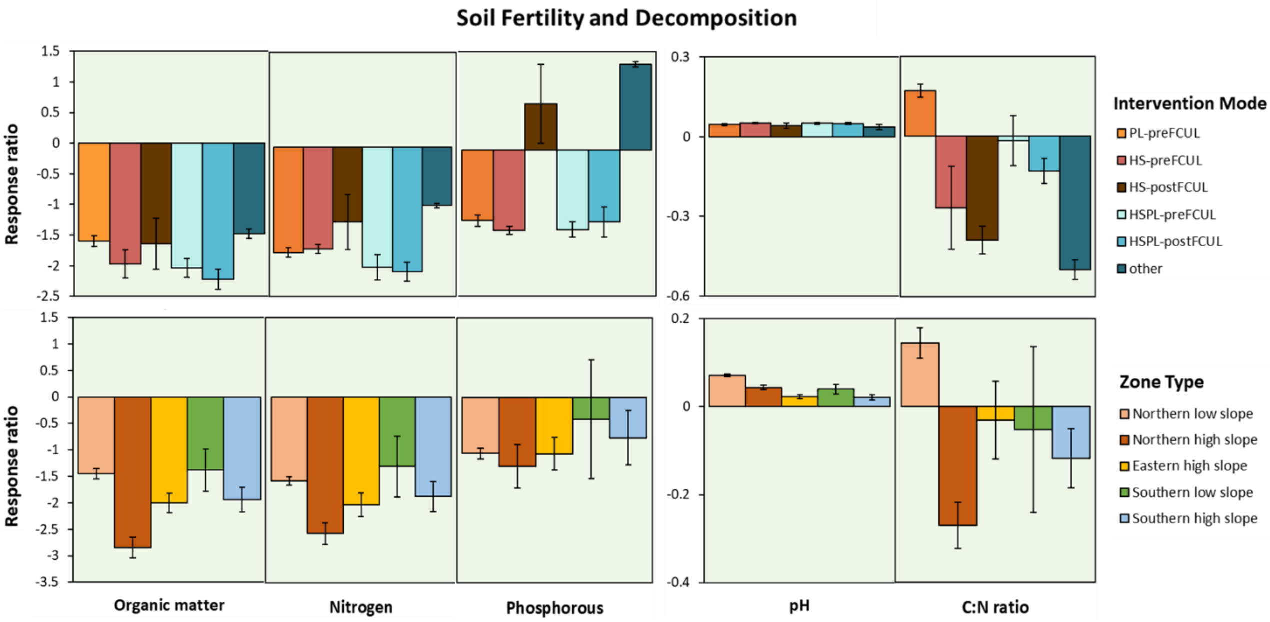


Figure 3.12. Response ratio (restoration success) of soil fertility (organic matter, nitrogen, phosphorous, pH) and decomposition (C:N ratio) variables, for the different intervention modes (top graphs) and zone types (bottom graphs), compared with the reference ecosystem. Positive values of the indicator under analysis indicate higher values in the restored sites than in the reference ecosystem, values close to zero indicate a high degree or full recovery, and the opposite (incomplete recovery) holds for negative values. Error bars correspond to the standard error of the mean.

3.3. Habitat Quality

3.3.1. Shrub, pine and herbaceous Cover

Shrub cover did not vary significantly with restoration age, despite showing a unimodal trend, with a peak around 15 years after restoration (Figure 3.13, top left image). Although shrub cover shows a clear trend of rise with restoration age in the hydroseeded (HS) and mixed (HSPL) plots, it then stabilizes and slightly decreases in the older restored plots where plantations of pines and shrubs occurred (PL-preFCUL). Pine cover, on the other hand, significantly increases with restoration age (Figure 3.13, top right image) and stabilizes after approximately 20 years. Contrarily, herbaceous cover had a negative relationship with restoration age after reaching a peak around 12 years (Figure 3.13, bottom image).

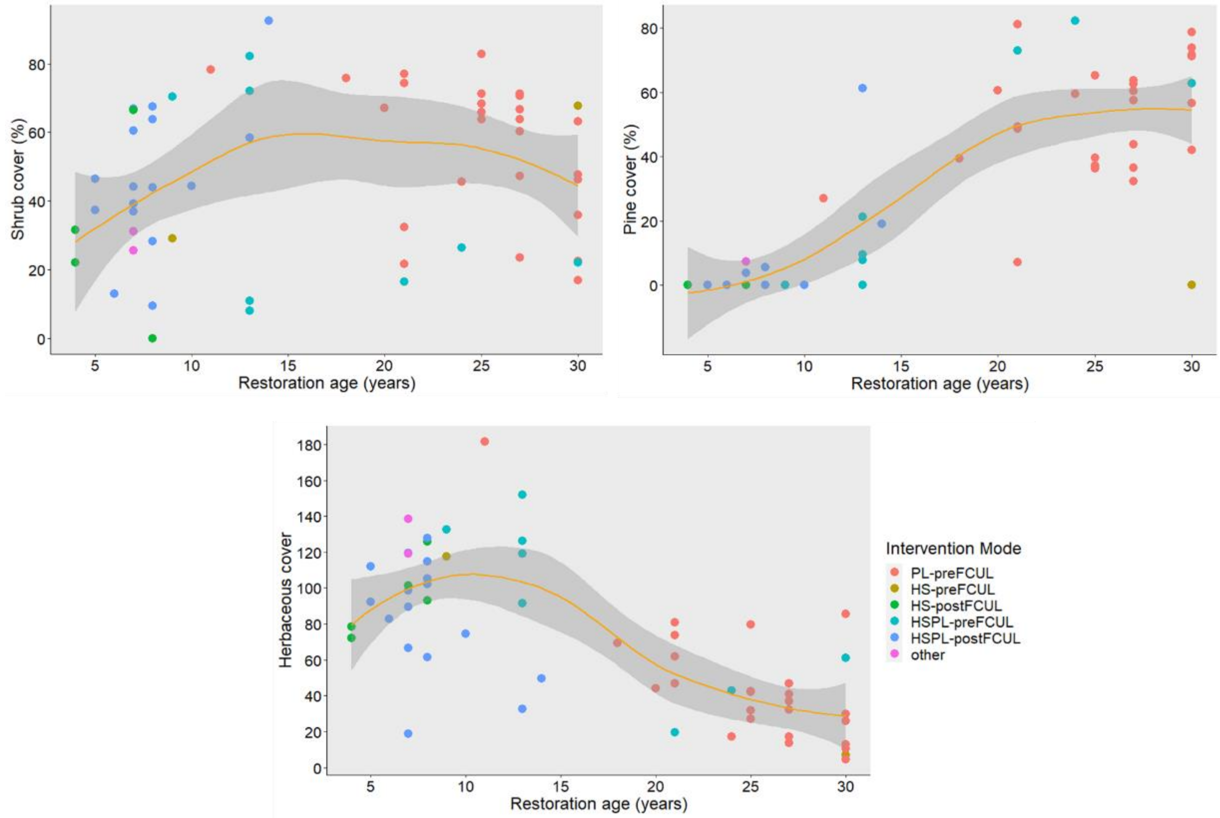


Figure 3.13. Top left image: Variation of shrub cover with restoration age (n.s.). Top right image: Variation of pine cover with restoration age (R^2= 0.67, p<0.001). Bottom image: Variation of herbaceous cover with restoration age (R^2= 0.55, p<0.05).

Shrub cover was significantly higher in NV than in the restored sites, and it showed significant differences across intervention modes, with plantations (PL-preFCUL) having the highest values and hydroseeding (HS-postFCUL) the lowest (Figure S2.23). No significant shrub cover predictors were identified for modelling.

Regarding pine cover, plantations (PL-preFCUL) had significantly higher values than the NV and the hydroseeded slopes (Figure S2.24). Furthermore, the best predictors of pine cover were the intervention mode (p<0.05) and restoration age (p<0.001), which together explained 79% of the pine cover variation (R^2= 0.79), with a significant interaction (p<0.001) between these two variables (Table S1.5).

The best spatially distributed predictors for mapping pine cover were $\frac{NDVI\ May}{NDVI\ August}$ (p<0.001) and slope (p<0.01), which explained 64% of the pine cover variation (R^2= 0.64) (Table S1.5). Pine cover is higher in places with low slope and low NDVI ratio. The model described above was used to map the pine cover in the restored area and is represented in Figure 3.14. While higher dominance of pine trees could be found in the north-facing benches and some of the east-facing slopes, lower pine cover is present in the north-facing and south-facing slopes.

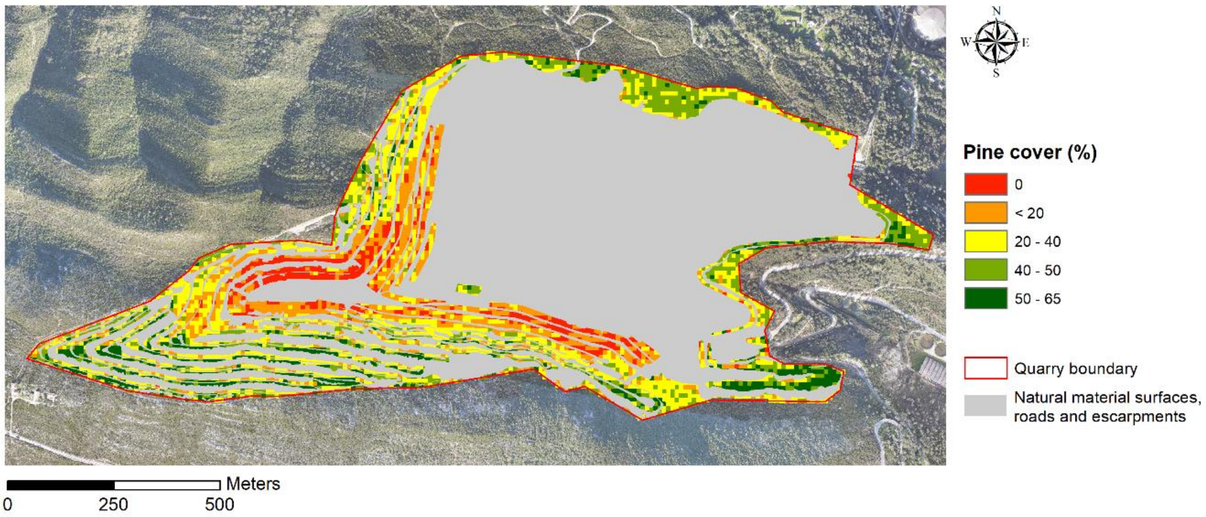


Figure 3.14. Pine cover (%) map, determined from the $\frac{NDVI\ May}{NDVI\ August}$ and slope ($R^2 = 0.64$). Orthophoto from SECIL (2020) with 5 cm resolution.

Herbaceous cover differs significantly between the intervention modes involving hydroseeding and plantations (HSPL) and plantations (PL-preFCUL), the former having significantly higher herbaceous cover than the latter (Figure S2.25). However, no statistical difference was found between the herbaceous cover in the quarry intervention modes and the reference ecosystem. The best predictors for explaining herbaceous cover were restoration age ($p<0.001$), intervention mode ($p<0.001$) and shrub density ($p<0.01$), with restoration age and shrub density being negatively correlated with herbaceous cover ($R^2= 0.76$) (Table S1.5).

The best spatial predictors for mapping herbaceous cover were the $\frac{NDVI\ September}{NDVI\ May}$ ratio ($p<0.01$), which was negatively associated with herbaceous cover, and slope ($p<0.01$), which was positively associated ($R^2= 0.49$) (Table S1.5). The herbaceous cover map, represented in Figure 3.15, indicates higher herbaceous cover over the east and south-facing marl and north-facing limestone slopes than on the north-facing limestone benches.

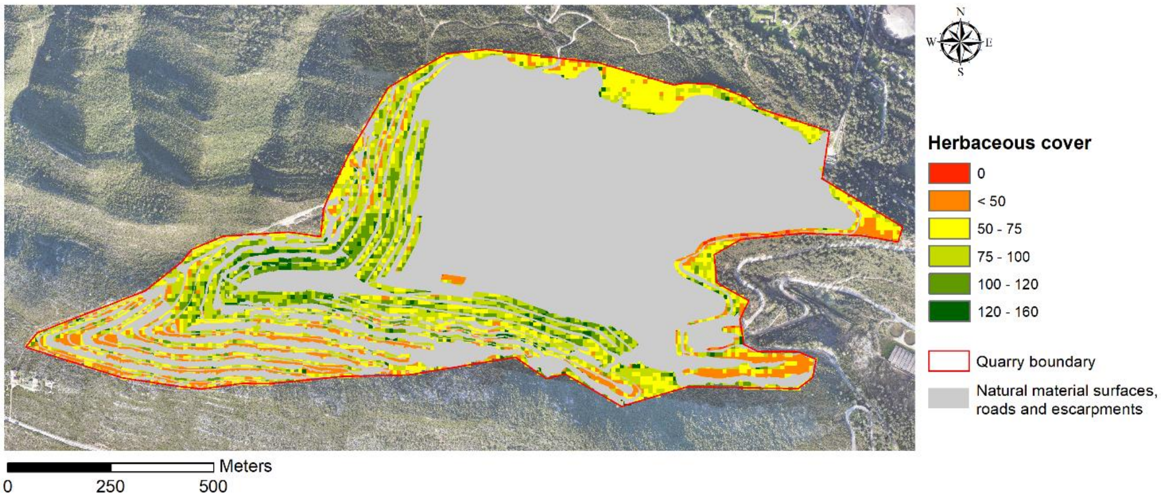


Figure 3.15. Herbaceous cover (%) map, determined from the $\frac{NDVI\ September}{NDVI\ May}$ and slope ($R^2 = 0.49$). Orthophoto from SECIL (2020) with 5 cm resolution.

3.3.2. Height

Tree and woody vegetation height (CWM) are higher in older restored plots, opposite to the trend observed with herbaceous vegetation height (CWM) (Figure S2.26 and Figure 3.16 left and right image, respectively). However, a slight increase in herbaceous height can be seen in the first 10 years of the restoration, where pine trees are absent or have low cover. Shrub height did not have a significant relationship with restoration age.

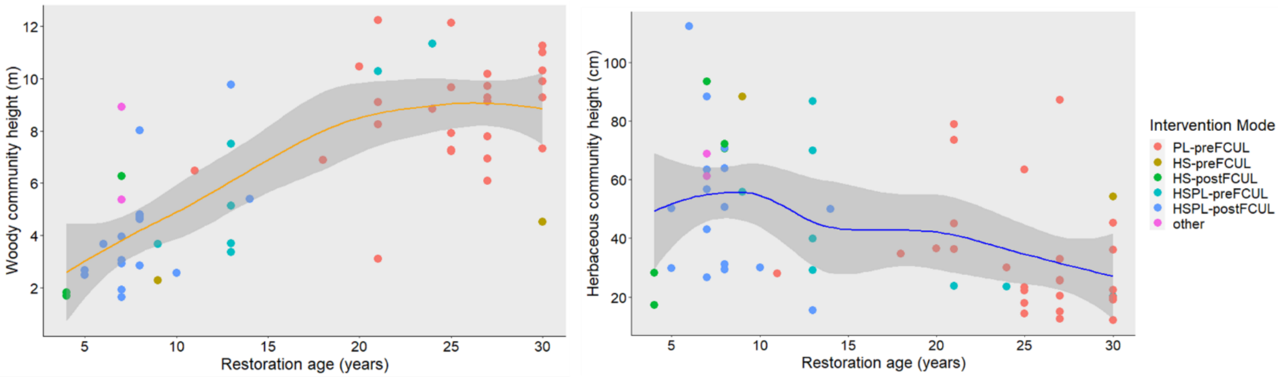


Figure 3.16. Left image: Variation of woody community height (CWM) with restoration age (left image) (R^2= 0.54, p<0.001). Right image: Variation of herbaceous community height (CWM) with restoration age (right image) (R^2= 0.27, p<0.05).

Tree and shrub height values did not significantly differ between intervention modes (Figure S2.27 and Figure S2.28, respectively). Tree height was best explained by the interaction ($p<0.05$) between pine cover ($p<0.001$), restoration age ($p<0.05$) and SOM ($p<0.01$) - all positively associated with tree height (R^2= 0.56) (Table S1.5). Its spatial distribution was based on the $\frac{NDVI\ February}{NDVI\ July}$ ratio (negative coefficient; $p<0.001$) and the soil index BI (negative coefficient; $p<0.001$), which explained 56% of the tree height variation (R^2= 0.56) (Table S1.5). The resultant map can be seen in Figure 3.17. In this map, higher tree height can be seen in the North-facing limestone benches, while the lowest tree height can be observed in the East and South-facing marl slopes and North-facing limestone slopes. The high values of tree height observed in the Northeastern and Southeastern parts of the quarry do not correspond to values in the revegetated area but are within with the official quarry boundary.

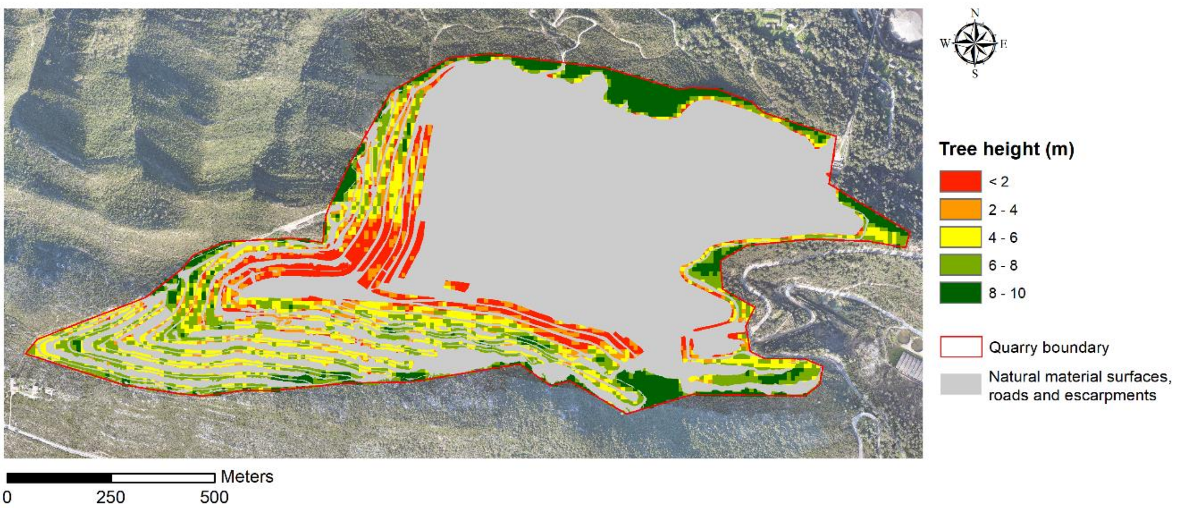


Figure 3.17. Tree height map, determined from the $\frac{NDVI\ February}{NDVI\ July}$ and BI (R^2 = 0.56). Orthophoto from SECIL (2020) with 5 cm resolution.

As expected, woody vegetation height (CWM) was significantly higher in plantations (old limestone benches restored with pine trees) than in hydroseeded plots or in the reference ecosystem (Figure S2.29). Height in the woody community was best explained by restoration age ($p<0.05$) and the interaction ($p<0.001$) between pine cover ($p<0.001$) and shrub height ($p<0.001$) (all predictors having positive relationships with height, and together, explaining 89% of the height variation (R^2= 0.89) (Table S1.5). Its spatial distribution was based on the $\frac{NDVI\ May}{NDVI\ July}$ ratio (negative coefficient; $p<0.05$), the BI2 soil brightness index (negative coefficient; $p<0.05$), slope (negative coefficient; $p<0.01$), and the interactions between these variables ($p<0.01$), explaining 68% of the height variation in the woody community, and its resulting map is presented in Figure 3.18 (R^2= 0.68) (Table S1.5).

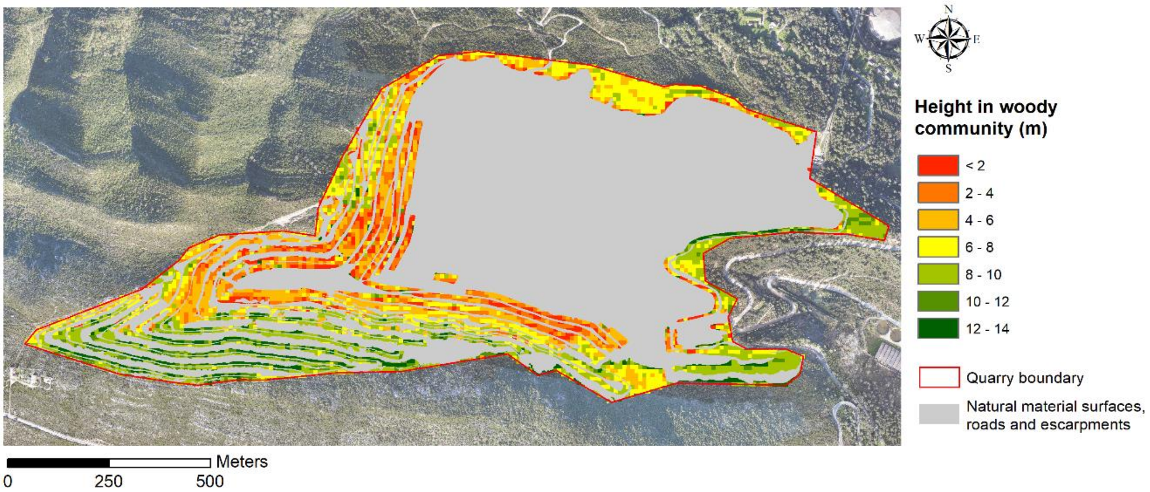


Figure 3.18. Height (CWM) in the woody community map, determined from the $\frac{NDVI\ May}{NDVI\ July}$, BI2 and slope ($R^2$ = 0.68). Orthophoto from SECIL (2020) with 5 cm resolution.

Herbaceous vegetation height (CWM) was similar between intervention modes (Figure S2.30). Its explanatory model was based on a negative relationship with pine cover ($p<0.001$) and shrub density ($p<0.05$) (R^2=0.30) (Table S1.5). Thus, in sites with greater pine cover and shrub density, the average herbaceous species' heights are significantly lower.

3.3.3. Ecological Restoration Success Index

Overall, habitat quality variables such as cover and height vary considerably between the quarry restored plots and the reference ecosystem (Figure 3.19). The most striking differences regarding cover are: (i) the shrub cover, which is below reference ecosystem's levels in all intervention modes and zone types, and (ii) the pine cover, which is overall much higher in the zone types of the restored plots than in the reference ecosystem. With lower differences from the reference ecosystem is the herbaceous cover, which is slightly below reference levels in the limestone benches plantations (PL-preFCUL and Northern low slope), and above reference levels in the other intervention modes and zone types. In comparison to the NV, hydroseeded sites (HS-postFCUL) and Sourthern slopes had the furthest shrub cover values from the reference. In contrast, sites with plantations (PL-preFCUL) had the closest shrub cover values, but the highest pine cover difference from the reference ecosystem along with the marl "Eastern high slopes".

The height of the different vegetation strata had more evident tendencies than cover, with (i) shrub height being below the reference ecosystem in all intervention modes and zone types (except for "other"), (ii) higher tree height in the restored plots than in the reference ecosystem, (iii) woody vegetation height (CWM) is generally higher in the restored plots and (iv) herbaceous species with a higher maximum height potential are more abundant in the quarry than in the NV, except for plantations (PL-preFCUL), Eastern-facing slopes and Southern-facing low slopes, which had inferior values than the NV.

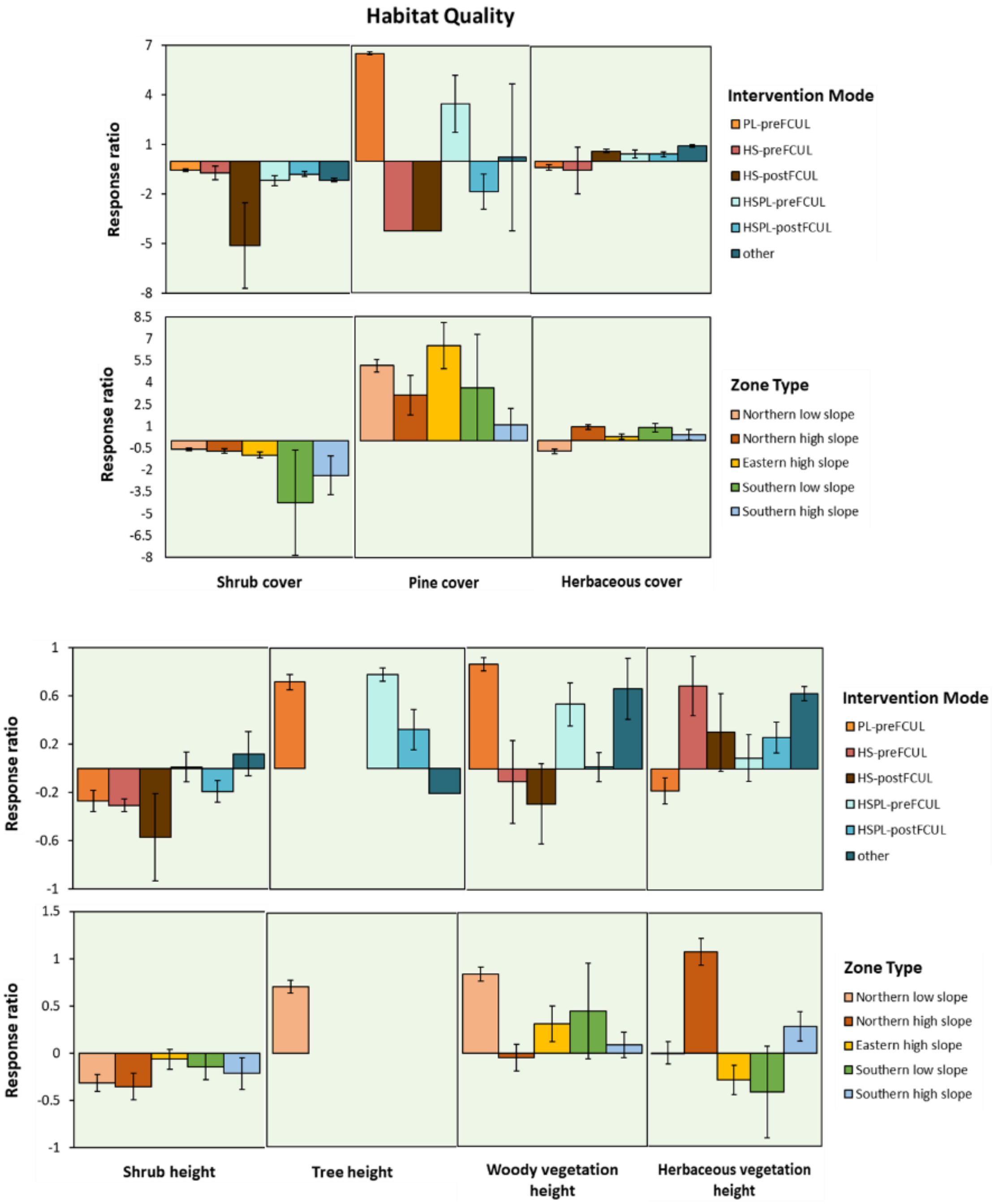


Figure 3.19. Response ratio (restoration success) of habitat quality variables, such as cover (shrub, pine and herbaceous – top graphs) and height (shrub, tree, woody and herbaceous – bottom graphs), for the different intervention modes and zone types, compared with the reference ecosystem. Positive values of the indicator under analysis indicate higher values in the restored sites than in the reference ecosystem, values close to zero indicate a high degree or full recovery, and the opposite (incomplete recovery) holds for negative values. Error bars correspond to the standard error of the mean.

3.4. Similarity to NV

Relationship with restoration age

The similarity of the species composition between the restored sites and the NV sites altogether (disregarding any zone type classification), which took into account the presence of species in the entire plant community (Sørensen–Dice index), increased significantly with restoration age ($p<0.001$) (Figure 3.20a, top panel). Nevertheless, after 30 years, the similarity is, on average, around 25%, corresponding to 75% dissimilarity. The most similar site to the NV has close to 40% similarity. The Bray-Curtis similarity, which considers not only species presences but also species abundances, does not present this linear pattern nor has a significant relationship with restoration age (Figure 3.20a, bottom panel). Considering abundances, similarity values are low, with an average of close to 10%, resulting in more than 90% of dissimilarity. The most similar site to the NV has close to 20% similarity.

When the comparison is made between zone types, i.e. comparing restored sites with their correspondent zone types in the reference ecosystem (in terms of surface exposure and slope), the pattern of change over time holds the same for Sørensen–Dice similarity (species presences) and shows a slight significant increase in Bray-Curtis similarity (species abundances) with restoration age ($p<0.05$) (Figure 3.20b). When the comparison of the restored areas with their zone types includes a comparison between the revegetated limestone benches (with high densities of pine) with a natural planted pine forest, the similarity values show a slight increase, remaining below 40% in Sørensen–Dice and 25% in Bray-Curtis similarity (which had an outlier of 43%) (Figure 3.20c).

The general pattern of change in similarity over time after restoration, represented based on chronosequence, shows that, for both metrics (Sørensen–Dice and Bray-Curtis similarity), there is an approximation to the composition and diversity of NV that is more evident in the first 15 years, after which there is a stabilization trend.

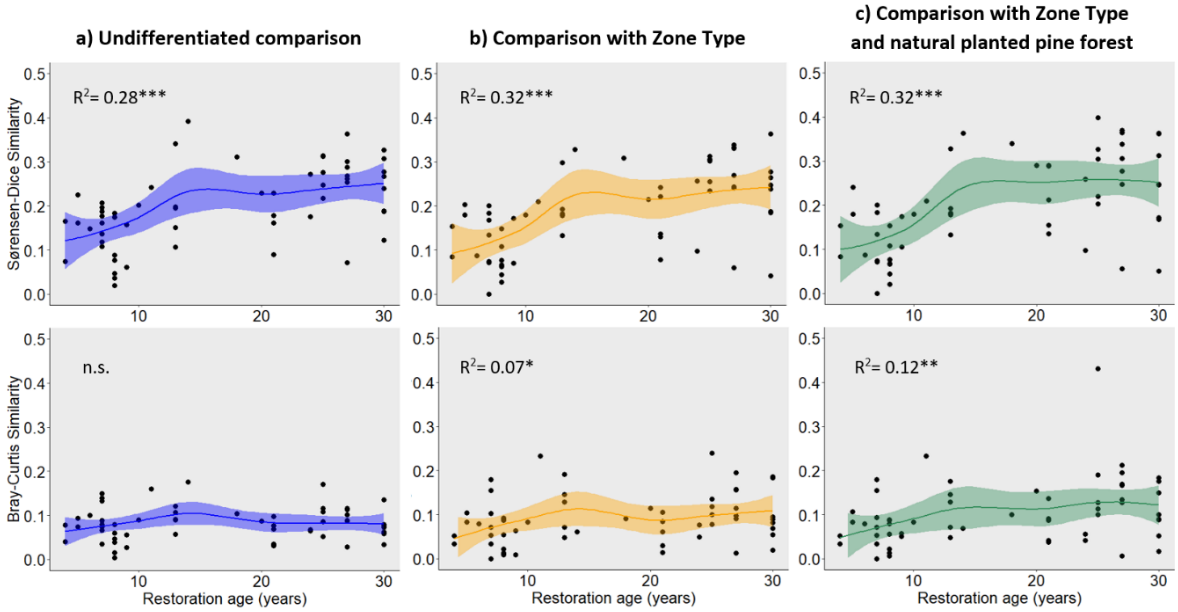


Figure 3.20. Variation with restoration age of Sørensen–Dice (based on presence/absence of species) and Bray-Curtis (based on species abundance) similarity indices between the restored quarry plots and reference ecosystem plots, considering all plant community (shrub, pines and herbaceous species), and taking into account three analysis with different sets of reference ecosystem controls. Model p-value: * $p<0.05$, ** $p<0.01$, *** $p<0.001$, n.s. non-significant.

Considering the presence of species only in the shrub community, the similarity to the NV (Sørensen–Dice index) increases with restoration age, showing a similarity, on average, of 25% and, at most, of about 45% (Figure 3.21). The Bray-Curtis similarity index does not show this significant linear pattern and has very low similarity values, with an average similarity of 15% and a maximum of 35% (Figure 3.21). Such was also observed in the field, for example, regarding the abundance of species such as *Q. coccifera* and *S. aspera*, which are abundant and form dense structures in the reference ecosystem but were not observed with the same density in the restored sites.

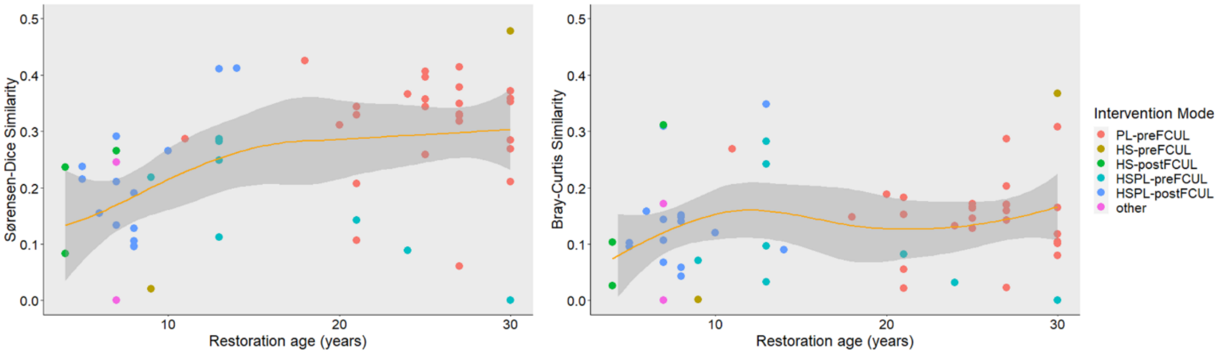


Figure 3.21. Variation with restoration age of Sørensen–Dice (based on presence/absence of species) (left graph) and Bray-Curtis (based in species abundance) (right graph) similarity indices between the restored quarry plots and reference ecosystem plots, considering the shrub community (R^2= 0.22, p<0.001 for Sørensen–Dice similarity, n.s. for Bray-Curtis similarity).

Similarity in the studied communities (shrub, woody, herbaceous and all plant community)

Moreover, similarity values between the restored plots and the reference ecosystem, regarding both species' presence (Sørensen–Dice index) and abundance (Bray-Curtis index), were relatively low for the different vegetation strata. The stratum with the highest similarity was the shrub community, although similarity was, on average, less than 30% regarding species presence and around 15% if their abundance was considered (Figure 3.22). The abundance of pines contributed to distancing the restored communities from the NV and thereby reducing similarity (Figure 3.22, right graph). This is most notorious in the Bray-Curtis index, which had lower similarity values (~0.1) in the woody community (shrub community and pines) than in the shrub community (without pines) (~0.15) (Figure 3.22). Furthermore, the herbaceous community was the studied community with the lowest similarity to NV, showing median values below 10% considering species presence and even lower, below 5%, when considering their respective abundance (Figure 3.22). When analysing all plant community, similarity (considering species presence) mainly varied between 0 and 0.35 with a median value of 0.2, corresponding to only 20% similarity with NV (Figure 3.22). Considering species abundance, similarity values vary between 0 and 0.2 with a median value of 0.1, indicating only 10% similarity with NV.

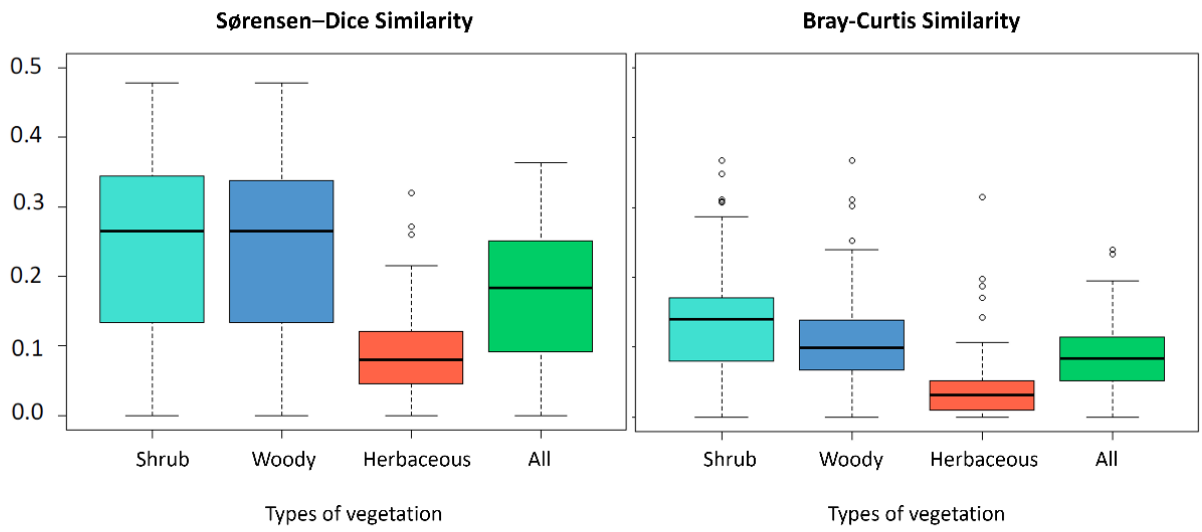


Figure 3.22. Variation of Sørensen–Dice (based on presence/absence of species) and Bray-Curtis (based in species abundance) similarity indices between the restored quarry plots and reference ecosystem plots, in the different types of communities studied. “All” includes shrub, woody and herbaceous vegetation.

Similarity between intervention modes and zone types

Considering the studied communities and regarding species presence (Sørensen–Dice similarity index), plantations (PL-preFCUL) correspond to the intervention mode most similar to the reference ecosystem. Generally, plantations had significantly higher values than hydroseeding and plantations (HSPL) but were not statistically different from hydroseeding (HS) values. In the shrub and woody community, the similarity to NV was significantly higher in plantations (PL-preFCUL) (~0.35) than in hydroseeding and plantations (HSPL) intervention modes (Figure S2.31 and Figure S2.33, respectively). In the herbaceous community, the maximum similarity was found in PL-preFCUL, which was significantly higher than HSPL-postFCUL but still with very low values (~0.1) (Figure S2.35). In all plant community, similarity found in PL-preFCUL (~0.25) was significantly higher than in hydroseeding (HS-postFCUL) and HSPL intervention modes (Figure S2.37). On the other hand, Bray-Curtis similarity values were very low (averages below 20%) and did not significantly differ across the intervention modes in none of the studied communities (Figure S2.32, Figure S2.34, Figure S2.36 and Figure S2.38).

Considering the similarity in the shrub community and the entire plant community in the different zone types (Figure 3.23), the highest similarity values considering species presences were found in the Northern low slope, which corresponds to the limestone benches, where plantations were made, in line with the previously stated statistical differences across intervention modes. However, the zone type with higher similarity considering species abundances was the “Southern high slopes" (marl slopes). The greatest dissimilarity to the NV was found on the limestone slopes (Northern high slopes) and in some of the marl slopes (Eastern high slope), which have less than 20% similarity in terms of species presences and around or less than 10% similarity in terms of species abundances.

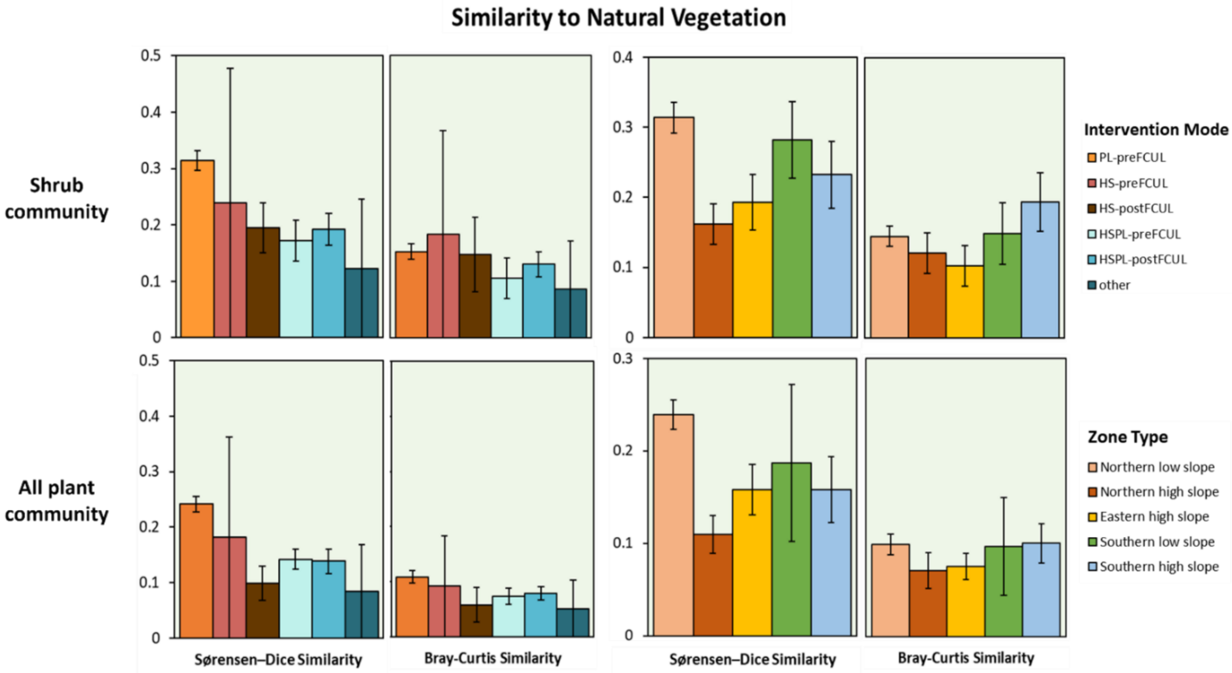


Figure 3.23. Similarity between the restored sites and the reference ecosystem (NV), of the shrub community (top panel) and all plant community (includes woody and herbaceous community - bottom panel), regarding species presence (Sørensen–Dice index) and species abundance (Bray-Curtis index), for the different intervention modes and zone types. Values of zero represent no similarity, while values of 1 represent maximum similarity. Error bars correspond to the standard error of the mean.

<u>Variation of the species composition and significant environmental factors (NMDS)</u>

The variation of the composition of the shrub community without and with pines (woody community) along the first axis (NMDS1) separates the communities under restoration from the reference areas, a separation which is more obvious when pines are considered (Figure 3.24). This variation is mainly related to soil characteristics, with lower pH and higher SOM content in NV areas, as well as a tendency towards greater potential surface exposure (PSR), associated with greater shrub cover. Soil characteristics are also associated with a restoration age gradient (visualised by the restoration decade colours), which may be related to soil succession and evolution with time after restoration. Along the second variation axis (NMDS2), variables such as slope, associated with greater herbaceous species cover and, in the opposite direction, pine trees cover, stand out, indicating a variation more associated with the restoration interventions carried out in the revegetated areas.

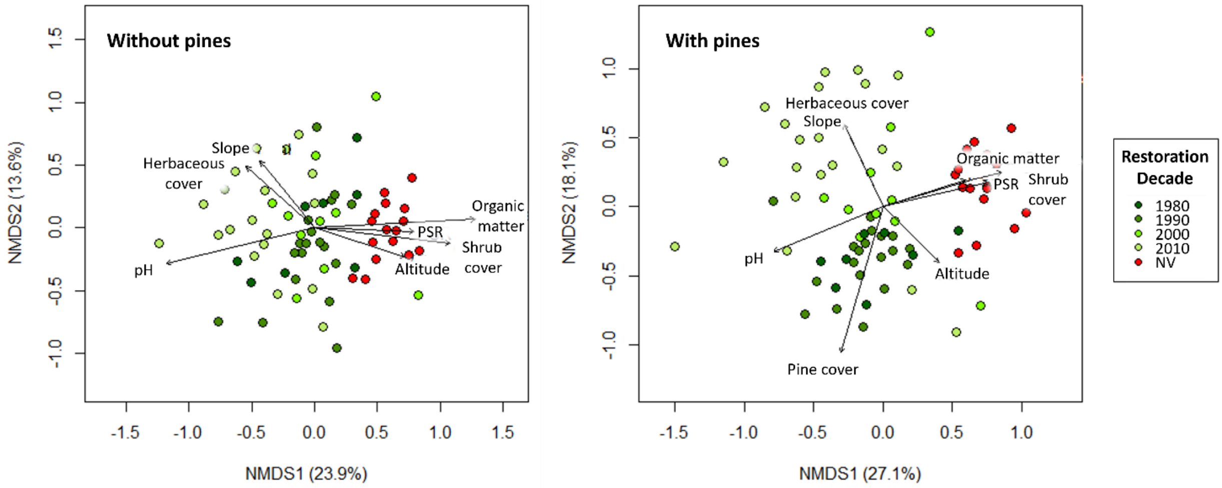


Figure 3.24. Ordinance analysis (NMDS – Non-metric multidimensional scaling) based on the abundance of the shrub community excluding pines (left) and the shrub community including pines (woody community) (right). The points represent the sampling sites classified according to the decade where the beginning of their revegetation started (in green scale) and the natural vegetation sites (NV, in red). Vectors represent the significant environmental variables that most strongly correlated with the NMDS axes.

Evaluation of the best predictors

Considering the entire plant community, the predictors that best explain the similarity to NV regarding species presence (Sørensen–Dice index) are restoration age ($p<0.05$), intervention mode ($p<0.001$) and shrub cover ($p<0.01$), which together explain 65% of the similarity variation ($R^2= 0.65$) (Table S1.5). Thus, the greater the shrub cover and the restoration age, the greater the similarity to NV considering all species. Additionally, the effect of restoration age on similarity depends on the intervention mode, which is evidenced by the significant interaction between the two variables in the model ($p<0.05$). Likewise, the predictors that best explain the similarity to NV considering species abundance (Bray-Curtis similarity) are shrub cover ($p<0.05$) and intervention mode ($p<1$) ($R^2= 0.37$) (Table S1.5). Therefore, a greater similarity to NV can be found in areas with higher shrub cover. The effect of shrub cover on Bray-Curtis similarity depends on the intervention mode resulting in the significant interaction ($p<0.05$) between the two variables in the model. However, restoration age was not a relevant factor in increasing Bray-Curtis similarity, contrarily to the Sørensen–Dice index.

Considering only the shrub community, shrub cover ($p<0.05$) and intervention mode ($p<0.001$) explained 71% of the variation observed in the similarity to the NV regarding species presence (Sørensen–Dice index) ($R^2= 0.71$) (Table S1.5). Furthermore, higher shrub cover was significantly associated with a similarity increase. An interaction between shrub cover and intervention mode was also found ($p<0.05$). However, no predictor was found to significantly explain the Bray-Curtis similarity in the shrub community.

Spatial mapping (landscape scale)

The best spatial predictors for the Sørensen–Dice index in the entire plant community were the $\frac{NDVI\ May}{NDVI\ July}$ ratio (negative coefficient; $p<0.01$), and slope (negative coefficient; $p<0.05$), which together explained 51% of the similarity variation ($R^2= 0.51$) (Table S1.5). In this case, the biggest the NDVI ratio, the more herbaceous cover present (herbaceous vegetation dries in summer and is less detectable in July than in May). As observed previously, there is a higher similarity to NV in sites with more shrub cover and less herbaceous cover, hence the negative relationship. This model was used to create the map of similarity of the revegetated areas to the reference ecosystem (Figure 3.25). It can be observed that similarity is higher (between 30 and 40%) in the north-facing limestone benches area where plantations were made, while lower similarity values are present in the south-facing marl slopes and some of the north-facing limestone slopes.

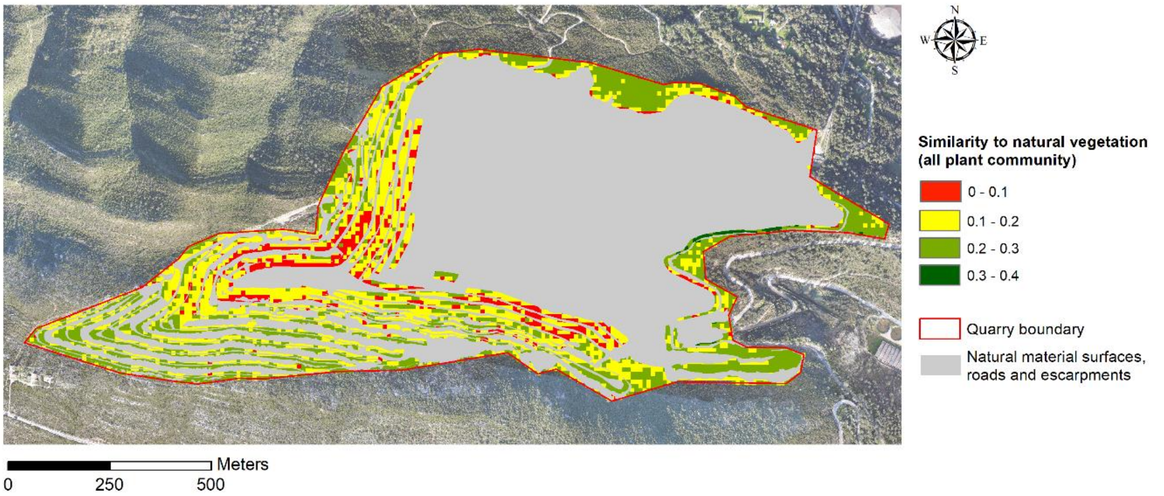


Figure 3.25. Similarity to NV (Sørensen–Dice index) map, considering all plant community, determined from the $\frac{NDVI\ May}{NDVI\ July}$ and slope ($R^2 = 0.51$). Orthophoto from SECIL (2020) with 5 cm resolution.

The similarity (Sørensen–Dice index) to the reference ecosystem of the shrub community was mapped using the same predictors as above, with $\frac{NDVI\ May}{NDVI\ July}$ (negative coefficient, $p<0.01$) and slope (negative coefficient, $p<0.05$) ($R^2= 0.37$) (Table S1.5), and can be seen in Figure 3.26. The spatial distribution of similarity in the shrub community follows the same pattern as in all plant community, and higher similarity values up to 50% can be observed in the north-facing limestone benches.

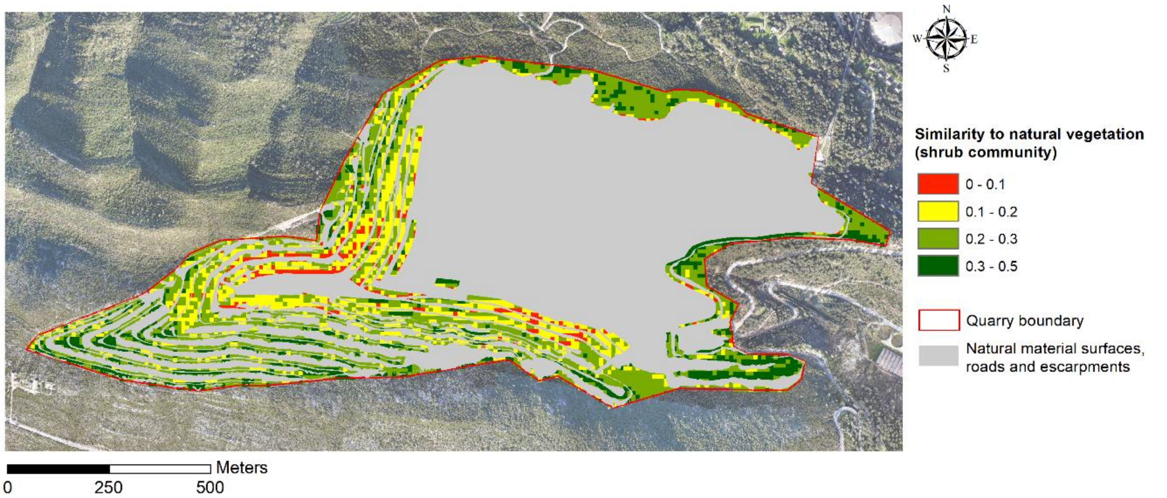


Figure 3.26. Similarity to NV (Sørensen–Dice index) map, considering the shrub community, determined from the $\frac{NDVI\ May}{NDVI\ July}$ and slope ($R^2 = 0.37$). Orthophoto from SECIL (2020) with 5 cm resolution.

3.5. Diversity

3.5.1. General characterisation of the vegetation in the study area

In total, 147 species were recorded, including 50 shrub species, four tree species (three species belonging to *Pinus* and one species of *Fraxinus*) and 93 herbaceous species, totalling 42 botanical families encountered in the field (Table S1.6). It should be highlighted that several species that are protected by the Habitats Directive and are predominantly found in the *Maquis* (NV – reference ecosystem), have already colonized or are close to colonizing some of the oldest revegetated limestone benches, namely *Arabis sadina*, *Iberis procumbens* subsp. *microcarpa* and *Ruscus aculeatus*. Although *A. sadina* was not sampled in the plots, its presence was found in the oldest restored bench, and *I. procumbens* and *R. aculeatus* were observed near the boundary separating the quarry's limestone benches and the surrounding NV. Other species of conservation concern (RELAPE species - Rare, Endemic, Localized, Threatened or Endangered) were recorded, such as *Serapias parviflora*, *Genista tournefortii* and *Gennaria diphylla* (the last two were not sampled, but were present in the oldest limestone benches, mostly near the path located in the middle of the benches).

3.5.2. Taxonomic Diversity

The taxonomic diversity (Shannon-Wiener index) considering all plant community (shrubs, pines and herbaceous species) did not significantly vary with restoration age. Nevertheless, the taxonomic diversity of shrub species significantly increased with restoration age, stabilising after approximately 23 years (Figure 3.27). Contrarily, the herbaceous community shows an opposite relationship by significantly decreasing with restoration age, reaching a peak in the first 10 years and decreasing progressively from that age onwards (Figure 3.27). Additionally, although there was no significant relationship between Shannon diversity and restoration age in the woody community (shrub and pine species) (Figure 3.27), a significant and positive relationship was found between the species richness of woody species and restoration age (Figure S2.39).

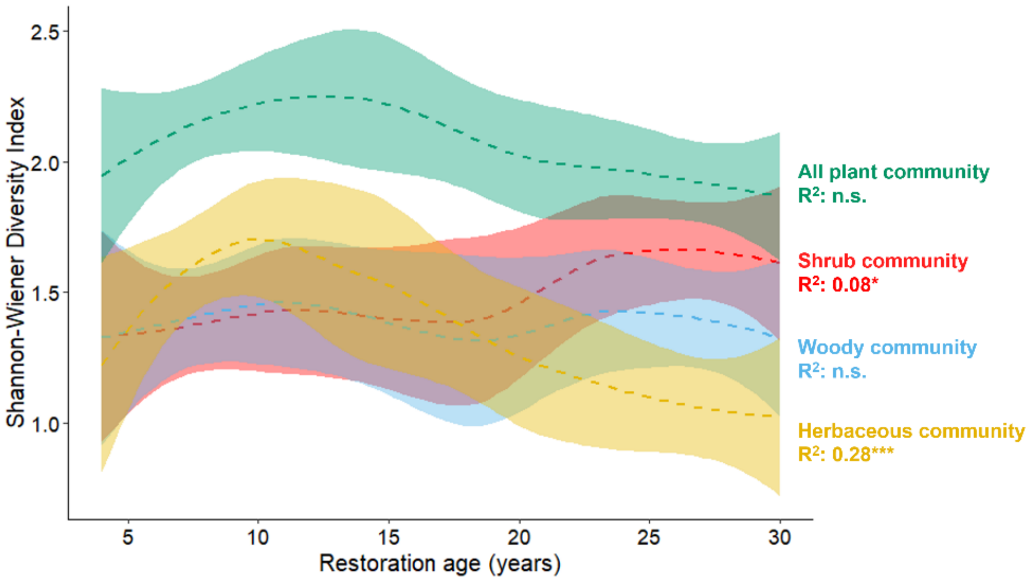


Figure 3.27. Variation of the Shannon-Wiener diversity index with restoration age, in all plant community, shrub community, woody community and herbaceous community. Model p-value: * p<0.05, ** p<0.01, *** p<0.001, n.s. non-significant.

In the shrub and woody community (Figure S2.41 and Figure S2.42, respectively), the Shannon-Wiener index did not significantly differ between the quarry intervention modes and the reference ecosystem, except for HSPL-preFCUL, which had significantly lower values than the NV. Likewise, the index did not significantly differ between the intervention modes and the NV in all plant community (Figure S2.40) and the herbaceous community (Figure S2.43). Species evenness in the woody community, on the other hand, was significantly lower in the PL-preFCUL than in NV (Figure S2.44).

The best predictors for estimating the Shannon-Wiener index in the shrub community were shrub cover ($p<0.001$), intervention mode ($p<0.05$) and slope ($p<0.05$), which explained 58% of the taxonomic diversity variation ($R^2= 0.58$) (Table S1.5). Thus, there is greater taxonomic diversity in places with higher shrub cover and lower slope. Shrub cover ($p<0.001$) was also the best predictor of woody species' taxonomic diversity, explaining 44% of the variation observed ($R^2= 0.44$) (Table S1.5). In the herbaceous community, the restoration age (negative coefficient; $p<0.05$) was the most important predictor of taxonomic diversity, which indicates that sites restored earlier via hydroseeding (most recent slopes) have higher taxonomic diversity ($R^2= 0.28$) (Table S1.5).

Furthermore, the best spatially continuous predictors of taxonomic diversity in the shrub community were the $\frac{NDVI\ February}{NDVI\ July}$ ratio ($p<0.01$) and slope ($p<0.01$), both negatively associated with the Shannon-Wiener diversity index ($R^2= 0.32$) (Table S1.5). From this model, the map in Figure 3.28 was produced. The map shows that the taxonomic diversity index is higher in the quarry areas where plantations exist (in limestone benches) and lower in the quarry areas where herbaceous cover and slope are higher (mostly in marl slopes and some areas in the limestone slopes).

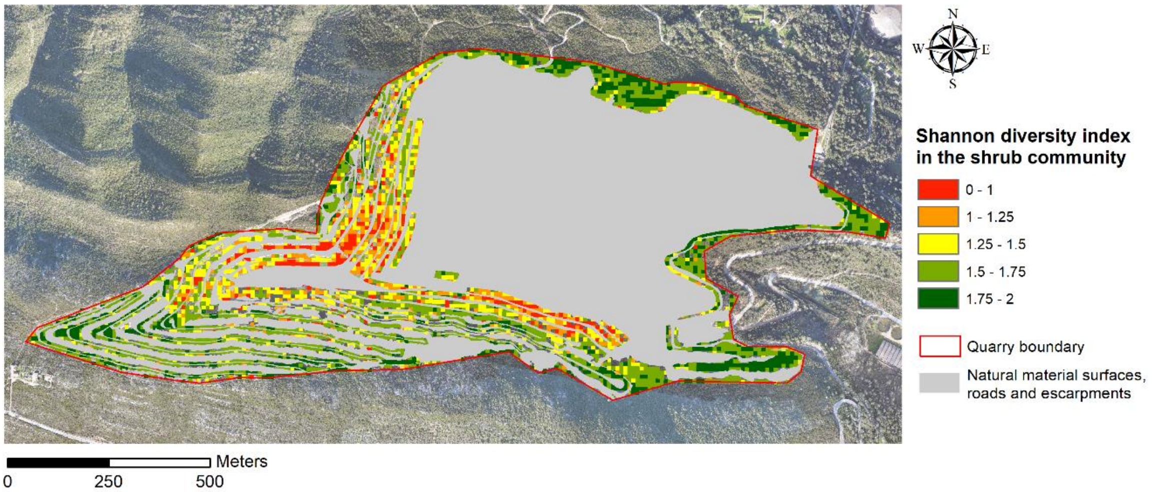


Figure 3.28. Shannon diversity index in the shrub community map, determined from the $\frac{NDVI\ February}{NDVI\ July}$ and slope ($R^2 = 0.32$). Orthophoto from SECIL (2020) with 5 cm resolution.

3.5.3. Functional Fiversity

The functional diversity (FDis) based on multiple functional attributes (multi-trait), which takes into account the variability of functional attributes, does not show a significant linear relationship with restoration age, either considering only shrub species or including pines (woody community) (Figure 3.29). Nevertheless, in the shrub community, it is possible to observe an increase in functional diversity starting approximately 20 years after restoration, suggesting that functional diversity is higher in areas undergoing restoration for a longer period of time, which correspond to the limestone benches where shrub species were planted. In the woody community, functional diversity decreased substantially from the most recent restored plots (maximum value) until around 20 years, and even though it increases thereafter, the older restored sites still had lower functional diversity values than the recently restored ones. Nonetheless, the woody community showed a significant increase in functional richness (which does not consider abundances) with restoration age (R^2= 0.30), such that higher values were found in older restored sites where plantations exist (Figure S2.45). None of the other studied communities showed a significant relationship between functional richness and restoration age. In what concerns the herbaceous community, functional diversity significantly decreased with restoration age. It first increased in the recently restored sites peaking at around 11 years, but then started to decrease and stabilized at low values in older restored sites with more than 20 years (Figure 3.29), as observed with herbaceous cover and taxonomic diversity.

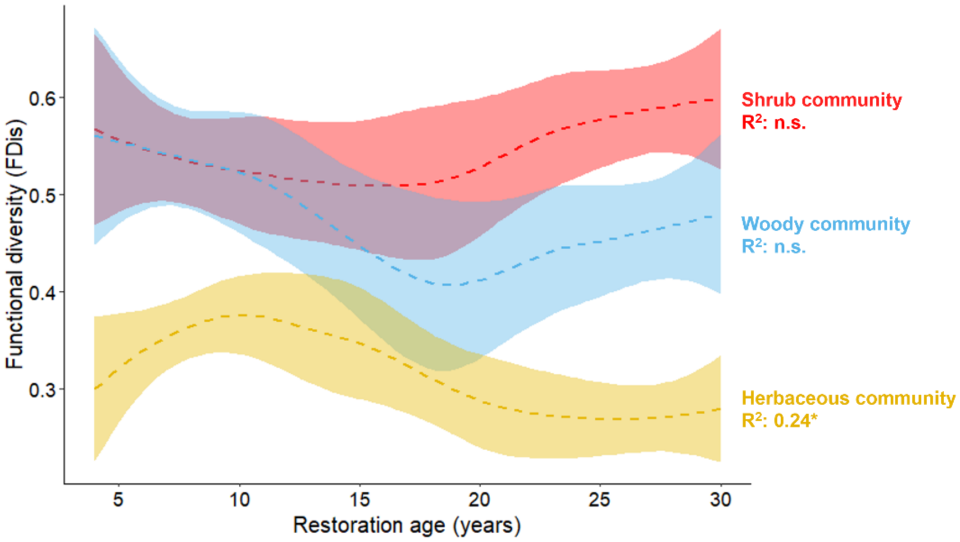


Figure 3.29. Variation of functional diversity (FDis – functional dispersion) with restoration age, in the shrub community, woody community and herbaceous community. Model p-value: * p<0.05, n.s. non-significant.

In the shrub community, functional diversity did not differ significantly between the intervention modes in the quarry and the reference ecosystem (Figure S2.46). However, plantations (PL-preFCUL) and some of the hydroseeded sites (HSPL-preFCUL) had significantly lower values than the reference ecosystem when the abundances of pines (woody community) was considered (Figure S2.47). Functional diversity did not differ significantly between the intervention modes in the herbaceous community (Figure S2.48).

Overall, the predictors that best explained the functional diversity in the restored quarry plots (in the three analysed communities) were linked to the woody layer (shrubs and pines) and restoration age. Functional diversity in the shrub community was best explained by a positive association with shrub density ($p<0.01$) (R^2= 0.14) (Table S1.5). The same pattern was observed for functional diversity in the woody community, which positively correlated with shrub density ($p<0.001$) but negatively correlated with pine cover ($p<0.001$), indicating that functional diversity increased with shrub density and decreased with pine cover in the restored areas (R^2= 0.33) (Table S1.5). In the woody community, functional richness significantly increased with restoration age ($p<0.001$) and shrub cover ($p<0.001$) (R^2= 0.53) (Table S1.5). In the herbaceous community, functional diversity was negatively associated with pine cover ($p<0.001$) (R^2= 0.16) (Table S1.5).

The $\frac{NDVI\ May}{NDVI\ July}$ ratio of the revegetated areas explained 36% of the variation of the functional richness in the woody community (negative coefficient, p<0.001) ($R^2 = 0.36$) (Table S1.5). Similar to functional diversity, the higher functional richness can be found in areas with higher shrub cover, and lower herbaceous and pine cover (Figure S2.50, Figure S2.51 and Figure S2.52). Thus, the negative relationship between functional richness and the NDVI ratio is caused by higher functional diversity values in areas with more shrub cover and less herbaceous cover, where the difference in NDVI between May and July is low. Based on this model, the map in Figure 3.30 was produced.

The limestone benches (North-facing low slopes), which were planted mostly with pine and shrub species, have estimated higher values of shrub cover and thus higher functional richness considering the woody community, which can be observed in Figure 3.30. In contrast, the marl slopes (south and east-facing) in which hydroseeding (with and without plantations) was performed, have higher values of herbaceous cover and, consequently, the lowest values of functional richness in the revegetated areas in the woody community (the same can be observed in some of the north-facing limestone slopes).

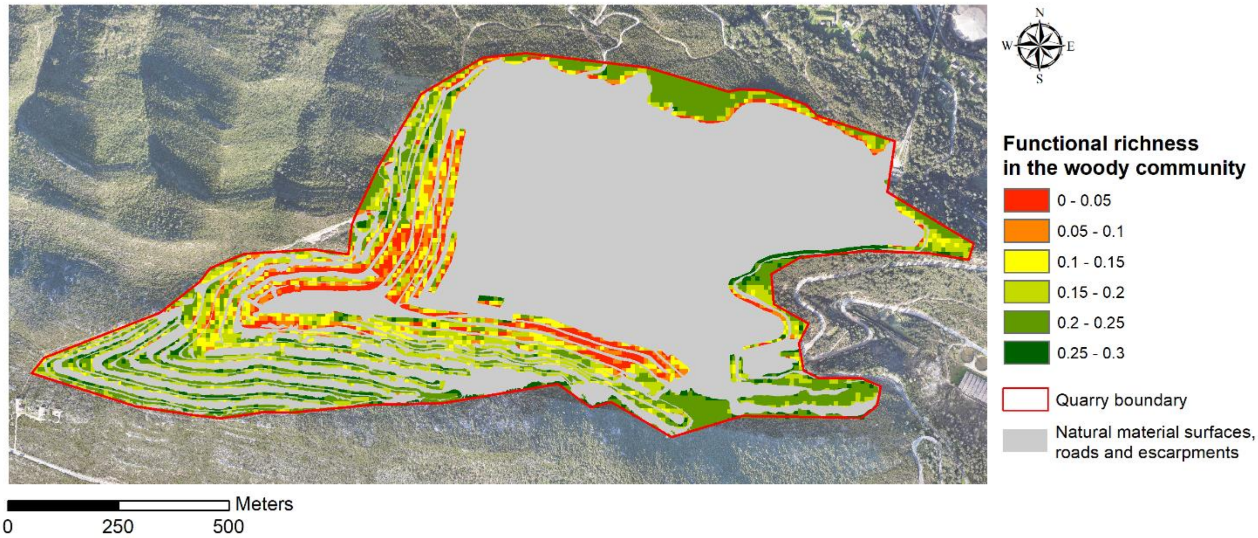
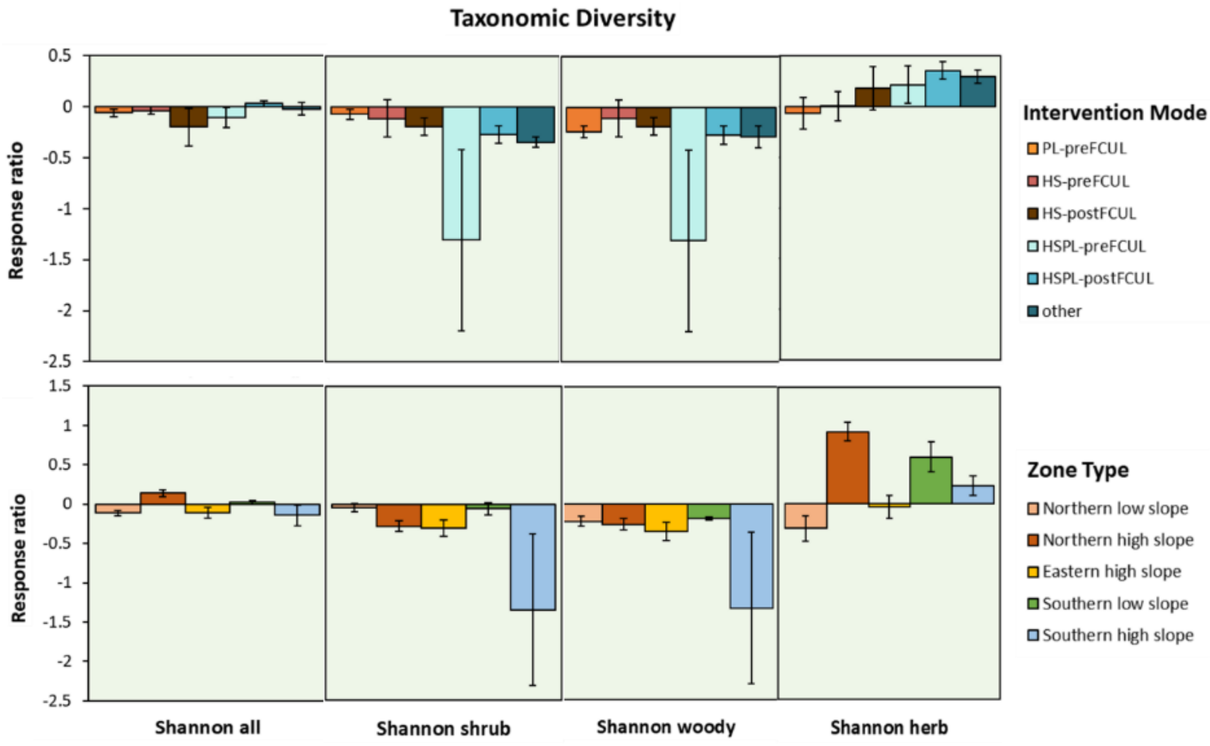


Figure 3.30. Functional richness in the woody community map, determined from the $\frac{NDVI\ May}{NDVI\ July}$ ($R^2 = 0.36$). Orthophoto from SECIL (2020) with 5 cm resolution.

3.5.4. Ecological Restoration Success Index

In general, the restored quarry areas have failed to attain reference ecosystem's levels of taxonomic diversity (Figure 3.31, top panel), especially regarding the shrub and woody communities, while being more diverse in herbaceous species than the NV. Considering the entire plant community, the restored plots have taxonomic diversity values close to the NV's, mostly due to the great diversity in herbaceous species found in the quarry where hydroseeding and mixed interventions were performed, which contributes to an overall higher taxonomic diversity when considering all the plant community. However, when analysing closely the shrub and woody communities, one can observe that taxonomic diversity has lower values than the reference ecosystem. Areas restored with hydrosseding and plantations (HSPL-preFCUL), or areas located in the marl slopes (Southern high slope) are especially distant from attaining recovery when considering this indicator. Nonetheless, the intervention mode and zone type closer to the reference ecosystem's taxonomic diversity values are the plantations (PL-preFCUL) on limestone benches (Northern low slope). The presence of pine species in the woody community, however, brings the diversity values down.

Concerning the functional diversity based on multiple functional attributes (multi-trait analysis) (Figure 3.31, bottom panel), values found in the quarry restored areas were very close to reference levels when considering the shrub community, with the exception of HSPL-preFCUL and some of the marl slopes (Southern high slope). The latter had values well bellow the reference, indicating incomplete recovery of functional diversity in the shrub community. Additionally, the presence of pines in the woody community decreased functional diversity values, especially in the plantations (PL-preFCUL) on the limestone benches (Northern low slope) and some of the marl slopes (Eastern high slope and Southern low slope). Regarding the herbaceous community, functional diversity values were successfully recovered in the restored quarry areas, with the exception of the limestone benches (Northern low slope), which were below the functional diversity values found in the reference ecosystem.



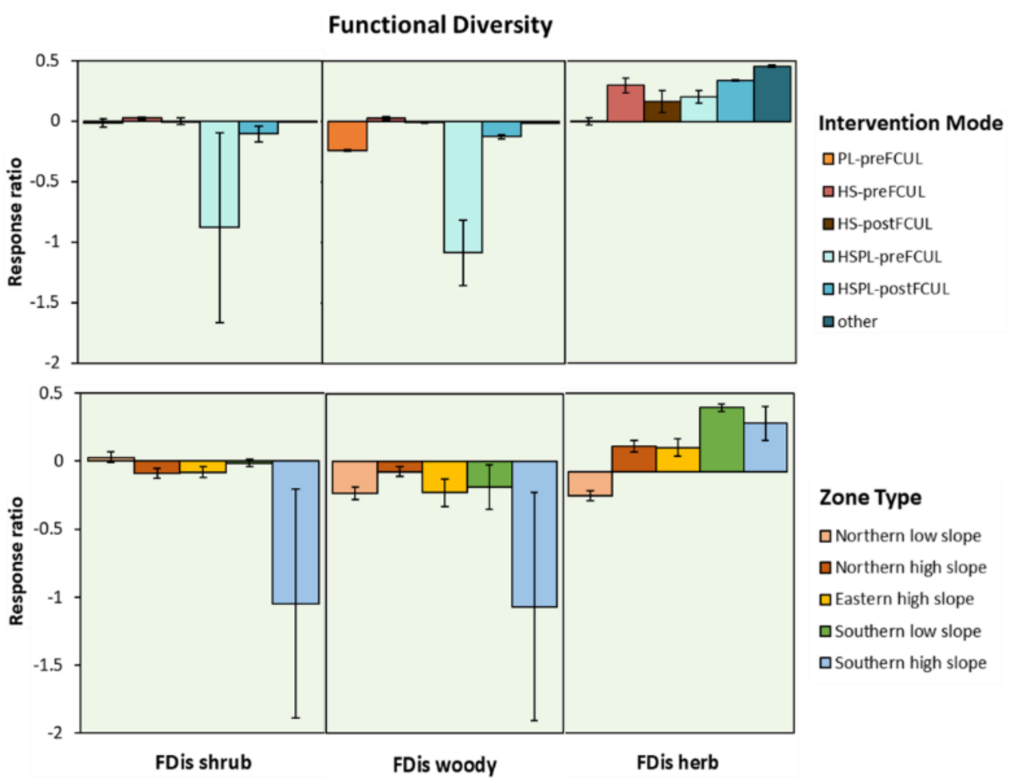


Figure 3.31. Response ratio (restoration success) of diversity variables, namely taxonomic diversity (Shannon-Wiener diversity index – top graphs) and functional diversity (functional dispersion (FDis) –bottom graphs), considering all plant community, shrub community, woody community and herbaceous community, and for the different intervention modes and zone types, compared with the reference ecosystem. Positive values of the indicator under analysis indicate higher values in the restored sites than in the reference ecosystem, values close to zero indicate a high degree or full recovery, and the opposite (incomplete recovery) holds for negative values. Error bars correspond to the standard error of the mean.

3.6. Ecosystem Functioning (functional traits)

Dominance of the functional attributes (categorical traits) considered relevant for assessing the state of the revegetated sites under study, considering their mode of intervention, for the shrub, woody and herbaceous community, can be seen in Figure S2.53, Figure S2.54 and Figure S2.55, respectively.

3.6.1. Pollination

3.6.1.1. Entomophilous pollination

The proportion of species that depend on entomophilous pollination (i.e. by insects) for reproduction, measured using the community weighted mean (CWM), significantly decreased with restoration age, being lower in older restored sites, considering the woody community alone or together with the herbaceous community (Figure 3.32). The proportion of entomophilous species did not differ significantly between intervention modes and the NV when considering both woody and herbaceous communities together (Figure S2.56). However, the proportion of entomophilous woody species was significantly lower in the limestone benches where plantations occurred (PL-preFCUL), which have a greater dominance of pines, than in some of the slopes (HSPL-postFCUL), but still with no statistical difference from the reference ecosystem (Figure S2.57). The influence of pines on this pattern is also reinforced by the fact that the proportion of species with entomophilous pollination decreases significantly with pine cover ($p<0.01$) ($R^2= 0.16$) (Table S1.5). Furthermore, anemophilous pollination predominates in the woody community of PL-preFCUL and HSPL-preFCUL, while in the NV, the proportion between entomophilous and anemophilous species is even (Figure S2.54). However, considering the herbaceous community, entomophilous pollination predominates in all intervention modes and NV (Figure S2.54).

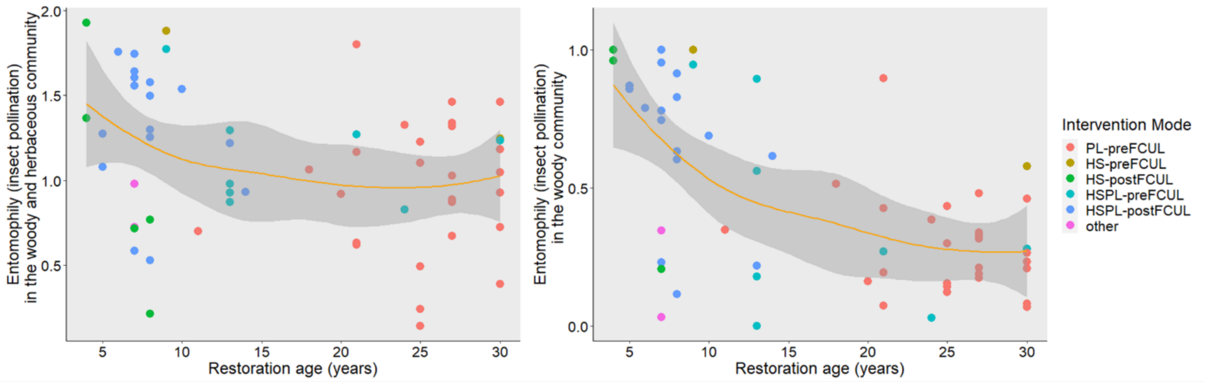


Figure 3.32. Variation with restoration age of the proportion (CWM) of species with entomophilous pollination in the woody and herbaceous community (left graph, $R^2= 0.08$, $p<0.05$) and only in the woody community (right graph, $R^2= 0.34$, $p<0.001$).

3.6.1.2. Flowering duration

Flowering duration (CWM) significantly decreased with restoration age in the woody community (Figure 3.33). Species with shorter flowering durations were more abundant in the oldest limestone benches where plantations occurred (PL-preFCUL), than in most recently restored slopes, where hydroseeding occurred and there is a higher proportion of woody species with longer flowering durations (Figure S2.58). Furthermore, the only intervention mode with significantly higher proportion of species with longer flowering duration than the NV were the hydroseeded plots with FCUL recommendations (HS-postFCUL) (Figure S2.58). However, when only considering the shrub community and excluding pines, flowering duration slightly increases in the plantations. Furthermore, the best predictors for estimating the CWM flowering duration of woody species were pine cover (negative coefficient; p<0.001) and intervention mode (p<0.01) (R^2= 0.49) (Table S1.5).

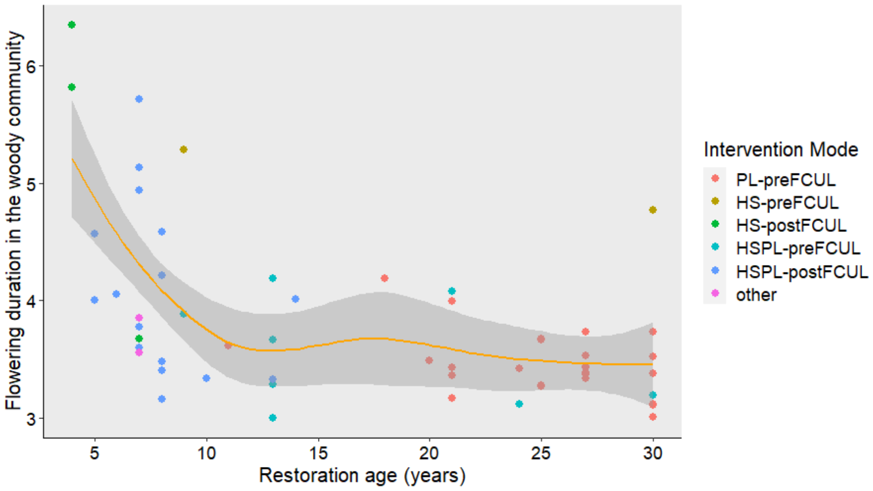


Figure 3.33. Variation of flowering duration (months) in the woody community with restoration age (R^2= 0.41, p<0.01).

3.6.2. Resilience and Adaptation to Fire

The proportion of woody species (shrubs and pines) with a post-fire resprouter strategy decreases with restoration age (Figure 3.34), being significantly lower in the older limestone benches (> 18 years) where plantations occurred (PL-preFCUL) than in the recently hydroseeded slopes (HS and HSPL-postFCUL). Additionally, the proportion of resprouter species in PL-preFCUL was three times lower than in the NV (Figure S2.61). However, the decreasing pattern with restoration age does not exist when analysing only the shrub community (without pines) (Figure S2.60), and the proportion of resprouters in the restored quarry plots does not differ significantly between the intervention modes and the NV (Figure S2.62). Additionally, the most significant predictor for estimating the proportion of resprouters in the woody community was pine cover, as greater pine cover was significantly correlated with smaller proportions of resprouters (R^2= 0.53, p<0.001) (Table S1.5).

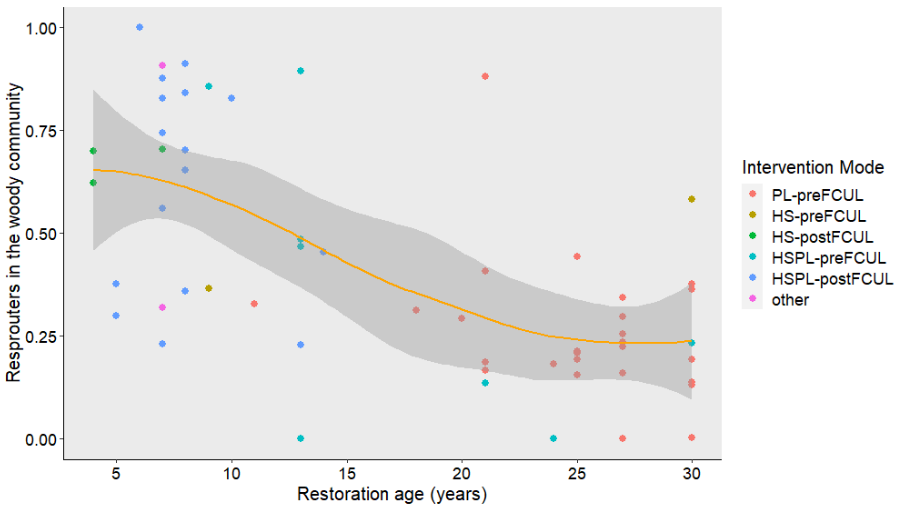


Figure 3.34. Variation of the proportion of resprouters in the woody community with restoration age (R^2= 0.40, p<0.001).

The spatially continuous predictors that best estimated the proportion of resprouters in the woody community were the $\frac{NDVI\ February}{NDVI\ October}$ ratio (p<0.001) and slope (p<0.05), both predictors with a positive correlation with the proportion of resprouters (R^2= 0.48) (Table S1.5). From this model, the map of Figure 3.35 was produced, where the higher proportion of resprouters, between 60 and 100%, can be observed in the South-facing marl slopes and North-facing limestone slopes, while the North-facing limestone benches have the lowest resprouters proportions, mostly from 20 to 40%.

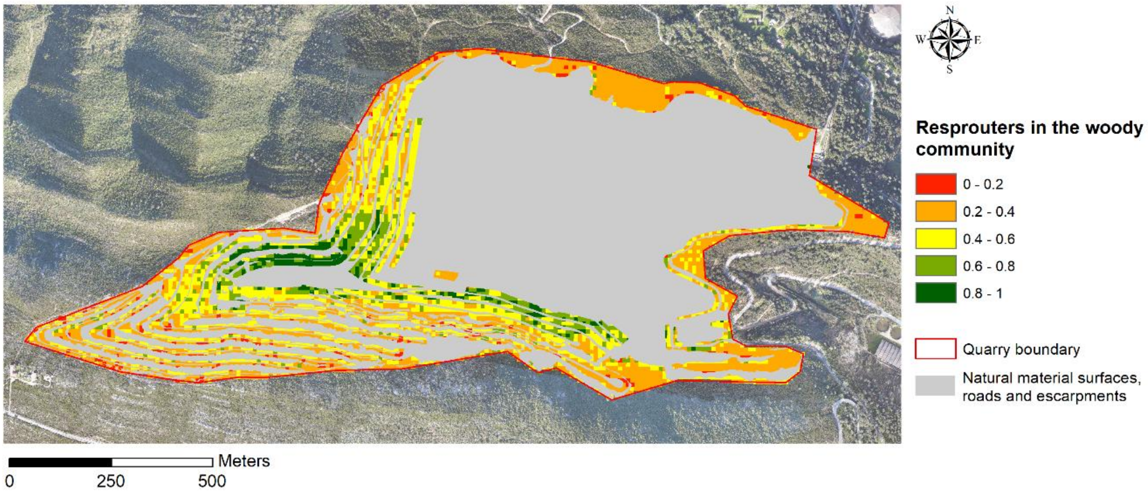


Figure 3.35. Resprouters (CWM) in the woody community map, determined from the $\frac{NDVI\ February}{NDVI\ October}$ and slope (R^2 = 0.48). Orthophoto from SECIL (2020) with 5 cm resolution.

3.6.3. Resilience and Adaptation to Drought

In the woody community, the SLA (CWM) decreased exponentially with restoration age (R^2=0.65) (Figure 3.36, left image), being significantly lower in the plantations of the limestone benches (PL-preFCUL) than in the reference ecosystem and some of the hydroseeded sites (HS and HSPL-postFCUL) (Figure S2.63). Its best explanatory model, which explains 76% of the variation in the values of SLA in the quarry restored plots, included as predictors the restoration age (negative coefficient, p<0.001), pine cover (negative coefficient, p<0.05) and intervention mode (p<0.01) (R^2= 0.76) (Table S1.5).

The proportion of sclerophyllous species in the woody community of the areas undergoing recovery also decreased with restoration age (Figure 3.36, right image). Its best predictor was pine cover (negative coefficient, p<0.001, R^2= 0.52) (Table S1.5). Considering other leaf types, only around 35% of sclerophyll vegetation was found, on average, in the limestone benches plantations (PL-preFCUL), remaining below the significantly higher average proportions found in the reference ecosystem (around 80%) and in the mixed hydroseeding-plantation plots (HSPL-postFCUL) (around 75%) (Figure S2.64). However, when only the shrub community is analysed and the presence and abundance of pines excluded, the proportion of sclerophyll vegetation is similar between the different intervention modes and NV (Figure S2.65).

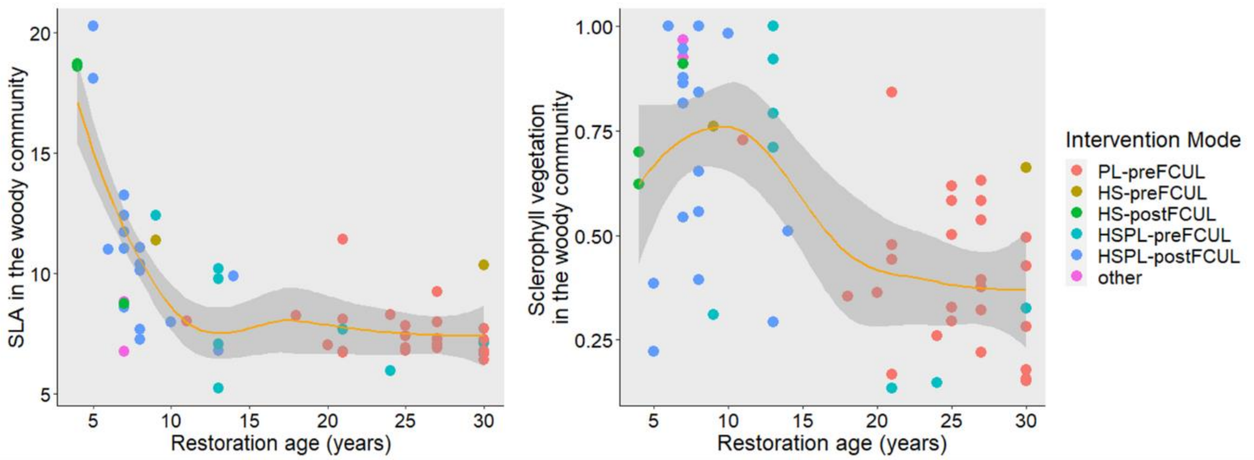


Figure 3.36. Left image: Variation of SLA (mm^2/mg) in the woody community with restoration age in quarry restored plots (R^2= 0.65, p<0.001). Right image: Variation of the proportion of sclerophyll vegetation in the woody community with restoration age in quarry restored plots (R^2= 0.35, p<0.05).

The best spatial predictor of SLA in the woody community was the soil brightness index BI2 (positive correlation, R^2=0.33, p< 0.001) (Table S1.5). More bare soil is present in sites with high SLA values and high density of herbaceous plants, than in sites with lower SLA and high density of pines where the BI2 index detects lower soil cover due to the overlapping of tree canopies. As such, the resulting map from this model (Figure 3.37) shows lower SLA values in limestone benches (facing North) and higher values in the most recently restored slopes (facing East).

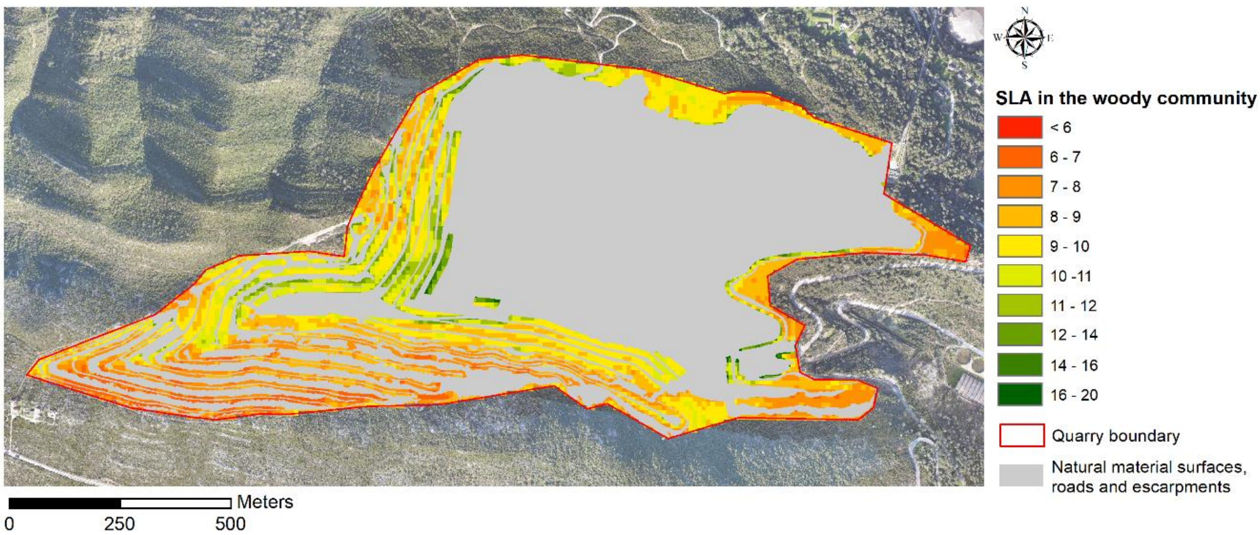


Figure 3.37. SLA (mm^2/mg) (CWM) in the woody community map, determined from the BI2 (R^2 = 0.33). Orthophoto from SECIL (2020) with 5 cm resolution.

The proportion of sclerophyll vegetation in the woody community was best explained by the $\frac{NDVI\ December}{NDVI\ October}$ ratio (p<0.001) and slope (p<0.01), both with a positive coefficient (R^2 = 0.50) (Table S1.5). The model was used to produce the map in Figure 3.38 and it shows from 60 to 100% sclerophyll vegetation in the limestone (facing North) and marl (facing South) slopes, contrasting with the lower values estimated in the marl slopes facing East (which have higher pine cover than most slopes) and in the limestone benches facing North.

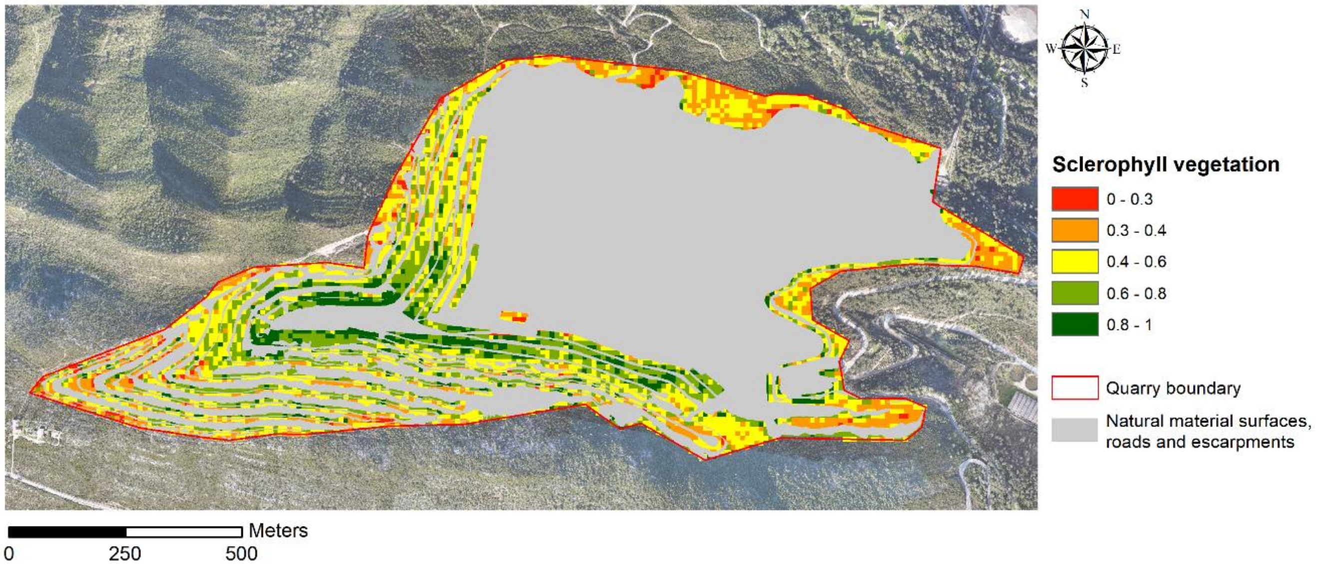


Figure 3.38. Sclerophyll vegetation (CWM) map, determined from the $\frac{NDVI\ December}{NDVI\ October}$ and slope (R^2 = 0.50). Orthophoto from SECIL (2020) with 5 cm resolution.

3.6.4. Dispersal capacity, Interaction with Fauna and Regeneration

3.6.4.1. Dispersal capacity

As can be seen in Figure 3.39, the biggest difference in the strategies responsible for seed dispersal, considering the abundance of species (CWM) composing the shrub community (top image) and woody community (bottom image), is that the proportion of anemochory (seed dispersal by wind) increases considerably when the abundance of pines is taken into consideration, from less than 10% (shrub community) to more than 50% (woody community), which is especially evident in the plantations (PL-preFCUL) intervention mode. This increase in anemochory dispersal reduces the proportion of zoochory (seed dispersal by animals) from around 60% to 30%. The effect of pines on the proportion of seed dispersal strategies is also noticeable in the mixed intervention mode (HSPL-preFCUL), which sees an increase in the proportions of anemochory from 2% to 30%. In the reference ecosystem, however, less than 5% of seed dispersal is carried out by the wind (barochory), as the existing plant species mostly rely on animals (zoochory) (80%) and to a lesser extent (15%) on gravity (barochory) for seed dispersal.

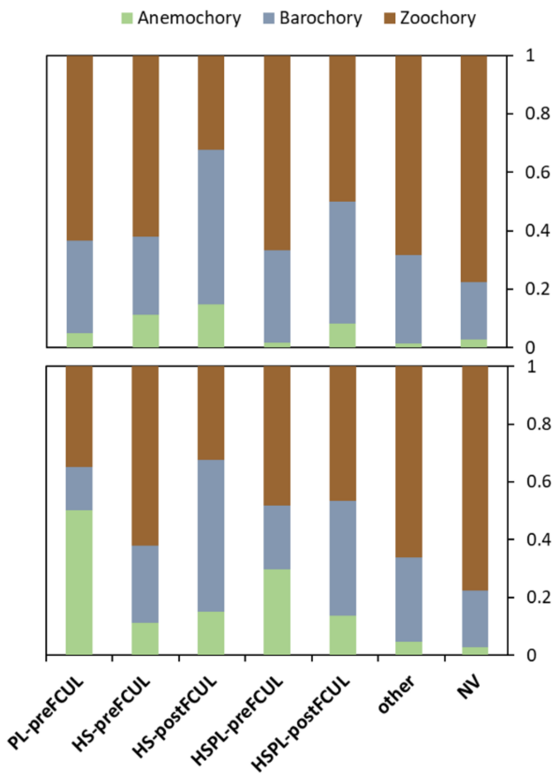


Figure 3.39. Proportion of the different seed dispersal strategies (CWM) in the different intervention modes and NV (reference ecosystem) in the shrub community which excludes pine abundances (top image) and woody community which includes pine abundances (bottom image).

3.6.4.2. Interaction with fauna

Shrub species that produce dry fruit predominate in most of the intervention modes in slopes (HS-preFCUL, HS-postFCUL, HSPL-postFCUL), while species producing fleshy fruits dominate in the plantations (PL-preFCUL) and some slopes (HSPL-preFCUL and “other”) (Figure S2.53). Despite fleshy fruits being the most dominant fruit type in the plantations considering shrub species (Figure S2.53), this proportion changes when taking into consideration the abundance of pines, which more than doubles the proportion of dry fruits in the community from 30% to almost 70% (Figure 3.40). This increase in dry fruits proportion when comparing the woody to the shrub community can also be observed in the mixed intervention mode (HSPL-preFCUL). On the other hand, the reference ecosystem shows a balanced proportion of fleshy and dry fruits, with approximately 50% of each fruit type. The fruit type is mainly dry in the herbaceous community in the quarry intervention modes and NV (Figure S2.55). Additionally, seed mass was significantly higher in the NV than in the quarry intervention modes (Figure S2.66).

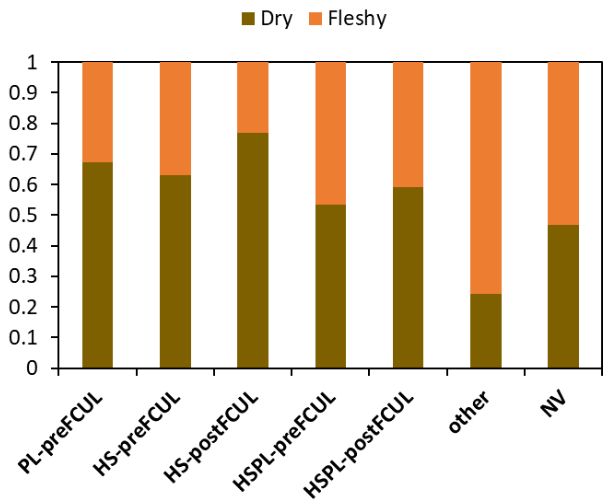


Figure 3.40. Proportion of the different fruit types (CWM) in the different intervention modes and NV (reference ecosystem) in the woody community.

3.6.4.2. Regeneration

Seedling density showed a significantly positive relationship with restoration age (Figure 3.41). Nevertheless, it did not differ significantly between intervention modes. The maximum seedling density was higher in the NV than in all the intervention modes at the quarry (Figure S2.67).

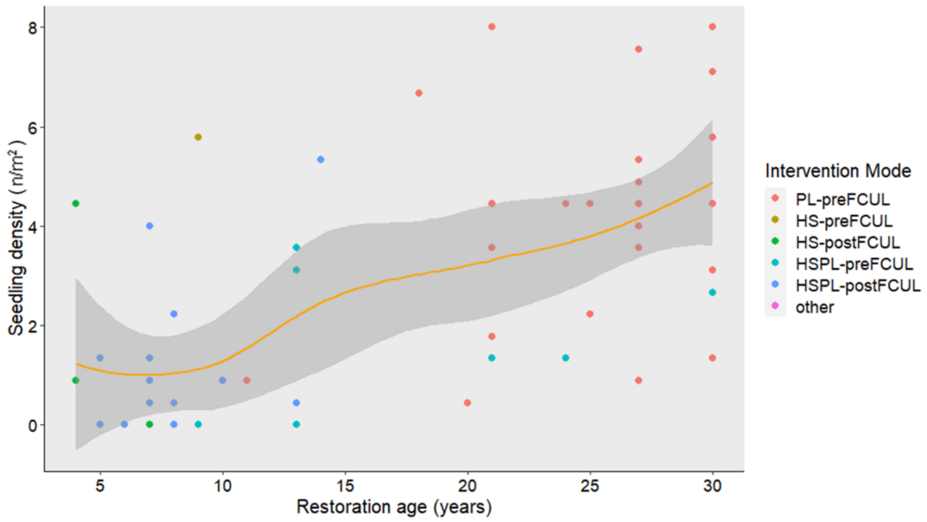


Figure 3.41. Variation of the seedling density (n/m^2) in the quarry restored plots with restoration age ($R^2 = 0.41$, $p<0.001$).

The most important factors for estimating seedling density were restoration age ($p<0.001$) and shrub density ($p<0.001$), both with a significant positive relationship with seedling density and with a negative interaction ($p<0.001$) between restoration age and shrub density ($R^2=0.67$) (Table S1.5).

The spatially distributed predictors that best estimated seedling density were the $\frac{NDVI\ February}{NDVI\ October\ 2}$ ratio ($p<0.001$) and slope ($p<0.01$), both with a negative relationship with seedling density ($R^2=0.56$) (Table S1.5). This negative relationship is due to seedling density being higher in the older restored limestone benches where less herbaceous cover (positively correlated with the NDVI ratio) is present and the slope is lower than in the marl and limestone slopes. The map in Figure 3.32 was produced from this model and shows higher seedling density values, up to seven seedlings per m^2, in the limestone benches (facing North). In contrast, the limestone and marl slopes (facing North and South/East, respectively) were estimated to have less than two seedlings per m^2.

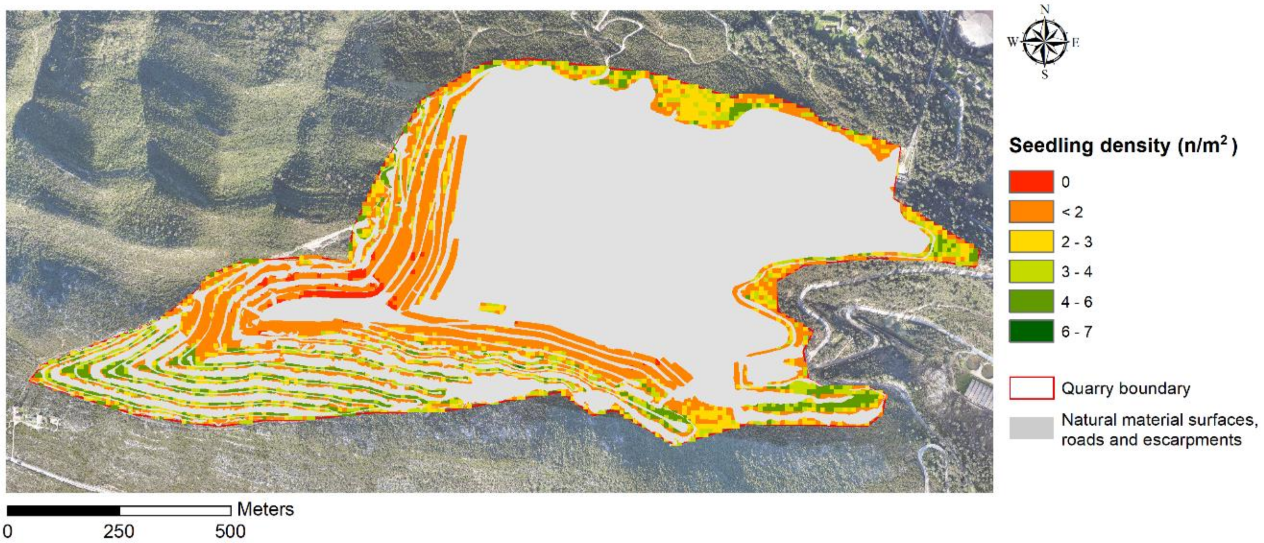


Figure 3.42. Seedling density (n/m^2) map, determined from the $\frac{NDVI\ February}{NDVI\ October\ 2}$ and slope ($R^2 = 0.56$). Orthophoto from SECIL (2020) with 5 cm resolution.

3.6.5. Ecological Restoration Success Index

In general, ecosystem functioning indicators have not recovered completely to the desired (reference ecosystem) levels in the intervention modes and zone types of the restored quarry areas (Figure 3.43).

The proportion of woody species with entomophily, which relates to pollination by insects, is below reference levels especially in the limestone benches' plantations (PL-preFCUL, Northern low slope), the hydroseeding/plantations intervention mode (HSPL-preFCUL) and the marl slopes facing South. Contrarily, the limestone slopes (Northern high slope) and sites where hydroseeding (HS) and mixed with plantations after FCUL recommendations (HSPL-postFCUL) were performed, had higher abundance of species with entomophily than the reference ecosystem. Furthermore, woody species with longer flowering duration are more abundant in the limestone slopes and hydroseeding (HS) intervention modes than in the reference ecosystem, and values remain close to reference levels in the other intervention modes and zone types.

However, the proportion of species in the woody community with resprouting strategy, which confers high ecosystem resilience to fires, is still well below the reference ecosystem's levels in all intervention modes, except for the hydroseeed HS-postFCUL plots. The plantations (PL-preFCUL) in limestone benches (Northern low slope) and mixed hydroseeding and plantations (HSPL-preFCUL) are the most distant from the restoration goals of attaining NV's resilience to fire.

The SLA, which is indicative of resilience and adaptation to drought, is lower than the reference ecosystem's in the limestone benches' plantations (PL-preFCUL, Northern low slope), HSPL-preFCUL and marl slopes facing South, and higher than the reference levels in the hydroseeded HS-postFCUL plots. The proportion of sclerophyll vegetation, which is also important in the resilience and adaptation to drought, is below the NV levels in all intervention modes and zone types, except for the South-facing marl slopes (Southern high slope) which have the most similar proportion of sclerophyllous vegetation to the reference ecosystem.

Seedling density is the ecological indicator with the highest difference from the reference ecosystem, with all intervention modes and zone types below reference levels. Nonetheless, the plantations in the limestone benches (PL-preFCUL) and some of the hydroseeded sites (HS-preFCUL) showed closer values to the seedling density found in the NV than the other intervention modes, indicating high self-sustainability in those areas. The marl slopes facing South (Southern high slopes) and HS-postFCUL are the most distant from offering the same seedling density as the reference ecosystem.

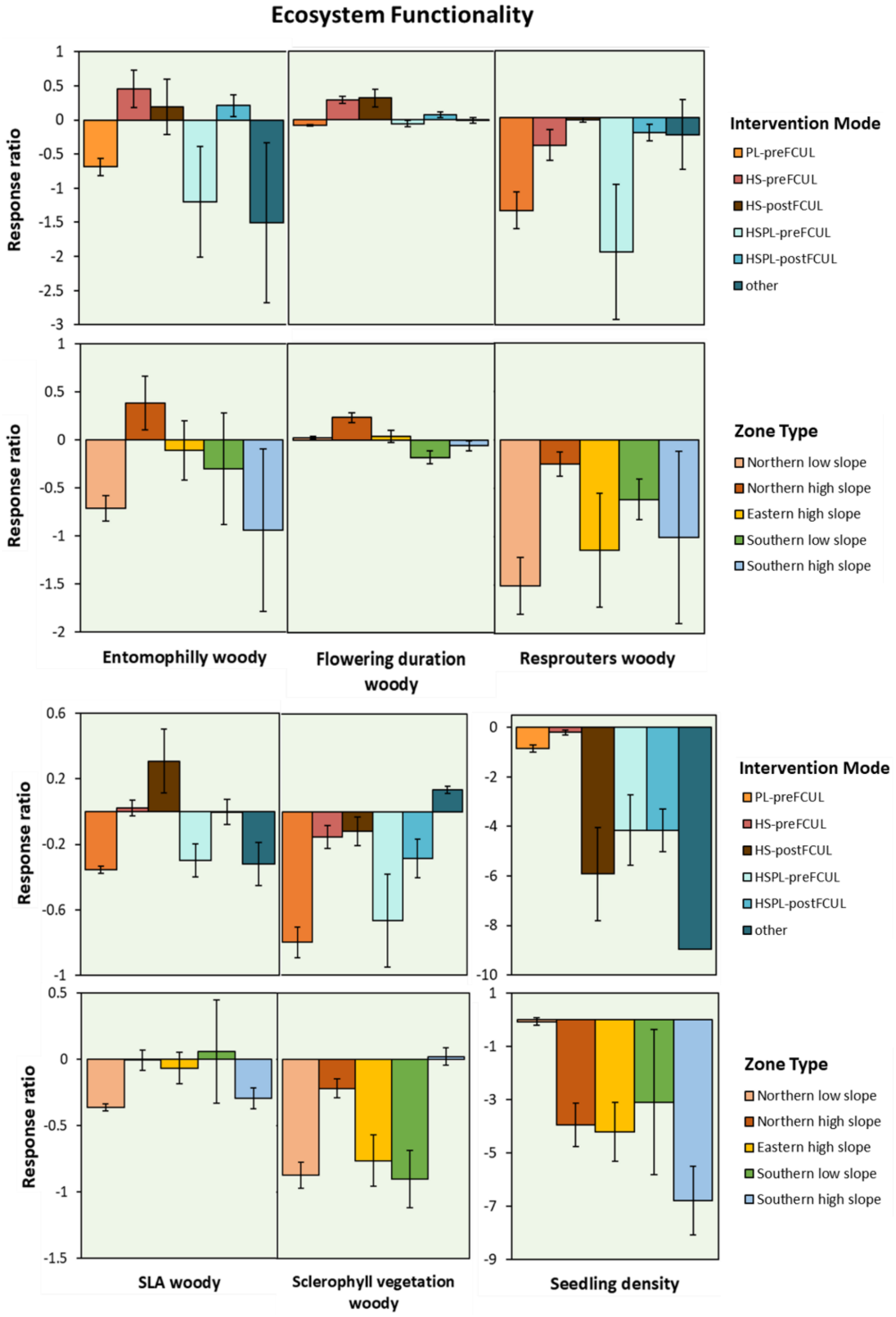


Figure 3.43. Response ratio (restoration success) of ecosystem functioning variables, such as pollination (entomophily and flowering duration in the woody community), resilience and adaptation to fire (resprouters in the woody community), resilience and adaptation to drought (SLA and sclerophyll vegetation in the woody community), and regeneration (seedling density), for the different intervention modes and zone types, compared with the reference ecosystem. Positive values of the indicator under analysis indicate higher values in the restored sites than in the reference ecosystem, values close to zero indicate a high degree or full recovery, and the opposite (incomplete recovery) holds for negative values. Error bars correspond to the standard error of the mean.

CHAPTER 4 | DISCUSSION

Natural colonisation in limestone quarries is very slow due to unfavourable climatic, edaphic and biotic conditions (Correia et al., 2001). Thus, ecological restoration of these degraded areas is necessary to increase biodiversity and ES delivery by creating new habitats or restoring existing ones (Moreno-Mateos et al., 2020; Salgueiro et al., 2020). The evaluation of ecological restoration practices is essential to determine the success of interventions and provide new guidelines or recommendations for adaptive ecosystem management. Furthermore, it is meaningful to do this assessment by analysing the maximum ecological components possible to capture the complexity and multidimensionality of ecosystems – not only focusing on single components, such as carbon sequestration, but also investigating other important ES and ecological indicators relevant in the context of ecological restoration (Ruiz-Jaen & Mitchell Aide, 2005).

This work aimed to assess the success of the ongoing ecological restoration at the SECIL-Outão limestone quarry, located within the protected area of the ANP, by using field and RS data to model and map several ES and their ecological indicators. Several ecosystem components, which can be associated with ES or relevant key ecosystem attributes used to evaluate ecological restoration, were analysed in this research, namely: 1) Productivity and Carbon Sequestration; 2) Soil Fertility and Decomposition; 3) Habitat Quality; 4) Similarity to Natural Vegetation; 5) Diversity (Taxonomic and Functional), and 6) Ecosystem Functioning (Pollination, Resilience and Adaptation to fire, Resilience and Adaptation to Drought and Dispersal Capacity, Interaction with Fauna and Regeneration).

Furthermore, the different revegetation ages in the quarry areas under recovery allow the study of the succession along a chronosequence of 30 years (Barbour et al., 1999; Correia et al., 2001). Chronosequences are a common approach to study plant succession, however, this approach is sometimes criticized, as many local factors can influence succession development in a site, from its specific geographic location to derived local climatic conditions (Walker & Moral, 2003). Additionally, differences in restoration techniques (or intervention modes) may constitute an obstacle to using chronosequences in this type of study (Majer, 1989). The best method to study succession would be observing it as it proceeds in time and in fixed areas (Majer, 1989; Walker & Moral, 2003). However, it might be difficult to do so due to the relatively long temporal scale in which the succession takes place (Barbour et al., 1999). Thus, the indirect method of chronosequence is often used, always taking into consideration the possible existence of different sources of variation from site to site (Walker & Moral, 2003).

As the analysis of key ecological indicators and ES based only on the chronosequence would be insufficient to understand the complexity of the factors influencing the restoration trajectory, statistical modelling was also performed. Modelling was the principal methodology to evaluate which variables most influence the studied ecological indicators and ES, and to identify limiting factors or promoters of recovery and suggest recommendations for adaptive ecosystem management. Then, models based on field and RS data were created to produce maps of the estimated spatial distribution of the ecological indicators and ES, to inform adaptive management decisions. Lastly, to compare the absolute values of the ecosystem attributes between the different restoration intervention modes and the reference ecosystem, statistical analysis and ecological restoration success indexes of each ecosystem component were calculated to determine if ecosystem recovery of the restored areas, based on the reference ecosystem, was achieved in the different intervention modes and zone types (zone types effect was only assessed in the ecological restoration success index).

4.1. Primary Productivity and Carbon Sequestration

Productivity (FAPAR) and soil carbon sequestration (chemical analysis)

Productivity and carbon sequestration are part of the "Climate regulation" ES group from the CICES framework and the "Ecosystem function" SER key ecosystem attribute for evaluating ecological restoration (*section 2.11*). Overall, the restored quarry sites studied have not recovered to the desired values of primary productivity and soil carbon sequestration of the reference ecosystem, regardless of the intervention mode or zone type. Nonetheless, the significant estimated contribution for carbon sequestration by the restored quarry areas (5313 Mg) is an encouraging example of how ecological restoration of degraded ecosystems can result in a positive contribution to sequestering carbon, effectively improving the regional sequestration capacity in mitigating climate change.

In this research, the RS biophysical index FAPAR was used as a proxy for primary productivity. Primary productivity is the process resulting from the photosynthetic activity of plants and determines biomass accumulation in forests (Holmes & Likens, 2016). In fact, primary production provides the organic carbon that ultimately supports the metabolism of autotrophs and heterotrophs in ecosystems, representing the energy that plants use for growth, development and reproduction, and the material basis for the survival and reproduction of biological groups (Pace et al., 2021; X. Wang et al., 2022). Ultimately, the fate of most primary production is conversion back to CO_2 via respiration; however, some residual carbon accumulates in biomass and organic matter in soils and sediments (Pace et al., 2021). The storage and fate of this unrespired carbon is important in understanding ecosystems as carbon sinks for atmospheric CO_2. Thus, productivity and carbon sequestration estimates are fundamental to evaluate the potential of forest ecosystems and their capacity to sequester carbon and help offset rising atmospheric CO_2 concentration (Gower et al., 1997).

Primary productivity (*section 3.1.1*) increased in the first 15 years of the chronosequence and then stabilized for the next 15 years. While the hydroseeded sites (HS and HSPL) with mostly herbaceous and some shrub species have a clear increasing productivity with restoration age, peaking at 15 years after restoration, the older restored sites with pine and shrub plantations (PL-preFCUL) have similar productivity values between 20 and 30 years after restoration, not increasing considerably in this period. Such suggests that the older plant communities in the plantations of the limestone benches are in a limiting environment in which productivity may not be far from its potential maximum level and the current biomass may not increase for the older restored sites unless some intervention is done (Whittaker et al., 1974). The significantly higher average FAPAR in the NV suggest better vegetation health, greater cover and higher primary productivity in the reference ecosystem than in the quarry revegetated areas. These results indicate that, despite the high abundance of pine trees in the quarry benches that have been under restoration for more than 30 years, primary productivity in the restored areas falls short of the productive dense maquis that characterizes the reference ecosystem. The lack of studies that assessed primary productivity using FAPAR in ecological restoration of Mediterranean ecosystems limits the comparison of FAPAR values between this research and others.

The soil chemical analysis (*section 3.1.2.1*), used to determine soil carbon sequestration, revealed that SOC slightly increased along the chronosequence, in accordance with a previous study by Correia et al. (2001) where SOM and, consequently, SOC, were higher on older revegetated benches than in recently revegetated ones. However, these values are still significantly below the reference ecosystem's values, which stores around three to up to 10 times more SOC, indicating that the restored sites are distant from giving the same contribution to soil carbon sequestration as their reference ecosystem. Carbon levels result from the interactions of several ecosystem processes, of which photosynthesis, respiration, and decomposition are key. Typically, arid regions have low SOC levels, mostly due to low primary production (Ontl & Schulte, 2012). Water availability is the main limiting factor for plant growth and productivity in drought-prone environments such as the Mediterranean region (Baquedano et al., 2008). In addition to water stress, limiting factors that could be hindering the productivity potential and consequent carbon sequestration in the restored areas are nutrient availability or minimum temperature (Gea-Izquierdo et al., 2015; Melillo et al., 1993).

Previous research has suggested that, in the medium term, the introduction of species with higher resource requirements (i.e. fast-growing conifers and late successional shrubs) might create a system with high competition and possibly slow growth rates (as observed with productivity after 15 years), due to exhaustion of limited resources (Nunes et al., 2014), as will be further discussed in *section 4.2*. Additionally, previous studies have found that higher temperatures and lower precipitation decreased productivity, a possible scenario for the potential consequences of climate change and CO_2 increase in Mediterranean forests, particularly for the restored areas in the quarry (Nunes et al., 2015). Since increases in temperature combined with a decrease in water availability could lead to additional drought zones (Christensen et al., 2007), a decline in biomass growth and primary production should be expected. As such, adaptive management interventions should be considered (Melillo et al., 1993). In the next *section 4.2*, factors such as soil fertility and decomposition variables limiting the primary productivity and carbon sequestration will be further discussed, as well as recommendations for possible interventions in the restored areas.

The positive correlation between pine and shrub cover with the FAPAR index suggests higher primary productivity and aboveground carbon sequestration in sites with higher shrub and pine cover. In addition, soil carbon was also positively associated with pine cover, although its effect was only significant in the plantations (PL-preFCUL). Such indicates that revegetation with woody species effectively contributed to increasing primary productivity and carbon sequestration in the quarry. In the future, revegetating new sites where carbon storage is currently absent should be considered, preferably with the native species present in the *Maquis* of the reference ecosystem. New revegetated sites could contribute to the decrease of atmospheric CO_2 rates by capture through plants and soil and also help mitigate the effects of climate change in the region by providing ES.

Even though reforestation is a necessary treatment in degraded quarries to fasten the delivery of ES (such as carbon sequestration), using the appropriate types of trees or shrubs is essential to increase resilience for future climate change-induced scenarios (Kowalska et al., 2020; Moreno-Mateos et al., 2020). Pine plantations contribute to ES, such as carbon sequestration, but they also negatively affect biodiversity and can cause environmental problems (Onaindia et al., 2013). Although areas restored with pine could potentially store more carbon than the NV (as will be discussed in the InVEST analysis below), these are still small areas that are not as dense and that differ from the NV in terms of soil fertility and species composition, having species that are not as prepared for climatic stresses (like drought and fire), which again reinforces that carbon cannot be the only indicator of restoration success (Gann et al., 2019). Thus, restoration actions should not focus solely on increasing forest carbon stocks, since there are trade-offs between protecting biodiversity and increasing carbon sequestration (Venter et al., 2009). For this reason, replacing pines with native species could be a positive intervention, not for carbon storage individually, but for multiple ES (Onaindia et al., 2013). The trade-offs between pine plantations, carbon storage, diversity, and others, will be further analysed in the following sections. Other adaptive management interventions that could improve soil carbon in the restored areas to meet the reference ecosystem levels should be investigated, such as increasing shrub density or the appropriate enrichment of soil (Kowalska et al., 2020). This applies especially to the limestone slopes

(Northern high slope), which had the furthest values of productivity and SOC from their reference. Such interventions will be discussed more thoroughly in the next *section 4.2*.

Carbon sequestration (InVEST analysis)

Cartography of the quarry revealed that more than 23 ha have been recovered in the quarry by revegetation, which corresponds to 25% of the quarry's area. The InVEST model, which calculates carbon sequestration based on the land cover type and carbon pool values, estimated that since restoration began more than three decades ago, a total of 5313 Mg of carbon has been sequestered from the atmosphere, which is equivalent to the sequestration of the annual CO_2 emissions of 311 cars. Such contribution positively impacts the restored areas' climate regulation by reducing atmospheric CO_2 concentrations.

Furthermore, revegetated areas where pines are dominant (limestone benches) were estimated to store higher values of carbon than areas with dominant herbaceous or sclerophyllous cover (marl and limestone slopes). Because conifers can store a large amount of carbon, mainly in the soil but also in their aerial and belowground parts, they would contribute more to sequestration than the NV (composed mainly of sclerophyllous vegetation). However, it is relevant to note that carbon sequestration estimates depend on the stipulated carbon pool values and the land cover type assigned to each land unit; therefore, it only serves as an estimating tool and further research to confirm and validate these values is lacking. The importance of validating these estimates is further demonstrated by the conflicting results found between the mapped soil carbon sequestration determined by the InVEST model and the carbon in soil determined by chemical analysis. The InVEST maps showed higher sequestration in the limestone benches' soils where pine trees are dominant than in the NV soils where sclerophyll vegetation dominates. However, chemical analysis revealed that the soil carbon storage of NV is significantly higher than in the limestone benches, which exposes the weakness of the InVEST model or the carbon pool values used for accurately predicting actual soil carbon storage values. Contrarily to the values in the InVEST carbon sequestration map, the results of soil carbon by chemical analysis are in accordance with the FAPAR map, which also shows higher FAPAR values in the NV than in limestone benches (North-facing), suggesting higher primary productivity and correspondingly higher carbon storage (Keeling & Phillips, 2007) in the undisturbed areas of the reference ecosystem.

Potentially, conifers would store higher amounts of carbon in soil than sclerophyll vegetation, but these are not the values found through the SOC chemical analysis. Despite the lack of studies regarding carbon sequestration in Mediterranean systems, it has been reported that tree type considerably affects the SOC stock capacity (Muñoz-Rojas et al., 2015). Coniferous species, for instance, have shown a remarkably lower capacity to store SOC than broad-leaved forests such as oaks (De Vries et al., 2003). Additionally, it is still uncertain how SOC sequestration under afforestation (i.e. establishment of a forest in an area where there was no previous tree cover) is affected by the soil type, as many factors are involved such as SOM decomposition and litter production (Paul et al., 2002; Vejre et al., 2003). As stated before, other reasons for the conflicting results may be that the soil carbon pool values in the literature do not adequately represent the vegetation and carbon storage found in the soils of the Maquis or the restored areas (e.g. soil carbon values for Aleppo pine specifically in the soils of the quarry limestone benches, which are poorer than the reference ecosystem soils). Additionally, attributing revegetated areas a single dominant land cover type results in an overestimation of carbon values. An example of this are the limestone benches, which have a dominance of conifers that leads to the attribution of the carbon pool value correspondent to "coniferous vegetation". This analysis ignores the existing shrub layer that contains a variety of shrub native species that contribute less than conifers to soil carbon values (Table 2.6), rendering soil lower carbon contents. Thus, the determination of carbon storage values for classes of mixed vegetation types should be addressed in the future for a more accurate estimation of carbon values in the restored areas.

4.2. Soil Fertility and Decomposition

Soil fertility and decomposition are part of the "Soil formation and composition" ES group from the CICES framework and the "Ecosystem function" and "Physical conditions" SER key ecosystem attributes for evaluating ecological restoration (*section 2.11*). Soil is an essential ecosystem component that supports biotic communities and largely determines ecosystem functioning. Soil fertility and nutrient availability indicators include SOM, SOC, N, P and pH. Mediterranean ecosystems are frequently nutrient-limited (Sardans et al., 2006), with N and P being the most limiting elements for plant productivity as they are essential nutrients for plant growth and development (Gong et al., 2020; Vitousek et al., 2010). The SOM, which mostly results from an accumulation of dead plant matter and partially decayed and resynthesized plant and animal residues (Bohn et al., 2001), improves soil quality through increased water retention and nutrients. In turn, these changes result in greater productivity of plants in natural environments and increased diversity of microorganisms by providing them with a source of nutrients (Kowalska et al., 2020). It also improves soil structure and reduces erosion, improving surface and groundwater quality (Ontl & Schulte, 2012). Additionally, soil microfauna such as bacteria and fungi, called decomposers, drive SOM decomposition, a fundamental process with great importance in biogeochemical cycles, especially C and N nutrient cycling (Glassman et al., 2018). Decomposers act by decomposing organic matter into mineral elements reused by plants (as a source of nutrients) or reintroduced into biogeochemical cycles. Furthermore, the availability and uptake of some plant nutrients are affected not only by decomposition rates but also soil pH (Gondal et al., 2021).

The results show that the condition and functioning of the soil are limited in the revegetated areas in terms of parameters such as SOM, SOC, N, P, and pH, and that these have not recovered to reference ecosystem levels (*section 3.2*). The functioning of the biogeochemical cycle also differs from the reference situation, although it depends on the intervention mode or zone type considering its dominant vegetation type, suggesting specific imbalances related to the decomposition (C:N ratio) and consequently the availability of soil nutrients.

The SOC, SOM, and the C:N ratio parameters significantly increased with restoration age. An increase of SOM with restoration age indicates development of soil quality since organic matter significantly improves the soil's capacity to store and supply essential nutrients (such as N and P). However, this increase is slow, as the recovery of soil components such as SOM is a slow process (Carpenter & Turner, 2000; Walker et al., 2012). Nonetheless, SOM allows the soil to cope with changes in soil acidity and helps soil minerals to decompose faster. Apart from serving as a reservoir of nutrients for plants, it also improves soil aggregation, increases nutrient exchange, retains moisture, reduces compaction, and increases water infiltration into soil. Thus, an increase of SOM and SOC levels benefits soil fertility and primary productivity and indicates a positive contribution from the restoration while also reflecting the importance of time in recovering soil health. However, as will be discussed next, the high C:N ratio in the older restored limestone benches may be negatively affecting soil nutrient availability and limiting the ecosystem's recovery.

Despite increasing with age, SOM in the restored plots was significantly lower than the SOM in the reference ecosystem. The high SOM values in the reference ecosystem might be due to its characteristic "terra rossa" soils, which are shallow in depth but are fertile and rich in SOM and SOC, contrasting with the quarry soils which started as a poor marl substrate devoid of soil microorganisms. The quarry soils have slowly evolved with the aid of vegetation but are still in need of considerable evolution for complete recovery of soil characteristics (Correia et al., 2001; Oliveira et al., 2011). Nitrogen levels were also significantly higher in the reference ecosystem than in the intervention modes. On the other hand, P levels far exceeded the reference levels in hydroseeded plots (HS-postFCUL) and were only significantly below reference levels in the limestone benches plantations (PL-preFCUL). In fact, calcium-rich/high-pH soils, such as the ones in the restored limestone benches (pH around 8.1), reduce the availability of P (Specht & Moll, 1983). Other soil characteristics like N content mainly depend on bacterial community activities, which are also dependent on soil pH. As discussed in Correia et al. (2001), the high pH in the quarry soils does not permit high nitrification rates and, consequently, larger retention of N in the soil. Ideally, soils of the restored areas should be around the reference ecosystem's pH (7.5-7.7), which allows most plant nutrients to be optimally available. Interestingly, the highest soil fertility measures in the quarry intervention modes, regarding SOM, N, and P, and the lowest pH value (closer to the reference) was found in the HS-postFCUL, which may be due to the FCUL interventions such as fertilizing the soil with chemical additives or increasing OM by addition of compost.

Furthermore, the shrub layer contributes to increasing SOM and N in the soil of areas under restoration, bringing them closer to NV levels. Shrub cover is especially important in the marl and limestone slopes (HS and HSPL interventions), where it appears to be a determinant factor for higher SOM values and in decreasing soil pH. Contrarily, higher slopes have more difficulty accumulating SOM. This can be explained by the soils in the marl and limestone slopes not being stabilized due to high erosion at the quarry caused by the exploration.

In limestone benches, where plantations exist, the presence and high abundance of pines may contribute to masking the real effect that shrub cover might have in the soil's SOM, N, P, and pH, possibly justifying why shrub cover does not show a significant association with these variables when analysing the quarry restored plots. However, shrub height (PR90) is significantly associated with increasing N concentration in the soil. This relationship is absent with pine variables such as pine cover or height, which gives evidence of the importance and potential of the shrub layer in increasing N levels and overall soil fertility. Since pine trees compete with the shrub layer (Nunes et al., 2014), the removal of pine trees could likely contribute to increasing the native shrubs' density, which could replace the role of the pines and contribute to achieving the SOM, N, and pH levels of the reference ecosystem, as will be further discussed at the end of this section.

SOM decomposition is essential to the functioning of terrestrial ecosystems because, through this process, energy is provided to organisms and nutrients are released for uptake by both plants and microorganisms (Van Veen & Kuikman, 1990). The C:N ratio, used in this research as an indicator of SOM decomposition, generally increases with decreasing decomposition rates, consequently influencing the release of mineral N into the soil and the mineralisation/immobilisation equilibrium (Rutigliano et al., 2009). Mineralisation refers to the conversion of organic matter to mineral nutrients, which are readily absorbed by plants, whereas immobilisation refers to mineral uptake by soil microbes and conversion to organic matter, which is unavailable to plants (Robertson & Groffman, 2007). In addition, litters with high N concentrations decompose faster than litters with low N concentrations due to the growth of soil microbes (Laskowski & Berg, 2006). Ultimately, high C:N ratios (>20-25) that stimulate immobilisation are associated with slower decomposition rates and less amount of N available to plants; C:N ratios between 15 and 20 indicate a good balance between N immobilisation and mineralisation and are associated with faster decomposition rates; and very low C:N ratios (<10) stimulate N mineralisation and can slow down decomposition rates (Craine, 2009; Gupta et al., 2020; Nemerow, 2005; Robertson & Groffman, 2007).

The C:N ratio increases with restoration age, with values below 15 (high decomposition rates) in more recently restored sites and close to or above 20 (low decomposition rates) in sites undergoing restoration for longer, whereas the reference ecosystem has an average C:N ratio of 17, which indicates a good decomposition rate (or equilibrium between the mineralisation and immobilisation processes). Compared to the reference ecosystem, the limestone benches' plantations (PL-preFCUL) have significantly lower decomposition rates. Contrarily, the more recently restored areas with hydroseeded plots (HS-postFCUL) have higher decomposition rates than the reference values. Furthermore, through statistical modelling, a higher C:N ratio (lower decomposition rate) was associated with older restored areas and high tree density. The C:N map confirmed this trend, showing higher C:N values, thus lower decomposition rates, in the limestone benches (older restored sites) where plantations of pine and shrub species exist, and the lowest C:N values and higher decomposition rates in the more recently revegetated steep marl slopes with lower vegetation cover.

These results indicate that the decomposition capacity of the soils in the restoration sites is significantly different from that of the soils in the NV areas, especially in the PL-preFCUL areas where there is a lower amount of N available to plants and low decomposition rates caused by immobilisation. As mentioned previously, N is often a limiting factor in Mediterranean ecosystems, restricting ecosystem productivity (Augustine & McNaughton, 2004; Clemente et al., 2004). In restored sites, soil pH is different from the reference, showing a slight trend towards lower values with increasing tree height (pines). These results are corroborated by Ruiz-Navarro et al. (2009), which demonstrated that recalcitrant pine needles contribute to low decomposition rates, as litter decomposition proceeds very slowly, and acidify the soil compared to Mediterranean shrubland or *Quercus* spp. species.

Decomposition rate is governed by three factors – the physical environment (temperature, moisture and soil properties), the quantity and quality of the dead material available to decomposers, and the nature of the microbial community itself (Chapin et al., 2002). In the limestone benches, the quality of the dead material – the recalcitrant pine needles – contributes to lower decomposition rates that result in a higher accumulation of SOM but less availability of N for plant uptake and overall low nutrient recycling (Nunes et al., 2014). Because C:N is too high and available N becomes less abundant, microbes consume it, which can lead to depletion of soil soluble N in different forms. As a result of that process, N deficiency occurs, and the decay of SOM is delayed (Mesic et al., 2012). As such, SOM slowly accumulates in the benches resulting from the balance between litter input and mineralization (Marchante et al., 2008). Thus, the difference in C:N ratio between the more recently and the old restored sites may reflect the qualitative differences in SOM due to the recalcitrant nature of pine needles and perhaps the microbial community since commonly reported effects of pine on topsoil include not only acidification and SOM accumulation but also a decrease in soil biological activity (Peltier et al., 2001).

Taking the Maquis as a reference, most indicators show that the condition and functioning of the soil are limited in the revegetated areas, reinforcing the soil conditions as a limiting factor for the evolution of the restoration process. This is evident in key parameters such as SOM, N, pH, and C:N ratio in soil, which, in turn, interfere with biogeochemical processes (decomposition and transfer of nutrients) and are a consequence of them. Previous studies in this quarry also concluded that soil is the ecosystem component that takes longer to recover in ecological restoration because it started as an inert marl substrate (Correia et al., 2001). Hence, despite vegetation being present in the restored areas, soil functionality has not yet been recovered, even in the older restored areas of the limestone benches.

When SOM contents are initially very low (as is the case), their substantial increase may take several decades (L. R. Walker & Moral, 2003). Minding that time is a very important factor in soil formation, and since the success of rehabilitation depends on soil quality, it will be essential to improve soil conditions. Thus, certain adaptive management interventions should be performed.

To improve soil conditions, interventions such as pine thinning should be considered for the limestone benches. The current nutrient-poor soil conditions in these restored sites are limiting productivity since nutrient levels are not enough to foster the dense community of pines and shrubs. Additionally, these soils lack the necessary structure and soil biota, which usually requires a long time to recover after disturbance (Heneghan et al., 2008). Since pine litter (recalcitrant pine needles) is slowing decomposition and reducing the availability of nutrients such as N, which are currently limiting primary productivity and growth, measures to increase native shrub density should be implemented. Pine thinning could reduce competition with native shrub species and improve soil properties, as studies have proposed that enriching the understory with shrubs would promote pine litter decomposition and C incorporation into the mineral soil (Ruiz-Navarro et al., 2009). Additionally, the presence of shrubs in arid regions creates resource islands where the chemical, microclimatic and physical status of the soil is improved, along with increasing soil microbes (Gratani et al., 2013). They also improve water retention, an important feature taking into consideration future climate change scenarios (Clemente et al., 2004).

Moreover, a previous study at the quarry showed that pine thinning increased the density of N-fixing species, a critical feature in nutrient-depleted soils (Nunes et al., 2014). Apart from increasing N and SOC contents due to the constant and rapid renewal of their leaves, N-fixing species also increase the biological activity at the top of soil layers which accelerates decomposition rates (Alegre et al., 2004), contrarily to pines, which have been shown to decrease soil biological activity (Peltier et al., 2001). Hence, higher SOC and N availability in the soil would significantly alter the quantity and quality of decomposer microorganisms, improving the soil's capacity to decompose and hold more plant species (Spehn et al., 2000). Thus, pine thinning would favour the soil microbial community, which plays a fundamental role in ecosystem functioning by decomposing SOM, determining the release of mineral nutrients and, consequently, influencing primary productivity and nutrient cycling (Rutigliano et al., 2005).

Furthermore, it would be important to promote the increase in native shrub density not only in the limestone benches, but also in the slopes, especially the limestone slopes (Northern high slope), which are the zone type with the lowest values of productivity and SOC, SOM, N and P. These slopes have low SOM and SOC contents, possibly due to the nature of the nutrient-poor substrate, low vegetation cover in some areas and slopes not being fully stabilized by the plant cover, which hinders SOM accumulation. In these slopes, plants have a critical role in preventing soil collapse and erosion by holding the soil in place with their root system (Zhang et al., 2018). Also, higher shrub density would minimize runoff and sediment movement along slopes (Puigdefábregas, 1996). Additionally, other adaptive management interventions that could improve soil fertility in the restored areas to meet the reference ecosystem levels should be investigated, such as the appropriate enrichment of soil through fertilisation or mulching (Kowalska et al., 2020), especially on the limestone slopes since they had the furthest values of productivity and soil fertility from reference values.

4.3. Habitat Quality

Habitat quality is part of the "Lifecycle maintenance, habitat and gene pool protection" ES group from the CICES framework and the SER key ecosystem attribute "Structural diversity" for evaluating ecological restoration (*section 2.11*). The habitat quality of the areas undergoing restoration is closely linked to the structural complexity of the vegetation and the representativeness of the tree, shrub, and herbaceous strata.

Vegetation structure encompasses the vertical arrangement of canopy layers and plants, as well as the horizontal variation in canopy closure and layers. In this way, structure is determined by the community's dominant strata and both the plants' height and the area of ground covered by the canopy (NASA, n.d.; Sealey & Logan, 2019). Vertical structure characterizes vegetation complexity, while horizontal diversity among stands represents vegetation heterogeneity (Hunter & Hunter Jr, 1999). Enhanced habitat complexity fosters species coexistence by offering diverse ecological niches, thereby reducing overlap and promoting species diversity through decreased competition for resources (Smith et al., 2014). Consequently, habitats with greater complexity and heterogeneity offer a wider array of niches and microhabitats, accommodating specialized species (Rutten et al., 2015). Plants contribute significantly to this heterogeneity due to their varied growth forms. This physical variation shapes diverse micro-environments and creates habitat niches, utilised by various organisms including birds, insects, reptiles, and small mammals (MacArthur & MacArthur, 1961; Rutten et al., 2015; Wiens, 1992). Habitat complexity can also be important in predator-prey dynamics, offering refuge to smaller organisms in cryptic habitats, thus reducing their vulnerability (Smith et al., 2014). Additionally, plant-provided structure serves essential ES such as enhancing resilience against erosion (Ludwig et al., 2005).

This research assessed habitat quality (*section 3.3*) by analysing the cover and height of the different vegetation strata at the restored sites, the latter including metrics such as shrub and tree height and other metrics related to the functional structure of the woody and herbaceous strata (CWM heights). While shrub cover has not recovered to reference ecosystem levels at the restored plots and decreases in older restored sites, pine cover increases with restoration age and is unusually higher than in the reference ecosystem, which may substantially increase the competition for water, nutrients and light by substantially reducing their availability and reduce shrub survival and growth (Bellot et al., 2004; Maestre et al., 2004). Herbaceous cover decreases along the chronosequence and is slightly below reference levels in the limestone benches' plantations. Regarding the vertical structure, shrub height is lower than in the reference ecosystem. Tree height and woody vegetation height (CWM) are higher in the restored plots, while the herbaceous vegetation height (CWM) is higher than the NV in some areas, but lower in others.

Analysing with more detail the relationship of the different strata with time after restoration, one can conclude that shrub cover was the only strata not having a significant relationship with restoration age. However, as expected (Frouz et al., 2008; Khater et al., 2003), shrub cover increased with time from the most recent to the intermediate restoration age, which corresponds to the recently restored hydroseeded slopes, peaking around 15 years after restoration. Nevertheless, shrub cover decreased slightly after that peak, between 15 and 30 years after restoration, in the older restored limestone benches' plantations (PL-preFCUL). Despite shrub cover values being higher on the plantations than on the hydroseeded slopes, the slopes show positive signs of development of the shrub layer with restoration age. The same cannot be said for the limestone benches, where shrub cover plantations appear to be currently limited by pine dominance; thus, interventions regarding this problem should be performed, as will be discussed throughout this section.

Pine cover, on the other hand, significantly increased with restoration age, a trend that was also observed in previous studies in this quarry that showed that pine's average cover increased along the chronosequence and was always higher than that of the remaining planted species at the limestone benches (Correia, 2000; Correia et al., 2001). Pine cover was significantly higher in the plantations than in the hydroseeded slopes and the NV. Thus, *P. halepensis*, which started to be planted more than three decades ago, is mainly responsible for the high tree cover of the limestone benches. Besides from increasing competition, pine stands have been reported to often generate monospecific and homogeneous stands, which potentially supply fewer ES than mixed forests with more heterogeneous structures (Moghli et al., 2022). Since pine cover is higher than shrub cover (including sclerophyllous species that were present from the beginning but belong to later successional stages), the latter will only begin to grow more rapidly when the soil has evolved in terms of SOM and pH (Correia et al., 2001). Thus, selective thinning of pine trees could help in reducing the inter-specific competition and accelerating the successional stages at the limestone benches to better resemble the reference ecosystem (Correia et al., 2001; Nunes et al., 2014).

Contrarily to pine cover, herbaceous cover significantly decreased with restoration age after reaching a peak of around 12 years. Such was expected since these species are characteristic of early successional stages, as observed in studies such as those by Khater et al. (2003) and Frouz et al. (2008). However, the herbaceous cover decrease seems to be due not so much to the natural process of succession in the vegetation, with the progressive replacement of herbaceous species by shrubs, but above all to the type of revegetation (intervention mode). Thus, hydroseeding (HSPL) of herbaceous species on the most recent slopes explains why herbaceous cover is higher there than at the limestone benches plantations (PL-preFCUL).

Regarding vertical structure, the height of the woody community (CWM) increases along the chronosequence, while the herbaceous community height (CWM) decreases over restoration age. Nevertheless, likewise to the herbaceous cover, herbaceous height slightly increases within the first 10 years of the restoration in the more recently restored areas, where pine trees are absent or have low cover, and competition with the tree and shrub strata is not as high as in the limestone benches plantations. The inverse relationship between the average height of the woody and the herbaceous communities with restoration age is a positive aspect concerning the structural complexity in the areas under restoration because it indicates high variety in the dominance of the different strata in the quarry area and, consequently, horizontal and vertical stand heterogeneity, which has a positive effect on the diversity of habitats and species (Borges Silva et al., 2022; Ruiz-Benito et al., 2012). Furthermore, previous studies found that afforestation with a pine stratum improved the vertical structure by establishing pluri-stratified communities (Chirino et al., 2006; Ruiz-Benito et al., 2012).

There is, therefore, a trade-off between greater structural complexity due to the presence of sites undergoing restoration with pines or herbaceous species, which may even provide habitat for other species compared to the NV where pines are absent, and the greater distance in the vegetation composition between areas under restoration and NV, resulting in low similarity with the reference ecosystem. Additionally, a decline in species richness and plant diversity in the understory of pine plantations has been previously reported (Chirino et al., 2006), an issue which, together with the similarity to the NV, will be further analysed in the next *section 4.4*.

Revegetation with Aleppo pine introduced the tree strata that was not present in the NV communities. Thus, height values in the woody community (CWM) differ significantly between intervention modes mainly due to past plantations of pine trees, being significantly higher in the older-restored limestone benches plantations than in the revegetated slopes or in the reference ecosystem since Aleppo pine is in much lesser abundance on the slopes and absent from the NV. Additionally, tree and shrub height values did not significantly differ between intervention modes, probably due to the high height variability in each intervention mode. The same was observed for the herbaceous community height (CWM).

Regarding the statistical models produced, the main predictors for estimating cover were the intervention mode, restoration age and shrub density. While pine cover increases with restoration age, herbaceous cover decreases with restoration age and shrub density. Similarly, tree height and woody vegetation height (CWM) were positively associated with restoration age, pine cover, shrub height and SOM, in contrast with herbaceous vegetation height (CWM), which decreased with pine cover and shrub density.

Several intervention modes and zone types deserve attention regarding their recovery level compared to the cover and height of the different strata in the ecosystem reference. Shrub cover was significantly higher in the reference ecosystem than in the restored sites, and more than 3 times as high as in hydroseeded plots (HS-postFCUL). Shrub height was also below the reference ecosystem in all intervention modes and zone types, suggesting a greater variety of growth forms present and higher habitat structural complexity in the more advanced successional stages of the reference ecosystem, characterized by dense and closed communities of Mediterranean maquis (Barbour et al., 1999; Majer, 1989; Pedro, 1998). Naturally, the denser and higher shrub layer in the NV most likely allows for more shelter, food, nesting sites and protection against agonistic encounters and predation than the restored areas (Monamy & Fox, 2000).

Considering the recovery level of habitat quality, hydroseeded sites (HS-postFCUL) were the least successful intervention mode in recovering shrub cover and height to reference levels. Interestingly, as discussed in the previous section, HS-postFCUL sites also had the highest soil fertility values found in the intervention modes. Although fertilisation often increases herbaceous productivity in the short term, it can prevent long-term vegetation development because of competition with spontaneous colonizers (García-Palacios et al., 2010). In general, native species from the ANP are used to poor scletic soils, and so, when soils such as the HS-postFCUL have high fertility conditions optimized for fast-growing herbaceous generalist and ruderal species, these outcompete and prevent the establishment of native ones (Clemente et al., 2016; García-Palacios et al., 2010; Matesanz & Valladares, 2007). Thus, the small shrub cover in these areas might be due to the very dense herbaceous cover, resultant from the commercial species used in hydroseeding, which impede the installation of native shrub species through interspecific competition. In fact, previous studies suggested it as one of the reasons to explain the poor performance of native species in the restoration of degraded Mediterranean areas (Oliveira et al., 2013).

Thus, it should be a priority to monitor these areas and determine the need to adopt adaptive management measures to increase native shrub cover at these sites and control herbaceous dominance. Nevertheless, as discussed before, shrub cover increased along the chronosequence in the most recently restored slopes, showing a positive development, a trend not seen at the older limestone benches, where pine dominance is limiting shrub growth. Thus, the decrease in shrub cover with restoration age at the limestone benches calls for the need of interventions to promote higher shrub cover, such as selective pine thinning, which has been reported to significantly increase spontaneous shrubs' density (Nunes et al., 2014). Thinning would possibly not only reduce the interspecific competition between species, and accelerate the successional stages at the limestone benches, but also contribute to a higher proximity to the NV in terms of species composition and thus increase similarity to the reference ecosystem, as will be further discussed in *section 4.4* (Correia et al., 2001; Nunes et al., 2014).

Contrarily to shrub cover, pine cover was practically absent in the NV. Therefore, plantations on the Northern low slopes and Eastern high slopes had the furthest values from the NV and the highest pine cover registered (also confirmed by the pine cover map). Thus, pine thinning at this zone types should be prioritized. Additionally, tree height and woody vegetation height (CWM) surpass the heights found in the reference ecosystem, with the north-facing limestone benches having the highest values (also visible in the maps).

No significant statistical differences were found between the restored sites and the reference ecosystem regarding herbaceous cover or height. Such could be due to the large and natural variability regarding herbaceous cover observed in the NV – while some of the NV's plots have a low herbaceous cover similar to those where plantations occurred, others have high cover values close to the HSPL sites. Still, the ecological restoration success index indicates that herbaceous cover is slightly below reference levels at the north-facing limestone benches' plantations, while all other intervention modes and zone types were above reference levels, which can be visualised in the herbaceous cover map. In fact, the north-facing benches have the lowest herbaceous cover values of the entire revegetated area (in accordance with the produced herbaceous cover map). Again, pine thinning in the limestone benches could increase herbaceous cover closer to reference levels, as the density of annual herbaceous species increased with thinning in a previous study at this quarry since they benefit from the enhanced light availability resulting from pine removal (Nunes et al., 2014). With successional progression, herbaceous species would become less abundant and other, more competitive species, such as the native sclerophylls, could replace them (Clemente et al., 1996). Meanwhile, the herbaceous strata could contribute to soil nutritional improvement since leaves from herbaceous species generally have higher decomposability than pine needles or sclerophyllous leaves of late-successional shrubs, which tend to immobilize nutrients (Dias et al., 2013).

Although the difference is not significant, there is a tendency on the north-facing limestone benches for not only herbaceous cover but also height to be typically smaller than in the NV. The insufficient recovery level in this indicator indicates that the herbaceous layer on the benches is scarcer and has characteristically smaller herbaceous species, which can probably be related to the presence of high pine cover and consequent shading and accumulation of pine litter which can compromise the development of the herbaceous stratum. The closed and dense canopies in the limestone benches are absent in slopes, where there is little shading and herbaceous species are typically taller. Furthermore, pine thinning could result in higher structural complexity of vegetation provided by an enhanced herbaceous layer, and higher habitat heterogeneity due to clearings created by thinning which may also provide additional refuge and food resources for animals coming from natural surroundings, thus contributing to increase biotic fluxes (e.g. Tews et al., 2004).

To conclude, the typical succession pattern of Mediterranean ecosystems, in which the initial dominance of pioneer species with a short life cycle progressively decreases, being replaced by species with slow growth and greater longevity (Kazanis & Arianoutsou, 2004) may never be observed on the restored limestone benches. In fact, the "artificial" presence of species planted in the initial stages and their influence on the development of the community are factors to which natural succession is not subjected to and which certainly influence the succession trajectory in the restored areas. Thus, a new type of succession of Mediterranean vegetation occurs at this quarry, with a relatively short first stage in which spontaneous pioneer species (herbaceous and shrub) coexist with introduced woody species but that are still underdeveloped. The current intermediate (and prolonged) phase where limestone benches would now be in, would be then characterized by the simultaneous presence of pioneering spontaneous shrub species (e.g. *Cistus* spp.) and shrub species (introduced or spontaneous) characteristic of the potential mature phase (sclerophyll species).

These results suggest that some stabilization is achieved after 15-20 years of revegetation, with the decrease in shrub cover and the increase in pine cover dominance which, in the future, may negatively impact habitat quality in the older restored areas. In turn, the stabilization can be related to the development of the soil, which is much slower than that of the vegetation, as discussed in *section 4.2*. Another possible cause for the slowdown in recovery will be a certain structural homogenization (cover and height) reached at some benches, since the spatial heterogeneity of the habitat is important for the establishment of new biotic interactions (e.g. with fauna) and, therefore, for the trajectory and successional speed (Alday et al., 2012; Šálek, 2012). Under these conditions, further significant community progression in older restored sites may depend mainly on external factors (e.g. a natural or anthropogenic disturbance).

4.4. Similarity to NV, Composition and Taxonomic Diversity

The similarity to the NV and taxonomic diversity are essential indicators for studying and comparing restored sites with the reference ecosystem. More specifically, considering the key ecosystem attributes for evaluating ecological restoration defined by SER, similarity to NV is part of the "Species composition" SER key ecosystem attribute, while taxonomic diversity belongs to the "Structural diversity" attribute (*section 2.11*).

Despite most measures of biodiversity seeking to assess restoration success being related to species abundance and richness, many authors admit that diversity indices such as the Shannon-Wiener index cannot be used alone (Noss, 1990). Indeed, completely different communities can be characterised by the same species richness and diversity values. Thus, similarity-dissimilarity indices (such as Sørensen–Dice or Bray-Curtis) are important to summarise more information between sites and distinguish different plant communities (Jaunatre et al., 2013). For that reason, similarity between the restored plots and the NV and taxonomic diversity will be discussed together in this section. Here, the Shannon-Wiener index characterized the α-diversity, which describes the species diversity within a community at a small or local scale (generally the size of one ecosystem), while the Sørensen–Dice and Bray-Curtis indices characterized the β-diversity, which describes the species diversity between two communities or ecosystems (Baselga & Orme, 2012; Jost, 2007; Wilson & Shmida, 1984).

Similarity to NV and Composition

Even though similarity to the reference ecosystem regarding species presence (Sørensen–Dice index) significantly increased with restoration age (considering all plant community), a pattern also observed by Correia et al. (2001), the restored sites were on average only 25% similar to the NV after 30 years (*section 3.4*). Similarity was even lower when species abundances (Bray-Curtis index) were taken into account, with average values smaller than 10%, indicating that species abundances are still very distinct from those found at the reference ecosystem, even at the oldest restored sites. Such should not be the case in a successful ecological restoration, in which the restored ecosystem should present a species composition similar to its reference ecosystem (SER, 2004). Nevertheless, the increase of both similarity indices with restoration age reveals the effect of time not only on plant succession but also on species migration from the surrounding areas (Correia et al., 2001).

Similarity values and pattern with restoration age were very similar when either comparing the restored sites to all the NV sites (capturing the variability in the species composition of the natural areas), or when comparing the restored sites exclusively to their correspondent zone types (which compares those to areas with the same surface exposure and slope). This indicates that regardless of the number of NV control plots and their variability in species composition, similarity remained quite low. The more evident approximation to the NV in terms of species similarity occurred in the first 15 years, with a tendency of stabilisation in the approximation to the reference ecosystem afterwards. This stabilisation calls for the need to ascertain its causes and develop strategies to ensure the evolution of the vegetation succession. Thus, the stabilisation of the approximation to NV should be investigated in more detail, as it may compromise the expected evolution of the ecological restoration process and the continuous approximation to the composition and diversity of the reference ecosystem.

Bearing in mind that Aleppo pine is one of the most abundant species in the restoration process, introduced as fast-growing species to allegedly facilitate the establishment of native species (despite being absent in the NV), the similarity analysis above was repeated but excluding pine and herbaceous species, to analyse the proximity of the shrub community to the NV. Although both Sørensen–Dice and

Bray-Curtis indices were higher when analysing shrub community instead of all plant community, the Bray-Curtis index did not show a significant relationship with restoration age and had low similarity values averaging around 15%, indicating that shrub species' abundances are still very different from those of NV even at the older restored sites. This is mainly due to the low abundances of the most characteristic species of the Maquis, like *Q. coccifera* which has a weak representation in the restored areas and other species such as *S. aspera*. The failure of these species to establish should be investigated, such as causes of mortality, poor development, germination and establishment seedlings, or the absence of effective dispersal mechanisms or dispersive agents (Correia et al., 2001). Thus, new strategies to improve their performance in the restored areas (in either the already established plants or future plantations) should be planned if the goal of the restoration follows SER's objectives of achieving the maximum similarity possible (species composition) to the reference ecosystem (SER, 2004). Additionally, previous research has suggested that the vegetation of the revegetated quarry clearly diverges from the NV due to the exclusive presence of not only *P. halepensis* but also *S. junceum*, *C. siliqua* and *P. pinea* (Correia et al., 2001). Thus, another SER goal needing careful consideration in future adaptive management decisions is having the greatest possible number of natives species (SER, 2004).

In fact, the older revegetated limestone benches (PL-preFCUL) are more similar to a natural planted pine forest (up to 43% similarity) than with their respective zone type in the reference ecosystem (up to 25% similarity), demonstrating the striking differences with their reference ecosystem which is characterised by Maquis vegetation. The contribution of the high pine abundances in distancing the restored communities from the NV can also be observed in the higher similarity values in the shrub community when compared to the woody community. Nonetheless, the herbaceous community was the stratum with the lowest similarity to the NV. Herbaceous species had a median of less than 5% similarity, showing that the restored sites are characterized by having very different herbaceous species and abundances than their reference ecosystem. Thus, one can conclude that the herbaceous strata is responsible for the main differences between sites when considering all plant community, which agrees to a previous study at the quarry by Correia et al. (2001).

Overall, the plant community in the restored sites is still very different from the NV. Nonetheless, generally, it presents higher similarity values in the limestone benches, which correspond to the zone type "Northern low slope" and the intervention mode "PL-preFCUL" where plantations exist. Therefore, even though there is lower herbaceous cover in this area than in other intervention modes or even the reference ecosystem (as discussed in the previous *section 4.4*), the similarity of herbaceous species presence to the reference is the highest at the limestone benches, which is also confirmed by the map. Contrarily, the lowest similarity values were found in the "HSPL" sites where hydroseeding and plantations occurred and where the herbaceous layer dominates, in the areas of the limestone and marl slopes. However, when considering species abundances, similarity did not differ significantly between intervention modes and was actually higher in the "Southern high slope" zone type, and lower in the other revegetated slopes, which shows that most slopes have considerably different abundances than their reference.

Visualising the ordination analysis, one could interpret that the composition variation in the plant community is primarily influenced by the characteristics of the soil in the restored areas, which have lower SOM contents and higher pH than the natural areas, as well as a tendency towards less potential sun exposure associated with a lower shrub cover. Nevertheless, a modest evolution of the soil and shrub cover can be observed with restoration decade, reflecting the effects of the succession. The restoration interventions carried out, depending on whether they were done through plantations (higher pine cover) or hydroseeding (higher herbaceous cover and slope), are the second most important factor dictating the differences in the composition of the plant community between sites under restoration with different ages. These results agree with the statistical modelling of similarity, which showed that similarity of species presence (considering all plant community) increased with restoration age and shrub cover, with this increase being also dependent on the intervention mode. However, when considering species abundances (Bray-Curtis index), restoration age was not a relevant predictor in statistical modelling, indicating that time after restoration does not have a major influence in the similarity of species abundances, which reinforces the necessity to adopt adaptive management strategies to further increase the proximity in species composition between the restored sites and the NV.

Taxonomic Diversity

The taxonomic diversity (*section 3.5.2*) of shrub species increases significantly with the years after restoration, showing a tendency towards stabilization approximately after 23 years. The stabilization is probably related to increased competition for space, light and nutrients, due to greater pine cover at later successional stages (Correia et al., 2001; Gómez-Aparicio et al., 2009). The closest diversity values to those of the NV can be found in the areas under restoration for a longer time period, that is, in the limestone benches plantations (PL-preFCUL), despite presenting greater variation, which could indicate that the maximum potential had been reached. Furthermore, even though the average species diversity at the different intervention modes is no different from that of the NV, the low similarity values of the plant community to the reference suggest that the species composition in the revegetated and natural sites is considerably different, therefore species and abundances different from those found in the NV are responsible for the diversity at the quarry's restored sites.

Additionally, although there was no significant relationship between the Shannon-Wiener diversity index and restoration age in the woody community, a significant and positive relationship was found between the species richness of woody species and restoration age. Thus, the pattern of species richness with restoration age changes when one also takes into consideration species evenness (species diversity). This can be explained by the high abundance of pine trees in the restored areas, which not only reduce the equitability of the distribution of abundances of shrub species (influencing the Shannon-Wiener index) but may also compete with the target native species, as indicated by previous studies (Nunes et al., 2014).

Although the Shannon-Wiener index did not show significant differences between the intervention modes and the NV, the NV has higher diversity values and significantly higher woody species evenness than "PL-preFCUL" (which had the lowest evenness values). Such indicates that species were more evenly distributed in the natural areas than in the restored sites, a finding also observed by Correia et al. (2001). Species evenness is important to biodiversity because it gives an indication of the ecosystem stability (Wittebolle et al., 2009). When communities are highly uneven, or there is extreme dominance by one or a few species, their functioning is less resilient to environmental stress. Such creates unstability because dominant species have high abundance relative to other species in a community, and, consequently, are the most important species in a layer, with the largest ecological role that determines the layer's basic characteristics (Zhang et al., 2018). Thus, if evenness is low and a dominant species is disturbed (e.g. fires, drought, pests), the ripple effects will affect the entire ecosystem and lower biodiversity. Therefore, the more species dominate (or the more evenness and diversity among species), the more productive and stable an ecosystem may be (Johnson et al., 1996). In a highly even ecosystem like the natural reference ecosystem, such disturbances are only felt in the niches of the disturbed species. A stable ecosystem has high species diversity and a strong ability to adapt to changes in the external environment (ecosystem reslience). Otherwise, even if the current vegetation cover is high, as is the case in the plantations of the limestone benches, but the species diversity or evenness are low, the ecosystem is prone to degradation (Yang et al., 2019). Such requires adptive management actions that will be further discussed at the end of this section.

The species diversity in the herbaceous community significantly decreased with restoration age, reaching a peak in the first 10 years and decreasing progressively from that age onwards, opposite to the trend observed with the shrub community. Similarly to what was discussed in *section 4.1* regarding the herbaceous cover, this decrease with restoration age seems to be due, not so much to the natural process of succession in the vegetation with the progressive replacement of herbaceous species by shrubs, but above all to the intervention mode of hydroseeding of herbaceous species carried out on the most recent slopes. Nevertheless, herbaceous diversity exceeded the ecosystem reference's values in most intervention modes (except for the plantations in the limestone benches), indicating that ecological restoration successfully achieved recovery in this indicator.

Comparably to the most relevant predictors for estimating similarity, the shrub species diversity index was positively influenced by shrub cover and negatively by slope (since herbaceous cover dominates the steeper slopes). Conversely, herbaceous species diversity was negatively correlated with restoration age, indicating that sites restored earlier via hydroseeding (most recent slopes) had higher herbaceous species diversity. Such is confirmed by the map where higher shrub taxonomic diversity index can be seen in the quarry areas where plantations exist (in the north-facing limestone benches) and lower values in the quarry areas where herbaceous cover and steeper slopes are higher (mostly in marl slopes and some areas in the limestone slopes).

The results of both similarity and species diversity confirm the need to adopt many of the adaptive management strategies or recommendations mentioned in the previous sections. As said before, higher pine abundances not only negatively impact soil fertility and decomposition (*section 4.2)* and habitat quality (*section 4.3),* they also increase the gap between restored sites and the reference in terms of similarity of species composition, and cause low evenness which compromises ecosystem resilience in the case of a disturbance. Additionally, previous studies (including this research) have shown that pine plantations strongly decrease resource levels in the understory (i.e., light, soil water, nutrients) and negatively affect diversity and performance of native plant species (Gómez-Aparicio et al., 2009). Thus, pine thinning should be performed to support reorientation of woody trees, pines in this case, towards species compositional states that are more similar to native habitats and with higher diversity and resilience (Luna et al., 2022). Futhermore, thinning has been considered a potentially useful tool to redirect pine plantations toward lower densities where facilitation mechanisms predominate, as in the case of oak regeneration (Gómez-Aparicio et al., 2009).

In addition, previous evidence was found in Mediterranean semiarid areas of a negative effect of pines on later-successional species and of failure to improve ecosystem functioning, as will be further discussed in next section 4.5 (Bellot et al., 2004; Goberna et al., 2007; Maestre et al., 2004). Management practices such as clearcutting, thinning and pruning are needed to redirect such systems, but scientific studies concerning the effects of these silvicultural treatments on biodiversity, ecosystem functioning, and successional trajectories are lacking (Nunes et al., 2014). Such studies are very relevant, since proper management of areas with *P. halepensis* is one of the major environmental challenges in the quarry's ecological restoration and will be necessary to avoid the negative impacts reported. This reasoning is also based on the assumption that the goal is to convert these restored areas into more diverse and resilient ecosystems, which are similar in species composition to their reference (Luna et al., 2022).

Regarding recovery in similarity and taxonomic diversity, some slopes are also in need of adaptive management actions. As observed with similarity, the intervention mode with lower diversity values when compared to the NV was the HSPL. Sites with this intervention mode or located in the marl slopes (Southern high slope) were the most distant from attaining recovery in this indicator, suggesting that these areas do not have a satisfactory level of diversity in shrub species, when compared to their reference. Additionally, the restored areas located in the limestone slopes (Northern high slope) and marl slopes (Eastern high slope) also deserve special attention due to being the zone types with greatest dissimilarity to NV in terms of species abundances.

In addition to increasing the diversity and evenness of woody species though pine thinning in the relevant areas, other adaptive measures applied to restored sites will be necessary to bring the similarity and diversity in the restored slopes closer to levels recorded in the natural vegetation. Restoration of semi-arid ecosystems with native Mediterranean shrub species has been shown to be an adequate method for improving plant diversity and ecosystem resilience (Chirino et al., 2006). Thus, the revegetation of bare slopes should go hand in hand with biodiversity targets, to create a diverse ecosystem with locally adapted native species and the closer similarity possible to the reference ecosystem (Hölzel et al., 2012; SER, 2004).

Lastly, promoting microhabitats for species of interest for conservation would also be an important goal. In order to do this, future research should test the creation of microhabitats to encourage the establishment of species such as *I. procumbens*, *A. sadina*, *R. aculeatus*, and RELAPE species. Greater abundances of RELAPE species were mainly observed in the oldest limestone benches, in areas undergoing recovery for a longer time period, at higher altitudes, and in close proximity to the surrounding NV. Such is a good indicator of the ability of these species to colonize through propagules from the NV, especially if more time has passed since restoration actions, which is probably favored by greater soil stabilization and recovery in these areas. Nonetheless, the introduction or transplantation of these species to restored areas with favourable environmental conditions for their installation is desirable once the challenges of assisted propagation have been overcome (of which there is still no experience at the SECIL-Outão quarry). For this, the characterisation of the species' microclimatic niches using variables such as altitude, PSR, slope, and restoration age should be done. Therefore, with the aim of increasing the niche or microhabitat areas to shelter species with conservation status, it is suggested to study in detail the ecology, microclimatic niches and methods of assisted propagation of each species, and provide indications on how to adapt restoration actions to recreate optimal conditions and to potentiate the existence of these species in the quarry area.

4.5. Ecosystem Functioning and Resilience to Climate Change

The increasing threats from climate change and land-use changes, including droughts, wildfires, and biodiversity loss, emphasise the critical importance of ecosystem stability and resilience (Bellard et al., 2012). Ecosystem stability involves maintaining properties over time, resisting, and recovering from environmental changes. It relies on populations, communities, and ecosystems buffering environmental effects while maintaining functions like productivity and pollination (de Bello et al., 2021). Ecosystem multifunctionality, encompassing various ecosystem functions contributing directly or indirectly to ecosystem services, is crucial for a successful ecological restoration (Garland et al., 2021). Furthermore, considering species' functional traits, not just diversity, is key in understanding ecosystem functioning (Lavorel & Garnier, 2002; Suding & Goldstein, 2008). Lastly, assessing ecological resilience involves understanding functional redundancy, where multiple species perform similar functions, enhancing resilience during disturbances (Loreau, 2004). Redundancy in functions can enhance system resilience, compensating for lost functions in cases of disturbances. However, a reduction in species within functional groups may signify a loss of biotic integrity (Pellant, 2005).

All functional traits that will be discussed in this section are indicators of ecosystem functioning and are part of the "Lifecycle maintenance, habitat and gene pool protection" ES group from the CICES framework and the SER key ecosystem attribute "Ecosystem function" for evaluating ecological restoration (*section 2.11*). Traits such as pollination, dispersal mode, and fruit type are important to assess reproductive strategies, dispersal capacity, and interaction with fauna, for example. Other functional traits are indicators of response to disturbances and resilience to climate change, such as species with a resprouter post-fire regeneration strategy which are resilient to fire, or evergreen sclerophylls characterized by higher drought tolerance features (Moghli et al., 2022; Nunes et al., 2014). Lastly, although not being a functional trait, seedling density was included in this section since it is a relevant indicator of natural regeneration of the plant community (Vickers et al., 2019).

4.5.1. Functional Diversity

Biodiversity includes species and functional diversity, both of which impact ecosystem functioning and likely ecosystem services (ES). Research indicates that as biodiversity decreases, the quality of ES provided by ecosystems diminishes (Balvanera et al., 2014; Garland et al., 2021; Wilson, 1999). Functional diversity, which considers the variety and abundance of functional traits in an ecosystem, proves to be a more reliable indicator of ecosystem vulnerability than species diversity alone (Díaz et al., 2007; Nunes et al., 2014). It measures the diversity of ecological roles prevailing in an ecosystem, highlighting the importance of species performing essential functions. The FDis, a multi-functional diversity index, was used here to assess the functional dissimilarity in each site. It reflects the degree of functional dissimilarity within a community (i.e., functional attributes) while simultaneously considering all the attributes selected in this study.

Although functional diversity (*section 3.5.3*) in the shrub community did not show a significant relationship with restoration age, it increased after 20 years. However, when considering shrubs abundances with pines (woody community), functional diversity decreases considerably, and although it increases after 20 years just like in the shrub community, it does not attain shrub's functional diversity values. Specifically, sites restored five years ago have higher functional diversity than sites restored 30 years ago, which reflects lower complementary levels in resource use within a community. This is in accordance with the findings in this research regarding productivity and soil fertility, which suggest limiting conditions in the older restored sites due to the high pine abundance. Additionally, the adverse effect of pines in the restored areas could also be detected in the statistical modelling, as higher pine cover was associated with lower functional diversity values both in the woody and herbaceous community, while shrub density positively affected functional diversity in the shrub and woody community.

The effects of the high abundance of pines on functional diversity were also confirmed by the positive significant relationship between functional richness (which does not consider species abundances) and restoration age, indicating that the abundance of Aleppo pine is responsible for the low functional diversity values (resultant from the dominance of pine functional traits to the detriment of other species' functional traits) and lack of significant relationship with restoration age. Ideally, functional diversity (FDis) would have been mapped and the real effects of pine abundance would be spatially visualised, however, no significant spatial predictors could adequately serve as indicators of the intervention mode or the type of plant species introduced or present in the restored areas, which, at the end, is what defines functional traits. Furthermore, functional diversity in the herbaceous community followed the same pattern as herbaceous cover and taxonomic diversity. This pattern can be explained by the type of revegetation done, since functional diversity in the herbaceous community decreased significantly with restoration age, even though it first increased in the recently restored sites where there was hydroseeding of herbs, but then decreased and stabilised at low values in sites with more than 20 years where pine and shrub plantations occurred.

The study revealed no significant differences in functional diversity between intervention modes and the reference ecosystem in both shrub and herbaceous communities. It is likely that, although the shrub or herbaceous species and respective abundances are different between the different intervention modes (*section 4.4*), there is a "compensation" between the present and absent species in each case, which determines a comparable functional diversity (de Bello et al., 2021). However, significant differences emerged in woody communities, where certain plantations and hydroseeded sites had lower functional diversity values compared to the reference ecosystem. This suggests that the reference ecosystem might have superior resource use differences, potentially enhancing ecosystem functioning, as higher functional trait diversity has been linked to increased resource-use efficiency (Cadotte et al., 2011; Díaz & Cabido, 2001).

Even though this study found that pines negatively impacted functional diversity in limestone benches, the intervention mode HSPL-preFCUL (dominated by herbaceous species and pines) and south-facing high slopes (dominated by herbaceous species) showed the lowest recovery levels (*section 3.5.4*), falling significantly below reference values. These areas exhibited incomplete functional diversity recovery in shrub and woody communities, indicating a need for interventions to enhance functional diversity and ecosystem functioning. Interventions should address the overdominance of herbaceous species, promoting native shrub abundance, as sites dominated by shrubs tend to have higher functional diversity and improved ecosystem functioning, resembling ANP reference ecosystems. While functional diversity in restored quarry areas was mostly recovered in herbaceous communities, limestone benches displayed values below the reference ecosystem, likely due to the dominance of slow-growing, conservative species. These species, retaining their niches and ecological traits over time, contribute to stability and resistance against extreme events like drought or fires, highlighting the importance of such species in ecosystem resilience (de Bello et al., 2021). Specific traits associated with certain functions will be discussed in subsequent sections.

4.5.2. Pollination

Pollination, facilitated by agents like insects (entomophily) or wind (anemophily), is vital for plant reproduction and seed production in Mediterranean ecosystems, where most plants rely on insect pollination (Mauseth, 2014; Petanidou & Vokou, 1990). In the maquis specifically, many of the woody plants are primarily anemophilous and temporarily visited by insects such as bees and flies (Petanidou & Vokou, 1990). Yet, in Mediterranean woodlands (with species such as *P. halepensis*), purely anemophilous plants are encountered. Pollinators are central to the success of restoration efforts as they provide essential ES, playing a crucial role in ecosystem processes and maintaining diversity and function (Potts et al., 2003; Young, 2000). Understanding insect communities and plant-pollinator interactions is crucial, given the current global pollination crisis. Factors like forage resources, habitat heterogeneity, and plant structural diversity influence pollination services (Goulson et al., 2015; Potts et al., 2003). Additionally, management that enhances floral resources (increasing density of flowers and promoting insect-wildflower visits) can be an effective way to support pollinators and pollination services (Hardman et al., 2016). Thus, assessing plant-pollinator community structure, including the proportion of insect-pollinated species and flowering duration, serves as indicators of pollination services, forming the basis for conservation and restoration efforts.

The results on pollination (*section 3.6.1*) revealed a decline in both the proportion of insect-pollinated species and flowering duration in the woody community along the restoration chronosequence, notably in older restored sites with pine plantations (PL-preFCUL). Pine cover was associated with decreased proportions of insect-pollinated species and shorter flowering durations, indicating a negative impact on pollination services. Excluding pines and analysing only the shrub community showed a slight increase in flowering duration, emphasising the role of pines (and their abundance), which have a relatively short flowering period (3 months), in reducing overall flowering duration. This might be undesirable since it has been assumed in previous studies that longer flowering might counterbalance any loss of pollination services from reduced pollinator richness (Petanidou et al., 2014). Interestingly, HS-postFCUL sites that have high herbaceous cover exhibited longer flowering durations and higher proportions of insect-pollinated species than the reference ecosystem, suggesting a positive contribution from herbaceous species to pollination services.

The proportion of insect-pollinated species in a plant community indicates its potential to support pollinators. Areas dominated by herbs (marl and limestone slopes) have higher proportions of insect-pollinated species with longer flowering, enhancing pollination services. However, these areas also show increased species dissimilarity compared to the shrub-dominated reference ecosystem (maquis), where the herbaceous stratum is, in general, scarce, due to the dominance and high density of shrubs. This trade-off impacts the balance between pollination service provision and ecosystem similarity with the reference ecosystem.

The proportion of insect-pollinated species in the woody community showed no significant differences among intervention modes and the natural vegetation (NV). However, the ecological restoration success index (*section 3.6.5*) revealed incomplete recovery in PL-preFCUL and HSPL modes. These areas had more wind-pollinated species and fewer insect-pollinated species compared to the reference ecosystem, where both types were evenly balanced. The HSPL intervention modes, which are dominated by both herbaceous species and pines, also had incomplete recovery regarding functional diversity, which might require adaptive management interventions. Furthermore, zone types that have incomplete recovery regarding pollination are the north-facing limestone benches and the marl slopes (eastern and south-facing). Thus, these intervention modes and zone types may have reduced pollination services and interactions with fauna. This could result from lower forage quantity or quality, and low abundance of insect-pollinated species. Additionally, structural homogenization, caused by habitat spatial homogeneity or by dominant vegetation, may hinder new biotic interactions, contributing to reduced pollination services (Potts et al., 2003). This will be further discussed in *section 4.5.5*.

4.5.3. Resilience and Adaptation to Fire

The Serra da Arrábida's typical vegetation, the Mediterranean maquis (composed by evergreen sclerophyllous species), is highly susceptible to wildfires since these are a common disturbance in Mediterranean dry sub-humid areas, occurring with some periodicity, which may increase according to climate change predictions (Clemente et al., 2004; Nunes et al., 2014). As a result of co-evolving with fire over time, native plant species have developed strategies to deal with this disturbance (which has considerable impacts on local biodiversity), and therefore exhibit a particularly strong fire resilience (Clemente et al., 1996; Vallejo et al., 2012). Two basic regeneration strategies have been widely recognised: resprouting from below-ground vegetative parts (resprouters) and recruitment of seedlings from soil-stored seeds (seeders) (Meira-Neto et al., 2011). Thus, the strong fire resilience observed in Mediterranean-like vegetation is driven by these two major functional groups of plants, and species of each group show different demographic patterns throughout the post-fire succession (Clemente et al., 2004). Obligate resprouters like evergreen sclerophyll species (e.g. *Q. coccifera*), are fire-resistant and can rapidly re-establish their aboveground biomass after disturbance through regeneration by vegetative regrowth. However, successful seedling establishment requires long disturbance-free periods. In contrast, obligate seeders (e.g. *Cistus* or *Pinus* sp.) are fire-sensitive. After a disturbance, these populations regenerate solely through postfire recruitment from a seed bank, lacking vegetative regeneration capacity (Clemente et al., 1996; Pausas et al., 2004; Zavala et al., 2000). Post-fire regeneration strategies are essential for the resilience of restoration sites, ensuring they can withstand future fire disturbances without compromising the ongoing restoration efforts. This is particularly important for the restored areas in the SECIL quarry, given past fire incidents have affected parts of the sites (Anjos, 2013).

The proportion of species with a post-fire resprouter strategy significantly decreased with restoration age (*section 3.6.2*). This pattern alerts for the vulnerability of these areas in the likely possibility of future fires since the most important feature of fire resilience in Mediterranean-like vegetation is the fast recovery of resprouters in the initial phase of post-fire succession, as the establishment of plants from this functional group improves general fire resilience (Meira-Neto et al., 2011).

The plantations (PL-preFCUL) at the older restored limestone benches had significantly lower resprouter proportions than the hydroseeded restored slopes and were three times lower than the NV. This agrees with the findings of Anjos (2013) that reported a significantly higher resprouter density in the reference ecosystem when compared to the restored benches. Such suggests that the presence of pines (seeders) in the PL-preFCUL sites significantly affects the proportion of the resprouter attribute in the communities. Additionally, the decreasing pattern of resprouters with restoration age was absent when analysing the shrub community (excluding pines), and no significant differences between intervention modes were found, which again indicates the influence pine abundances have in the proportion of resprouters. Besides, pine cover was the most significant predictor to estimate the proportion of resprouters in the woody community, showing a negative relationship, reinforcing the role of pines in decreasing the proportion of resprouters in the ecosystem. Thus, pines have negative implications for fire-resilience capacity of the restored areas, and an eventual fire could hinder the post-restoration succession in areas where pine cover is higher and resprouter species are less abundant, since resprouter capacity is one of the main fire response traits of Mediterranean Basin plants (J G Pausas & Verdú, 2005).

Furthermore, the ecological restoration success index (*section 3.6.5*) indicated that there is incomplete recovery regarding resilience to fire in practically all intervention modes, as the proportion of resprouters is below ecosystem reference levels. Therefore, areas with higher pine abundance are the most distant from the restoration goals of attaining the fire resilience of the reference ecosystem due to having lower resprouter capacity.

On the one hand, it is important that all post-fire regeneration strategies are present given that these are complementary. On the other hand, it is also important that they are well represented (high functional redundancy) so that if species loss occurs due to disturbances, other species will perform the same functional role (Loreau, 2004). Thus, the low proportions of resprouter species in the plantations at the limestone benches indicate low functional redundancy which might compromise the resilience and recovery of these restored areas in the case of future fire disturbances. Additionally, post-fire regeneration of Aleppo pine forests often generates overstocked and vulnerable stands as they accumulate a high fuel load, increasing the risk of further fires (Moghli et al., 2022). Therefore, apart from having lower fire resilience and being susceptible to short-fire intervals, high pine densities support low species diversity, reduce species similarity with the reference ecosystem, and decrease ecosystem functioning, as found by this study and discussed in the previous sections.

If the restoration plan aims to follow the SER goals of ecological restoration, such as “being sufficiently resilient to withstand normal periodic stress and disturbance events like fires” or “be self-sustainable to the same degree as its reference ecosystem and have the potential to persist indefinitely under existing environmental conditions” (SER, 2004), then it would be desirable to reduce pine dominance through progressive thinning and in this way increase the abundance of resprouter native species and boost the current low resilience of the vegetation community in the restored areas (especially in the limestone benches). Furthermore, a combination of thinning and resprouter plantations has been shown to improve ecosystem functions, as plantations of resprouter species helped to maximize ES and disturbance resilience (Moghli et al., 2022).

Climate change projections indicate an increase in the frequency and intensity of megafires due to more extended and severe seasonal droughts, which most likely will impact ecosystems’ species composition and functioning (Stephens et al., 2013; Viana-Soto et al., 2020). Thus, ecosystems must adapt not only to changes in average climatic variables, but also to a wide variability with higher risk of extreme climatic events, such as prolonged droughts (which will be further discussed in *section 4.5.4*). Hence, it is of utmost importance that restored sites can withstand disturbances and retain their functionality. Careful planning must be made before restoring new sites at the quarry, including choosing species with relevant functional traits regarding fire resilience.

For future restoration of sites after quarrying, it is recommended to plant a high diversity of indigenous species from the resprouter fire-responsive functional group (Meira-Neto et al., 2011). However, germination from a pre-existing seed bank is another important resilience feature of Mediterranean vegetation. Therefore, although resprouting species are mainly responsible for fire resilience, it is also highly recommended to have a seed bank settlement with a high diversity of indigenous species from the seeder fire-responsive functional group to achieve wider fire resilience of Mediterranean-like ecosystems in the earlier stages of post-quarry sites. If the intensity and frequency of fires increases, the usual post-fire succession dynamics can be compromised. Hence, there is a need for a diversity of complementary strategies so that responses to disturbance can be more diverse. Additionally, different responses from different plant functional types allow faster regeneration of Mediterranean-like vegetation after fire (Meira-Neto et al., 2011).

Therefore, in future sites targeted for restoration, both the resprouting capacity of resprouter species and the seed germination of seeder species should be supported to: (i) increase biodiversity, becoming more similar to natural sites, (ii) increase the presence of indigenous species, (iii) include more than one functional group to increase chances of long-term stability, and (iv) improve resilience to natural disturbances (resistance and recovery from environmental change and perturbation) (de Bello et al., 2021; Meira-Neto et al., 2011). Furthermore, conserving and maintaining the surrounding Mediterranean-like ecosystem is fundamental for natural seed rain (Meira-Neto et al., 2011). High-intensity fires tend to destroy the seed bank and eliminate the possibility of recolonization by this mechanism, which can lead to ecosystem disequilibrium. Therefore, it is essential to preserve natural zones adjacent to the quarry so as not to limit seed rain and ensure a seed bank with indigenous species that can promote a greater resilience to fire.

4.5.4. Resilience and Adaptation to Drought

Mediterranean climate zones are, by definition, zones where there is a xeric or dry period in summer, with high temperatures and water scarcity which impose a high hydric stress on the vegetation (Rana & Katerji, 2000). Furthermore, climate change projections predict an increase in water stress conditions in the Mediterranean as a result of more extended and severe droughts, since increased temperatures and a decrease in precipitation will further decrease soil water-availability in these already water-limited ecosystems (Sardans et al., 2006; Stephens et al., 2013; Vicente-Serrano et al., 2010). This pattern is expected to greatly impact ecosystems' species composition and functioning, since water is considered to be the most important factor limiting production, growth, development, and distribution of plants in Mediterranean ecosystems (Acherar & Rambal, 1992; Baquedano et al., 2008). Therefore, plant strategies to deal with water availability limitations are crucial for the resilience of these ecosystems to drought and, consequently, to maintain essential attributes in the plant communities undergoing restoration. Here, SLA and sclerophyll vegetation were used as indicators of drought response or resilience.

The SLA corresponds to the unilateral area of a fresh leaf divided by its oven-dried weight and is used as an indicator of drought response, and often positively correlated with plant growth rate, photosynthetic rate, and water loss (Ackerly et al., 2002; Lopez-Iglesias et al., 2014). In general, species tend to have lower SLA in resource-poor environments, but species with different SLA values can also coexist, co-occurring different "complementary" strategies (Schenk & Jackson, 2002). In woody evergreen species, greater drought or aridity may favour water stress-tolerant strategies, such as perennial leaves (plants maintain green leaves during the summer drought) with low SLA. However, in very arid conditions, these species can be replaced by (or co-dominate with) relatively shorter-lived stress-avoidant species with semi-deciduous leaves (plants that lose part of their leaves during the summer) and high SLA (Ackerly et al., 2002; Ehleringer & Mooney, 1983; Gross et al., 2013).

Furthermore, evergreen sclerophylls are a type of Mediterranean vegetation characteristic of the mature phase of the succession. They are important for the future because they are adapted to periods of drought based on deep root systems, hard leaves covered by a protective waxy cuticle, and oily secondary compounds that prevent water loss during the dry season (Gullo & Salleo, 1988). Sclerophylly is generally associated with species with great longevity and slow growth as a form of adaptation to long periods of drought. Furthermore, sclerophylly is very common in the species used in the restoration of the SECIL quarry sites and is an important strategy to ensure the resilience of plant communities to drought.

In the restored sites, both the SLA and the proportion of sclerophylls in the woody community decreased with restoration age (*section 3.6.3*), with the older plantations having significantly lower values than the NV, most likely due to the high pine cover, which was a significant predictor in estimating both variables. Nonetheless, SLA values were also influenced by the restoration age and the intervention mode, probably due to the species planted or hydroseeded in each mode which conferred different SLA values across intervention modes. Thus, the plantations in the limestone benches have potentially less drought resilience due to the low proportions of sclerophyll vegetation due to pine dominance and competition.

Regarding recovery success (*section 3.6.5*), SLA values were lower in the north-facing limestone benches plantations and some of the marl slopes than the other intervention modes, which was also confirmed by the produced map. These areas had values below reference levels, indicating that species with low SLA dominate, while in other sites and in the ecosystem reference there may be species with complementary strategies including both high and low SLA species (which explains the high SLA variability observed in the NV). Even though plants with lower SLA are thought to be more resilient to drought, the complementarity of strategies can also be important for resilience (Granda et al., 2018). Furthermore, some hydroseeded plots (HS-postFCUL) had species with higher SLA than the reference ecosystem's species, which might be explained by the semi-deciduous, short-lived, and stress-avoidant species with high SLA that occur in higher abundances in the hydroseeded sites than in the reference.

Additionally, the proportion of sclerophyll vegetation is below reference levels in all intervention modes and zone types, except for the marl slopes facing South (Southern high slope) which have the highest and most similar proportions of sclerophyllous vegetation to the reference ecosystem. Because the reference ecosystem is characterized by having 80% sclerophyll species and 20% semi-deciduous species, one can conclude that there is incomplete recovery regarding drought resilience since most restored sites are characterised by having higher proportions of fast-growing species (conifer or herbaceous), that are not as resilient to drought as the native sclerophyll and semi-deciduous species found in the reference. These results emphasise the importance of using native shrub species in the recovery of all quarry locations as a way to ensure greater drought resilience of the communities in the restored sites, since functional traits related to drought resilience contribute to the species response to climate events (de la Riva et al., 2017).

The predicted increasing drought in Mediterranean ecosystems due to climate change is expected to enhance nutrient deficiencies besides decreasing water availability, which will greatly affect nutrient cycles and, ultimately, plant growth and survival, reinforcing the need to readjust the current plant community at the restored sites (Sardans et al., 2006). Although *P. halepensis* is described as the most drought-tolerant species of its genus, the future of pine forest dynamics remains uncertain due to global warming (de Luis et al., 2011; Miguel A Zavala & Zea, 2004). A temperature increase may give rise to an extended growing season but a longer and more intense summer season, which may negatively impact growth and survival rates. This pattern could threaten the survival of *P. halepensis* forests in the most arid regions, as chronic limiting water conditions caused by warming, and the severe and prolonged droughts common in the region, could cause extensive forest dieback (Vicente-Serrano et al., 2010). Although hydraulic systems of *P. halepensis* are highly plastic (Klein et al., 2011), oak sclerophyllous species such as *Q. coccifera* are expected to endure water stress further than less drought-tolerant conifers. In fact, sclerophyllous behave as a drought-tolerant species (drought-induced stomatal closure does not limit carbon assimilation) and *P. halepensis* is a drought-avoiding species (carbon assimilation is inhibited because of stomatal closure) (Baquedano et al., 2008; Guillermo Gea-Izquierdo et al., 2014; Martínez-Ferri et al., 2000).

Thus, to achieve the SER goal of "being sufficiently resilient to withstand normal periodic stress and disturbance events" (SER, 2004), adaptive management interventions should be performed in the areas with lower drought resilience, which are the north-facing limestone benches where pine abundances are higher. Similarly to what was discussed in previous sections (e.g. resilience to fire), pine thinning and efforts to increase native shrub species are recommended to increase drought resilience in the restored areas. Pine thinning should be performed as indicated in a previous study at the SECIL quarry, which found that thinning in the limestone benches increased the basal growth of evergreen sclerophylls and the density of species with semi-deciduous drought strategy, both having high drought-tolerance features (Nunes et al., 2014; Bellot et al., 2004; Espelta et al., 1995; Maestre et al., 2004). Thus, pine thinning could favour species that exhibit higher drought tolerance and increase the restored ecosystems' resilience in future climate change scenarios.

Regarding new future sites targeted for restoration, it is strongly recommended to use native sclerophyllous species, as they have the advantage of being highly adapted to the local edaphic and climatic conditions characterized by strong seasonality and can withstand years of extreme climatic events (Correia et al., 2001). The use of fast-growing commercial species should be discontinued because these strongly depend on irrigation, which is incompatible with sustainable water use in arid and semi-arid environments and can outcompete native species and constrain vegetation dynamics in the long term (García-Palacios et al., 2010; Matesanz & Valladares, 2007).

4.5.5. Dispersal Capacity, Interaction with Fauna and Regeneration

Seed dispersal is a key ecosystem function that allows plants to occupy newly available niches, avoid sibling competition and attack by natural enemies (e.g. herbivores, pathogens), find suitable conditions for germination, and expand their distribution range (Donoso et al., 2022; Traveset et al., 2014). Furthermore, seed dispersal is one of the key phases in the regeneration process of plant populations, as it determines the potential area of recruitment and, simultaneously, acts as a template for the subsequent stages of plant growth and spatial patterns. Seeds can be dispersed by wind (anemochory), gravity (barochory), and by a wide assemblage of animals (zoochory). Besides moving seeds across the landscape, animals that ingest fruits and pass viable seeds through their digestive tracts can further play an important role in plant establishment, as they can modify the seed germination rate and seedling growth (Donoso et al., 2022; Rogers et al., 2021). Furthermore, poor seed dispersal (often assessed by measuring seedling density) is a major limiting factor for forest recovery. Thus, the recovery of biological interactions is critical for the long-term function of a restored ecosystem (Ruiz-Jaen & Mitchell Aide, 2005).

Moreover, seed dispersal influences the biodiversity and functioning of plant communities and the ES they provide (Traveset et al., 2014). However, dispersal ability faces considerable challenges in locally fragmented areas such as the SECIL quarry. Some of the main limitations to plant establishment and growth in such fragmented landscapes are: the lack of dispersers due to high local fragmentation resulting from the exploitation process; the low-intensity continuous disturbance from remnant quarry activities; and the reduced biotic interactions with fauna due to structural homogenisation (Nunes et al., 2014; Ruiz-Benito et al., 2012). Furthermore, the lack of spatial habitat heterogeneity hinders the development of new biotic interactions due to providing fewer niches, less diverse ways of exploiting environmental resources, and lower species diversity than more structurally complex habitats (Tews et al., 2004).

Overall, the proportion of seed dispersal mechanisms in the restored areas differed from the reference ecosystem (*section 3.6.4.1*). In the reference ecosystem, 80% of the woody community relies on animals for seed dispersal, a percentage higher than in the intervention modes, where some have more than 50% of the community relying on wind for seed dispersal. The effect of Aleppo pine (whose seeds are dispersed by wind) on the proportion of seed dispersal strategies was mostly notorious. One could suggest that, in comparison with the restored areas, the traits in the reference ecosystem reflect the lower levels of environmental severity and higher levels of resource availability for reproduction and growth. In the natural areas, more plants invest energy in dispersal for attracting animals, as most of the species have fleshy fruits and primarily zoochorous long-distance seed dispersal (Gilardelli et al., 2015). The restored areas, on the other hand, have higher levels of environmental severity and a higher proportion of species with small seeds and anemochorous short dispersal distance, which may be due to the existing plantation of anemochoric species, but may also reflect the propensity of plants to reduce their resource requirements for dispersal in more selective environments with low nutrient availability (Pellissier et al., 2010).

Nevertheless, there is a considerable percentage of species dispersed by gravity in the restored areas. This dispersal strategy is more energetically costly for plants than anemochory (Pellissier et al., 2010). However, again, its proportions are lower in the plantations at the limestone benches than in the other intervention modes and NV, possibly suggesting lower energy investment of the species of this intervention mode. Additionally, a study by Nunes et al. (2014) showed that species with non-zoochorous dispersal (anemochorous and barochorous), which may be more tolerant to disturbance than small mammals or birds (Herrera, 1989), could benefit from the resources made available by pine thinning at the limestone benches. The same study showed that pine thinning decreased the proportion of zoochorous (endozoochorous) species, suggesting these were possibly limited by the infrequency or scarcity of dispersers in the quarry benches. This was confirmed by a later study by Sampaio et al. (2021) which revealed that the number of birds in the limestone benches plantations able to provide seed dispersal services was below the expected numbers reported in neighbouring natural areas. The study concluded that seed dispersal services at the quarry are depleted, possibly due to the sparse native vegetation cover and reduced food resources that induced changes in bird communities by hindering habitat suitability for seed-dispersing species, clearly suggesting a low ecological restoration success regarding dispersal capacity.

Moreover, species with fleshy fruit are valuable in biotic interactions, mainly because they are a food source for different animal groups, including birds (Whitford & Duval, 2019). Although animals can also consume dry fruits, which is evident in the NV that has mostly zoochorous dispersal and an even proportion (50%) of fleshy and dry fruits, the dominance of dry fruits in the intervention modes might reduce biotic interactions in the restored areas and be responsible for the reduced food resources that are affecting bird communities (*section 3.6.4.2*). This is the case at the limestone benches plantations, where dry fruits predominated (70%) mostly due to the high cover of Aleppo pine. Aleppo pine cover associated with poor soil quality and shallow soil depth in the restored area might constrain the development of native vegetation and, consequently, plant species with ripe fleshy fruits, which negatively influences the abundance of seed-dispersing birds, as reported by Sampaio et al. (2021). Therefore, the negative effects of the Aleppo pine reveal a bottom-up cascading effect of the revegetation practices in seed dispersal services provided by birds. For this reason, adaptive management interventions, such as pine thinning, should be performed to improve habitat suitability for fauna, promote biotic interactions, and enhance seed dispersal. Improving these aspects is essential to achieve the SER goal that the restored ecosystem should be capable of “sustaining reproductive populations of the species necessary for its stability or development along the desired trajectory”.

Furthermore, among dispersal traits, seed mass is a key functional trait related to many biological factors like dispersal ability (Weiher et al., 1999). Wind-dispersed seeds tend to have lower mass and are associated with large seed production, enhancing dispersal. However, seedling establishment is often positively associated with seed mass (Weiher et al., 1999). One of the most abundant species with dry fruit in the reference ecosystem, the zoochorous *Q. coccifera,* is responsible for the significantly higher seed mass in the NV sites than in the restored areas, where this species has not successfully established yet (ideally, it would have abundances similar to the reference ecosystem). Its heavy seeds, which have a remarkably higher seed mass than other typical species of the restored areas, are packed with nutrients that aid seedlings surviving and establishment, especially in the face of environmental stresses (Trubat et al., 2008). Furthermore, *Q. coccifera* edible acorns are an important high-protein food source for wild fauna, including foxes, rodents, and wild boars (Aronson et al., 2012; Pons & Pausas, 2007), which might suggest that natural areas have potentially greater food resources than the restored sites.

Additionally, seedling recruitment is a vital component if restoration efforts are to achieve a self-sustaining and resilient ecosystem (Florentine et al., 2013; Gijsbers et al., 1994; SER, 2004). Thus, it is important to understand whether the introduced plants reproduce and regenerate naturally, which can be translated into the seedling density of species, used in this research as an indicator of the self-sustainability and regeneration of the restored areas.

The density of woody seedlings increased significantly along the chronosequence and no significant differences between the intervention modes and the NV were found, indicating satisfactory self-sustainability capacity of the restored sites, with similar densities to the reference ecosystem (*section 3.6.4.2*). Nevertheless, the reference ecosystem had the highest maximum seedling density, potentially indicating that some sites in the NV have healthier populations than the restored sites. However, the ecological restoration success index (*section 3.6.5*) showed that seedling density was below reference ecosystem levels in all intervention modes, possibly due to the high variability and maximum seedling density values that inflated the average NV values. Nevertheless, plantations and some hydroseeded sites had the highest success index, which agrees with the produced map of seedling density that shows highest seedling density estimates for the north-facing limestone benches where plantations occurred. Such might indicate that regeneration by seed dispersal and seedling recruitment is not the problem in the limestone benches, but rather the competition between the Aleppo pines and the native shrub layer, hindering the latter from achieving the same cover, diversity, functional attributes, and consequently ES delivery, as the native shrub layer in the natural areas. As discussed in previous sections, pine thinning would potentially improve these aspects.

On the other hand, the south-facing marl slopes had the lowest seedling density estimates of the map and the lowest success index, suggesting that this area is still distant from offering the same self-sufficiency and regeneration capacity as the reference ecosystem. Such might support the idea that ecological recruitment barriers, which negatively impact the seedling density and self-sustainability of the system (Elgar et al., 2014; Suganuma & Durigan, 2015), may be present in the quarry slopes. Barriers to native plant regeneration might relate to competitive and dispersive barriers (Elgar et al., 2014). For example, the higher seedling densities in the limestone benches and the natural areas, when compared to the low values of the south-facing marl slopes, may be related to a lower density of herbs in these areas, as was also observed by Mexia (2008), facilitating colonization of woody species (Ruiz-Jaen & Mitchell Aide, 2005). Thus, the high herbaceous cover in the south-facing slopes might represent a competitive barrier to native seedling recruitment. Additionally, poor seed dispersal of the existent native woody shrubs in this area might be limiting seedling recruitment (Ruiz-Jaen & Mitchell Aide, 2005). Since seed-dispersing birds are attracted by ripe fleshy fruits (Sampaio et al., 2021), the dominance of dry fruits of the herbaceous species might be responsible for the lack of dispersers, which in turn might result in low seedling density in the south-facing marl slopes.

Moreover, restoration age and the density of adult shrubs showed to be crucial in aiding the establishment of the newly recruited woody seedlings, as they were the most relevant predictors when modelling seedling density, suggesting that these improve the self-sustainability of the restored sites and contribute to their resilience. This agrees with previous studies that demonstrated that shrubs facilitate seedling establishment by habitat amelioration, positively affecting restoration success (Gómez-Aparicio et al., 2004). Furthermore, the negative interaction between restoration age and shrub density essentially indicates that the effect of shrub density on seedling density is greater in the most recently restored sites, which correspond to the recently restored slopes characterised by their high herbaceous cover, emphasizing the importance of using native shrub species in these areas' restoration interventions. Since plantations close to seed sources could be expected to have more active recruitment and successional dynamics than plantations far from seed sources (Gómez-Aparicio et al., 2009), and the majority of slopes is more distant from natural sources than the limestone benches, the former should have interventions to promote their seedling recruitment and density.

Ultimately, the goal of ecological restoration is to create a self-sustainable ecosystem (similar to its reference ecosystem), resilient to perturbation without further assistance, and with the potential to persist indefinitely under existing conditions (Ruiz-Jaen & Mitchell Aide, 2005; SER, 2004). Thus, efforts should be made to increase seedling recruitment in the slopes where seedling density is currently low, minding that quarry revegetation with non-native species must be avoided, since these may have a negative effect on native fruit-bearing vegetation reestablishment and alter bird composition and the seed dispersal ES. Alternatively, the use of native shrubs should be favoured since shrub density is positively correlated with seedling density, assuring the development of suitable habitats to attract fauna and take advantage of the ES it provides. In the limestone benches, pine thinning should be performed to reduce competition with the native shrub species, improve habitat suitability for fauna and promote biotic interactions.

CHAPTER 5 | FINAL CONSIDERATIONS

Evaluating the success of ecological restoration actions is essential to determine the need for adaptive management strategies, to ensure both the efficacy of restoration efforts and the achievement of the SER's restoration goals. The main aim of this study was to evaluate the success of the ecological restoration of a quarry located within the Arrábida Natural Park, a protected area in Portugal with high ecological value. This was achieved by using field and RS data to study, model and map key ecosystem attributes associated with several ES.

Overall, ecosystem recovery in the SECIL-Outão quarry was incomplete. Full recovery is only achieved when "all key ecosystem attributes closely resemble those of the reference model" (Gann et al., 2019), which was not the case for the restored sites, as most studied attributes and associated ES did not reach those of the reference ecosystem (e.g. soil carbon sequestration, soil fertility, habitat quality, resilience to fire and drought). Moreover, the hypothesis that older restored quarry areas would be closer than recent ones to providing the same regulating ES as the reference ecosystem was only observed for some attributes, as this trend was not so clear or was the opposite for several of the studied attributes.

The analysis of the ecosystem attributes over time, along a chronosequence, supported the proposed hypothesis that the recovery trajectory would differ between ecosystem attributes, since each recovered at different speeds due to their inherent natural dynamics and other factors, such as the intervention mode applied. While some variables recovered more quickly and continued increasing with restoration age (e.g. seedling density – regeneration), showing positive signs of improvement, others showed a slower recovery (e.g. SOC - soil fertility). Some ecosystem attributes show signs of stabilisation at 15-20 years after restoration actions occurred (e.g. productivity, soil decomposition, similarity with the reference ecosystem), indicating limitations in the ecosystem, possibly due to the high competition and exhaustion of limited resources caused by the high Aleppo pine cover. Furthermore, the observed decrease over time in ecosystem functioning variables, such as the proportion of entomophilous species (pollination), the proportion of resprouters (resilient to fire) or sclerophyll vegetation (resilient to drought), might negatively affect the restored ecosystem's resilience. This tendency is particularly worrying in a climate change scenario, where higher fire frequency and intensity and more extended and severe droughts are expected. Thus, older restored sites might be in a vulnerable state due to their decreasing resilience, which could jeopardize restoration efforts in case of future disturbances, calling the need for adaptive management actions.

Furthermore, it was not possible to determine which intervention mode (plantations, hydroseeding, or mixed) was the most adequate for successful ecological restoration, given that they all have positive and negative aspects, with trade-offs occurring between attributes as well (e.g. pollination service provision and ecosystem similarity with the reference ecosystem). Overall, soil fertility values were below reference levels in the restored sites regarding SOC, SOM, and N. Furthermore, the natural soils stored around three times more and up to 10 times more carbon than the restored sites' soils. Such reveals the limitations of the restored sites in contributing to carbon sequestration to the same degree as their reference ecosystem. Additionally, the recalcitrant pine needles in the plantations are thought to contribute to the low decomposition rates, limiting primary productivity and growth in the older restored sites.

Overall, the intervention modes differed from the reference ecosystem due to the dominance of Aleppo pine in the plantations and the dominance of the herbaceous layer where hydroseeding was performed. Thus, the restored areas had low recovery levels regarding shrub cover, which is dominant in the reference ecosystem. These factors contributed to distancing the restored areas from the reference ecosystem in terms of species composition (low similarity indices) and habitat quality. Additionally, functional diversity and the proportion of sclerophyll vegetation and resprouters were below reference ecosystem levels in the old restored benches plantations. On the other hand, hydroseeded sites appear to have problems regarding seedling density, possibly due to the lack of dispersers and food resources from native shrubs' fleshy fruits, which might decrease biotic interactions.

Additionally, modelling the ecosystem attributes' variables allowed the identification of promoters or limiting factors of recovery. Pine cover contributed to the slow decomposition rates in the soil and decreased functional diversity in the restored areas, which might have negative implications for ecosystem functioning and the quality of ES provided by the restored areas. Pine cover was also found to decrease the resilience to drought (sclerophyll vegetation) and fire (resprouters), apart from negatively influencing the pollination service (flowering duration and entomophily). Since the high pine cover became an obstacle to the progression of the plant community in the direction of the reference system, pine thinning interventions are highly recommended. Such interventions are expected to promote key functional traits and contribute, in the medium term, to improving soil and nutrient cycling, ecosystem resilience, and biotic fluxes (Nunes et al., 2014). On the other hand, shrub density in the restored sites proved to be very important in increasing species similarity with the reference ecosystem, taxonomic and functional diversity, and seedling density. Thus, interventions that promote higher native shrub densities are highly suggested as these results emphasise the importance of using native shrub species in the recovery of all quarry locations to improve the functioning, resilience, and self-maintenance of the restored areas.

Furthermore, multiple maps created by statistical models based on RS, topographical variables and field data were successfully produced. Their estimates were overall comparable with the statistical analysis of the different intervention modes and the recovery success indices calculated for each ecosystem attribute (which considered the different intervention modes and zone types). Thus, the produced maps supported the hypothesis that RS data, together with field data, allow upscaling information and the creation of valuable models that can be used to produce spatial information at the landscape scale. These maps provided the spatial distribution of relevant ecological indicators, indicating which areas are more in need of adaptive management actions and, in this way, contributed to the planning of future adaptive management actions.

In conclusion, this study shows that the restored ecosystem differs from its reference model and the ES it provides, even after 30 years of restoration. These findings support the idea that ecological restoration should not be viewed as a substitute for conservation, nor should the promise of restoration be used to justify further destruction or unsustainable use, as restoration may not succeed in re-establishing the full assemblage of native species or the full extent of the original ecosystem's structure and function (Aronson & Alexander, 2013; SER, 2023).

Contribution, Limitations and Recommendations for Future Research

Positive aspects of this study include its contribution to the assessment of ecological restoration success by using multiple attributes and measures, with at least two variables within each of the ecosystem attributes studied and at least two reference sites to capture the variation that exists in the natural ecosystems, as recommended by Ruiz-Jaen & Mitchell Aide (2005). Additionally, the results presented in this study give evidence of the necessity for adaptive management strategies to promote recovery and achieve the SER goals of resembling the ecosystem functioning, self-maintenance, and resilience of the reference ecosystem.

Nevertheless, although the recovery trajectory was studied based on a chronosequence (a common approach to studying ecological succession), it would be important to complement the present study with future work that analyses individual sites over time using data time series, taking also advantage of the data collected in SECIL over the past decades. Furthermore, future studies concerning fauna would be relevant since zoological studies can directly support the botanical aspects of restoration as seed dispersers, pollinators, and grazers are central to the success of restoration efforts (Young, 2000). Additionally, the functional diversity of individual traits should be assessed in the future to reflect the variety of strategies and traits in the restored areas. Such a study would bring further insight into resource complementary use and resilience to future climate change stresses, such as fires and drought in the restored sites.

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Appendix I – Supplementary Tables

Table S1.1. Description of the extracted remote sensing indices (date of the Sentinel-2 image: 03-07-2020).
Labels: Vegetation Radiometric Indices, Water Radiometric Index, Soil Radiometric Indices, Biophysical Variables.
Information was obtained from SNAP's "Help Content" section.

	Index	General Description
Vegetation Radiometric Indices	NDVI *Normalized Difference Vegetation Index*	Sensitive to chlorophyll, index of plant "greenness", enables you to monitor density and relative vigour of vegetation growth using the spectral reflectivity of solar radiation. NDVI is calculated from the visible and near-infrared light reflected by vegetation. Healthy vegetation absorbs most of the visible light that hits it and reflects a large portion of the near-infrared light. Unhealthy or sparse vegetation reflects more visible light and less near-infrared light.
	SAVI *Soil Adjusted Vegetation Index*	Huete (1988) presents a theoretical basis for this index based on simple radiative transfer, so SAVI probably has one of the better theoretical backgrounds of the vegetation indices; designed to minimize the soil brightness influences that occur in indexes like NDVI. Adjustment factor L: 0- very high vegetation 0.5- intermediate vegetation density 1- very low vegetation
	TSAVI *Transformed Soil Adjusted Vegetation Index*	This index assumes that the soil line has arbitrary slope and intercept, and it makes use of these values to adjust the vegetation index. This would be a nice way of escaping the arbitrariness of the L in SAVI, if an additional adjustment parameter had not been included in the index. The parameter X was adjusted so as to minimize the soil background effect.
	MSAVI *Modified Soil Adjusted Vegetation Index*	The adjustment factor L for SAVI depends on the level of vegetation cover being observed. This leads to the circular problem of needing to know the vegetation cover before calculating the vegetation index, which is what gives you the vegetation cover. The basic idea of MSAVI was to provide a variable correction factor L. The correction factor used is based on the product of NDVI and WDVI.
	MSAVI 2 *second Modified Soil Adjusted Vegetation Index*	Basically, they use an iterative process and substitute 1 - MSAVI(n-1) as the L factor in MSAVI(n). They then inductively solve the iteration where MSAVI(n) = MSAVI(n-1). In the process, the need to precalculate WDVI and NDVI and the need to find the soil line are eliminated.
	DVI *Difference Vegetation Index*	This is the simplest vegetation index. Sensitive to the amount of vegetation; distinguishes between soil and vegetation
	RVI *Ratio Vegetation Index*	Simplest ratio-based index (also called Simple Ration (SR)) High for vegetation; low for ice, water, etc; indicates amount of vegetation; eliminates various effects such as: irradiance (topography), transmittance (atmospheric effects)
	PVI *Perpendicular Vegetation Index*	Generalization of the DVI which allows for soil lines of different slopes

	Index	General Description
	IPVI *Infrared Percentage Vegetation Index*	IPVI is functionally equivalent to NDVI and RVI, but it only ranges in value from 0.0 - 1.0; IPVI eliminates one mathematical operation per image pixel which is important for the rapid processing of large amounts of data
	WDVI *Weighted Difference Vegetation Index*	This has a relationship to PVI similar to the relationship IPVI has to NDVI. WDVI is a mathematically simpler version of PVI, but it has an unrestricted range.
	TNDVI *Transformed Normalized Difference Vegetation Index*	Indicates a relation between the amount of green biomass that is found in a pixel. It has a higher coefficient of determination (correlation) for the same variable and this is the difference between TNDVI and NDVI. The formula of TNDVI (square root of NDVI) has always positive values and the variances of the ratio are proportional to mean values.
	GNDVI *Green Normalized Difference Vegetation Index*	GNDVI is more sensible than NDVI to identify different concentration rates of chlorophyll, which is highly correlated at nitrogen. This is an indicator of the photosynthetic activity of the vegetation cover; it is most often used in assessing the moisture content and nitrogen concentration in plant leaves. The use of green spectral band was more efficient than the red spectral band to discriminate nitrogen. Used in assessing depressed and aged vegetation.
	GEMI *Global Environmental Monitoring Index*	Attempts to eliminate the need for a detailed atmospheric correction by constructing a stock atmospheric correction for the vegetation index. A paper by Leprieur et al. (1994) claims to find that GEMI is superior to other indices for satellite measurements.
	ARVI *Atmospherically Resistant Vegetation Index*	Resistance to atmospheric effects. This index takes advantage of the different scattering responses from the blue and red band to retrieve information regarding the atmosphere opacity.
	NDI45 *Normalized Difference Index*	This algorithm is more linear, with less saturation at higher values than the NDVI. Some algorithms have already been presented in work by Delegido et al. (2011b) which specifically investigated the optimal bands to use in the NDVI formula with synthesised Sentinel-2 data. Research found that bands 4 and 5 were the optimal combination.
	MTCI *Meris Terrestrial Chlorophyll Index*	Designed to estimate chlorophyll content especially from Merris datasets. This index is sensitive to a wide range of chlorophyll concentrations since the reflectance from the NIR, red-edge and red bands are used in the calculation (aims at estimating the Red Edge Position (REP) – maximum slant point in the red and near-infrared region of the vegetal spectral reflectance). Useful for observing chlorophyll contents, vegetation senescence, and stress for water and nutritional deficiencies (2004).
	MCARI *Modified Chlorophyll Absorption Ratio Index*	Gives a measure of the depth of chlorophyll absorption and is very sensitive to variations in chlorophyll concentrations as well as variations in Leaf Area Index (LAI). MCARI values are not affected by illumination conditions, the background reflectance from soil and other non-photosynthetic materials observed.
	REIP *Red-Edge*	Developed for applications in biomass and nitrogen (N) uptake measurement in heterogeneous fields (1988). Red edge, as the inflection point of the strong

	Index	General Description
	Inflection Point Index	red absorption to near infrared reflectance, includes the information of both crop N and growth status. The reflectance around red edge is sensitive to wide range of crop chlorophyll content, N content, LAI and biomass.
	S2REP *Sentinel-2 Red-Edge Position Index*	S2REP should provide better characterisation of the Red Edge slope compared to application of the method using the MERIS or future Sentinel-3 sensors.
	IRECI *Inverted Red-Edge Chlorophyll Index*	Incorporates the reflectance in four bands to estimate canopy chlorophyll content. Increases in the amount of chlorophyll visible to the sensor, either through an increase in leaf chlorophyll content, or Leaf Area Index (LAI), result in a broadening of a major chlorophyll absorption feature centred around 680 nm. The effect is to cause a movement of the point of maximum slope, termed the red edge position (REP). The position of the red edge has been used as an indicator of stress and senescence of vegetation
	PSSRa *Pigment Specific Simple Ratio*	Chlorophyll index. It investigates the potential of a range of spectral approaches for quantifying pigments at the scale of the whole plant canopy. When applying existing narrow-band pigment indices, the PSSR algorithms have the strongest and most linear relationships with canopy concentration per unit area of Chl a (Chlorophyll a), Chl b (Chlorophyll b) and Cars (carotenoids).
Biophysical Variables	LAI: Leaf Area Index	Dimensionless quantity that characterizes plant canopies. It is defined as half the developed area of photosynthetically active elements of the vegetation per unit horizontal ground area. It determines the size of the interface for exchange of energy (including radiation) and mass between the canopy and the atmosphere. LAI is directly related to the amount of light that can be intercepted by plants. It is an important variable used to predict photosynthetic primary production, evapotranspiration and as a reference tool for crop growth. The satellite-derived value corresponds to the total green LAI of all the canopy layers, including the understory which may represent a very significant contribution, particularly for forests. Practically, the LAI quantifies the thickness of the vegetation cover. LAI is recognized as an Essential Climate Variable (ECV) by the Global Climate Observing System (GCOS).
Biophysical Variables	FAPAR: Fraction of Absorbed Photosynthetically Active Radiation	The FAPAR quantifies the fraction of the solar radiation absorbed by live leaves for the photosynthesis activity. Then, it refers only to the green and alive elements of the canopy. The FAPAR depends on the canopy structure, vegetation element optical properties, atmospheric conditions, and angular configuration. To overcome this latter dependency, a daily integrated FAPAR value is assessed. This biophysical variable is directly related to the primary productivity of photosynthesis and some models use it to estimate the assimilation of carbon dioxide in vegetation. FAPAR can also be used as an indicator of the state and evolution of the vegetation cover; with this function, it advantageously replaces the Normalized Difference Vegetation Index (NDVI), provided it is itself properly estimated. FAPAR is recognized as an Essential Climate Variable (ECV) by the Global Climate Observing System (GCOS).

	Index	**General Description**
	FCOVER: Fraction of vegetation cover	The Fraction of Vegetation Cover (FCover) corresponds to the fraction of ground covered by green vegetation. Practically, it quantifies the spatial extent of the vegetation. Because it is independent from the illumination direction and it is sensitive to the vegetation amount, FCover is a very good candidate for the replacement of classical vegetation indices for the monitoring of ecosystems.
	Cab: Chlorophyll content in the leaf	Cab (leaf cholrophyll content (µg / cm ^ 2)) corresponds to the content of chlorophyll a, chlorophyll b and carotenoids per unit of leaf area.
	CWC: Canopy Water Content	Since radiation is absorbed significantly by water in the near and middle infrared, the spectral configuration of SENTINEL2 allows accessing this variable. Water represents between 60 % and 80% of the living plant mass. The variable that is the best related to the remote sensing signal is defined as the mass of water per unit ground area (g.m-2). One of the difficulties in retrieving this variable is the possible confusion with soil moisture effects
Water Radiometrix Index	NDWI *Normalized Difference Water Index*	Measure of liquid water molecules in vegetation canopies that interacted with the incoming solar radiation. NDWI is sensitive to changes in liquid water content of vegetation canopies. It is less sensitive to atmospheric effects than NDVI.
Soil Radiometric Indices	BI *Brightness Index*	Represents the average of the brightness of a satellite image. This index is sensitive to the brightness of soils which is highly correlated with the humidity and the presence of salts in surface
	BI2 *Second Brightness Index*	Represents the average of the brightness of a satellite image. This index is sensitive to the brightness of soils which is highly correlated with the humidity and the presence of salts in surface
	RI *Redness Index*	Developed to identify soil colour variations
	CI *Colour Index*	Developed to differentiate soils in the field. Low valued CIs have been shown to be correlated with the presence of a high concentration of carbonates or sulfates and higher values to be correlated with crusted soils and sands in arid regions. In most cases the CI gives complementary information with the BI and the NDVI. Used for diachronic analyses, they help for a better understanding of the evolution of soil surfaces.

Table S1.2. Complete list of the variables and predictors used in modelling and their description, units and sources. Remote sensing indices used as predictors in the models are shown in Table S1.3. Notes: SECIL's quarry DTM was used for the quarry area.

Variable/predictor	**Description**	**Units**	**Sources**
Restoration age	Years after the end of the restoration intervention	Years	Information from SECIL databases
Slope	Slope	°	SECIL DTM and ASTER GDEM
Elevation	Elevation above mean sea level	m	SECIL DTM and ASTER GDEM
PSR	Potential solar radiation	WH/m^2	ASTER GDEM
Latitude	Latitude	°	ASTER GDEM
Organic matter	Organic matter in soil	%	Field
Shrub cover	Shrub cover	%	Field
Pine cover	Pine relative cover	%	Field
Pine absolute cover	Pine absolute cover (sum of canopy areas)	%	Field
Herbaceous cover	Absolute cumulative herbaceous cover (sum of herbaceous species cover in each plot)	-	Field
Shrub height	Average shrub height in each plot	cm	Field
PR90 shrub height	Average 90 percentile of shrub height in each plot	cm	Field
Tree height	Average tree height in each plot	m	Field
PR90 tree height	Average 90 percentile of tree height in each plot	m	Field
Intervention mode	Mode of intervention performed in the revegetated areas	Categorical unit	Information from the natural of the intervention (hydroseeding, plantation, both or unknow) and application of the recommentations (or pilot tests) of the FCUL team
Zone type	Zone type based on classes of aspect and slope	Categorical unit	ASTER GDEM
Carbon in soil	Total organic carbon in soil	%	Field
C:N ratio	Carbon-to-nitrogen ratio: ratio of carbon to nitrogen in a substance	-	Field
Height woody (CWM)	Community weighted mean of height in woody vegetation (shrubs and *pine* trees)	m	Functional trait analysis
Height herbs (CWM)	Community weighted mean of height in herbaceous vegetation	cm	Functional trait analysis
Shannon shrubs	Shannon diversity index in the shrub community	-	Taxonomic diversity analysis
Shannon woody	Shannon diversity index in the woody community	-	Taxonomic diversity analysis
Shannon herbs	Shannon diversity index in the herbaceous community	-	Taxonomic diversity analysis
Seedling density	Density of seedlings in each plot	Nr of individuals / m^2	Field
Sørensen all	Sørensen–Dice similarity in all plant community (shrubs, trees and herbs)	-	Similarity analysis
Bray-Curtis all	Bray-Curtis similarity in all plant community (shrubs, trees and herbs)	-	Similarity analysis
Sørensen shrubs	Sørensen–Dice similarity in shrub community	-	Similarity analysis

Variable/predictor	Description	Units	Sources
Bray-Curtis shrubs	Bray-Curtis similarity in shrub community	-	Similarity analysis
FRic shrubs	Multi-trait functional richness in the shrub community	-	Functional diversity analysis
FRic woody	Multi-trait functional richness in the woody community (shrubs and pine trees)	-	Functional diversity analysis
FDis woody	Multi-trait functional dispersion in the woody community (shrubs and pine trees)	-	Functional diversity analysis
FDis herbs	Multi-trait functional dispersion in the herbaceous community	-	Functional diversity analysis
Entomophily woody and herbs (CWM)	Community weighted mean of entomophily in the woody (shrubs and pine trees) and herbaceous community	-	Functional trait analysis
CWM flowering duration woody	Community weighted mean of flowering duration in the woody community (shrubs and pine trees)	months	Functional trait analysis
Resprouters woody (CWM)	Community weighted mean of resprouters in the woody community (shrubs and pine trees)	-	Functional trait analysis
Resprouters shrubs (CWM)	Community weighted mean of resprouters in the shrub community	-	Functional trait analysis
SLA woody	Community weighted mean of SLA (specific leaf area) in the woody community (shrubs and pine trees)	mm^2/mg	Functional trait analysis
Sclerophyll vegetation woody (CWM)	Community weighted mean of sclerophyll vegetation in the woody community (shrubs and pine trees)	-	Functional trait analysis
FAPAR (proxy for aboveground productivity)	Fraction of Absorbed Photosynthetically Active Radiation	-	Sentinel-2

Table S1.3. List of the remote sensing indices and ratios used as predictors in the models and the dates of the images that derived them.

Remote sensing indices and ratios	Sentinel-2 images dates
$\frac{NDVI\ December}{NDVI\ July}$	December: 05-12-2020 July: 08-07-2022
$\frac{NDVI\ December}{NDVI\ October}$	December: 05-12-2020 October: 11-10-2020
$\frac{NDVI\ February}{NDVI\ July}$	February: 19-02-2020 July: 28-07-2020
$\frac{NDVI\ February}{NDVI\ November}$	February: 19-02-2020 November: 15-11-2020
$\frac{NDVI\ February}{NDVI\ October\ 2}$	February: 19-02-2020 October 2: 16-10-2020
$\frac{NDVI\ February}{NDVI\ October}$	February: 19-02-2020 October: 11-10-2020
$\frac{NDVI\ May}{NDVI\ January}$	May: 24-05-2020 January: 20-01-2020

Remote sensing indices and ratios	**Sentinel-2 images dates**
$\frac{NDVI\ May}{NDVI\ August}$	May: 24-05-2020 August: 27-08-2020
$\frac{NDVI\ May}{NDVI\ July}$	May: 24-05-2020 July: 28-07-2020
$\frac{NDVI\ May}{NDVI\ October\ 2}$	May: 24-05-2020 October 2: 16-10-2020
$\frac{NDVI\ September}{NDVI\ February}$	September: 06-09-2020 February: 19-02-2020
$\frac{NDVI\ September}{NDVI\ May}$	September: 06-09-2020 May: 24-05-2020
BI	03-07-2022
BI2	03-07-2022
NDVI June	09-06-2020

Table S1.4. List of the predictors present in the final explanatory models. Remote sensing indices and NDVI ratios used as predictors in the models are shown in Table S1.3.

Predictors
BI
BI2
Remote sensing indices and NDVI ratios
Latitude
Elevation
Slope
Intervention mode
Zone type
Restoration age
Organic matter
Pine absolute cover
Pine cover
Tree density
Shrub cover
Shrub density
Shrub height
Herbaceous cover

Table S1.5. Final explanatory models for 3 different analyses: 1) Models based on restoration age, to analyse the behaviour of the variables over time (based on a chronosequence); 2) Models based on all predictor variables (biological, environmental, topographical, restoration age, intervention mode) to understand which predictors best explain the variables of interest; 3) Models for upscaling and mapping at a landscape scale, based on spatially distributed predictors (environmental and topographical variables, remote sensing indices and ratios) to estimate and map the variables for the entire quarry area. For these models, data from 59 restored quarry sites was used (excluding data from NV). Notes: For each model, coefficient estimates are shown only for the continuous variables. In the case of categorical variables and interactions involving them, it was decided to replace the coefficients (one for each category) with the symbol "$" to simplify the visualization of the formulas. n.s. - statistically non-significant.

Results Index	**Variables**	**Models based on restoration age**	**Pseudo R^2**	**Models based on all predictors**	**Pseudo R^2**	**Models for upscaling and mapping at a landscape scale**	**Pseudo R^2**
3.1. Productivity and carbon	**FAPAR (proxy for aboveground productivity)**	$-1.502 \times 10^{-1} + 8.708 \times 10^{-2}$ Restoration age $- 4.405 \times 10^{-3}$ Restoration age^2 $+ 7.003 \times 10^{-5}$ Restoration age^3	0.32	-0.3449 + 0.4835 Pine absolute cover + $ Intervention mode + 0.2751 Shrub cover	0.45	FAPAR is already a spatially distributed variable	
	Carbon in soil*	$10^{(0.324575 + 0.004818 \text{ Restoration age})} - 1$	0.10	$10^{(1.891 \times 10^{-16} + 4.604 \times 10^{-1} \text{ Pine absolute cover} + \$ \text{ Intervention mode} + \$ \text{ Pine absolute cover} * \text{Intervention mode})} - 1$	0.46	$10^{(0.19105 + 0.52142 \text{ NDVI June})} - 1$	0.17
3.2. Soil fertility and decomposition	**Organic matter in soil**	$10^{(0.458290 + 0.005936 \text{ Restoration age})} - 1$	0.10	$10^{(0.623992 + 0.028635 \text{ Shrub height} + 0.118151 \text{ Pine absolute cover} + \$ \text{ Intervention mode} + \$ \text{ Pine absolute cover} * \text{Intervention mode})} - 1$	0.51	$10^{(0.9113 - 1.7828 \text{ BI2})} - 1$	0.17
	C:N ratio	16.3540454 - 1.0372320 Restoration age + 0.1000250 Restoration age^2 - 0.0020794 Restoration age^3	0.54	-0.6396 + 0.2454 Tree density + 0.3817 Restoration age + $ Intervention mode	0.64	28.86933 -47.98524 BI2 - 0.12093 Slope	0.50

Results Index	**Variables**	**Models based on restoration age**	**Pseudo R^2**	**Models based on all predictors**	**Pseudo R^2**	**Models for upscaling and mapping at a landscape scale**	**Pseudo R^2**
3.3 Habitat quality	**Pine cover**	-70.248 + 86.010 $\log_{10}$(Restoration age)	0.67	$-9.852 \times 10^{-1} + 1.888 \times 10^{-14}$ Restoration age + \$ Intervention + \$ Restoration age * Intervention mode	0.79	96.6310 -39.6030 $\frac{NDVI\ May}{NDVI\ August}$ -0.7126 Slope	0.64
	Shrub cover	n.s.	n.s.	n.s.	n.s.	79.643 -20.516 $\frac{NDVI\ February}{NDVI\ July}$	0.25
	Herbaceous cover	$10^{(1.8966677 + 0.0207664 \text{ Restoration age } -0.0013607 \text{ Restoration age}^2)} -1$	0.55	4.9853446 + 2.9347949 Restoration age -0.0318342 Restoration age^2 + 0.0006325 Restoration age^3 -0.2030771 Shrub density + \$ Intervention mode	0.76	99.3453 -72.3907 $\frac{NDVI\ September}{NDVI\ May}$ + 1.3304 Slope ₦	0.49
	Tree height	-8.7609318 + 2.7714363 Restoration age - 0.1551590 Restoration age^2 + 0.0027063 Restoration age^3	0.43	-0.18334 -0.38410 Restoration age + 0.01165 Organic matter + 0.87291 Pine cover - 0.01863 Restoration age*Organic matter -0.15172 Restoration age*Pine cover - 0.24243 Organic matter*Pine cover + 0.69650 Restoration age*Organic matter*Pine cover	0.65	14.189 - 3.336 $\frac{NDVI\ February}{NDVI\ July}$ - 36.456 BI	0.56

Results Index	Variables	Models based on restoration age	Pseudo R^2	Models based on all predictors	Pseudo R^2	Models for upscaling and mapping at a landscape scale	Pseudo R^2
	Height woody (CWM)	2.3284 + 0.2494 Restoration age	0.54	$10^{(-0.02933 + 0.66467 \text{ Pine cover} + 0.34291 \text{ Shrub height} + 0.21350 \text{ Restoration age} – 0.18380 \text{ Pine cover} * \text{Shrub height})} - 2$	0.89	$40.9876 - 49.8246 \frac{NDVI\ May}{NDVI\ July} - 260.9106\ BI2 - 3.2361\ Slope + 263.1639 \frac{NDVI\ May}{NDVI\ July} * BI2 + 2.7684 \frac{NDVI\ May}{NDVI\ July} * Slope + 15.8085\ BI2*Slope - 13.3176 \frac{NDVI\ May}{NDVI\ July} * BI2 * Slope$	0.68
	Height herbs (CWM)	$10^{(1.806897 - 0.012267 \text{ Restoration age})} - 2$	0.27	$10^{(3.406 \times 10^{-16} - 4.725 \times 10^{-1} \text{ Pine cover} - 2.254 \times 10^{-1} \text{ Shrub density})} - 2$	0.30	$10^{(1.26465 + 0.28754 \frac{NDVI\ May}{NDVI\ October\ 2})} - 2$	0.26
3.4. Similarity to NV	**Sørensen all**	- 0.917431+ 0.005863 Restoration age + 1	0.32	-1.5601 + 0.9030 Restoration age + $ Intervention mode + $ Restoration age * Intervention mode + 0.3650 Shrub cover + 1	0.65	$-0.5972678 - 0.1306172 \frac{NDVI\ May}{NDVI\ July} - 0.0023736\ Slope + 1$	0.51
	Bray-Curtis all	-0.938373 + 0.001609 Restoration age + 1	0.07	0.4502 + 1.0209 Shrub cover + $ Intervention mode + $ Shrub cover * Intervention mode + 1 ~~N~~	0.37	n.s.	n.s.

Results Index	Variables	Models based on restoration age	Pseudo R^2	Models based on all predictors	Pseudo R^2	Models for upscaling and mapping at a landscape scale	Pseudo R^2
	Sørensen shrubs	(-1.364810 + 0.002366 Restoration age)^2 + 1	0.22	0.4502 + 1.0209 Shrub cover + $ Intervention mode + $ Shrub cover * Intervention mode + 1	0.71	-0.498767 - 0.153402 $\frac{NDVI\ May}{NDVI\ July}$ - 0.002917 Slope + 1	0.37
	Bray-Curtis shrubs	n.s.	n.s.	n.s.	n.s.	n.s.	n.s.
3.5. Diversity	**Shannon shrubs**	1.27284 + 0.01373 Restoration age	0.08	0.43918 + 0.60346 Shrub cover + $ Intervention mode - 0.45153 Slope	0.58	-0.3714 + 2.6265 × $\log_{10}$(4.9137 - $\frac{NDVI\ February}{NDVI\ July}$) + 0.4364 × $\log_{10}$(39.8559165749 - Slope)	0.32
	Shannon woody	1.4117236 - 0.0005952 Restoration age	0.0001 (n.s.)	-0.04881 + 0.70163 Shrub cover	0.44	n.s.	n.s.
	Shannon herbs	0.1863161 + 0.3272616 Restoration age - 0.0208931 Restoration age^2 + 0.0003659 Restoration age^3	0.28	0.1863161 + 0.3272616 Restoration age -0.0208931 Restoration age^2 + 0.0003659 Restoration age^3	0.28	0.06503 + 1.03466 $\frac{NDVI\ February}{NDVI\ November}$	0.29

Results Index	**Variables**	**Models based on restoration age**	**Pseudo R^2**	**Models based on all predictors**	**Pseudo R^2**	**Models for upscaling and mapping at a landscape scale**	**Pseudo R^2**
	Fdis multi-trait shrub	n.s.	n.s.	-0.02113 + 0.39091 Shrub density	0.14	$0.67274 - 0.09981 \frac{NDVI\ May}{NDVI\ October\ 2}$ ₦	0.13
	FRic multi-trait woody	-0.0697251 + 0.0289798 Restoration age - 0.0006775 Restoration age^2	0.30	0.651579 + 1.041141 Restoration age - 0.002052 Restoration age^2 + 0.591445 Shrub cover	0.53	$0.43935 - 0.21602 \frac{NDVI\ May}{NDVI\ July}$	0.36
	Fdis multi-trait woody	n.s.	n.s.	-0.007703 -0.437576 Pine cover + 0.422297 Shrub density ₦	0.33	n.s.	n.s.
	Fdis multi-trait herbs	1.250×10^{-1} + 5.553×10^{-2} Restoration age - 3.683×10^{-3} Restoration age^2 + 6.708×10^{-5} Restoration age^3 ₦	0.24	3.296×10^{-16} -4.003×10^{-1} Pine cover ₦	0.16	$0.44073 - 0.18142 \frac{NDVI\ September}{NDVI\ February}$ ₦	0.18

Results Index	**Variables**	**Models based on restoration age**	**Pseudo R^2**	**Models based on all predictors**	**Pseudo R^2**	**Models for upscaling and mapping at a landscape scale**	**Pseudo R^2**
3.6.1. Pollination	**Entomophily woody and herbs (CWM)**	1.318642 - 0.013055 Restoration age	0.08	3.380×10^{-16} -3.986×10^{-1} Pine absolute cover	0.16	$0.3418 + 0.9959 \frac{NDVI\ May}{NDVI\ January}$	0.17
	Flowering duration woody (CWM)	$10^{(9.285 \times 10^{-1} - 4.355 \times 10^{-2}\ \text{Restoration age} + 2.184 \times 10^{-3}\ \text{Restoration age}^2 - 3.503 \times 10^{-5}\ \text{Restoration age}^3)} - 1$	0.41	$10^{(1.6025 - 0.2715\ \text{Pine cover} + \$\ \text{Intervention mode})} - 1$	0.49	n.s.	n.s.
3.6.2. Resilience and Adaptation to Fire	**Resprouters woody (CWM)**	0.747093 - 0.019039 Restoration age	0.40	0.02521 -0.72943 Pine cover	0.53	$-0.110920 + 0.280449 \frac{NDVI\ February}{NDVI\ October} + 0.006393\ \text{Slope}$	0.48
	Resprouters shrubs (CWM)	n.s.	n.s.	n.s.	n.s.	n.s.	n.s.
3.6.3. Resilience and Adaptation to Drought	**SLA woody (CWM)**	28.3469814 - 3.4749928 Restoration age + 0.1788803 Restoration age^2 - 0.0028937 Restoration age^3	0.65	-15.37 -12.01 Restoration age + 6.7×10^{-2} Restoration age^2 - 1.089×10^{-3} Restoration age^3 -1.764×10^{-1} Pine absolute cover + $ Intervention mode	0.76	$10^{(0.6574 + 1.6864\ \text{BI2})} - 1$	0.33
	Sclerophyll vegetation woody (CWM)	0.847813 - 0.017128 Restoration age	0.35	$2.7788 + 1.7309 \times \log_{10}(83.265 - \text{Pine cover})$	0.52	$-0.051516 + 0.260751 \frac{NDVI\ December}{NDVI\ October} + 0.007545\ \text{Slope}$	0.50

Results Index	**Variables**	**Models based on restoration age**	**Pseudo R^2**	**Models based on all predictors**	**Pseudo R^2**	**Models for upscaling and mapping at a landscape scale**	**Pseudo R^2**
3.6.4. Dispersal capacity, interaction with fauna and regeneration	**Seedling density**	$(0.298420 + 0.047038\ \text{Restoration age})^3$	0.41	$(0.11843 + 0.47804\ \text{Restoration age} + 0.51432\ \text{Shrub density} - 0.34560\ \text{Restoration age*Shrub density})^3$	0.67	$\left(2.345257 - 0.586836\ \frac{NDVI\ February}{NDVI\ October\ 2} - 0.019073\ \text{Slope}\right)^3$	0.56

Notes:

*: Organic matter is directly proportional to Carbon in soil. Both are indicators of Productivity and Soil fertility.

~~N~~: General linear model assumptions are not met by this model because errors have a non-normal distribution, and transformation of the response variable and predictor variables did not solve this problem.

Table S1.6. Botanic families recorded in the field sampling.

BOTANIC FAMILIES	
Amaryllidaceae	Lamiaceae
Anacardiaceae	Malvaceae
Apiaceae	Myrtaceae
Aristolochiaceae	Moraceae
Asparagaceae	Oleaceae
Aspleniaceae	Orchidaceae
Asteraceae	Pinaceae
Boraginaceae	Plantaginaceae
Brassicaceae	Poaceae
Caprifoliaceae	Primulaceae
Caryophyllaceae	Ranunculaceae
Cistaceae	Rhamnaceae
Convolvulaceae	Rosaceae
Crassulaceae	Rubiaceae
Cupressaceae	Rutaceae
Cyperaceae	Santalaceae
Ericaceae	Smilacaceae
Euphorbiaceae	Solanaceae
Fabaceae	Thymelaeaceae
Gentianaceae	Valerianaceae
Iridaceae	Xanthorrhoeaceae

Appendix II – Supplementary Figures

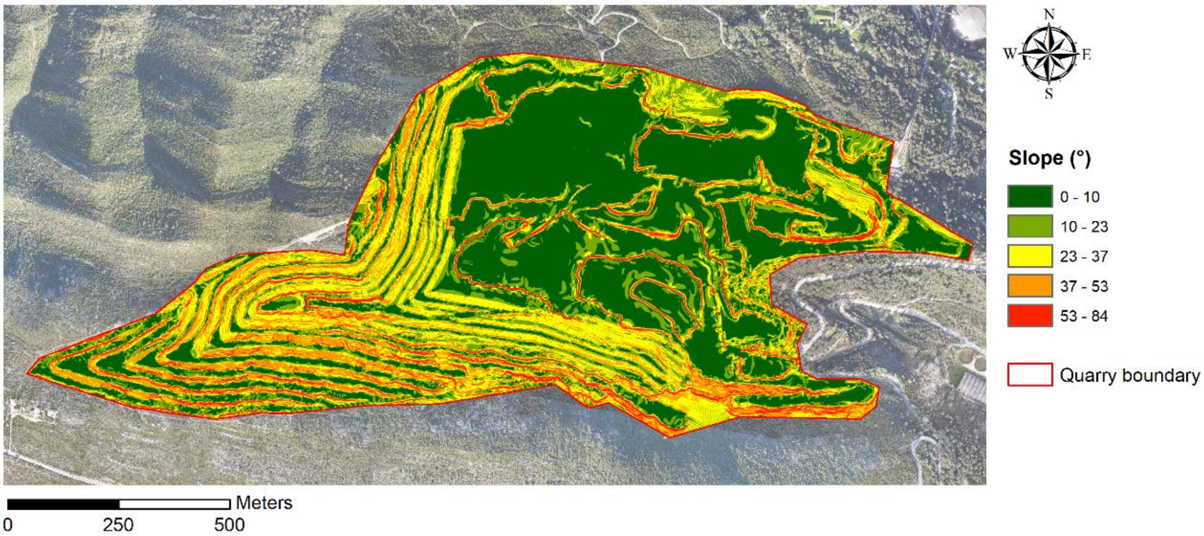

Figure S2.1. Slope (°) calculated with a DTM of 20 cm resolution.

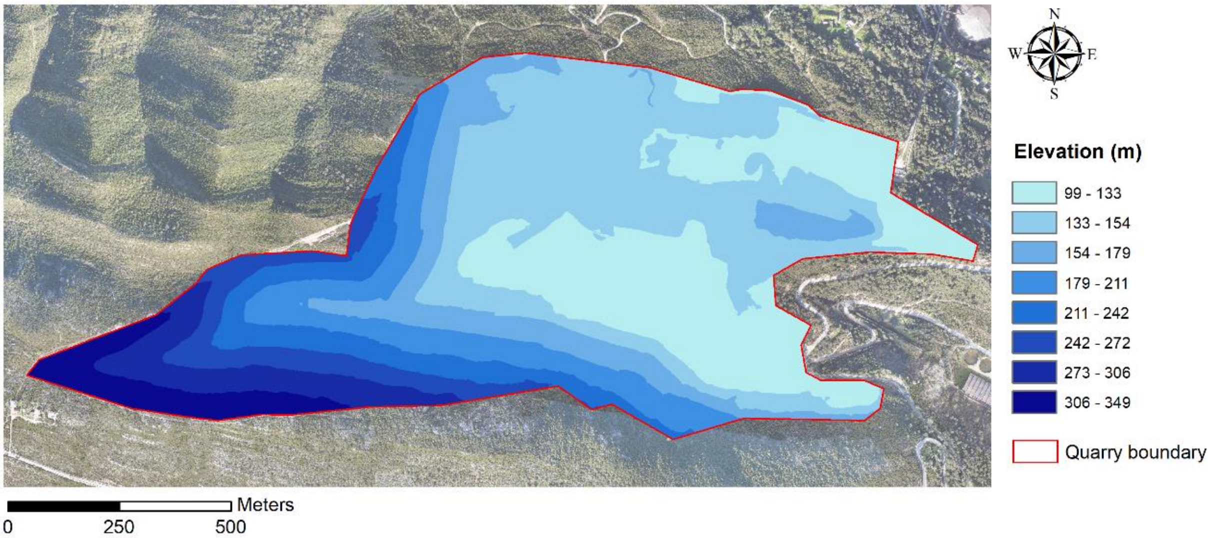

Figure S2.2. Elevation (m) calculated with a DTM of 20 cm resolution.

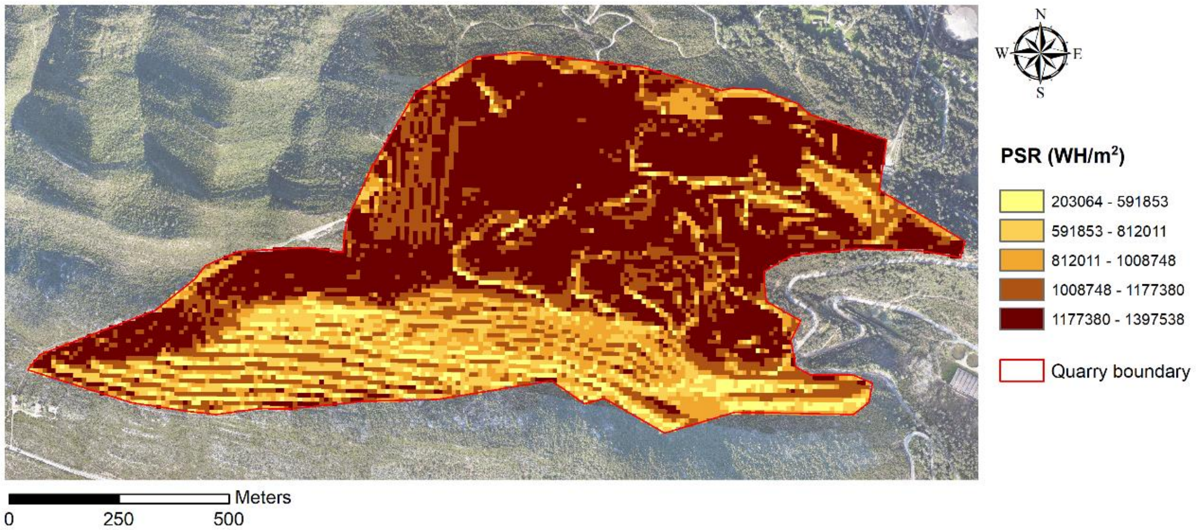

Figure S2.3. PSR (WH/m^2) calculated with a DTM of 20 cm resolution

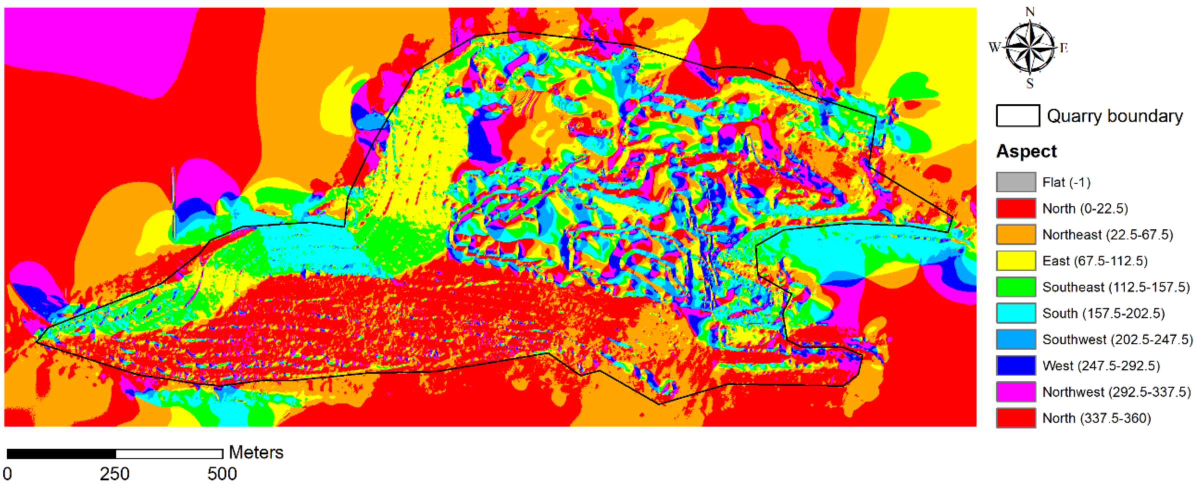

Figure S2.4. Aspect calculated with a DTM of 20 cm resolution.

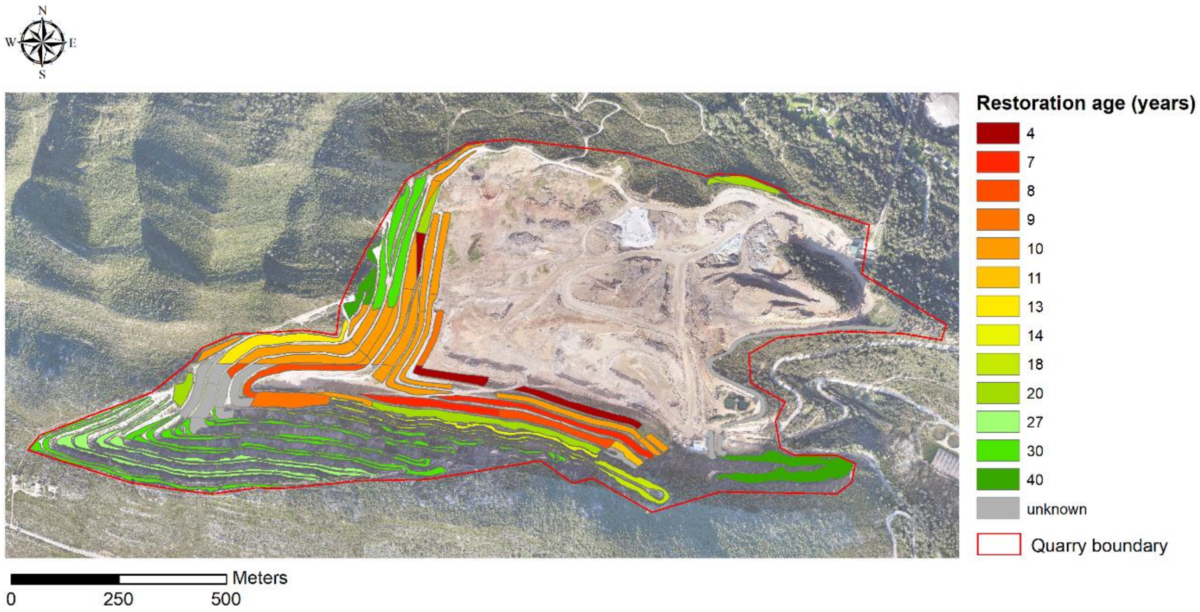

Figure S2.5. Restoration age (years after end of restoration intervention) of the different revegetated areas in the SECIL-Outão quarry.

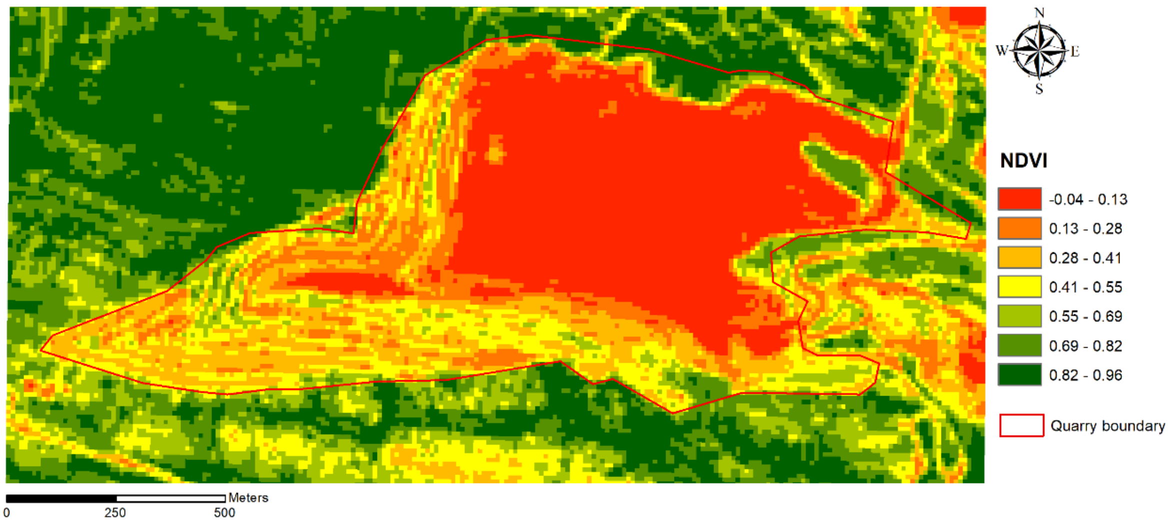

Figure S2.6. NDVI obtained through a Sentinel-2 image from 03-07-2020.

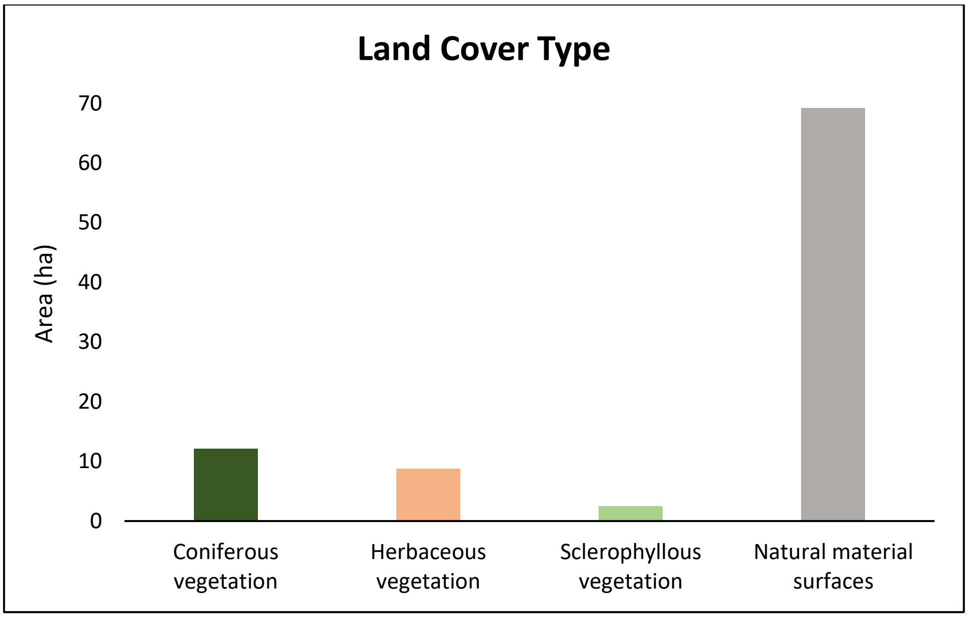

Figure S2.7. Area (ha) of each land cover type in the SECIL-Outão quarry.

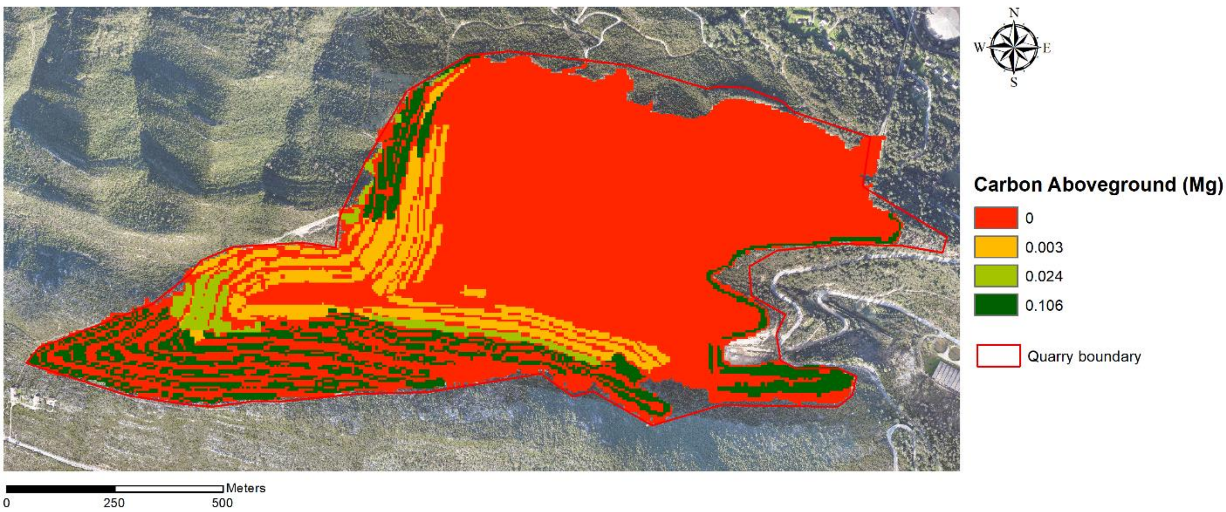

Figure S2.8. Map of the carbon sequestration aboveground provided by the quarry's revegetated areas. Natural material surfaces (rock), roads and escarpments have a value of 0. The Maquis vegetation surrounding the quarry (reference ecosystem), mainly composed of sclerophyllous vegetation, has a value of 0.024 Mg/pixel.

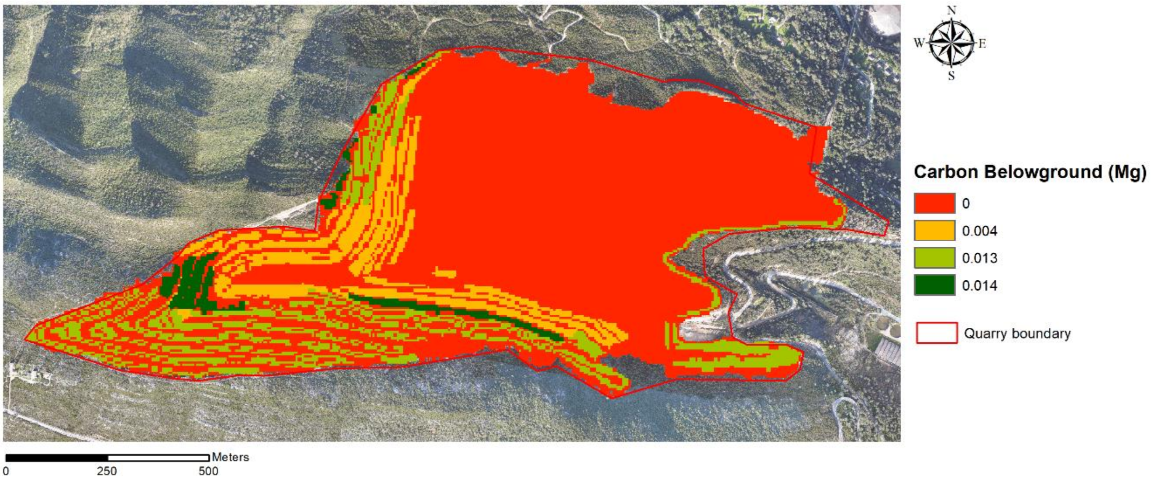

Figure S2.9. Map of the carbon sequestration belowground provided by the quarry's revegetated areas. Natural material surfaces (rock), roads and escarpments have a value of 0. The Maquis vegetation surrounding the quarry (reference ecosystem), mainly composed of sclerophyllous vegetation, has a value of 0.014 Mg/pixel.

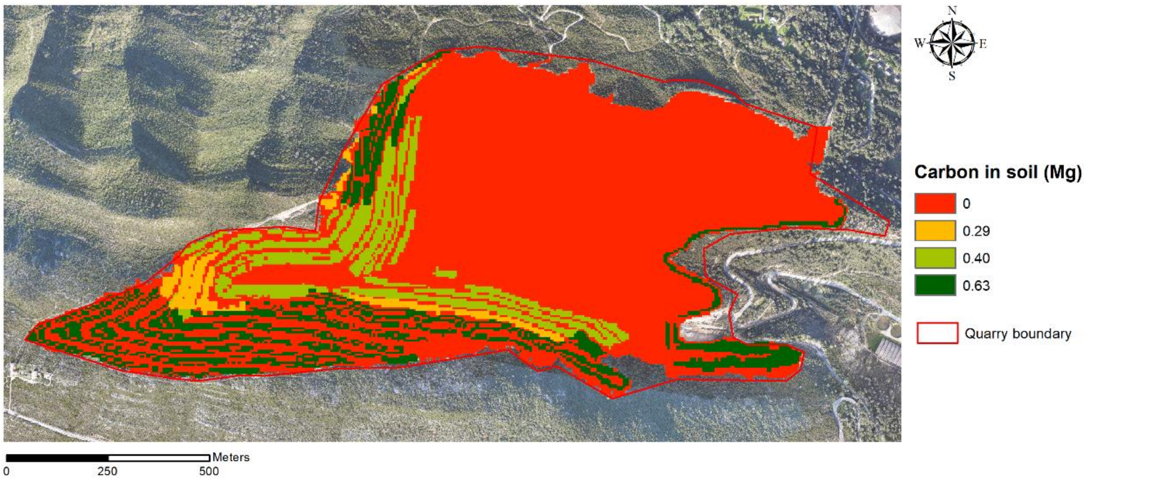

Figure S2.10. Map of the carbon sequestration in soil provided by the quarry's revegetated areas. Natural material surfaces (rock), roads and escarpments have a value of 0. The Maquis vegetation surrounding the quarry (reference ecosystem), mainly composed of sclerophyllous vegetation, has a value of 0.3 Mg/pixel.

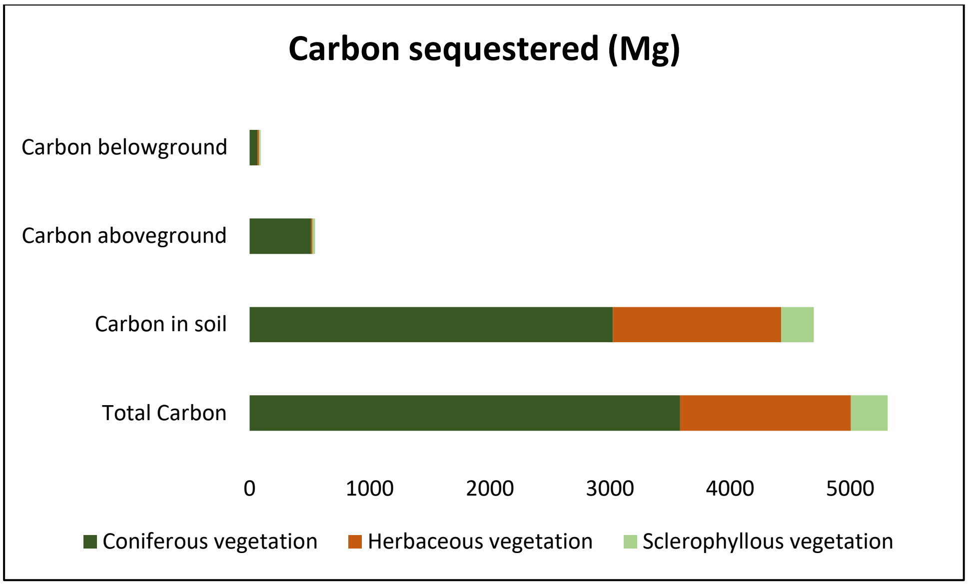

Figure S2.11. Carbon sequestered by the dominant vegetation types present at the quarry.

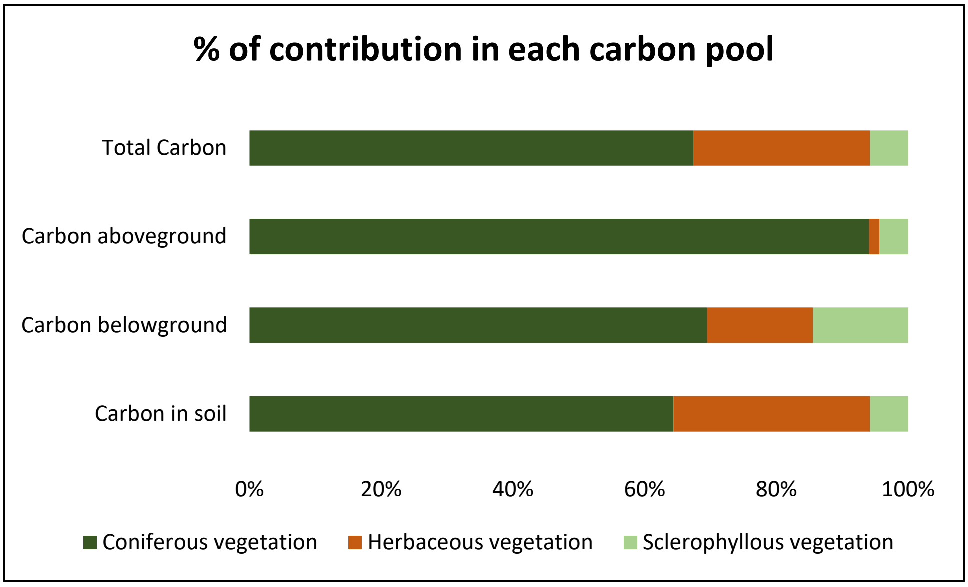

Figure S2.12. Contribution of each vegetation type in the quarry to the different carbon pools (%).

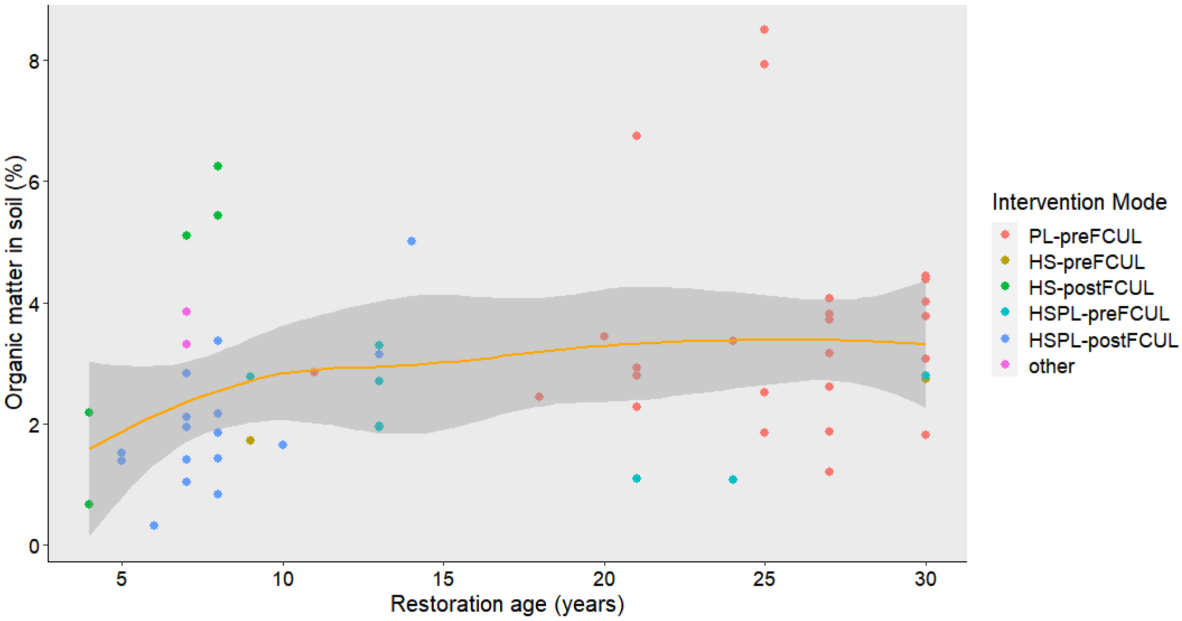

Figure S2.13. Variation of organic matter in soil (%) with restoration age (years) in quarry restored plots (R^2= 0.10, p<0.05).

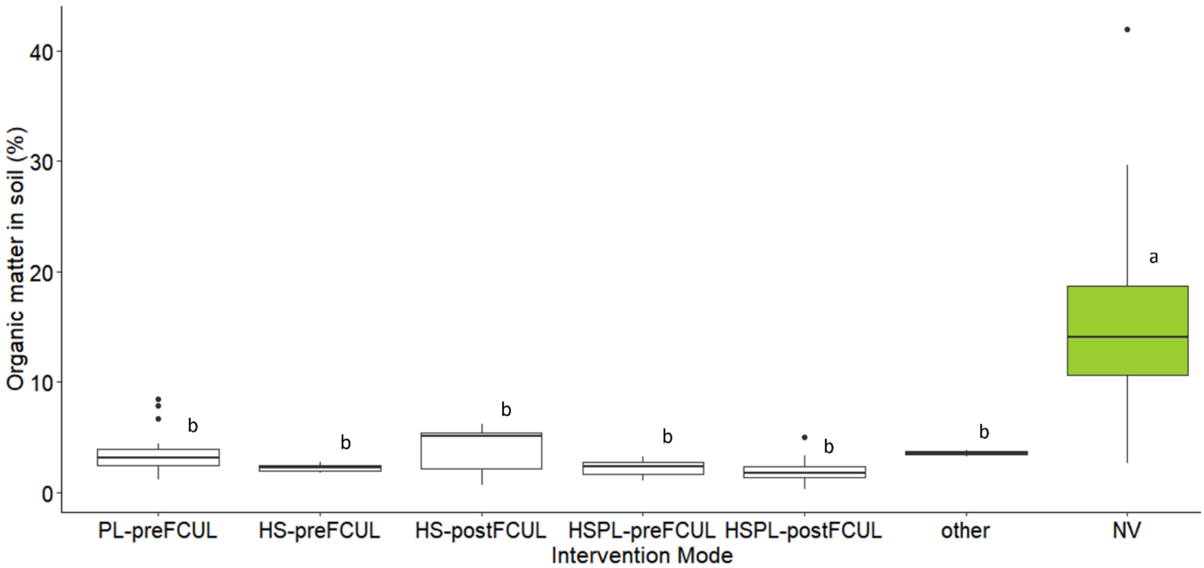

Figure S2.14. Variation of organic matter in soil (%) in the different intervention modes in quarry restored plots and the NV (reference ecosystem).

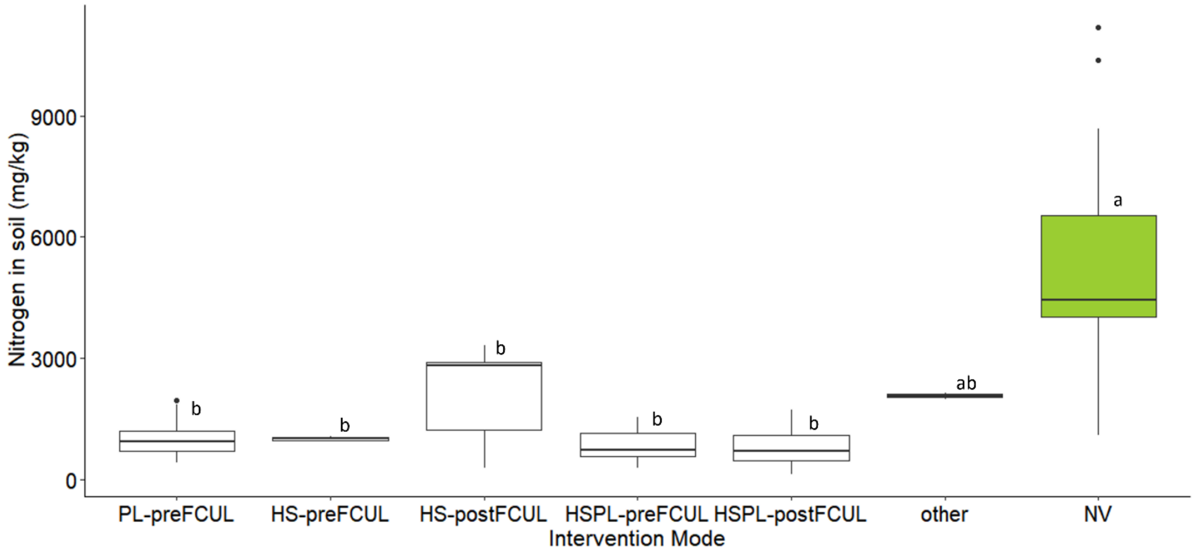

Figure S2.15. Variation of nitrogen in soil (mg/kg) in the different intervention modes in quarry restored plots and the NV (reference ecosystem).

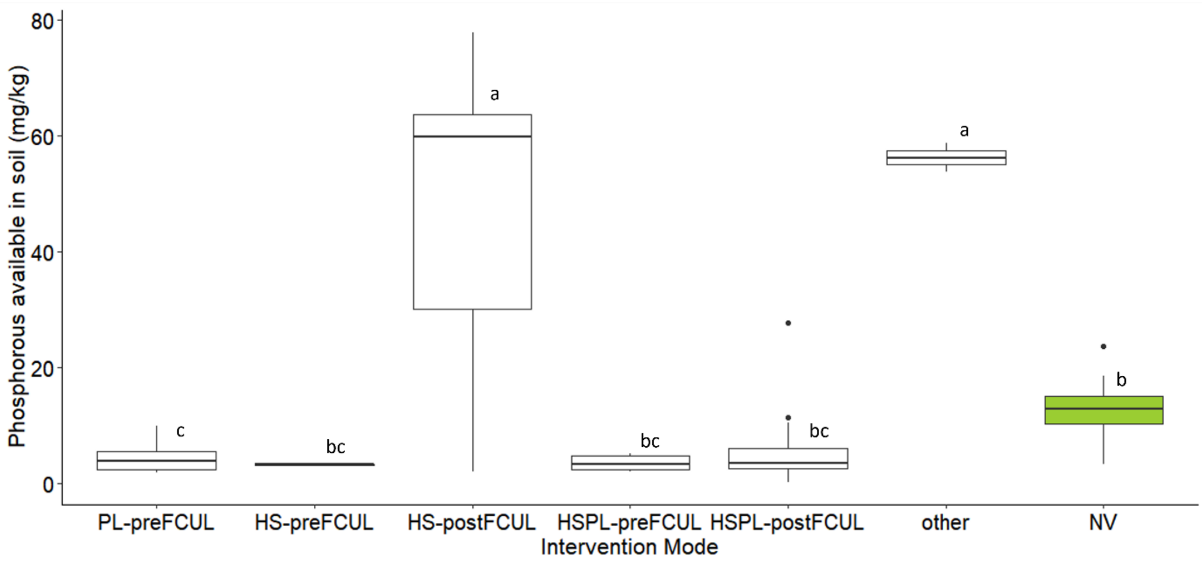

Figure S2.16. Variation of phosphorous available in soil (mg/kg) in the different intervention modes in quarry restored plots and the NV (reference ecosystem).

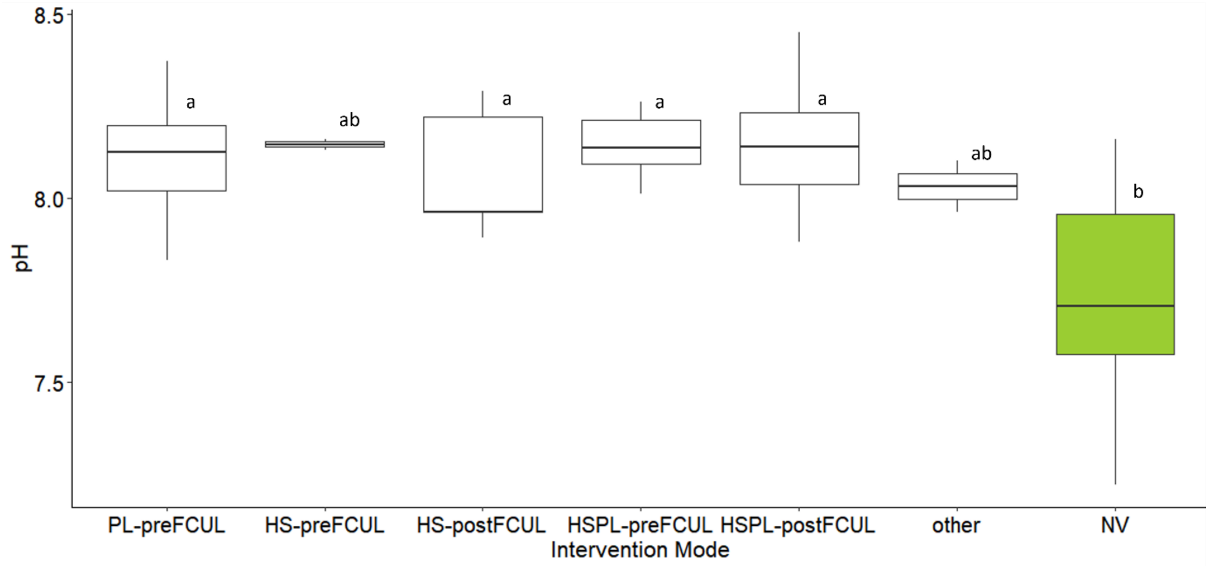

Figure S2.17. Variation of pH in the different intervention modes in quarry restored plots and the NV (reference ecosystem).

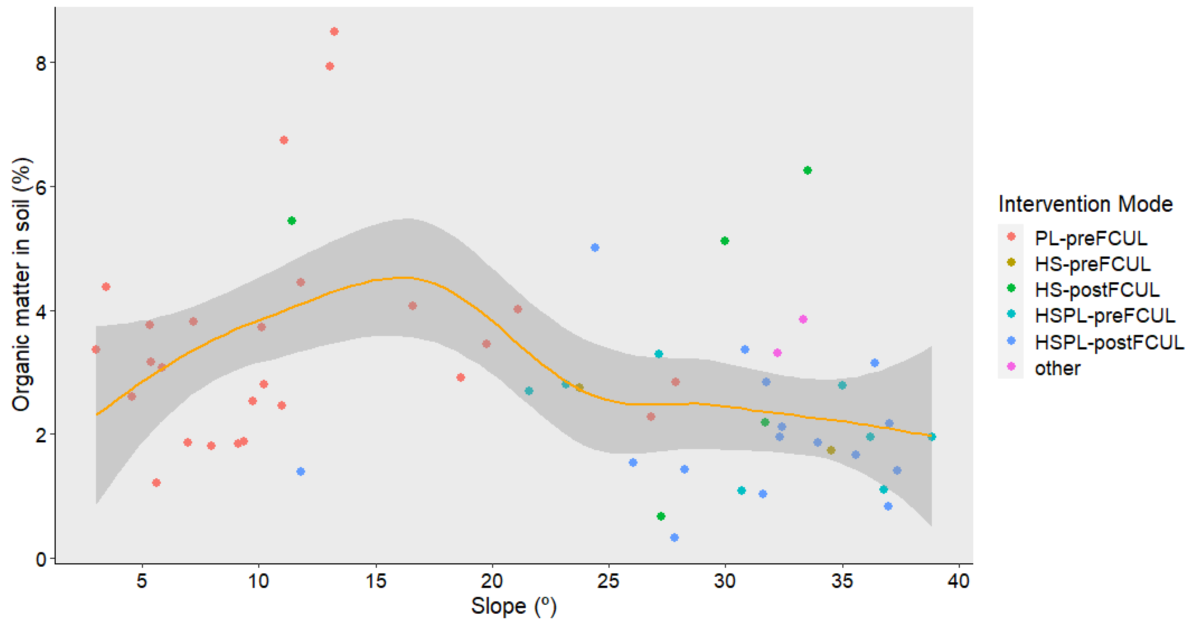

Figure S2.18. Variation of organic matter in soil (%) in soil with slope (º) in quarry restored plots (R^2= 0.09, p<0.05).

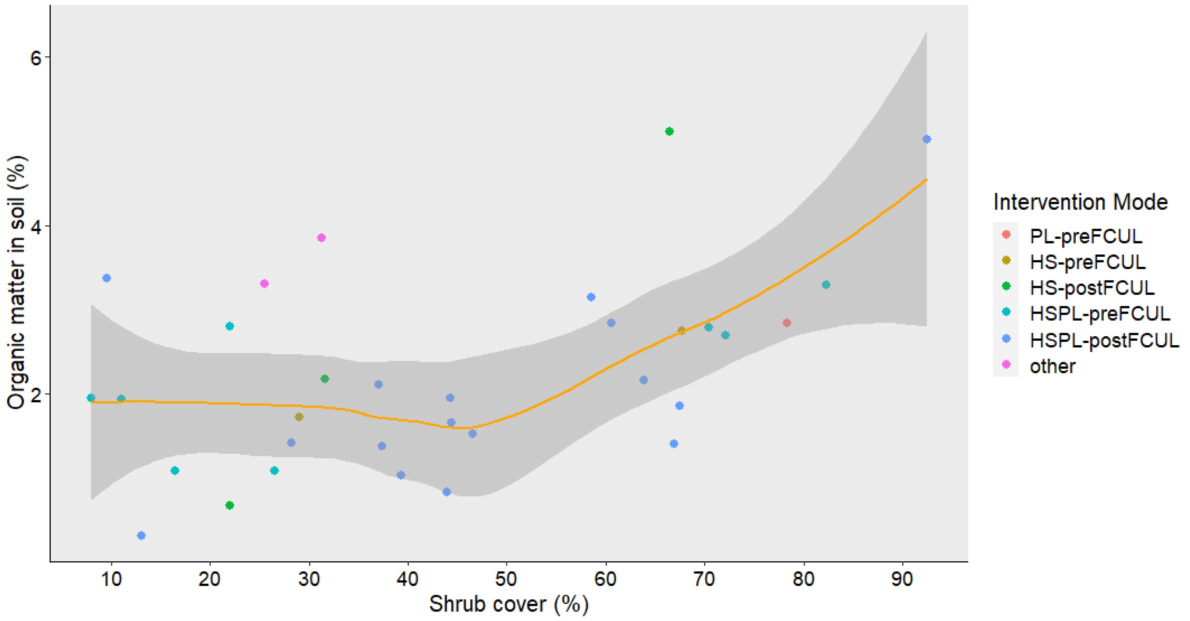

Figure S2.19. Variation of organic matter in soil (%) in soil with shrub cover (%) in quarry restored plots in limestone and marl slopes (excluding limestone benches) (R^2= 0.23, p<0.01).

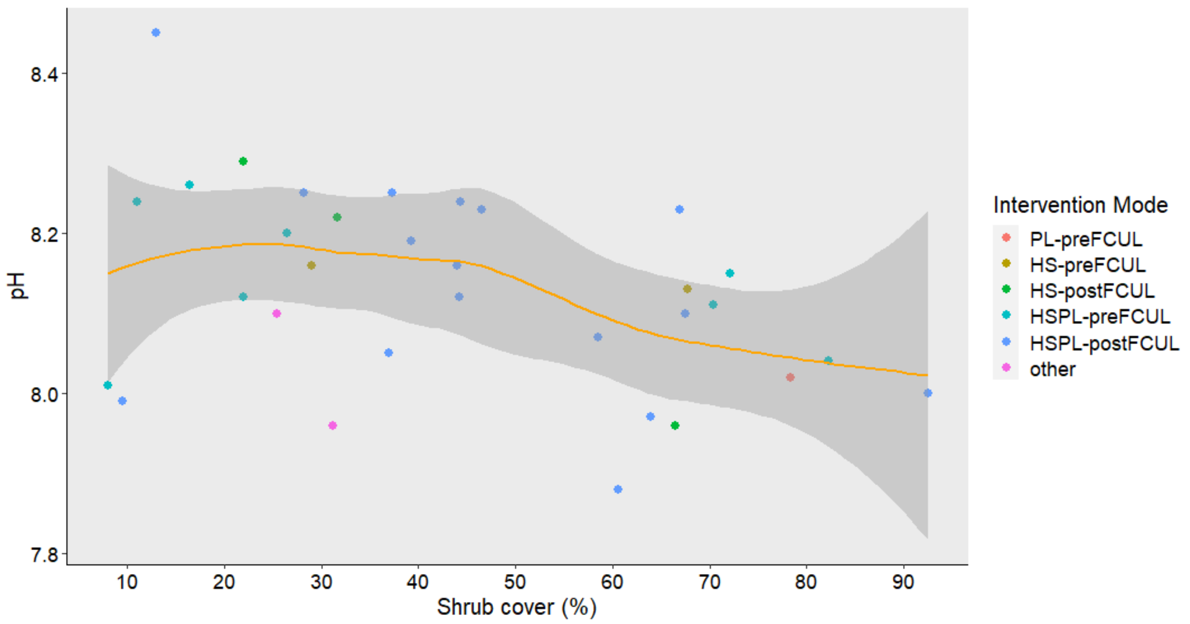

Figure S2.20. Variation of soil pH with shrub cover (%) in quarry restored plots in limestone and marl slopes (excluding limestone benches) (R^2= 0.1, p<0.05).

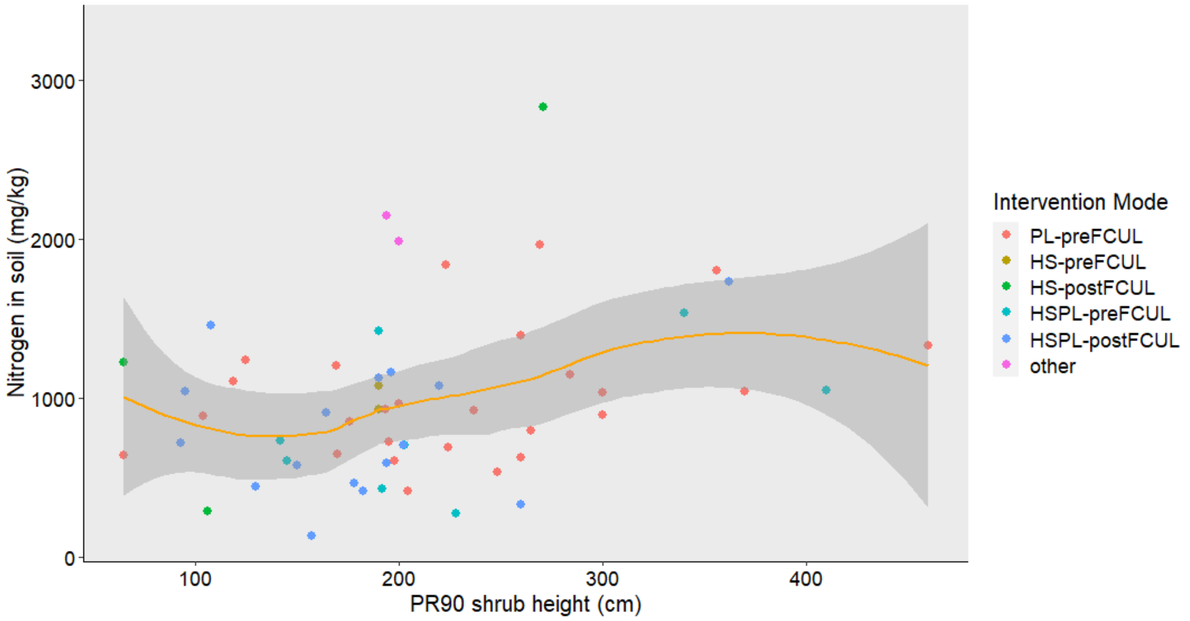

Figure S2.21. Variation of nitrogen in soil (mg/kg) with PR90 shrub height (cm) in quarry restored plots (R^2= 0.10, p<0.05).

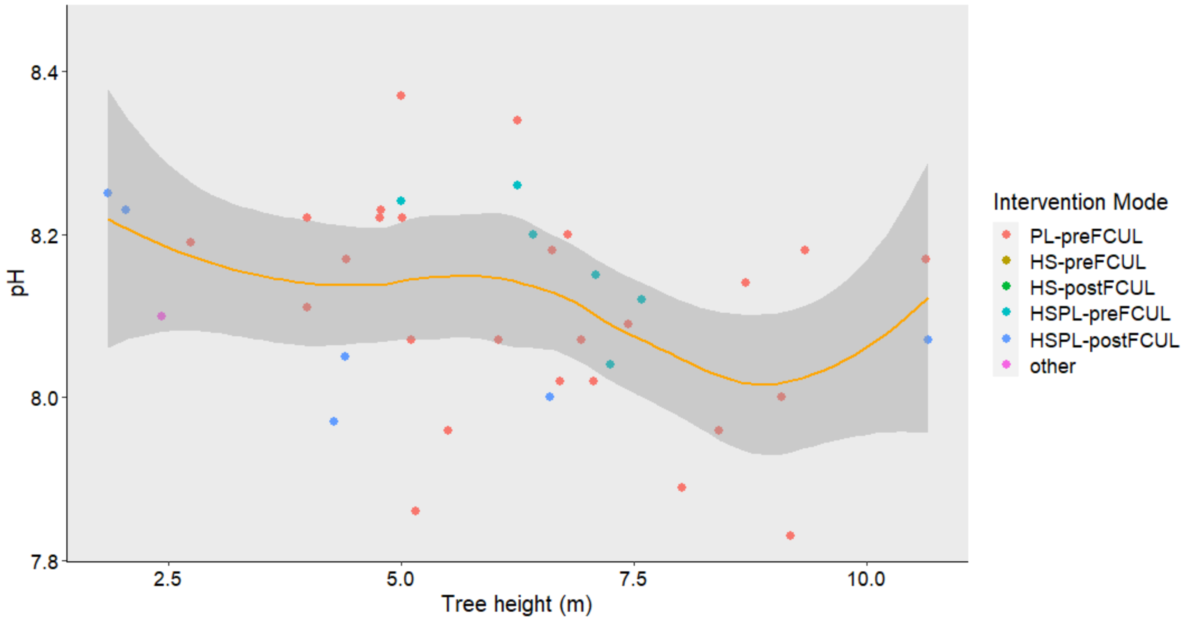

Figure S2.22. Variation of pH with tree height (m) in quarry restored plots (R^2= 0.11, p<0.05).

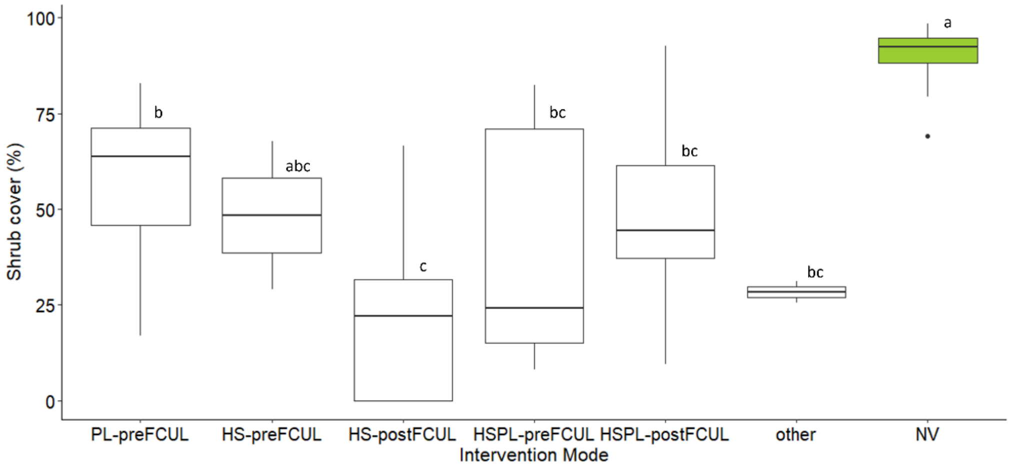

Figure S2.23. Variation of shrub cover in the different intervention modes in quarry restored plots and the NV (reference ecosystem).

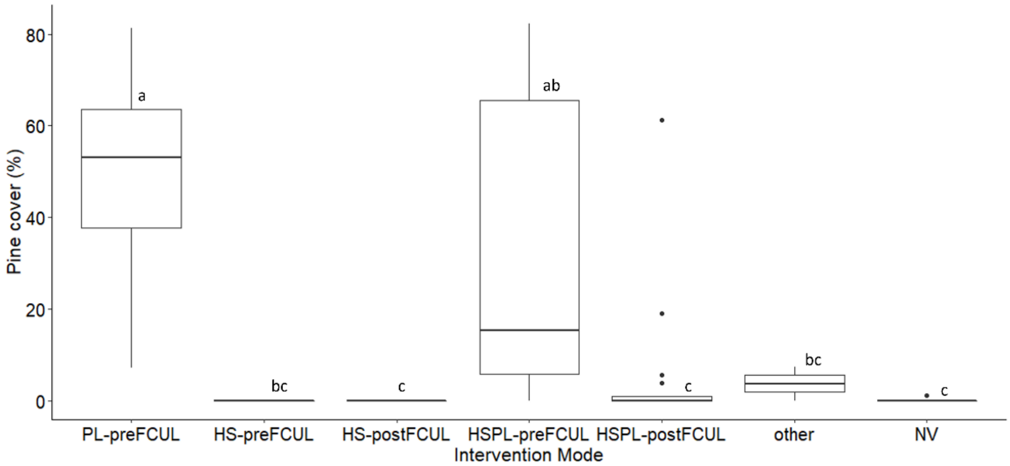

Figure S2.24. Variation of pine cover in the different intervention modes in quarry restored plots and the NV (reference ecosystem).

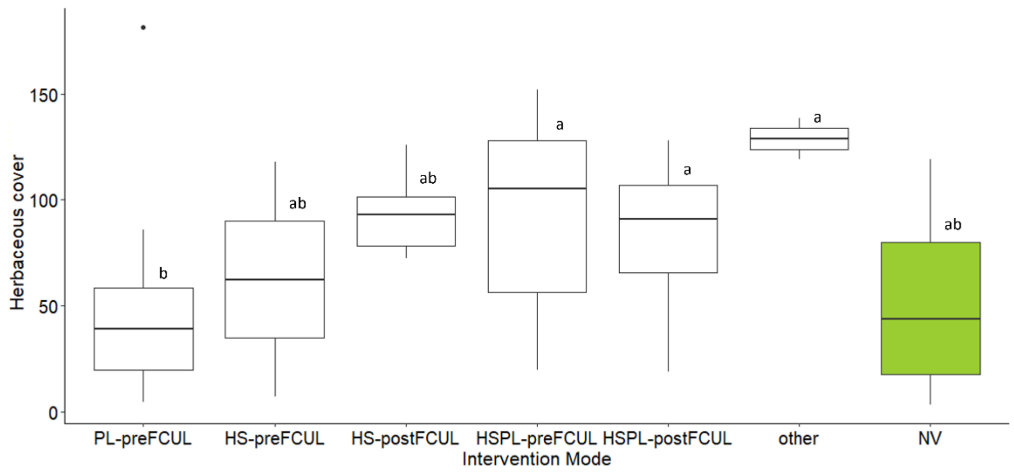

Figure S2.25. Variation of herbaceous cover in the different intervention modes in quarry restored plots and the NV (reference ecosystem).

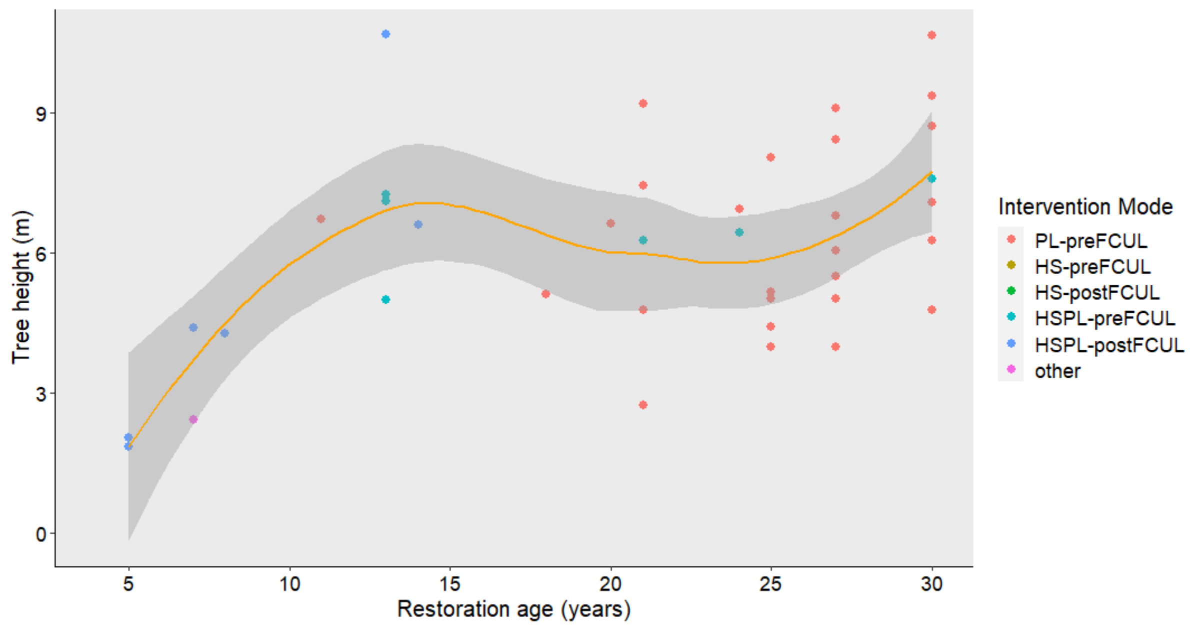

Figure S2.26. Variation of tree height with restoration age in quarry restored plots (R^2= 0.43, p<0.01).

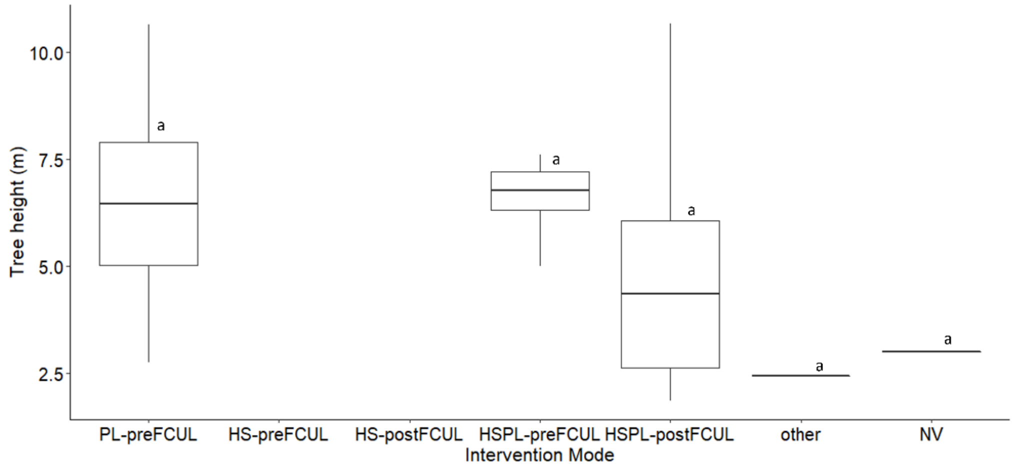

Figure S2.27. Variation of tree height in the different intervention modes in quarry restored plots and the NV (reference ecosystem).

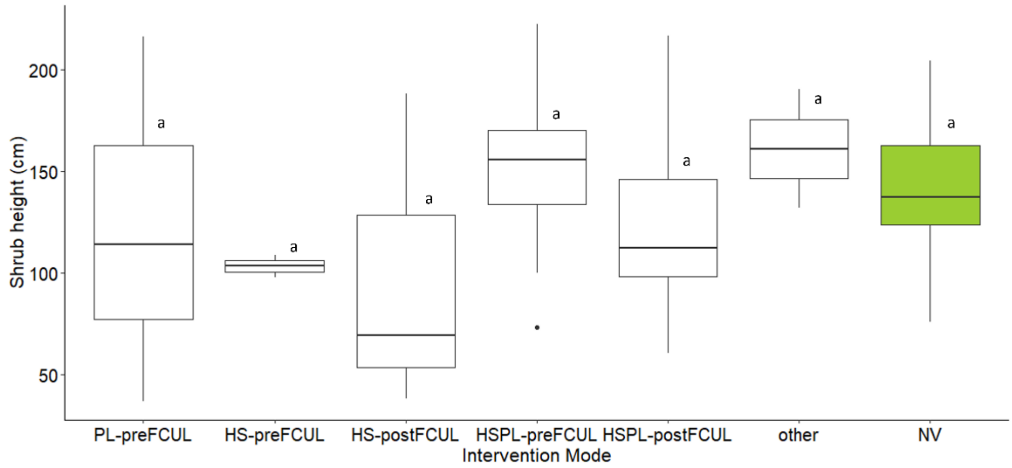

Figure S2.28. Variation of shrub height in the different intervention modes in quarry restored plots and the NV (reference ecosystem).

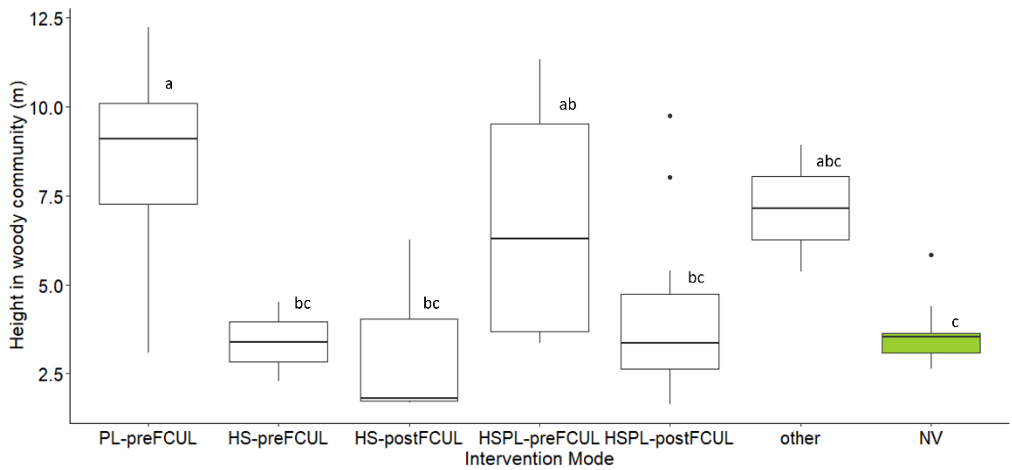

Figure S2.29. Variation of woody community height (CWM) in the different intervention modes in quarry restored plots and the NV (reference ecosystem).

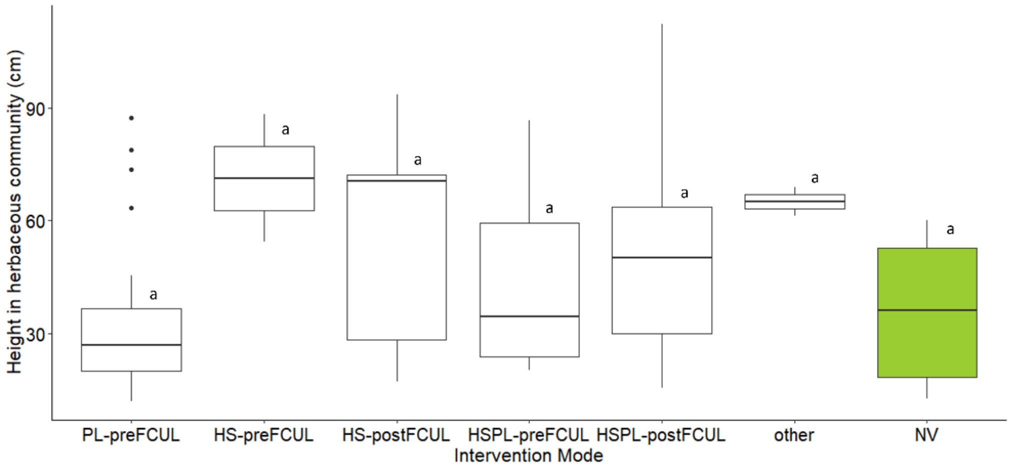

Figure S2.30. Variation of herbaceous community height (CWM) in the different intervention modes in quarry restored plots and the NV (reference ecosystem).

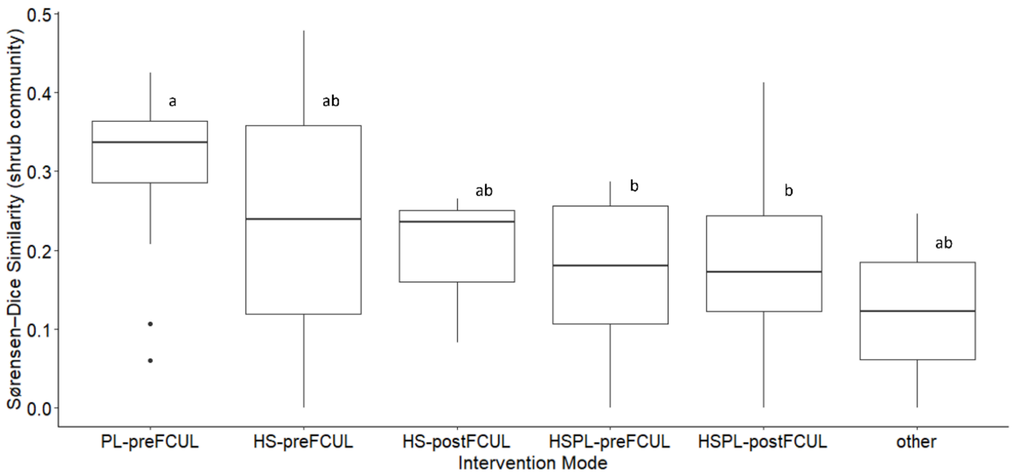

Figure S2.31. Variation of Sørensen–Dice similarity to the reference ecosystem, considering the shrub community, in the different intervention modes in quarry restored plots.

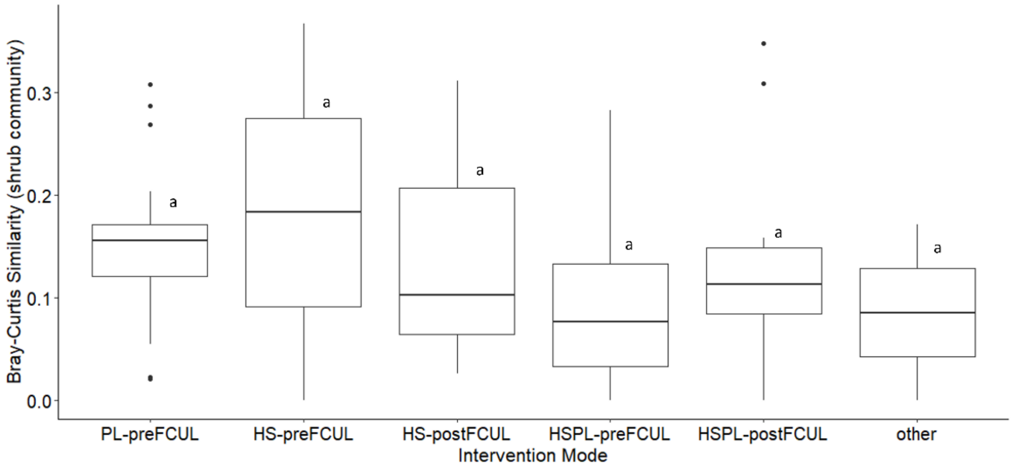

Figure S2.32. Variation of Bray-Curtis similarity to the reference ecosystem, considering the shrub community, in the different intervention modes in quarry restored plots.

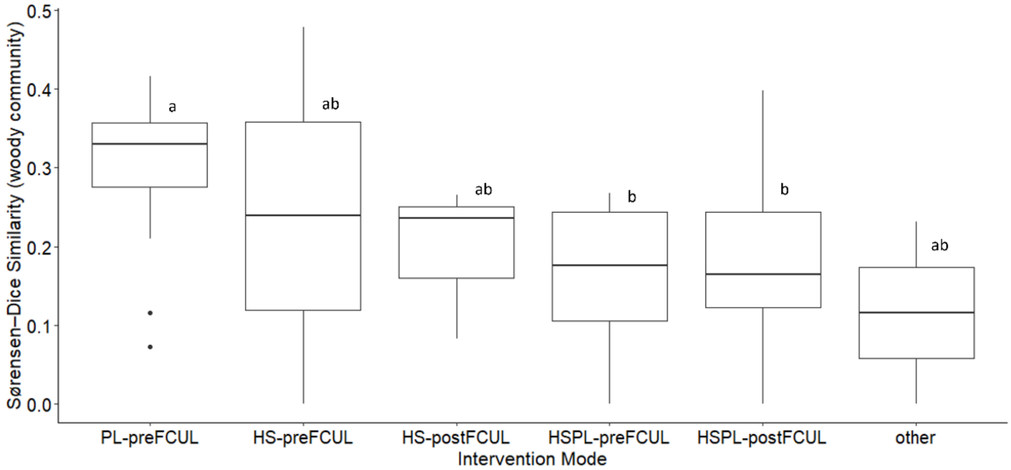

Figure S2.33. Variation of Sørensen–Dice similarity to the reference ecosystem, considering the woody community (shrubs and pine trees), in the different intervention modes in quarry restored plots.

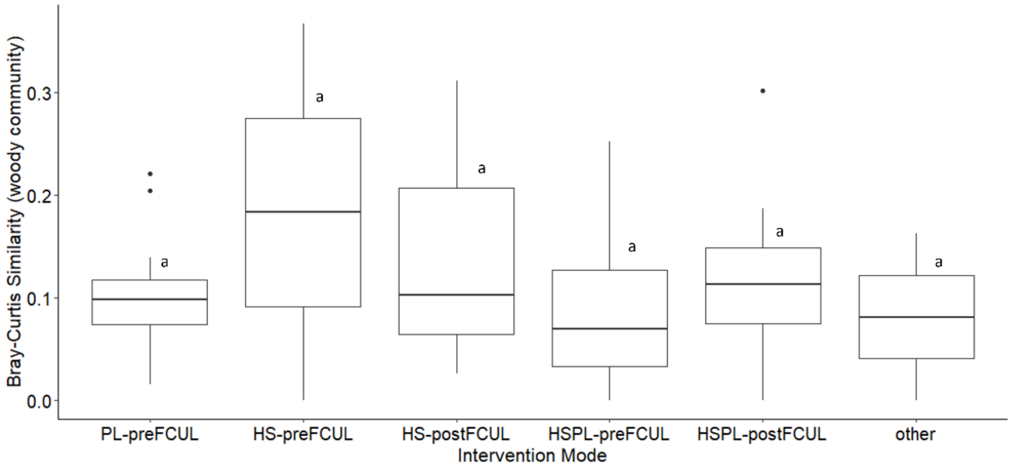

Figure S2.34. Variation of Bray-Curtis similarity to the reference ecosystem, considering the woody community (shrubs and pine trees), in the different intervention modes in quarry restored plots.

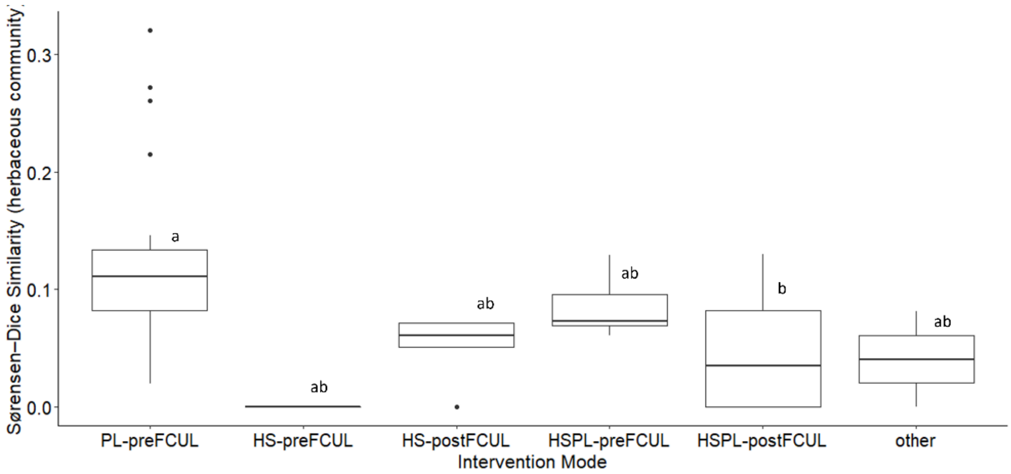

Figure S2.35. Variation of Sørensen–Dice similarity to the reference ecosystem, considering the herbaceous community, in the different intervention modes in quarry restored plots.

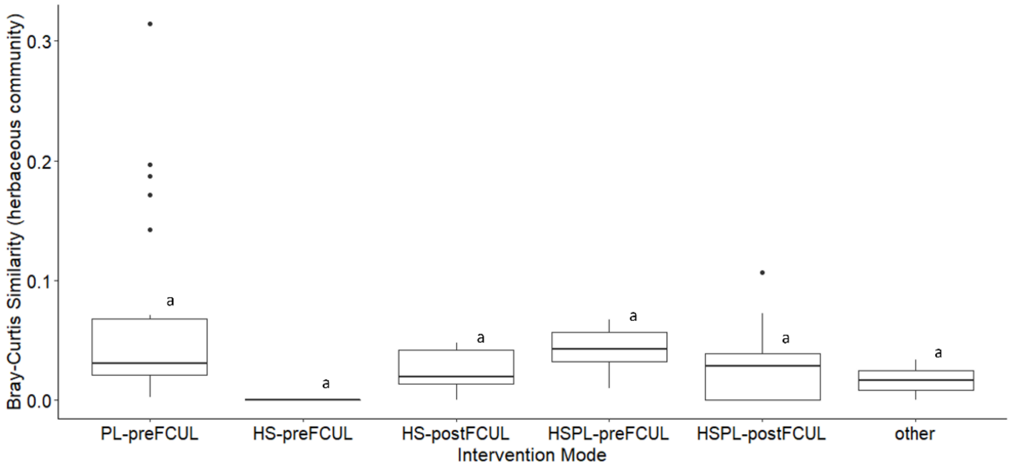

Figure S2.36. Variation of Bray-Curtis similarity to the reference ecosystem, considering the herbaceous community, in the different intervention modes in quarry restored plots.

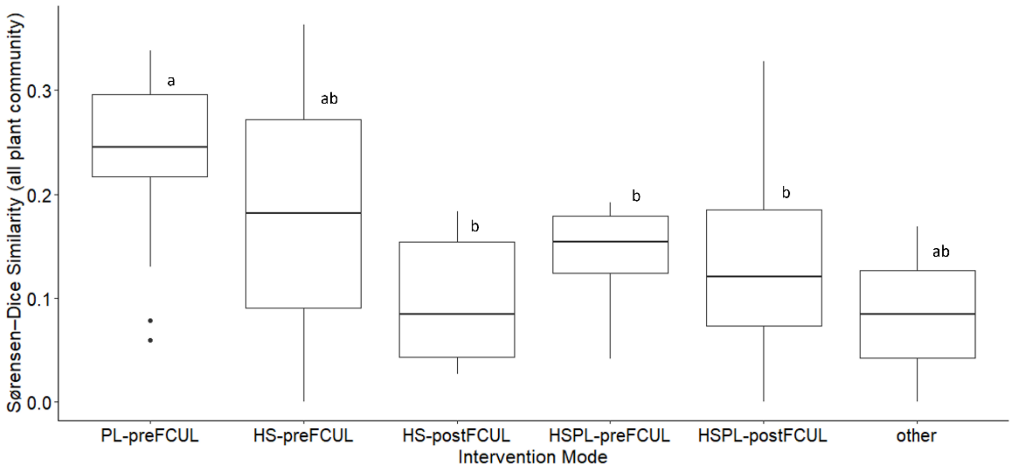

Figure S2.37. Variation of Sørensen–Dice similarity to the reference ecosystem, considering all plant community (shrubs, pine trees and herbaceous species), in the different intervention modes in quarry restored plots.

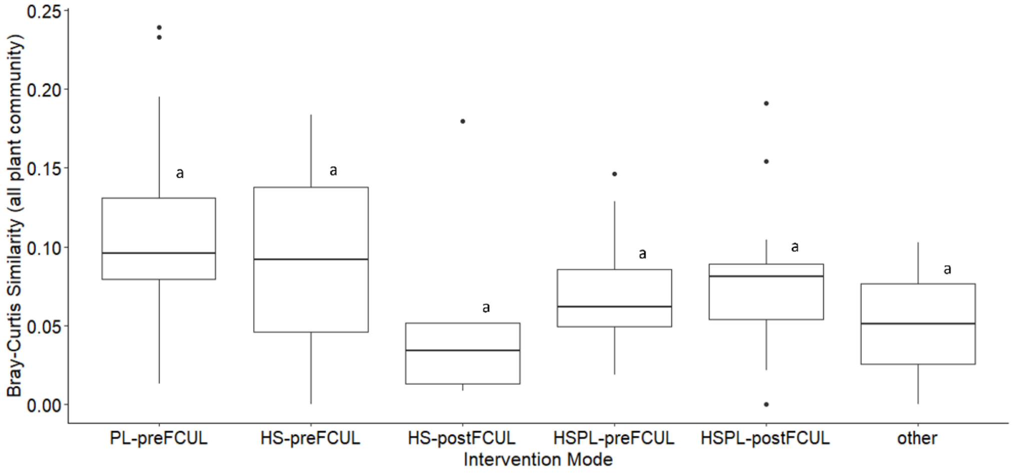

Figure S2.38. Variation of Bray-Curtis similarity to the reference ecosystem, considering all plant community (shrubs, pine trees and herbaceous species), in the different intervention modes in quarry restored plots.

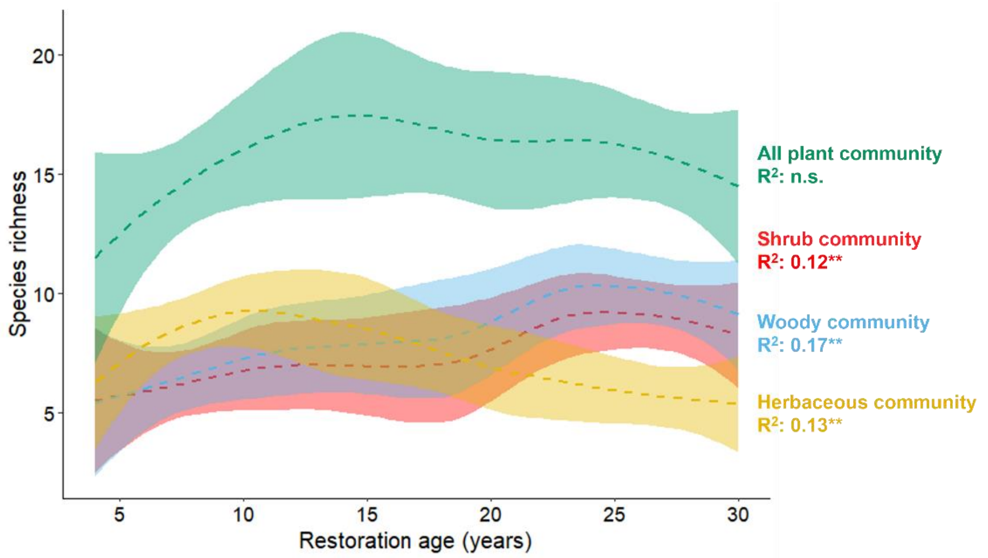

Figure S2.39. Variation of species richness with restoration age, in all plant community, shrub community, woody community and herbaceous community. Model p-value: * p<0.05, ** p<0.01, *** p<0.001.

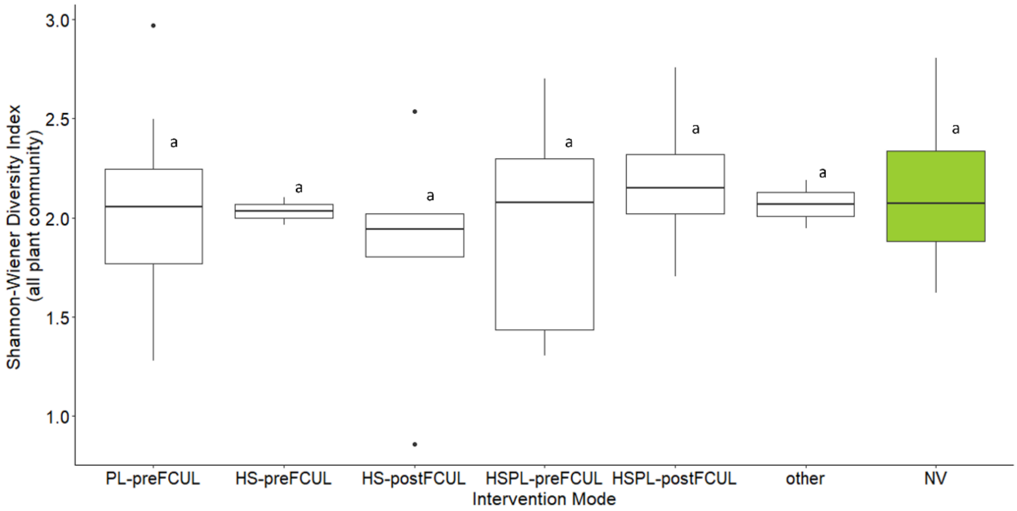

Figure S2.40. Variation of Shannon-Wiener index (taxonomic diversity) in the different intervention modes in quarry restored plots and the NV (reference ecosystem), considering all plant community (shrubs, pine trees and herbaceous species).

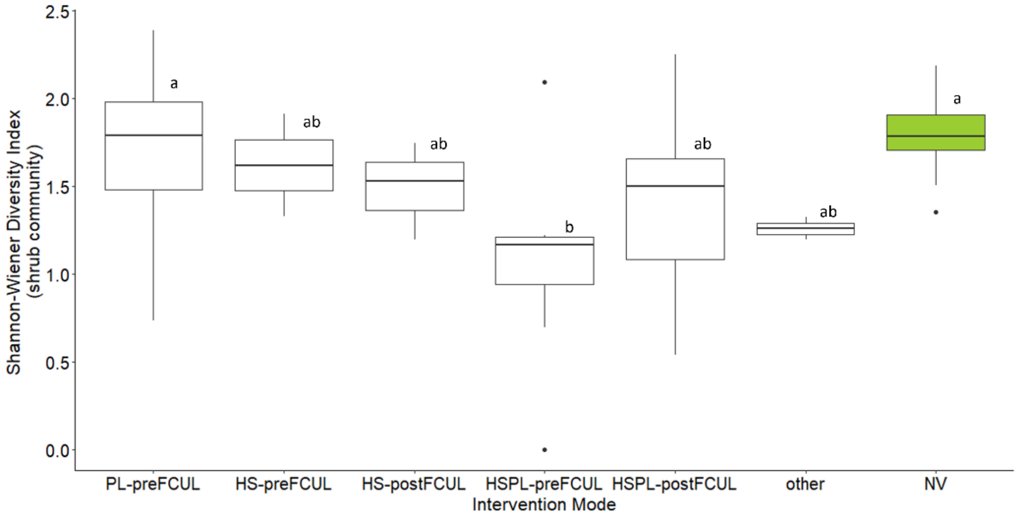

Figure S2.41. Variation of Shannon-Wiener index (taxonomic diversity) in the different intervention modes in quarry restored plots and the NV (reference ecosystem), considering the shrub community (shrubs, pine trees and herbaceous species).

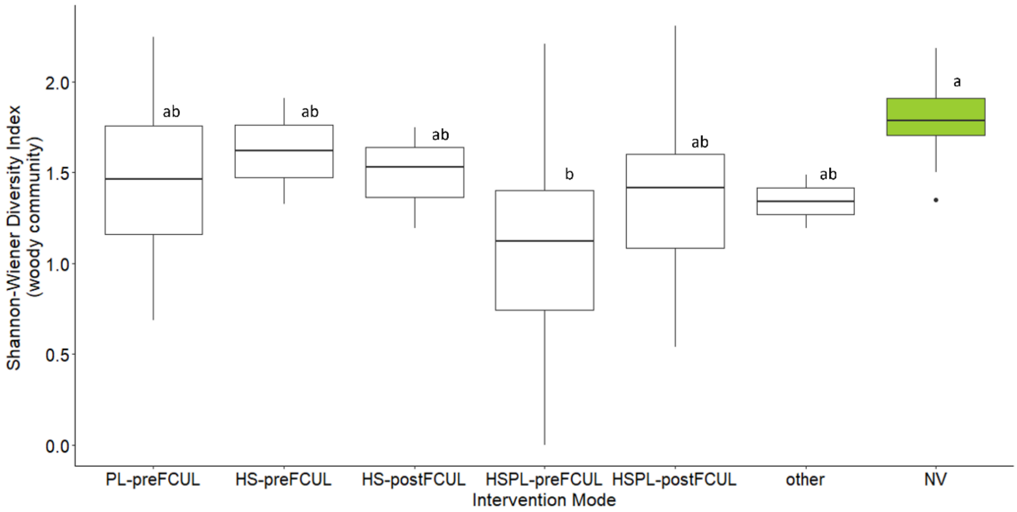

Figure S2.42. Variation of Shannon-Wiener index (taxonomic diversity) in the different intervention modes in quarry restored plots and the NV (reference ecosystem), considering the woody community (shrub and pine species).

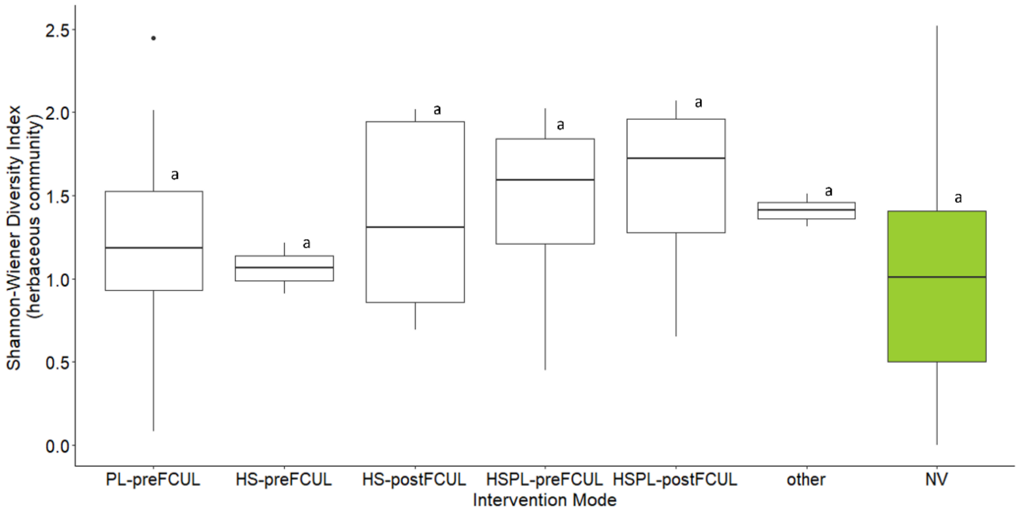

Figure S2.43. Variation of Shannon-Wiener index (taxonomic diversity) in the different intervention modes in quarry restored plots and the NV (reference ecosystem), considering the herbaceous community.

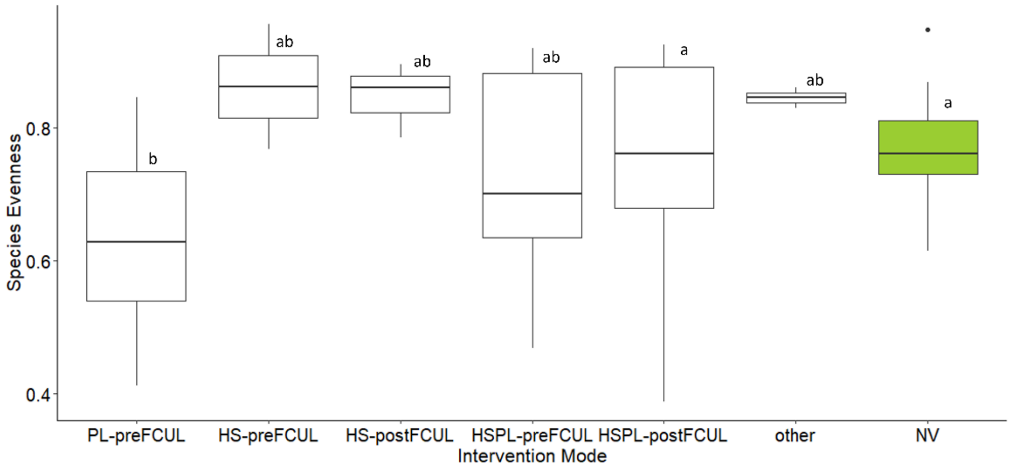

Figure S2.44. Variation of species evenness in the different intervention modes in quarry restored plots and the NV (reference ecosystem), considering the woody community.

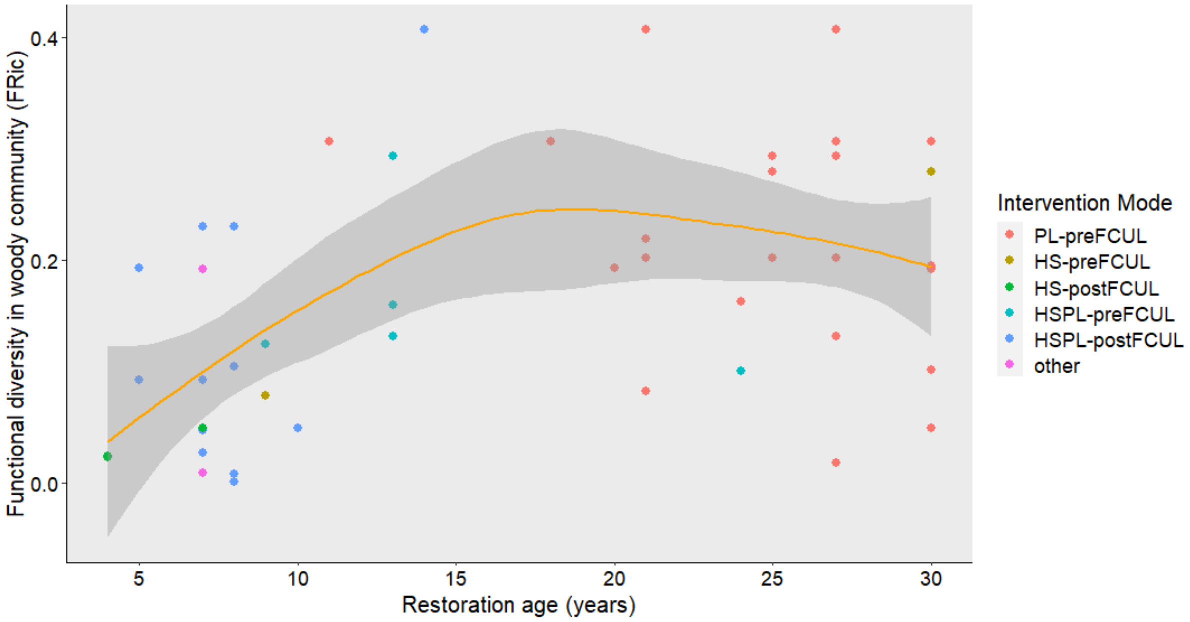

Figure S2.45. Variation of functional diversity (FRic – functional richness) with restoration age in quarry restored plots, considering the woody community (R^2= 0.30, p<0.001).

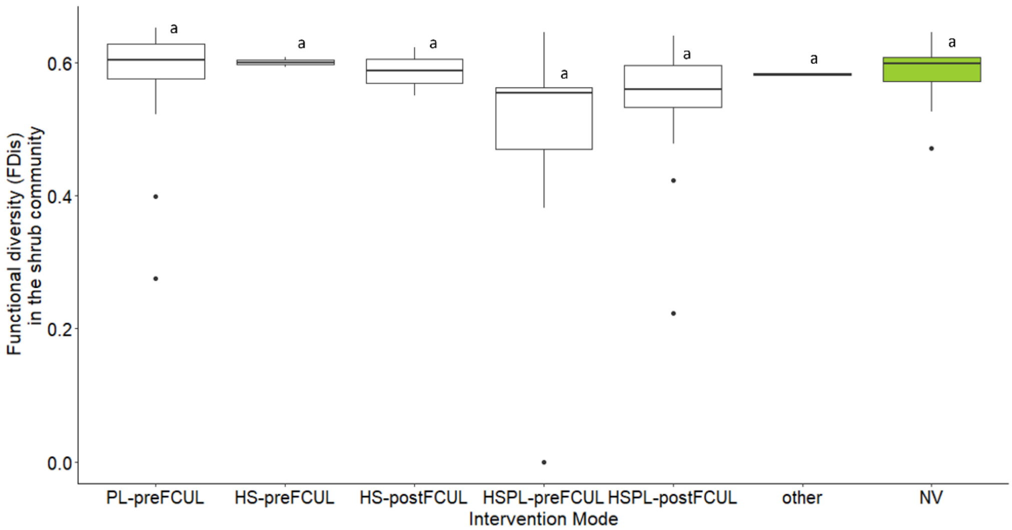

Figure S2.46. Variation of functional diversity (FDis – functional dispersion) in the different intervention modes in quarry restored plots and the NV (reference ecosystem), considering the shrub community.

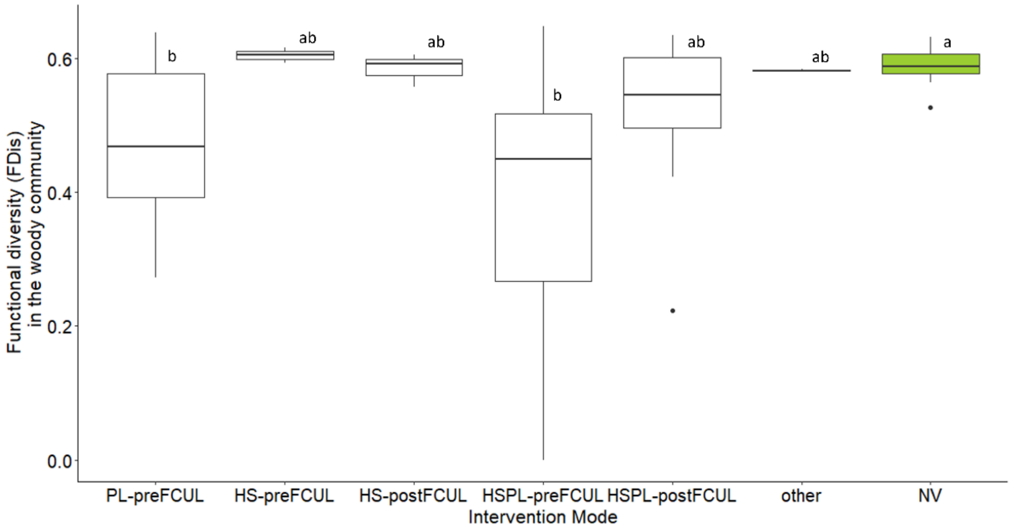

Figure S2.47. Variation of functional diversity (FDis – functional dispersion) in the different intervention modes in quarry restored plots and the NV (reference ecosystem), considering the woody community.

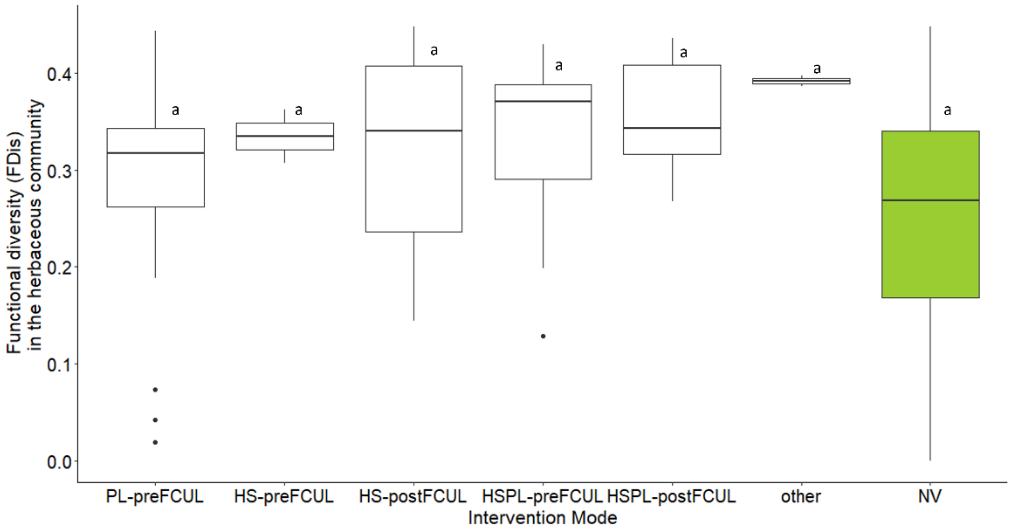

Figure S2.48. Variation of functional diversity (FDis – functional dispersion) in the different intervention modes in quarry restored plots and the NV (reference ecosystem), considering the herbaceous community.

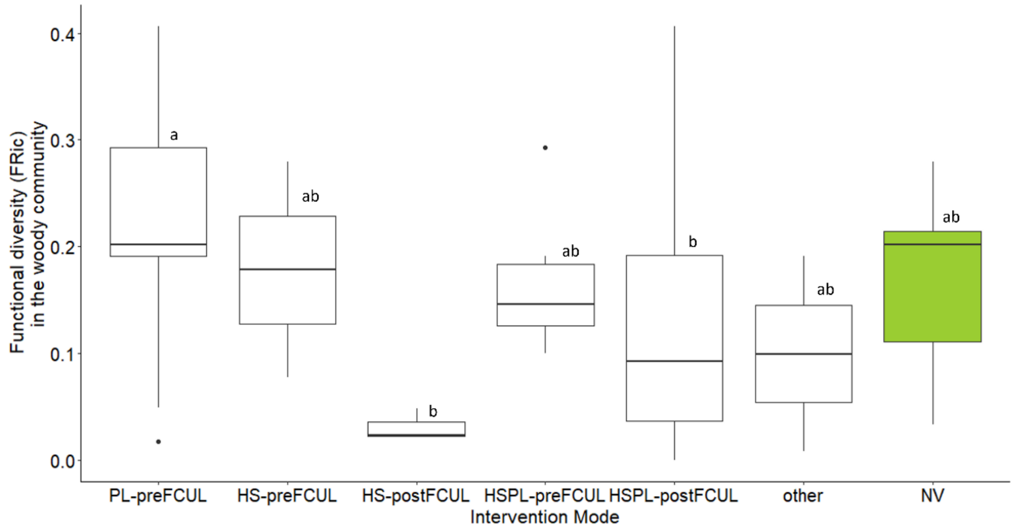

Figure S2.49. Variation of functional diversity (FRic – functional richness) in the different intervention modes in quarry restored plots and the NV (reference ecosystem), considering the woody community.

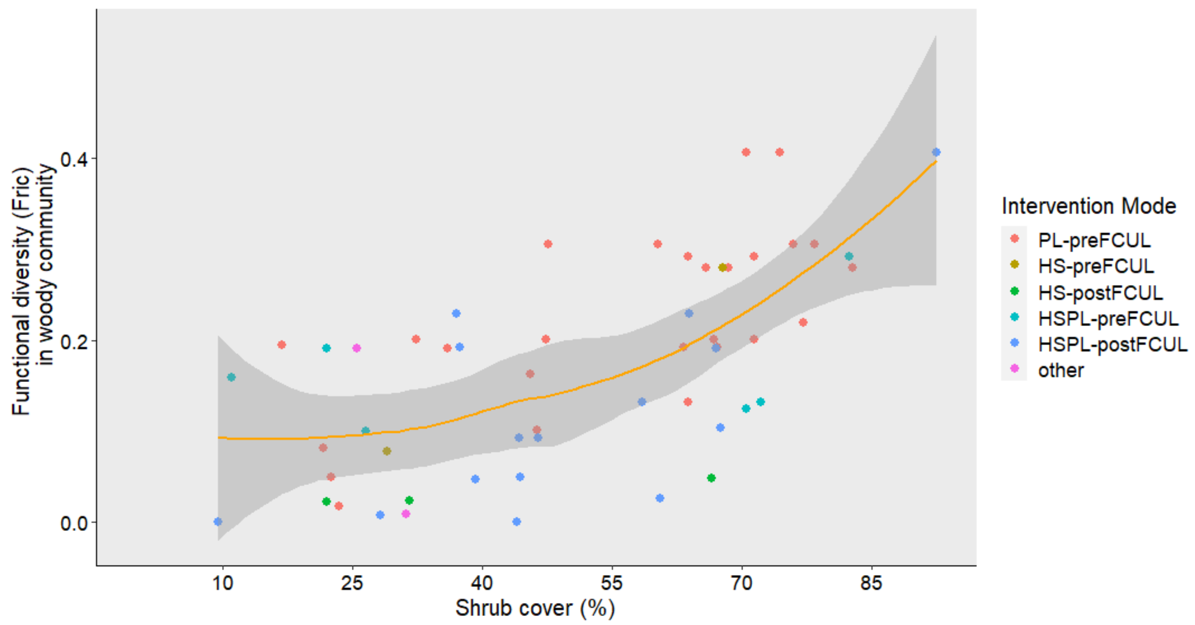

Figure S2.50. Variation of functional diversity (FRic – functional richness) with shrub cover in quarry restored plots, considering the woody community (R^2=0.38, p<0.001).

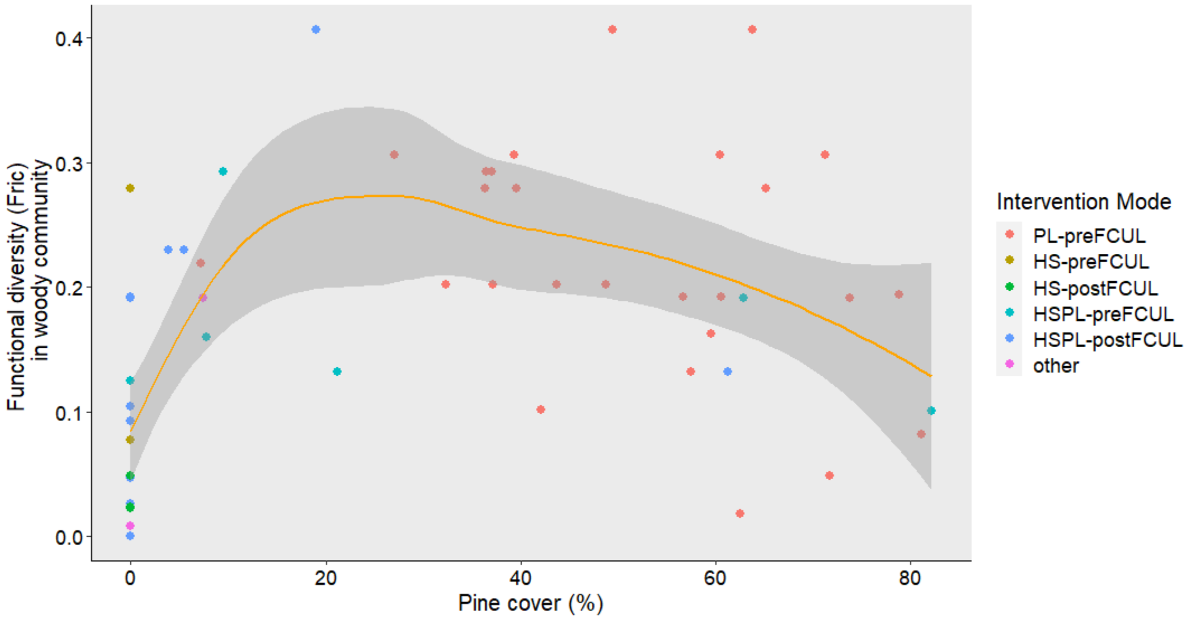

Figure S2.51. Variation of functional diversity (FRic – functional richness) with pine cover in quarry restored plots, considering the woody community (R^2=0.11, p<0.05).

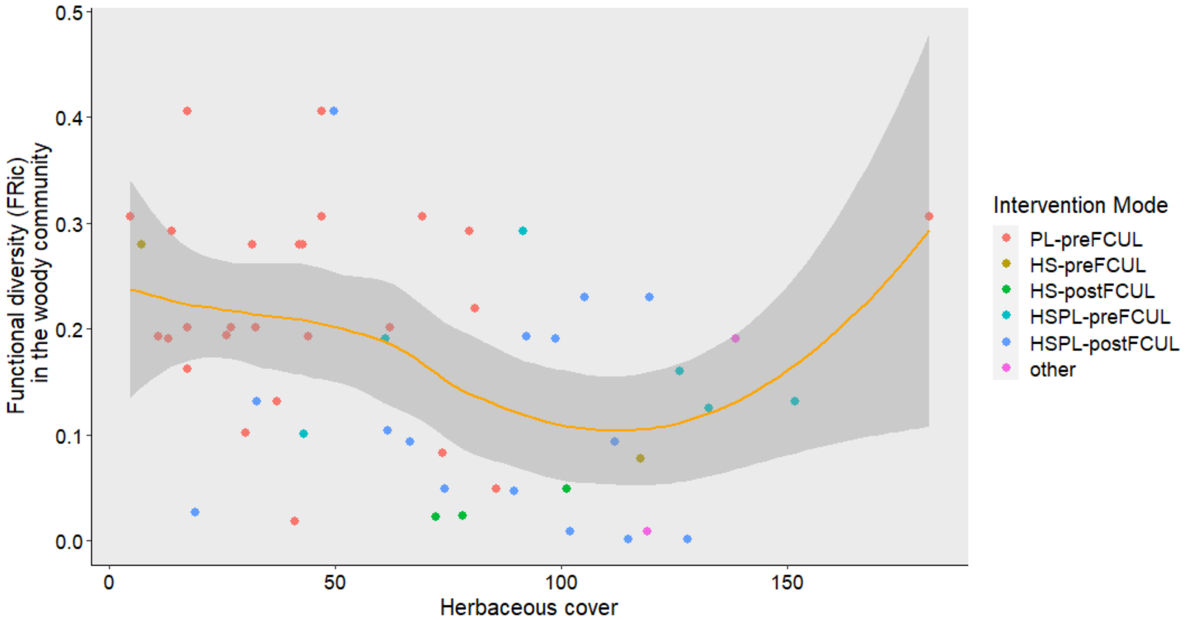

Figure S2.52. Variation of functional diversity (FRic – functional richness) with herbaceous cover in quarry restored plots, considering the woody community (R^2=0.09, p<0.05).

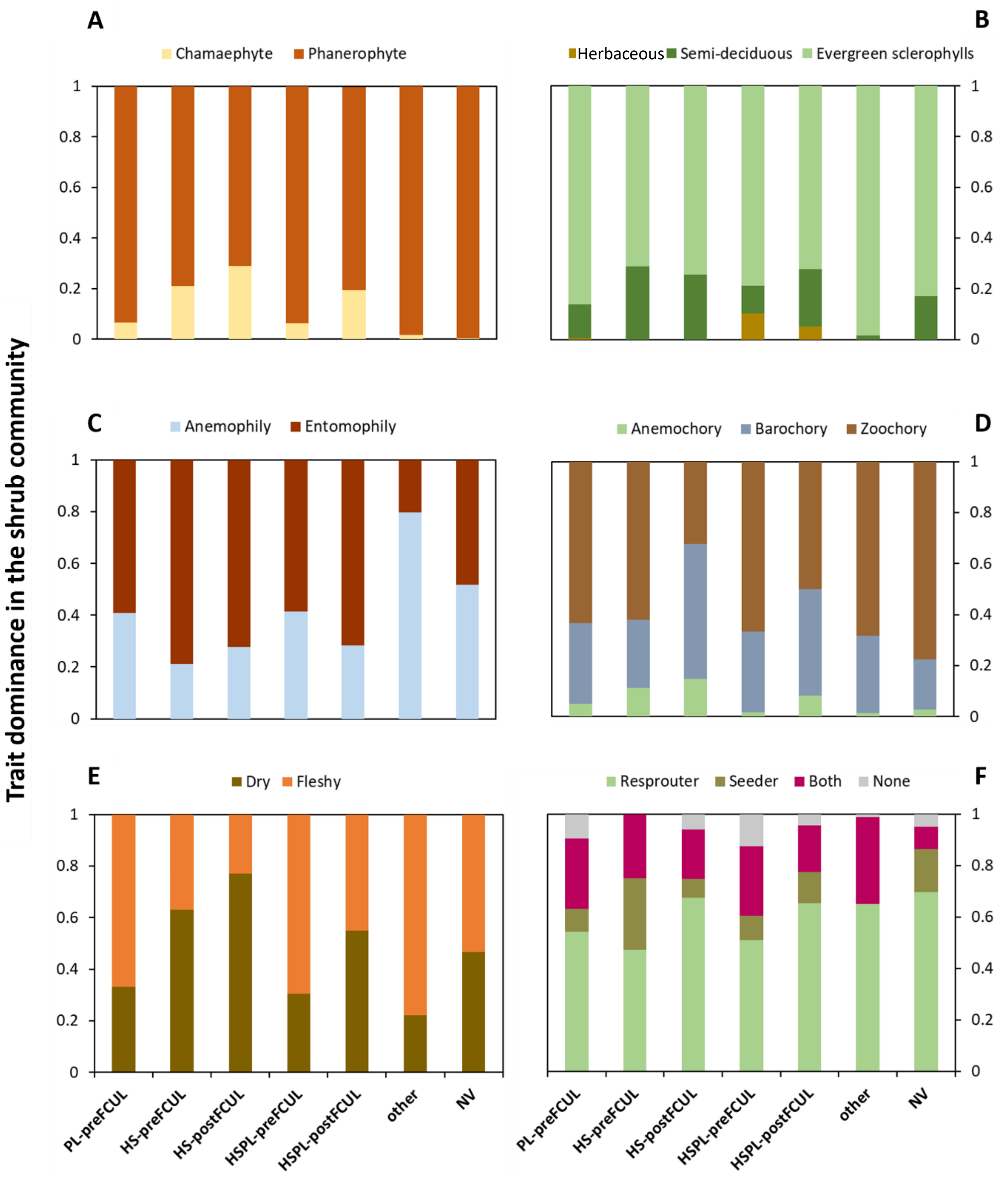

Figure S2.53. Dominance, in the shrub community, of the functional attributes (categorical traits) considered relevant for assessing the state of the revegetated sites under study, taking into account their mode of intervention in quarry restored plots and the NV (reference ecosystem). For each intervention mode, the relative average weight (CWM) of the categories considered in each attribute is presented: (A) Raunkiaer life forms, (B) leaf life cycle, (C) type of pollination, (D) type of dispersal of seeds, (E) type of fruit and (F) post-fire regeneration strategy.

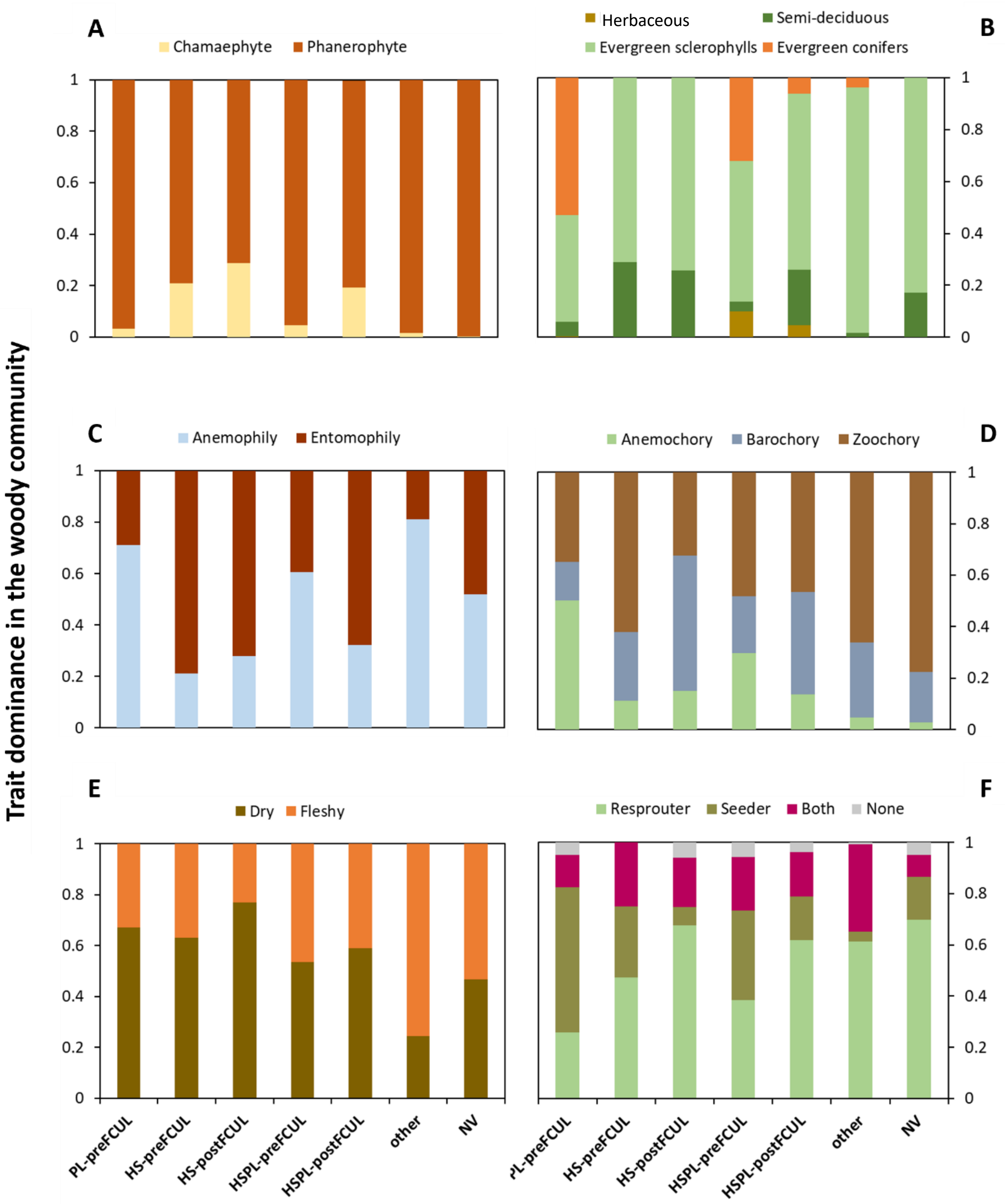

Figure S2.54. Dominance, in the woody community, of the functional attributes (categorical traits) considered relevant for assessing the state of the revegetated sites under study, taking into account their mode of intervention in quarry restored plots and the NV (reference ecosystem). For each intervention mode, the relative average weight (CWM) of the categories considered in each attribute is presented: (A) Raunkiaer life forms, (B) leaf life cycle, (C) type of pollination, (D) type of dispersal of seeds, (E) type of fruit and (F) post-fire regeneration strategy.

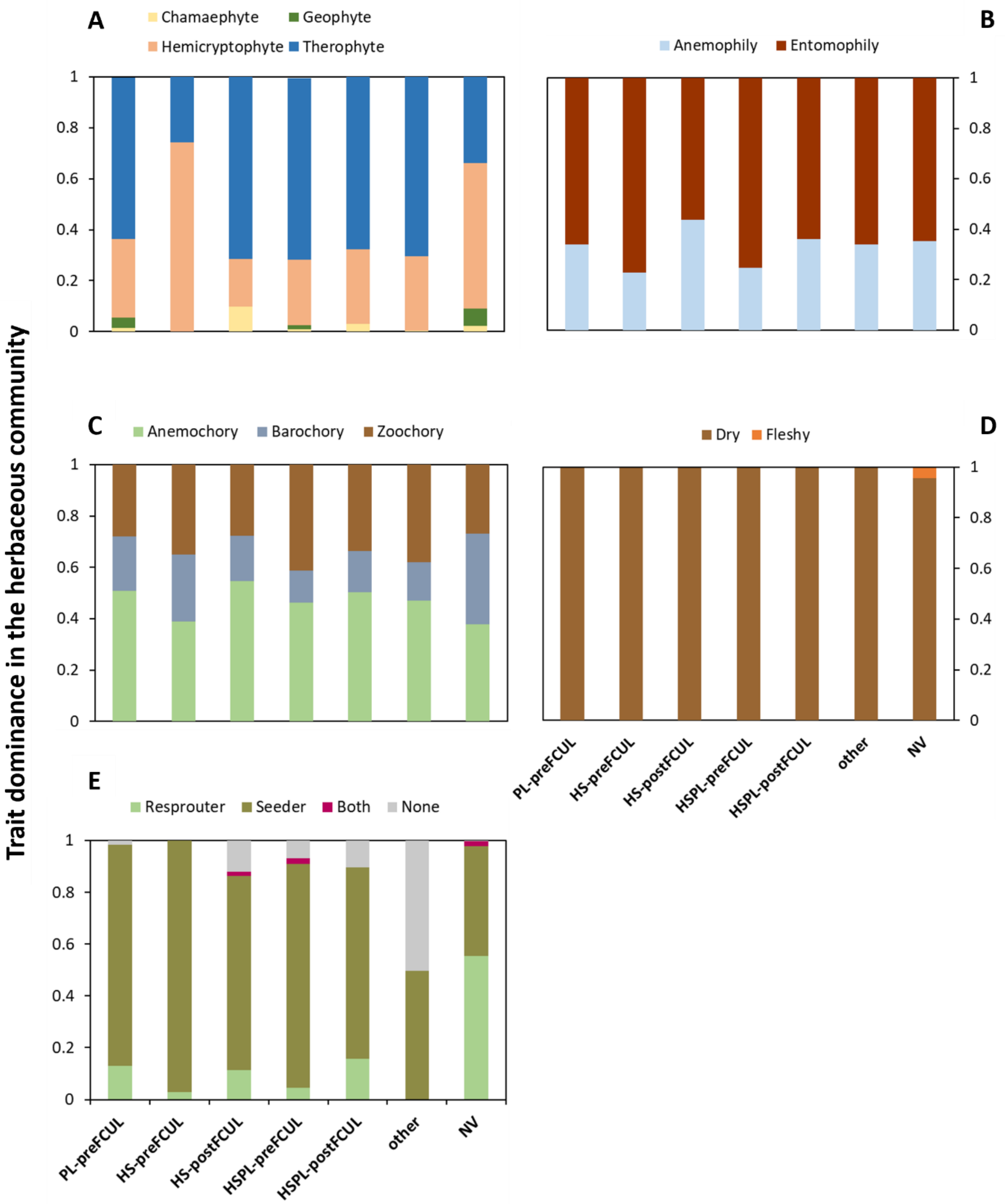

Figure S2.55. Dominance, in the herbaceous community, of the functional attributes (categorical traits) considered relevant for assessing the state of the revegetated sites under study, taking into account their mode of intervention in quarry restored plots and the NV (reference ecosystem). For each intervention mode, the relative average weight (CWM) of the categories considered in each attribute is presented: (A) Raunkiaer life forms, (B) type of pollination, (C) type of dispersal of seeds, (D) type of fruit and (E) post-fire regeneration strategy.

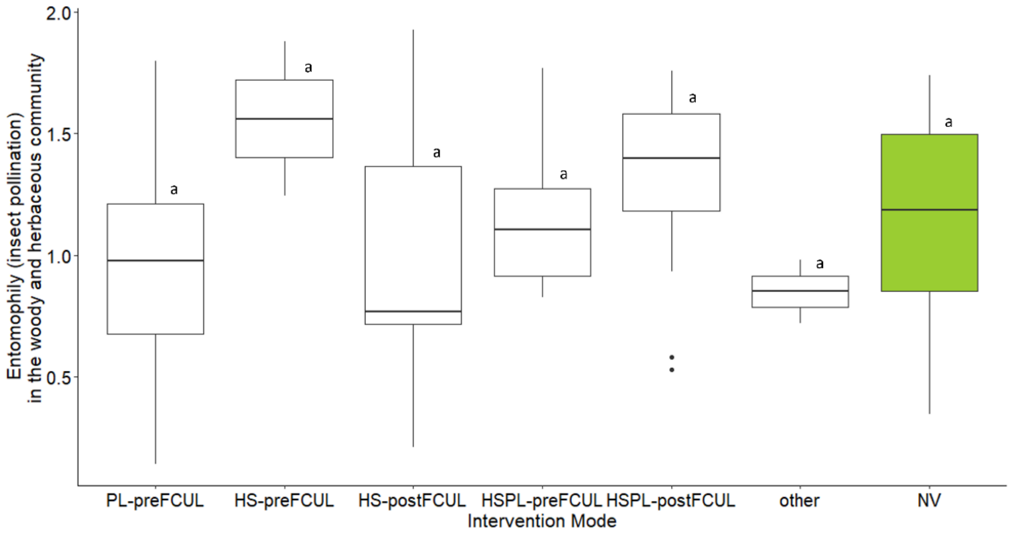

Figure S2.56. Variation of the proportion (CWM) of entomophily (insect pollination) in the different intervention modes in quarry restored plots and the NV (reference ecosystem), considering the woody and herbaceous community.

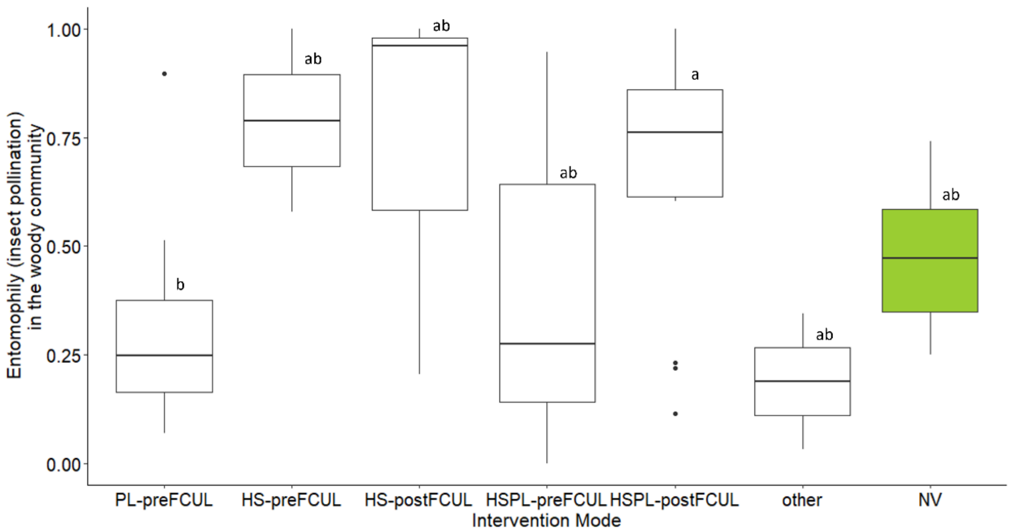

Figure S2.57. Variation of the proportion (CWM) of entomophily (insect pollination) in the different intervention modes in quarry restored plots and the NV (reference ecosystem), considering the woody community.

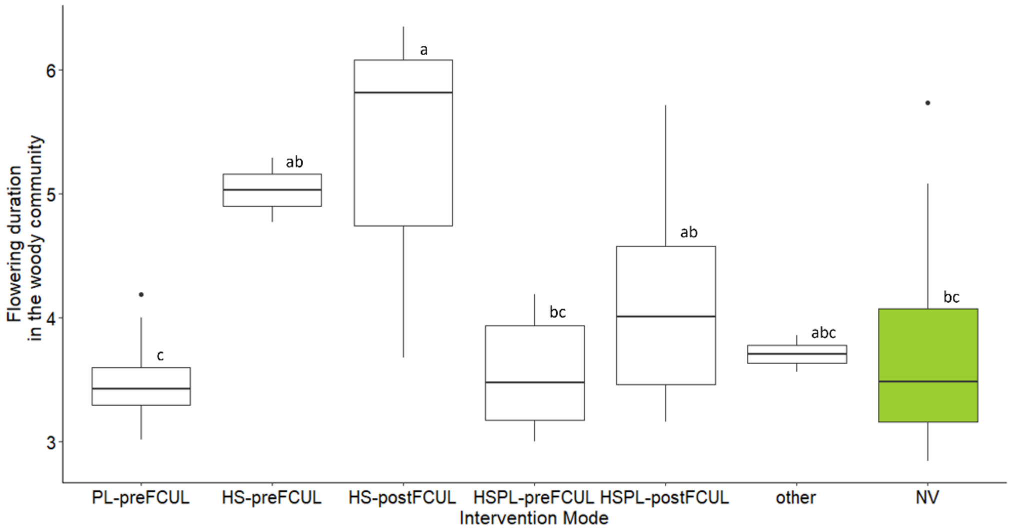

Figure S2.58. Variation of the flowering duration (months) in the different intervention modes in quarry restored plots and the NV (reference ecosystem), considering the woody community.

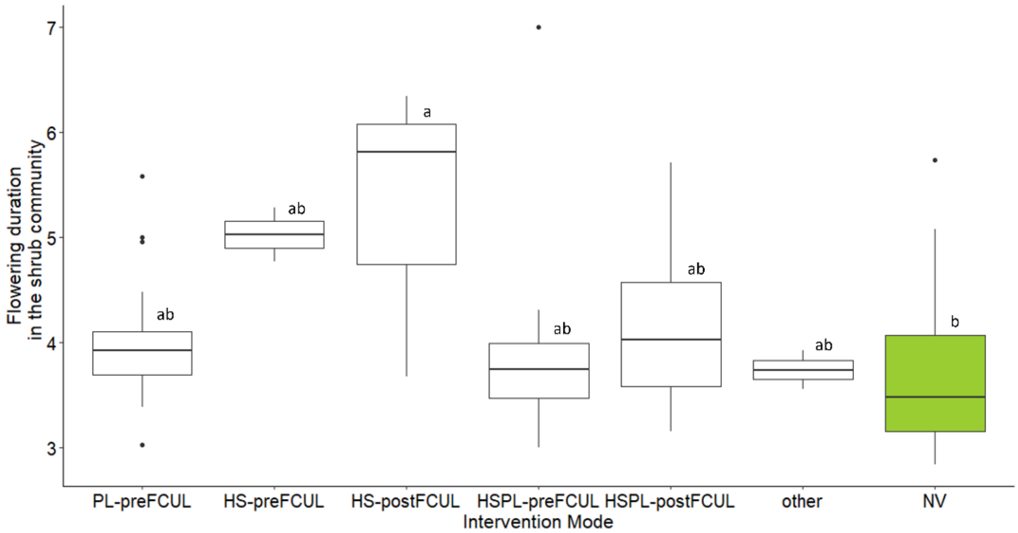

Figure S2.59. Variation of the flowering duration (months) in the different intervention modes in quarry restored plots and the NV (reference ecosystem), considering the shrub community.

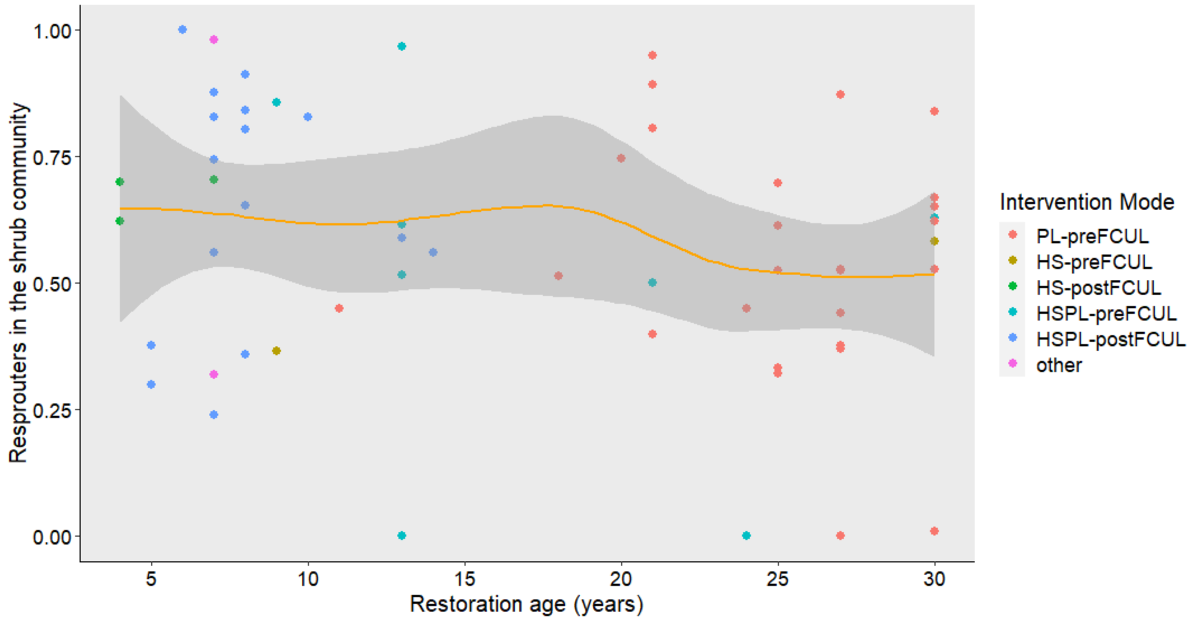

Figure S2.60. Variation of the proportion of resprouters in the shrub community with restoration age in quarry restored plots (n.s.).

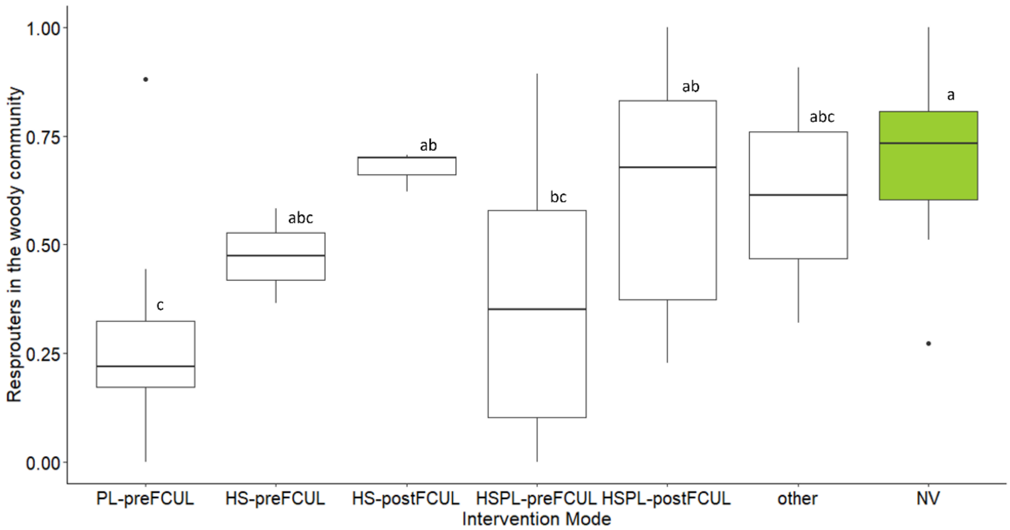

Figure S2.61. Variation of the proportion of resprouters, considering the woody community, in the different intervention modes in quarry restored plots and the NV (reference ecosystem).

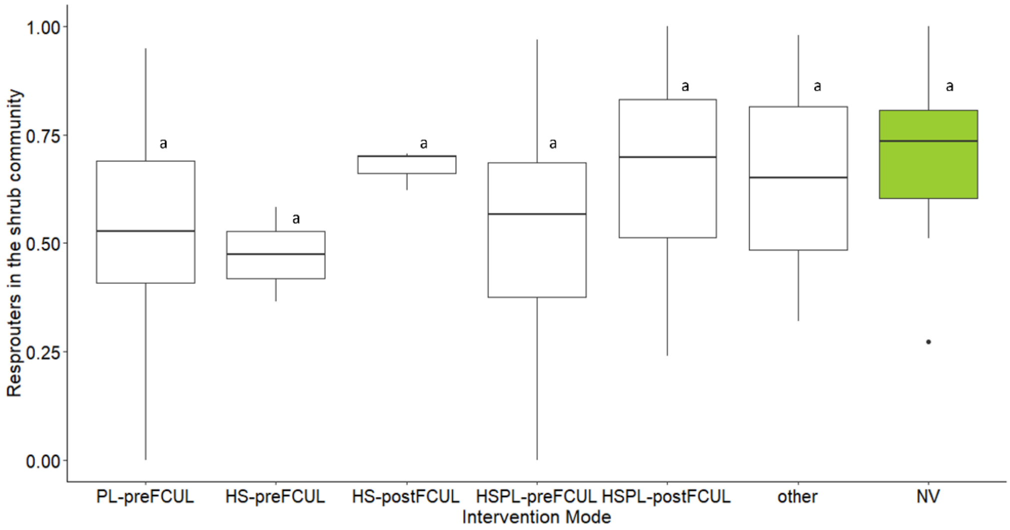

Figure S2.62. Variation of the proportion of resprouters, considering the shrub community, in the different intervention modes in quarry restored plots and the NV (reference ecosystem).

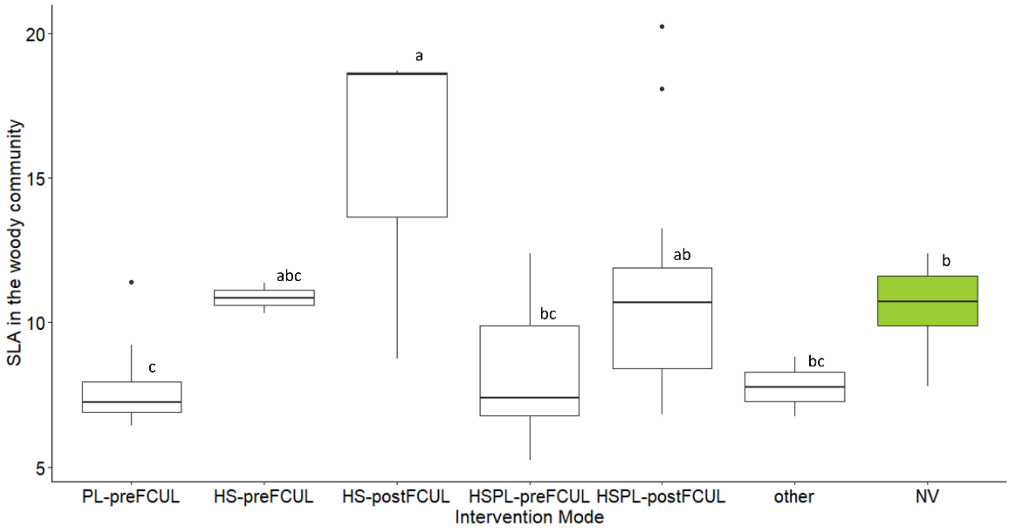

Figure S2.63. Variation of SLA (mm^2/mg), considering the woody community, in the different intervention modes in quarry restored plots and the NV (reference ecosystem).

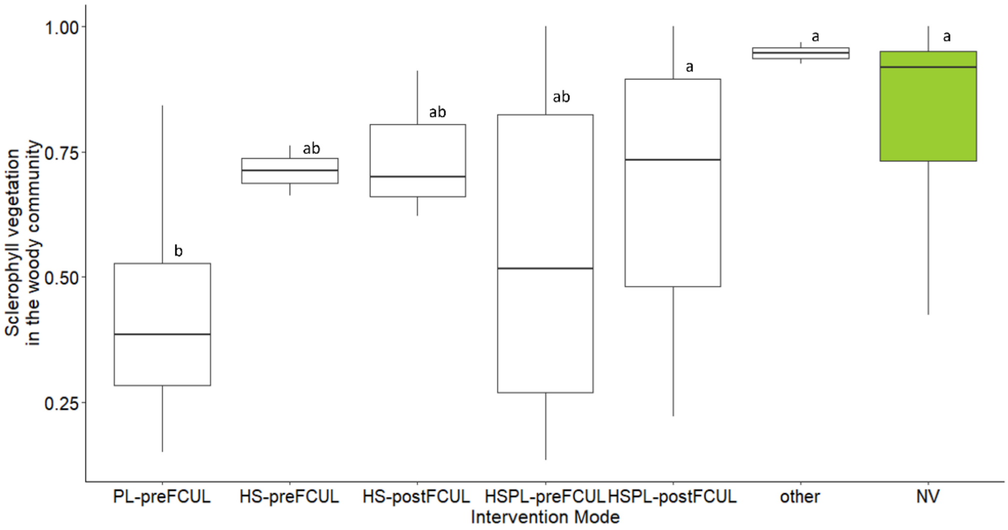

Figure S2.64. Variation of the proportion of sclerophyll vegetation, considering the woody community, in the different intervention modes in quarry restored plots and the NV (reference ecosystem).

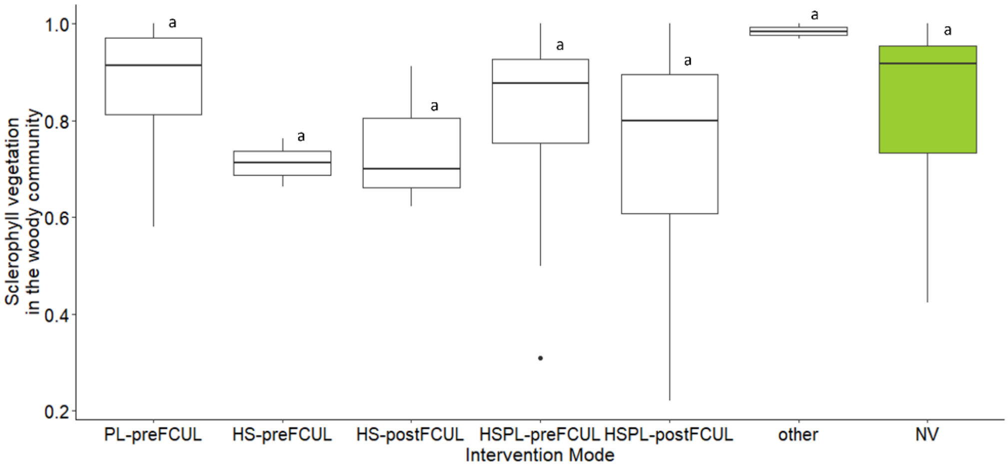

Figure S2.65. Variation of the proportion of sclerophyll vegetation, considering the shrub community, in the different intervention modes in quarry restored plots and the NV (reference ecosystem).

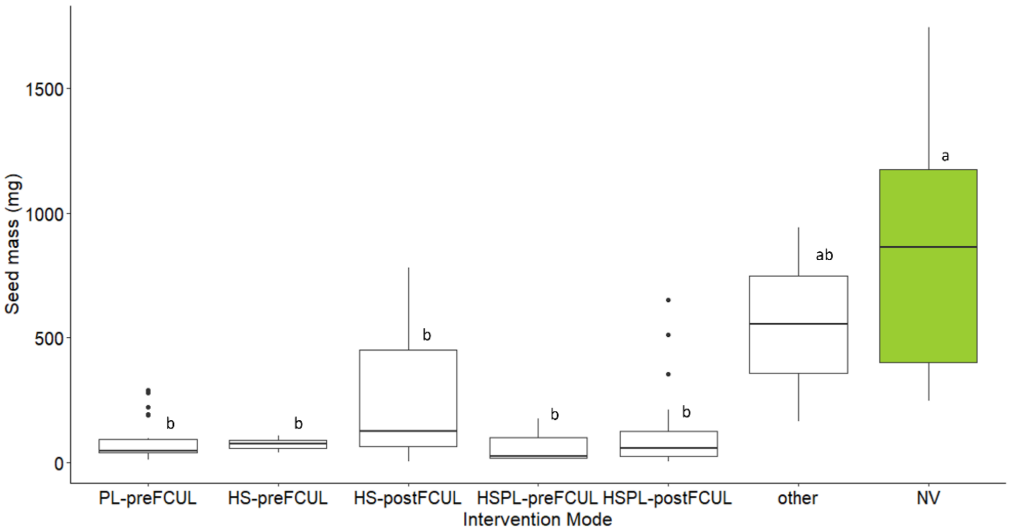

Figure S2.66. Variation of the seed mass (mg), considering the woody community, in the different intervention modes in quarry restored plots and the NV (reference ecosystem).

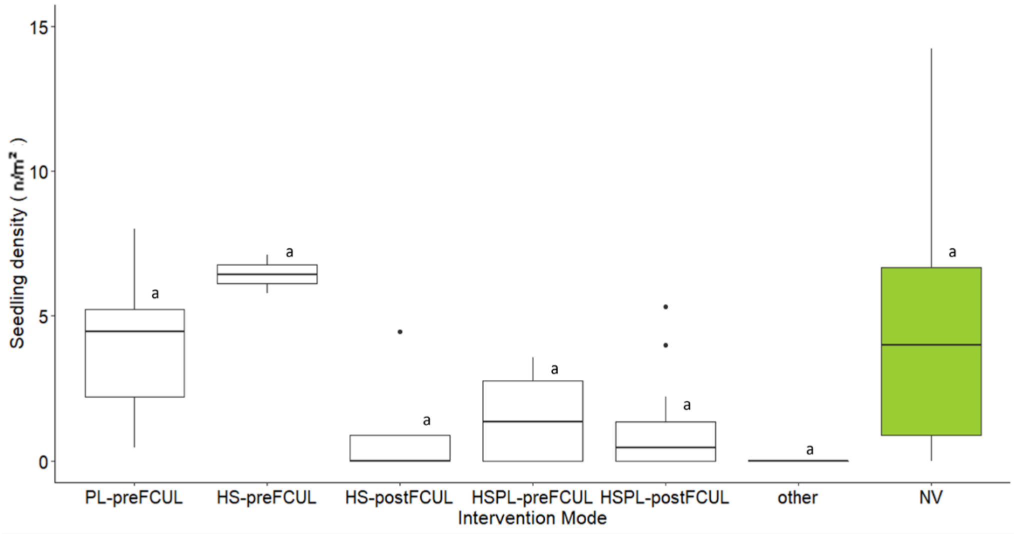

Figure S2.67. Variation of seedling density (n/m^2) in the different intervention modes in quarry restored plots and the NV (reference ecosystem). One outlier of NV (seedling density of 54.6 n/m^2) was omitted to enable the visualisation and comparison of the values between the intervention modes.