Fois, M., Murgia, L., & Bacchetta, G. (2023). Plant diversity and species composition of the abandoned mines of the Iglesiente mining district (Sardinia, Italy): A restoration perspective. *Ecological Engineering*, *188*, 106879.

The publisher's version is available at:
https://doi.org/10.1016/j.ecoleng.2022.106879

Plant diversity and species composition of the abandoned mines of the Iglesiente mining district (Sardinia, Italy): A restoration perspective

Mauro Fois [*], Luca Murgia, Gianluigi Bacchetta

Centre for Conservation of Biodiversity (CCB), Department of Life and Environmental Sciences, University of Cagliari, Viale S.Ignazio da Laconi, 11-13, Cagliari 09123, Italy

ARTICLE INFO

Keywords:
Endemic vascular plants
Geoparks
Herbivory
Natural vegetation succession
Phytoremediation
Post-mining vegetation

ABSTRACT

The 'metalliferous ring' of Iglesiente was for centuries the most important mining district of Italy. After a marked decline in mining activities, alternative management strategies were proposed to face issues mainly related to unemployment and environmental pollution. We focused on the coastal section of the mining district, in order to evaluate the development of vegetation after disturbance and assess the conservation value of the study area. The floristic composition in terms of species richness, number of endemics, evenness, life form diversity, vegetation and cryptogamic covers were analysed for 96 vegetation samples. Three main plant assemblages with 227 vascular plant species and 18 endemics were described. The most influential variable in determining species composition was the herbivory, which negatively affects species richness, endemic species richness and life form diversity and, positively, the evenness. Other influential variables were elevation, positively associated with species and endemic species richness, and the degree of debris/rock flow, which negatively influences species richness and life form diversity. Soil grain size positively influences evenness and negatively life form diversity and vegetation cover. The number of years of mining inactivity was only positively influencing life form diversity. Each identified community, especially those characterised by perennial herbs and small shrubs, can provide surrogate habitats for many endemic plants, which can be in turn used for phytostabilization and remediation. Further indications for strategic management of these mining environments, where policy and long-term monitoring are necessary interrelated strategies, were provided.

1. Introduction

Mining activities represent an important source of environmental disturbance and contamination which often leaves a landscape with extreme conditions for the vegetation development with, among others, poor soil and high heavy metals concentration (Alday et al., 2012). In such damaged systems, investigating the establishment and development of natural vegetation development is useful for guiding ecological restoration to healthy, long-term and self-sustaining ecosystems (Alday et al., 2011). A number of studies have identified different factors, mainly related to soil-forming processes, which control vegetation dynamics in mining areas (Bacchetta et al., 2007a; Alday et al., 2011; Pallavicini et al., 2015). However, deeper investigations on environmental factors at different scales (landscape or local site factors) that control vegetation dynamics are needed (Rehounková and Prach, 2006; Alday et al., 2011; Pallavicini et al., 2015). Several parameters can be measured to evaluate the natural vegetation succession after any disturbance and, in particular, after mining activities. Among them, species composition and richness have been the most investigated (Martínez-Ruiz and Fernández-Santos, 2005; Pallavicini et al., 2015; Řehounková et al., 2020) but other aspects, such as the number of endemic and alien species or the functional trait diversity, were considered informative for plant and ecosystem conservation and recovery (Jost, 2010; Fois et al., 2020; Bonari et al., 2021). Various factors influence the recovery of biodiversity in abandoned mines and in disturbed environments in general; these can be summarised as natural and anthropogenic. For instance, high slopes and low soil nutrient content and moisture generally slow the vegetation establishment and successional process (Du et al., 2021). Among human-induced factors, herbivory has been found to negatively affect endemic species (eg. Lozano et al., 2021). However, such effects do not generally compromise the recovery of the entire site, but promote micro-scale heterogeneity generating diverse ecosystems for plants and animal as well (Farris et al., 2010; Alday et al., 2012; Pallavicini et al., 2015).

---

* Corresponding author.
E-mail address: mfois@unica.it (M. Fois).

Sardinia is one of the few regions in Italy with historical mining activity extending over four thousand years, from the first Nuragic excavations (Bronze and Early Iron Ages), to the exploitation in Punic and Roman times – when first wells and galleries were made – and from Middle Ages, under the Aragoneses and Pisans, up to recent days. A large variety of metalliferous and industrial minerals were mined, but the most important activity was devoted to the exploitation of Pb-Ag-Zn (Ba-F) ores in the Iglesiente region (SW Sardinia), where mining activities represent the main anthropogenic landscape feature (Ardau and Runddeddu, 2001). Over the last four decades, there was a marked decline of Sardinian mining activities, especially in the metalliferous sector, largely because of the competition of mines in other countries. In the Iglesiente, hundreds of abandoned mines exist, from which a variety of heavy metals are dispersed into the environment, affecting the terrestrial and aquatic ecosystems in surrounding areas (Cidu et al., 2009; Bacchetta et al., 2012). Due to the extension of the area, the use of native plants for remediation was identified as the most suitable solution (Jim´enez et al., 2011, 2014; Bacchetta et al., 2012, 2015; De Giudici et al., 2015). The floristic and vegetation uniqueness of this mining sector has been already highlighted (e.g. Angiolini et al., 2005; Bacchetta et al., 2007a, 2007b; Angius et al., 2011), and this has enriched the debate on the best strategies for their management. In several cases, they represent degraded environments with issues related to public health, which need extensive active interventions to initiate, accelerate, or direct recovery of a damaged ecosystem (Alday et al., 2012; Macdonald et al., 2015). In other cases, they constitute irreplaceable historical landscapes and even biodiversity heritages, which might be conserved and/or used as a new form of sustainable economy, mainly related to tourism development (Muntoni et al., 2020). In such situations, ongoing researches have revealed that passive restoration is a cost-effective method of encouraging the conservation potential of post-mining landscapes, since spontaneously revegetated post-mining sites may act as important secondary habitats for many threatened vascular plant species (Dekoninck et al., 2010; Řehounková et al., 2020; Salgueiro et al., 2020).

In this paper, we investigated the floristic composition of mining plant communities in terms of species richness, number of endemics, evenness, life form diversity and vegetation and cryptogamic covers along 96 plots positioned in the coastal mining area of Iglesiente. We tested the hypothesis that floristic composition patterns were influenced by both natural (topography and soil texture) and anthropogenic factors (soil disturbance, herbivory and estimated age since mining abandonment).

The main objectives of this study were to:(1) identify and describe plant species composition and assemblage that characterise the abandoned mines of the Iglesiente; (2) measure drivers of plant composition in each plant cluster; (3) use all this information to plan the restoration of abandoned mines.

2. Materials and methods

2.1. Study area

Investigations were conducted in the coastal mining area of the Iglesiente district (southwestern Sardinia, Italy; Fig. 1), a territory covering a total surface of 8433 ha, including around 25 km of coastal strip within the municipalities of Iglesias, Gonnesa, Portoscuso and Buggerru. The area of this study is characterised by a Mediterranean pluviseasonal bioclimate, within the thermo-Mediterranean thermotype, and ombrotypes between the upper dry and the lower sub-humid (Bacchetta et al., 2009). Rainfall ranges from 400 to 600 mm per year; the mean annual temperature ranges from 17 to 19 °C (SardegnaARPA, 2020). This area is characterised by lead and zinc metallic vein abun- dance that was, from the middle of XIX to the middle of XX century, the most important mining district of Italy (Ardau and Rundeddu, 2001). Following the general decline of Sardinian mining activities in the nineties of the past century, the communities living in this territory started to face social and environmental issues, mainly related to un- employment and pollution, especially due to the excess of toxic compounds, such as sulphides and sulphates, in soils, which may cause the

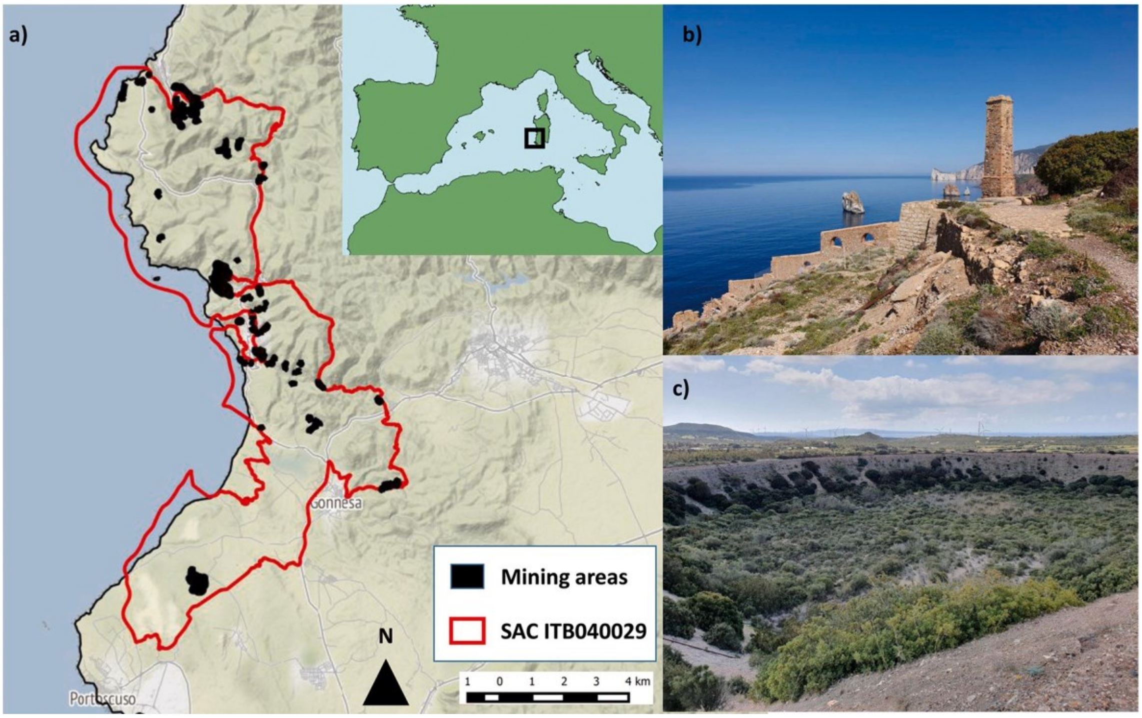

Fig. 1. a) The coastal mining area of the Iglesiente mining district within the Special Area of Conservation (SAC) and some typical landscapes: b) the station of 'Laveria La Marmora' (Nebida) used to screen and select the material extracted from the subsoil of the lead-zinc mine of Nebida, and c) the abandoned and naturally revegetated tailing dams of 'Seruci' (Gonnesa). Base map tiles by Stamen Design, under a CC BY 3.0, data by OpenStreetMap, under CC BY SA.

production of sulphuric acid and the release of toxic quantities of iron, aluminium and other heavy metals (Mannu et al., 2020). Nonetheless, goat and sheep husbandry still plays a role in small-scale farming systems.

The novel coastal landscape elements provided by a long-standing mining activity are, however, attractive after their abandonment, with several industrial architectures and scenic galleries (Arisci et al., 2003; Muntoni et al., 2020). This scenario is contextualised within a natural landscape of high value, with coastal cliffs alternated to long and pocket beaches, surrounded by small cliff islands and islets, such as the 'Pan di Zucchero' (Bocchieri, 1990). The consideration of green tourism as a new opportunity for economic growth is indirectly leading to an increasing interest to the high biodiversity and the general environmental value of this territory. Accordingly, the entire coastal area of the Iglesiente mining district is part of a valuable cultural heritage recognised by UNESCO in the Environmental and Mining Historical Geopark of Sardinia and it is included in a Special Area of Conservation (SAC 'Costa di Nebida', ITB040029, Habitat Directive 92/43/EEC).

The vestiges of past activities are mainly represented by three mining landforms: mining dumps and surface mines, generally characterised by coarse soil textures and unstable landforms, and tailing dams, while characterised by fine-grained soil textures in depositional landforms. Special vegetation series linked to the particular features of the substrata were already described in this context (Bacchetta et al., 2007a, 2007b). From the first stage of the therophytic grasslands, shrub communities can spread in a relatively short time, even on scraps and on extremely leached substrata. Of particular interest are the hemicryptochamaephytic and nanophanerophytic garrigues with several endemic and rare species that have colonised such artificial habitats shaped by a millennial mining activity (Angiolini et al., 2005). Accordingly, it has been defined as a 'micro hotspot' of biodiversty for its high endemicity of vascular plants: 136 over the 295 endemics to Sardinia are hosted here (Fois et al., 2018). Due to the peculiarity and relevance of such plant communities, these have been recently used to describe additional habitat types to be included in the Annex I of European 'Habitat' Directive (92/43/EEC) (Casavecchia et al., 2021; Fois et al., 2021).

2.2. Vegetation plot surveys

Vegetation surveys were conducted during spring-early summers of 2019 and 2020 in 96 plots of 5 × 5 m each, scattered in the coastal area of the Iglesiente mining district (Supplementary S1). The location of the plots was selected in an ad-hoc stratified way before fieldwork to ensure that samples represented each mining landforms: mining dumps (36 plots), surface mines (27 plots) and tailing dams (33 plots). Each landform was photointerpreted and delimitated. Then, the 'Random Selection Within Subsets' tool of the Hawth's Tools (http://www.spatialecology.com/htools) was used in Quantum GIS 2.4.0 (Quantum GIS Development Team, 2014) environment to select 10% of 100 × 100 m grid centroids of each mining landform. Plots were located from strictly coastal to more internal areas, ranging from 42 to 5422 m from the shoreline and an elevation from 19 to 439 m asl. For each plot, elevation, aspect and slope were recorded in the field with a global positioning system (GPS) receiver and a digital compass, while the linear distance from the coast was calculated in GIS environment on the base of the recorded coordinates.

We estimated the number of years since mining by interpreting aerophotos from 1954 to 2013 available from the from the national open access website Sardegna Geoportale (www.sardegnageoportale.it). Photointerpretations were supported by online site descriptions, available at http://www.minieredisardegna.it.

In the field, percentages of stoniness and rockiness, were visually estimated. To measure grain size, we estimated the prevalent textural category of the surface deposits on a scale of 1–5 as described in Gentili et al. (2013) and adapted to the mining context. A human-induced geomorphological disturbance index, ranging from 0 to 4, was used to interpret landform morphodynamics: 0 corresponds to a plot located on a stable landform and 4 on a landform affected by high instability (Gentili et al., 2013; Giaccone et al., 2019). The index is composed of three criteria: frequency of disturbances that affect the deposit (i.e., debris flow), movement of the surface and soil development. Herbivory presence was also visually estimated referring to percentage of damaged leaves: 0 (null), 1 (<25%), 2 (≥25 to <50%), 3 (≥50%). Percentage cover of vegetation, bryophytes and lichens were visually estimated. Each vascular plant species in the plot was listed (nomenclature following Bartolucci et al., 2018) and a cover-abundance value was assigned to every species in accordance with the Braun-Blanquet scale (Rivas-Martinez, 2005). Species cover in each plot is reported in Supplementary S2. For each surveyed plot, species richness (number of species per plot), endemic richness (number of species exclusive to the Italo-Tyrrhenian biogeographic superprovince, sensu Fois et al., 2022), life form diversity (number of Raunkiaer life forms) and Pielou's evenness (biodiversity index measuring the dominance of one species above the others in the same survey; Pielou, 1966) were calculated to characterise the biodiversity within the plots. The details for the attribution of each value are given in Table 1. Spearman's rank coefficient was calculated to assess the relationship between species richness, endemic richness, life form diversity, evenness, vegetation and cryptogamic covers variables and their correlations plotted using the R package *corrplot* (Supplementary S3; Wei et al., 2017).

2.3. Defining and comparing plant species composition among clusters

First, cluster analysis was performed to classify the plots according to their floristic similarity. Following Giaccone et al. (2019), Braun-Blanquet species cover was numerically transformed according to the mean cover of Wildi (2017). The Bray–Curtis dissimilarity index was used to build the dendrogram using the 'hclust' function (Ward's method) of the *fastcluster* R package (Müllner, 2013). Three clusters were retained to classify the plant assemblages and the environmental factor values were specified for each cluster. Non-Metric Multidimensional Scaling (NMDS) in the in the *vegan* R package (Oksanen et al., 2017) was used to visualize the relationship in our study area between plant species composition, sites and explanatory variables. Two dimensions were calculated for NMDS. Then, we incorporated the explanatory factors (Table 2) into the NMDS analysis with the 'envfit' function using a permutation approach (999 permutations), to test the relations between the ordered plant communities and environmental factors. The linear fit for each variable along the NMDS axis was determined and its significance tested. Non-parametric smoothed surfaces were calculated for each significant factor and reproduced on the ordination plot by using the 'ordisurf' function.

To find differences of plant diversity parameters among the identified clusters were graphically represented by box plots and tested by Dunn's multiple comparison with Bonferroni correction with the *dunn.test* R package (Dinno, 2017).

2.4. Plant composition and diversity gradients within clusters

To gain information on the characteristic species of each vegetation cluster, an indicator species analysis (Dufr^ene and Legendre, 1997) was performed with the R package *indicspecies* (De Caceres and Jansen, 2010). The group-based approach within the function 'signassoc' was applied to test the null hypothesis, 999 permutations were used to compute *p* values for each cluster. Sidak's p value correction was preferred to the more common Bonferroni correction due to its better statistical performance (Gossner et al., 2014). Indicator values (IndVal), ranging from 0 (no association) to 1 (complete association), were calculated using the function 'strassoc' of the same *indicspecies* package.

Generalized Linear Models (GLMs) were used to estimate the effect of each explanatory variable on plant diversity in each cluster. All statistical analyses were conducted using R v. 3.2.3 (R Development Core

Table 1
Descriptions of the ecological variables surveyed and measured for each 5 × 5 m plot.

Variable	Abbreviation	Description	Type
Elevation	elev	Elevation above sea level (metres)	Continuous
Aspect	asp	Categorical index of aspect: from 1° to 45° = 1; from 46° to 90° = 2; from 91° to 135° = 4; from 136° to 225° = 6; from 226° to 270° = 5; from 271 to 315 = 3; from 316° to 360° = 1	Class
Slope	slope	Slope angle degrees from horizontal	Continuous
Years of inactivity	yi	Years since last mining activity: 66 (last activity detected in the aerophotos of 1954), 52 (aerophotos of 1968), 43 (aerophotos of 1977), 17 (aerophotos of 2003), 7 aerophotos of 2013)	Semi-continuous
Vegetation cover	v_cover	Percent of plot surface covered by vascular plants	Continuous
Cryptogamic cover	c_cover	Percent of plot surface covered by bryophytes and lichen covers	Continuous
Herbivory	Herbiv	Estimated percentage of damaged leaves: 0 (null), 1 (<25%), 2 (≥25 to <50%), 3 (≥50%)	Class
Distance from the coast	d_coast	Shoreline distance (metres)	Continuous
Grain size	grain	Estimated prevalent textural category: clay/sand (Ø < 0,5 mm) = 1, sand (0,5 mm to 2 cm) = 2, mixed gravel clay/sand = 3, gravel (2 cm to 1 dm) = 4 and loose debris (Ø > 1 dm) = 5	Class
Stoniness	stones	Percent of plot surface covered by stones	Continuous
Rockiness	rocks	Percent of plot surface covered by rocks	Continuous
Disturbance	disturb	Estimated degree of human-induced geomorphological processes (mainly debris flow, debris/rock fall) action: stable/undisturbed = 1, low disturbance = 2, medium disturbance = 3, and high disturbance = 4	Class

Team, 2015). To select variables containing maximum information, univariate models were compared by means of corrected Akaike Information Criterion (AICc; Burnham et al., 2011) with the R package *AICcmodavg* (Mazerolle, 2019). All variables in models that had ΔAICc ≤2 were considered for multivariate analysis. Therefore, effects supported by the univariate GLM selection were combined and evaluated with automated multivariate model selection with *glmulti* R package (Calcagno and de Mazancourt, 2010). Averaged parameter estimates from top-ranked models within ΔAICc ≤2 (Burnham et al., 2011) and the relative variable importance (RI) of each explanatory variable, as the sum of the relative evidence weights of the candidate, were produced with *glmulti* R package (Calcagno and de Mazancourt, 2010).

Table 2
Correlations between NMDS axes and explanatory variables. NMDS1 and NMDS2 are the axis scores about the 2D ordination space.

	NMDS1	NMDS2	R^2	Signif.
Vegetation cover	−0.442	−0.897	0.217	***
Grain size	0.961	−0.274	0.152	***
Stoniness	0.856	0.517	0.151	**
Disturbance	0.982	−0.189	0.069	*
Herbivory	0.581	0.813	0.049	n.s.
Slope	0.997	−0.082	0.055	ns
Aspect	0.852	0.524	0.045	ns
Elevation	0.468	0.883	0.031	ns
Cryptogamic cover	0.041	−0.999	0.028	ns
Distance from the coast	−0.921	−0.388	0.028	ns
Years of inactivity	−0.853	−0.522	0.029	n.s.
Rockiness	0.625	−0.780	0.007	ns

R^2 = linear fit of correlation. Signif.: indicates the *p*-value. *** ≤ 0.001; ** ≤ 0.01; * ≤ 0.05; n.s. = not significant. See Table 1 for abbreviations.

3. Results

3.1. Plant composition and differences among clusters

The total number of vascular plant species found in the coastal area of the Iglesiente mining district was 227. *Arundo donax, Carpobrotus acinaciformis*, *C. edulis* and *Oxalis pes-caprae* were the only surveyed alien species. The most frequent taxa were *Rumex bucephalophorus* (in 46% of all plots), *Pistacia lentiscus* (43%), *Helichrysum microphyllum* subsp. *tyrrhenicum* and *Carlina corymbosa* (42% each). Among the 18 endemic taxa that were found within plots, apart from the already mentioned *H. microphyllum* subsp. *tyrrhenicum*, *Euphorbia pithyusa* subsp. *cupanii* (in 21% of all plots), *Limonium merxmuelleri* subsp. *sulcitanum* (15%) and *Santolina corsica* (12%) were also relatively frequent. Conversely, other endemics, such as *Bellium crassifolium*, *Dianthus insularis* or *Linum muelleri* (only 1 plot each), appeared relatively rare in the investigated area.

The Ward classification resulted in three distinct clusters, which were plotted in an NMDS ordination (Fig. 2, Supplementary S4; final stress = 0.22). Among significant variables ($p < 0.05$), the most correlated with NMDS1 were grain size, stoniness and disturbance, while the variable most correlated with NMDS2 was vegetation cover (Fig. 2, Supplementary S5; Table 2).

The first cluster comprised 66 plots with undifferentiated ecology while Cluster 2 and 3 were different in degrees of stoniness, disturbance and grain size (higher in Cluster 2), and vegetation cover (higher in Cluster 3). No statistical differences ($p > 0.05$) among clusters were found in terms of number of years since mining. Despite the relatively large standard error, the number of years since mining was generally lower in Cluster 2 (21 ± 3), compared to Cluster 1 (24 ± 2) and, especially, to Cluster 3 (29 ± 4). Cluster 2 appeared significantly more diverse than the rest. Species richness and life form diversity was higher, especially in comparison with Cluster 1 ($p < 0.001$), while endemic richness was significantly higher in Cluster 2 than that in Cluster 3 ($p < 0.01$). The dominance of one species above the others (i.e. evenness) decreased from Cluster 1 to Clusters 2 and 3. No significant differences were found in vegetation and cryptogamic covers among clusters (Fig. 3).

3.2. Plant composition and diversity gradients within clusters

From a vegetation viewpoint, Cluster 1 was characterised by early successional species of the *Tuberarion guttatae* alliance with a high frequency of *Rumex bucephalophorus* and *Reichardia picroides*, in some cases with high coverages of *Plantago bellardii* (Fig. 4). All the 18 endemic taxa were found at least once, even though none of them was particularly frequent or representative of the cluster. Cluster 2 grouped 16 plots of garrigue communities that were more complex and mature than those

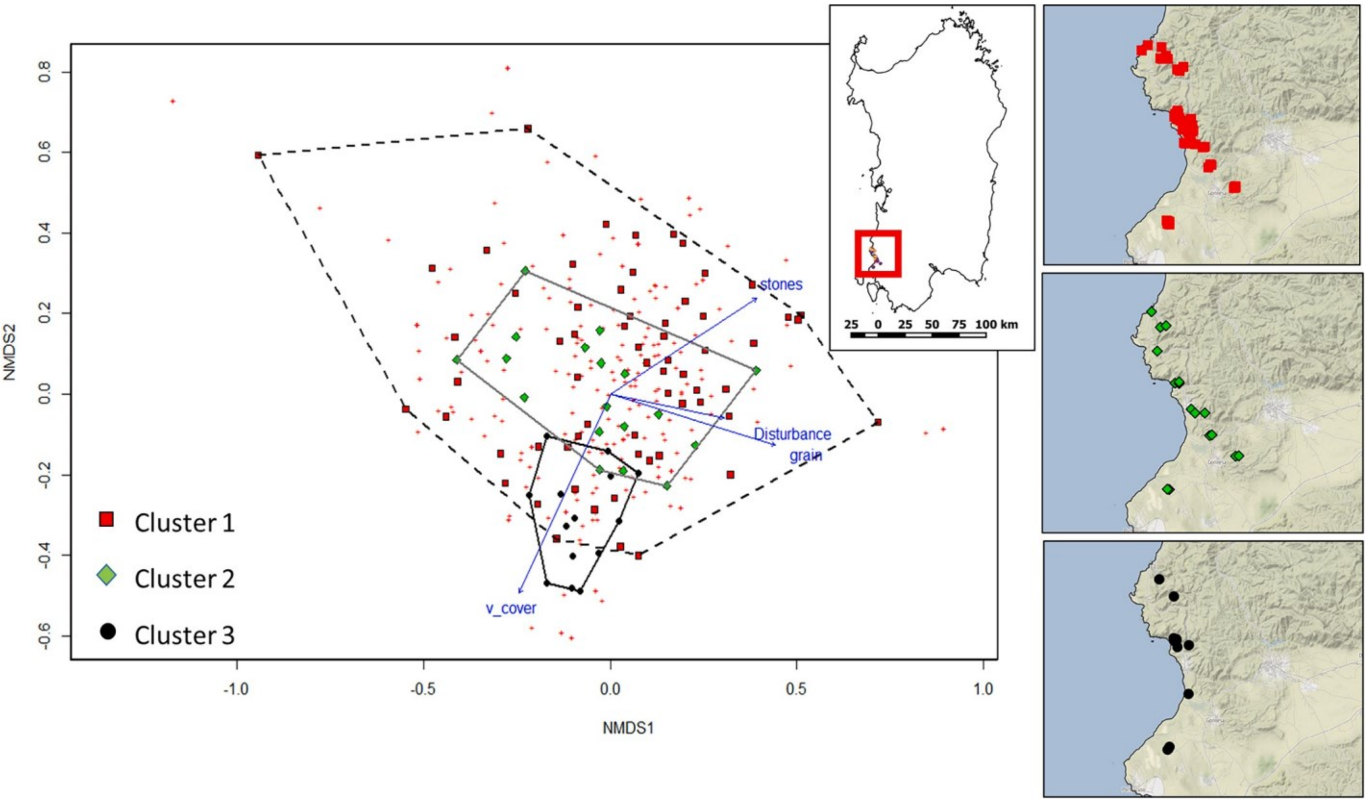

Fig. 2. NMDS biplot of the vegetation plots, based on the Bray–Curtis dissimilarity matrix, with fitted vectors of all significant explanatory factors Ordination stress = 0.22. The plot data are regrouped in 3 clusters, which are also mapped using the same symbols.

belonging to Cluster 1. Floristically, Cluster 2 is mainly represented by perennial herbs and small shrubs, such as *Carlina corymbosa*, *Helichrysum microphyllum* subsp. *tyrrhenicum*, *Ptilostemon casabonae* and *Santolina corsica* (Fig. 4). The latter were three of the eight endemic taxa that were present in this cluster. Last, Cluster 3 grouped 14 plots of the typical Mediterranean maquis with *Pistacia lentiscus* and some lianas, such as *Smilax aspera* and *Rubia peregrina*. Only six endemic taxa were here present.

With the exception of evenness, all the rest of response variables were correlated with one another. The highest correlation was between species richness and life form diversity, while the lower coefficients were between vegetation and cryptogamic covers and the rest of variables. The most influential variable was herbivory, which negatively influences species richness of Cluster 2, endemic richness of Cluster 1 and life form diversity of Clusters 1 and 3. The same herbivory positively affects the evenness of Clusters 1 and 3, and the cryptogamic cover of Cluster 1. Another influential variable was elevation, which was positively associated with vegetation cover of Cluster 1, species richness and endemic richness of Cluster 2, and negatively with life form diversity of Cluster 2. Disturbance was negatively correlated with species richness of Cluster 1, endemic richness of Cluster 2 and evenness of Cluster 1. Grain size was also an influencing factor, mainly for increasing the evenness of Cluster 1 and negatively affecting life form diversity and vegetation cover of Clusters 1 and 2. The number of years of mine inactivity was positively associated with evenness of Cluster 1 and vegetation cover of Cluster 3 (Fig. 5).

4. Discussion

This study confirms the great plant diversity interest of the coastal area of the Iglesiente mining district. Considering the overall plot surface (2400 m^2), the native and endemic vascular species richness, 227 and 18 taxa respectively, is similar to the already identified 'nano hotspots' of biodiversity along the coastal strip of the Iglesiente 'micro hotspot', such as the uninhabited islet of Pan di Zucchero (Fois et al., 2016) or the sand dunes of Buggerru (Fois et al., 2018), with six endemics in around 4000 m^2 and 24 endemics in 12,000 m^2, respectively.

Although our plot surveys did not allow to define the complete vegetation dynamic, which was in any case extensively described in other specific studies (Angiolini et al., 2005; Bacchetta et al., 2007a; Angius et al., 2011), our identified clusters were explicative of the current vegetation status. Cluster 1 was representing an early successional stage, which might develop, especially when the topography and environment are suitable, towards more evolved stages represented by the assemblages grouped in Clusters 2 and 3. Otherwise, it is the only cluster were all the recorded endemic plant taxa are present, with species richness and endemic species richness second only to Cluster 2. Cluster 3 appears to represent a stage similar to the one included in the special thermo-Mediterranean series of mining dumps, described in Bacchetta et al. (2007a), which can develop to micro-forests, mainly belonging to the *Oleo-Ceratonion* alliance. Currently, assemblages of Cluster 3 have already reached a relatively mature stage of vegetation that might need low interventions to facilitate the transition to micro-forests recovery. Their higher vegetation cover and lower stoniness, grain size and disturbance suggest that this is the best stage for soil recovery and stabilisation. The garrigues under the Cluster 2 were well differentiated from Cluster 3, both floristically and in terms of grain size, disturbance and stoniness. This stage, which is likely to be stable in the absence of any external intervention, resulted to be the richest in plant diversity. The variable disturbance facilitates the development of these communities, creating a secondary glareicolous environment with several indicator endemic species (*Helichrysum microphyllum* subsp. *tyrrhenicum*, *Ptilostemon casabonae*, *Santolina corsica*).

Three of the four species composition parameters, namely species richness, endemic species richness and life form diversity, were positively correlated to each other. This means that favouring one of these

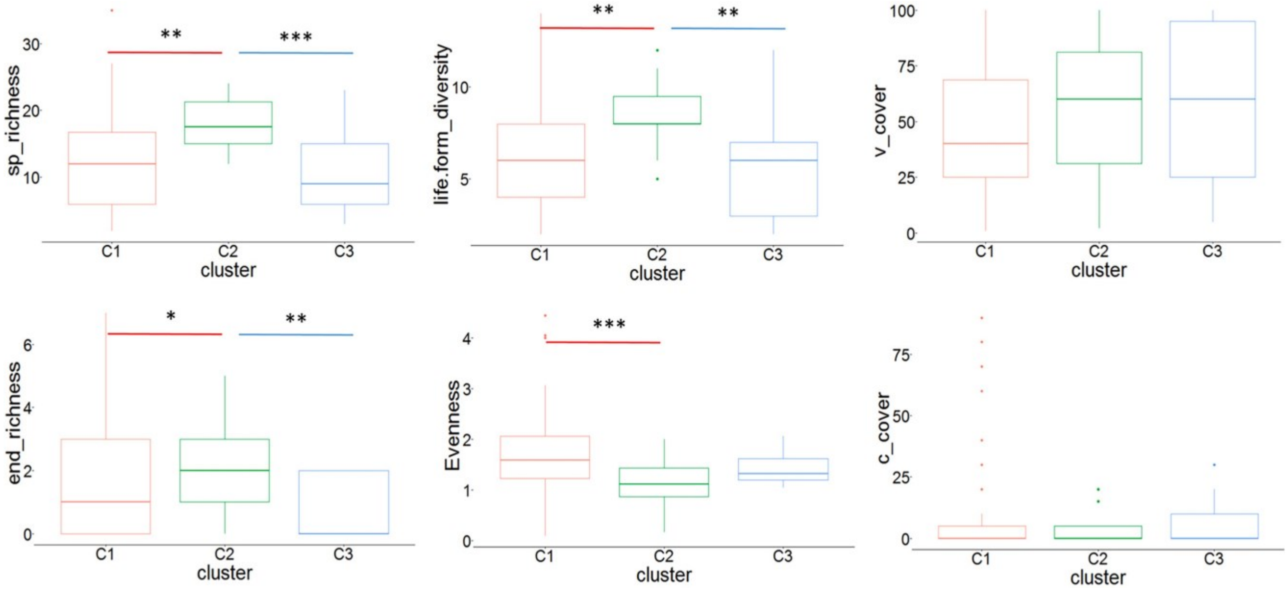

Fig. 3. Boxplots showing the distribution of six biodiversity parameters among the three clusters. Significant differences after Dunn's test with Bonferroni corrections are reported; *: 0.05 > p > 0.01, **: 0.01 > p > 0.001; ***: p < 0.001.

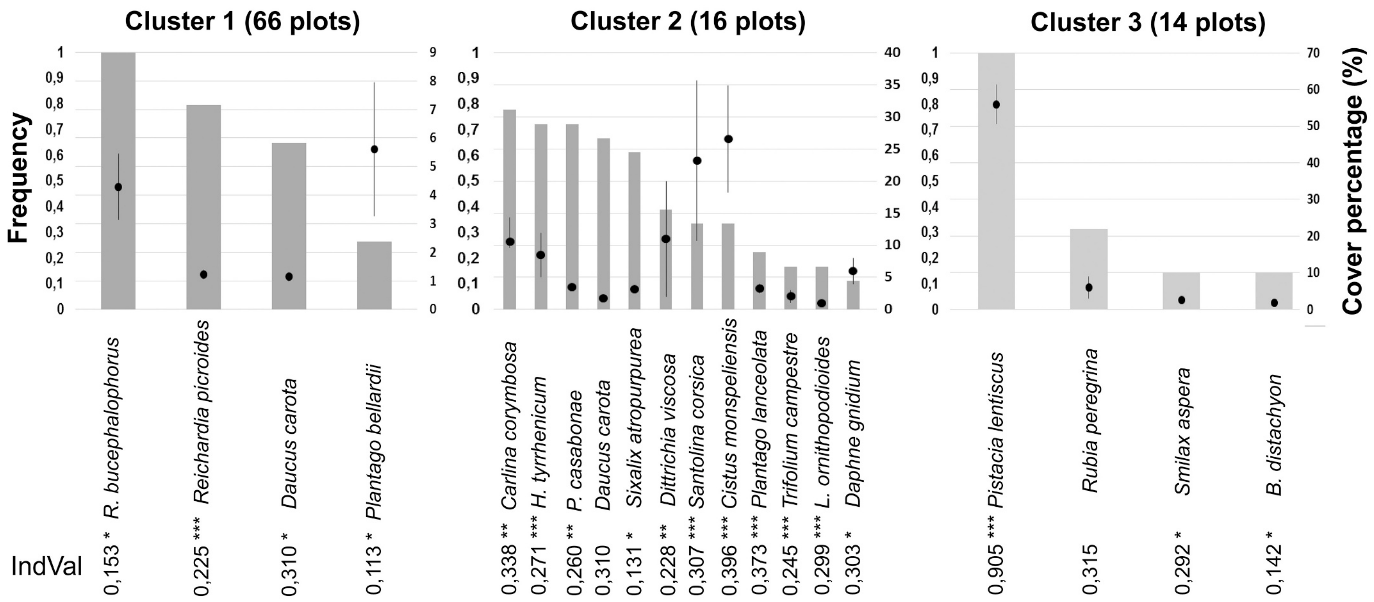

Fig. 4. Graph showing frequencies (bars) and cover (dots ± SE). Only species with frequency > 0.30 and Indicator Values (IndVal) > 0.30 or significant (p < 0.05) are reported; *: 0.05 > p > 0.01, **: 0.01 > p > 0.001; ***: p < 0.001. Abbreviated species: *Rumex bucephalophorus* (*R. bucephalophorus*), *Helichrysum microphyllum* subsp. *tyrrhenicum* (*H. tyrrhenicum*), *Ptilostemon casabonae* (*P. casabonae*), *Lotus ornithopodioides* (*L. ornithopodioides*), *Brachypodium distachyon* (*B. distachyon*).

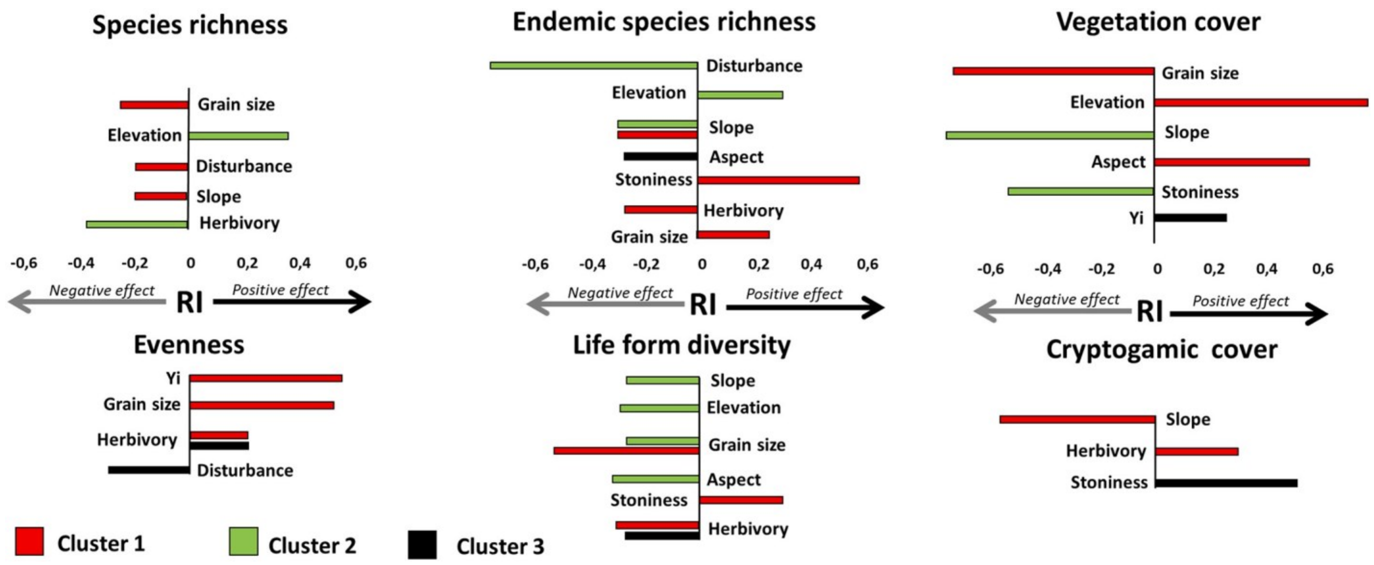

Fig. 5. Predictor variables included in the ΔAICc <2 set of models with significant correlations (alpha = 0.05) for the four plant diversity variables within each cluster. The relative variable importance (RI), as the sum of the relative evidence Akaike weights, represents the effect (positive or negative) of each predictor in each cluster.

parameters is possibly positively influencing the rest. Differently, the evenness, which is an important measure of inequality (Jost, 2010), and thus often related to irreplaceability (Belbin, 1995; Fois et al., 2020), increased inversely to all other parameters. This is quite common in nature and especially when rare and endemic species richness are considered (e.g. Fois et al., 2017; Kontopanou and Panitsa, 2020; Lozano et al., 2021). Similarly, we did not find the generally accepted unimodal relationship between richness-cover similar to the one between richness and biomass (e.g. Casado et al., 2004). The lack of such pattern in the study case is common in extreme conditions, such as in high mountains, screes and mining environments, where several plants with specific environmental requirements – often rare – tend to colonise such places with low diversity and competition (e.g. Magiera et al., 2018; Dalrymple et al., 2021).

As regards the variables explaining the species composition, our results suggest that both natural and anthropogenic factors are important. Among natural factors, the elevational gradient found for the entire island has been confirmed (Fois et al., 2017). The almost ubiquitous hump-shaped diversity gradients, with diversity peaking at mid-elevations, thus exponentially or linearly increasing in the coastal belt, might be related to several causes, such as climate, productivity, human and livestock accessibility, and topography diversity (e.g., Bunn et al., 2010; Sciandrello et al., 2020). Differently, species diversity is decreasing with slope, unlike general trends in Sardinian endemic plants, where richness is particularly high in cliffs, determined by low competition and accessibility (Fois et al., 2017). The here found trend might be thus contextualised in the peculiar mining environment, where natural revegetation after abandonment might be slower in steep slopes, generally poor in soil fertility (Cullen et al., 1998); controlled restoration projects can be necessary in these cases (Martínez-Ruiz et al., 2007; Gentili et al., 2011).

Anthropogenic factors are those that might be planned to be managed and these can be therefore the object of special focus. Herbivory is confirmed to be an influential factor in plant composition, leading to lower levels of species richness and endemic species richness on a small scale and favour homogenization of plant composition (i.e., evenness). Agro-pastoral practices, such as overgrazing or soil ploughing, can lead to degradation of natural vegetation, particularly where soils are shallow, and the climate is dry, like in the thermo-Mediterranean bioclimate (Lozano et al., 2021). However, Mediterranean plants are subject to a millennial history of grazing/browsing pressures herbivory that has driven the evolution of particularly adapted structures, such as plant spinescence (Bagella et al., 2019). Moreover, there is also increasing evidence that maintaining low or sustainable levels of grazing is crucial to preserve the typical Mediterranean landscape with several plant species and habitat types linked to pastures (Farris et al., 2010; Caria et al., 2021; Lozano et al., 2021). Similarly, herbivory on the investigated sites was favouring cryptogamic covers, thus facilitating other aspects of biodiversity. Human-induced geomorphological processes (i.e., disturbance), also resulted a limitation factor for plant diversity, especially for early successional stages (i.e., clusters 1 and 2). The passive limitation of disturbance for soil restoration, e.g. by fencing off an area, has been shown to be an effective strategy (e.g. Santoro et al., 2012; Li et al., 2014), although a case-by-case evaluation is also needed. Contrasting responses were found, finding positive effects, for example, in vegetation cover and diversity (Santoro et al., 2012) but also negative effects, such as in the seed dispersal capacity of some species (Lorite et al., 2021). However, in highly disturbed locations, soil re-establishment could be facilitated by providing artificial soil mixtures or by redistributing or transplantating soil and plant litter in order to favour the presence of microorganisms, which are crucial to stimulate increased soil organic matter, plant growth, and plant species richness (Chenot et al., 2017; Benetkova´ et al., 2020; Farrell et al., 2020). Nevertheless, caution must be paid when covering quarries with top soil, because it seems to encourage plant communities that are very different from the natural ones, with the intrusion of alien species (Gentili et al., 2011; Ballesteros et al., 2014). Differently from other places modified by human activities, such as lowland rivers, gypsum or marble quarries (Mota et al., 2004; Gentili et al., 2011; Angiolini et al., 2013) a low number of invasive species was found, suggesting that current plant assemblages serves as a barrier to their spread. Indeed, plants colonising the investigated environments where coming from the surrounding grasslands or woodlands, some of them of high natural value. As suggested by other studies (e.g. Mota et al., 2004; Gentili et al., 2011), the vicinity of natural or semi-natural grasslands or woodlands can significantly enhance natural revegetation processes with local flora.

In summary, the high biodiversity of the study area is likely ensured by the coexistence of the three clusters. The lack of significance of the number of years of mines inactivity in differentiating the three clusters suggest that these secondary formations tend to be stable, without any change from the current state. A wise management of the studied area, considering cultural, environmental elements, and also plant conservation needs, should thus maintain the current landscape heterogeneity represented by the here-found plant clusters. Punctual restoration activities, rather than passive or extensive, might be useful for specific areas (e.g. near urbanisations) or purposes (e.g. to protect particularly endangered species). For instance, the protection from ungulate herbivory with fences or the stabilisation of highly disturbed slopes might increase both biodiversity and health safety. Even though decreasing soil disturbance may increase endemic species richness, this could be patchily preserved to ensure the maintenance of less evolved successional stages (i.e. Clusters 1 and 2) where some endemic taxa found optimal conditions. Any passive or active restoration should however be complemented by monitoring, which is necessary for site managers and scientists to understand whether the above-mentioned suggestions effectively produce the desired results (Roz´e and Lemauviel, 2004).

In accordance with other studies (e.g. Khater et al., 2003; Gentili et al., 2011), species used in restoration projects should be chosen from local vegetation types according to their abundance and adaptability to local conditions. Some of the characteristic species of Clusters 1 and, especially, 2, such as *Helichrysum microphyllum* subsp. *tyrrhenicum* and *Scrophularia canina* subsp. *bicolor*, which have already been found to be useful for phytostabilisation and phytoextraction (Bacchetta et al., 2012, 2018; Boi et al., 2020), should be considered for restoration projects. Other rare or endemics, which are likely being specialised for colonising extremely disturbed environments, such as *Iberis integerrima*, *Linum muelleri*, *Limonium merxmuelleri* subspp. *merxnuelleri* and *sulcitanum* or *Epipactis helleborine* subsp. *tremolsii* (Angiolini et al., 2005; De Agostini et al., 2020), are further promising candidates for this scope.

This research offers a series of snapshots that might assist strategic guidelines for a wise management of mining environments. Otherwise, our findings might be supplemented by the following actions: (a) improve legislation to recognise post-mining sites as important secondary habitats to be conserved; (b) improve knowledge about the relative importance of substrate characteristics and pollution levels in shaping the biological diversity; and (c) understand soil and vegetation development over a longer-term time scale to provide sufficient information about the history of the site and the effect of grazing and other factors on vegetation development.

Data availability statement

The primary datasets are available in electronic supplementary materials.

Funding information

This study has not been financially supported.

CRediT authorship contribution statement

Mauro Fois: Conceptualization, Methodology, Software, Validation, Investigation, Data curation, Formal analysis, Writing – original draft, Writing – review & editing, Visualization. **Luca Murgia:** Conceptualization, Investigation, Data curation, Writing – review & editing. **Gianluigi Bacchetta:** Conceptualization, Methodology, Validation, Writing – review & editing, Funding acquisition, Supervision.

Declaration of Competing Interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

Data availability

All data used for this research are in the Supplementary Material

Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.ecoleng.2022.106879.

References

Alday, J.G., Pallavicini, Y., Marrs, R.H., Martínez-Ruiz, C., 2011. Functional groups and dispersal strategies as guides for predicting vegetation dynamics on reclaimed mines. Plant Ecol. 212, 1759–1775. https://doi.org/10.1007/s11258-011-9947-6.

Alday, J.G., Marrs, R.H., Martínez-Ruiz, C., 2012. Soil and vegetation development during early succession on restored coal wastes: a six-year permanent plot study. Plant Soil 353, 305–320. https://doi.org/10.1007/s11104-011-1033-2.

Angiolini, C., Bacchetta, G., Brullo, S., Casti, M., Giusso del Galdo, G., Guarino, R., 2005. The vegetation of mining dumps in SW-Sardinia. Feddes Repert 116, 243–276. https://doi.org/10.1002/fedr.200411072.

Angiolini, C., Nucci, A., Landi, M., Bacchetta, G., 2013. Distribution of endemic and alien plants along Mediterranean rivers: a useful tool to identify areas in need of protection? C R Biol. 336, 416–423. https://doi.org/10.1016/j.crvi.2013.07.001.

Angius, R., Bacchetta, G., Pontecorvo, C., 2011. Floristic and vegetation features of Monte Marganai (SW Sardinia). In: Nardi, G., Whitmore, D., Bardiani, M., Birtele, D., Mason, F., Spada, L., Cerretti, P. (Eds.), Biodiversity of Marganai and Montimannu (Sardinia). Research in the framework of the ICP Forests network. Centro Nazionale per lo studio e la conservazione, Verona, Italy, pp. 57–132.

Ardau, C., Rundeddu, L., 2001. Environmental problems and industrial archaeology in the Iglesiente mining district. Rend Fac. Sci. Univ. Cagliari 71, 91–108.

Arisci, A., De Waele, J., Di Gregorio, F., Ferrucci, I., Follesa, R., 2003. Integrated, sustainable touristic development of the karstic coastline of SW Sardinia. J. Coast. Conserv. 9, 81–90. https://doi.org/10.1652/1400-0350(2003)009[0081:ISTDOT]2.0.CO;2.

Bacchetta, G., Casti, M., Zavattero, L., 2007a. Integration of vegetational and multitemporal analysis: a case study in the abandoned mine district of Montevecchio (South-western Sardinia). Ann. Bot. 7, 164–173.

Bacchetta, G., Casti, M., Zavattero, L., 2007b. Analisi della vegetazione del distretto minerario di Montevecchio (Sardegna sud-occidentale). Fitosociol. 44, 83–108. https://doi.org/10.1080/00837792.2007.10670815.

Bacchetta, G., Bagella, S., Biondi, E., Farris, E., Filigheddu, R., Mossa, L., 2009. Vegetazione forestale e serie di vegetazione della Sardegna (con rappresentazione cartografica alla scala 1:350.000). Fitosociologia 46, 3–82.

Bacchetta, G., Cao, A., Cappai, G., Carucci, A., Casti, M., Fercia, M.L., Lonis, R., Mola, F., 2012. A field experiment on the use of *Pistacia lentiscus* L. and *Scrophularia canina* L. subsp. *bicolor* (Sibth. et Sm.) Greuter for the phytoremediation of abandoned mining areas. Plant Biosyst. 146, 1054–1063. https://doi.org/10.1080/11263504.2012.704886.

Bacchetta, G., Cappai, G., Carucci, A., Tamburini, E., 2015. Use of native plants for the remediation of abandoned mine sites in Mediterranean semiarid environments. Bull. Environ. Contam. Toxicol. 94, 326–333. https://doi.org/10.1007/s00128-015-1467-y.

Bacchetta, G., Boi, M.E., Cappai, G., De Giudici, G., Piredda, M., Porceddu, M., 2018. Metal tolerance capability of *Helichrysum microphyllum* Cambess. subsp. *tyrrhenicum* Bacch., Brullo & Giusso: a candidate for phytostabilization in abandoned mine sites. Bull. Environ. Contam. Toxicol. 101, 758–765. https://doi.org/10.1007/s00128-018-2463-9.

Bagella, S., Filigheddu, R., Benesperi, R., Giordani, P., Minuto, L., Viciani, D., Caria, M. C., Pisanu, S., Casazza, G., 2019. Thorn, spine and prickle patterns in the Italian flora. Plant Biosyst. 153, 118–133. https://doi.org/10.1080/11263504.2018.1474961.

Ballesteros, M., Cañadas, E.M., Foronda, A., Peñas, J., Valle, F., Lorite, J., 2014. Central role of bedding materials for gypsum-quarry restoration: an experimental planting of gypsophile species. Ecol. Eng. 70, 470–476. https://doi.org/10.1016/j.ecoleng.2014.06.001.

Bartolucci, F., Peruzzi, L., Galasso, G., Albano, A., Alessandrini, A., Ardenghi, N.M.G., et al., 2018. An updated checklist of the vascular flora native to Italy. Plant Biosyst. 152, 179–303. https://doi.org/10.1080/11263504.2017.1419996.

Belbin, L.E.E., 1995. A multivariate approach to the selection of biological reserves. Biodivers. Conserv. 4, 951–963. https://doi.org/10.1007/BF00058206.

Benetková, P., Tichý, L., Háněl, L., Kukla, J., Vicentini, F., Frouz, J., 2020. The effect of soil and plant material transplants on vegetation and soil biota during forest restoration in a limestone quarry: a case study. Ecol. Eng. 158, 106039 https://doi.org/10.1016/j.ecoleng.2020.106039.

Bocchieri, E., 1990. Some herborizations on the islet of Pan di Zucchero (Sardinia W-SW). G. Bot. Ital. 124, 615–621. https://doi.org/10.1080/11263509009430261.

Boi, M.E., Porceddu, M., Cappai, G., De Giudici, G., Bacchetta, G., 2020. Effects of zinc and lead on seed germination of *Helichrysum microphyllum* subsp. *tyrrhenicum*, a metal-tolerant plant. Int. J. Environ. Sci. Tech. 17, 1917–1928. https://doi.org/10.1007/s13762-019-02589-9.

Bonari, G., Padullés Cubino, J., Sarmati, S., Landi, M., Zerbe, S., Marcenò, C., Scoppola, A., Angiolini, C., 2021. Ecosystem state assessment after more than 100 years since planting for dune consolidation. Restor. Ecol. 29, e13435 https://doi.org/10.1111/rec.13435.

Bunn, W.A., Jenkins, M.A., Brown, C.B., Sanders, N.J., 2010. Change within and among forest communities: the influence of historic disturbance, environmental gradients, and community attributes. Ecography 33, 425–434. https://doi.org/10.1111/j.1600-0587.2009.06016.x.

Burnham, K.P., Anderson, D.R., Huyvaert, K.P., 2011. AIC model selection and multimodel inference in behavioral ecology: some background, observations, and comparisons. Behav. Ecol. Sociobiol. 65, 23–35. https://doi.org/10.1007/s00265-010-1029-6.

Calcagno, V., de Mazancourt, C., 2010. Glmulti: an R package for easy automated model selection with (generalized) linear models. J. Stat. Softw. 34, 1–29. https://doi.org/10.18637/jss.v034.i12.

Caria, M.C., Grillas, P., Rivieccio, G., Bagella, S., 2021. Effect of interannual meteorological fluctuations and wild boar disturbance on the vegetation of Mediterranean temporary ponds. Appl. Veg. Sci. 24, e12624 https://doi.org/10.1111/avsc.12624.

Casado, M.A., Castro, I., Ramírez-Sanz, L., Costa-Tenorio, M., de Miguel, J.M., Pineda, F. D., 2004. Herbaceous plant richness and vegetation cover in Mediterranean grasslands and shrublands. Plant Ecol. 170, 83–91. https://doi.org/10.1023/B:VEGE.0000019021.62054.62.

Casavecchia, S., Allegrezza, M., Angiolini, C., Biondi, E., Bonini, F., Del Vico, E., Fanfarillo, E., Foggi, B., et al., 2021. Proposals for improvement of Annex I of Directive 92/43/EEC: Central Italy. Plant Sociol. 58, 99–118. https://doi.org/10.3897/pls2021582/08.

Chenot, J., Jaunatre, R., Buisson, E., Dutoit, T., 2017. Long-term effects of topsoil transfer assessed thirty years after rehabilitation of dry alluvial quarries in Southeastern France. Ecol. Eng. 99, 1–12. https://doi.org/10.1016/j.ecoleng.2016.11.010.

Cidu, R., Biddau, R., Fanfani, L., 2009. Impact of past mining activity on the quality of groundwater in SW Sardinia (Italy). J. Geochem. Explor. 100, 125–132. https://doi.org/10.1016/j.gexplo.2008.02.003.

Cullen, W.R., Wheater, C.P., Dunelavy, P.J., 1998. Establishment of species-rich vegetation on reclaimed limestone quarry faces in Derbyshire. UK Biol. Conserv. 84, 25–33. https://doi.org/10.1016/S0006-3207(97)00089-X.

Dalrymple, S.E., Hopkins, J., Carter, S.P., Slingsby, D.R., 2021. Nutrient additions three decades on: potential interactions of nutrients and climate in the recovery of a high latitude serpentine system. Plant Biosyst. 155, 64–72. https://doi.org/10.1080/11263504.2019.1701578.

De Agostini, A., Caltagirone, C., Caredda, A., Cicatelli, A., Cogoni, A., Farci, D., Guarino, F., Garau, A., Labra, M., Lussu, M., Piano, D., Sanna, C., Tommasi, N., Vacca, A., Cortis, P., 2020. Heavy metal tolerance of orchid populations growing on abandoned mine tailings: a case study in Sardinia Island (Italy). Ecotoxicol. Environ. Saf. 189, 110018 https://doi.org/10.1016/j.ecoenv.2019.110018.

De Caceres, M., Jansen, F., 2010. Package 'indicspecies'. Function to assess the strenght and significance of relationships of species site group associations. http://CRAN.R-project.org/package=indicspecies (accessed 16 February 2018).

De Giudici, G., Medas, D., Meneghini, C., Casu, M.A., Giannoncelli, A., Iadecola, A., Podda, S., Lattanzi, P., 2015. Microscopic bio mineralization processes and Zn bioavailability: a synchrotron-based investigation of *Pistacia lentiscus* L. root. Environ. Sci. Pollut. Res. 22, 19352–19361. https://doi.org/10.1007/s11356-015-4808-9.

Dekoninck, W., Hendrickx, F., Dethier, M., Maelfait, J.P., 2010. Forest succession endangers the special ant fauna of abandoned quarries along the river Meuse (Wallonia, Belgium). Restor. Ecol. 18, 681–690. https://doi.org/10.1111/j.1526-100X.2008.00499.x.

Dinno, A., 2017. Dunn.Test: Dunn's test of multiple comparisons using rank sums. R package version 1.3.5. https://cran.r-project.org/package=dunn.test.

Du, H.D., Cao, Y.C., Zhang, Y.Y., Ning, B.Y., 2021. Plant community development in a coal mining subsidence area: active versus passive revegetation. Écosci. 28, 185–197. https://doi.org/10.1080/11956860.2021.1901425.

Dufrêne, M., Legendre, P., 1997. Species assemblages and indicator species: the need for a flexible asymmetrical approach. Ecol. Monogr. 67, 345–366. https://doi.org/10.1890/0012-9615(1997)067[0345:SAAIST]2.0.CO;2.

Farrell, H.L., Barberán, A., Danielson, R.E., Fehmi, J.S., Gornish, E.S., 2020. Disturbance is more important than seeding or grazing in determining soil microbial communities in a semiarid grassland. Restor. Ecol. 28, S335–S343. https://doi.org/10.1111/rec.13156.

Farris, E., Filigheddu, R., Deiana, P., Farris, G.A., Garau, G., 2010. Short-term effects on sheep pastureland due to grazing abandonment in a Western Mediterranean island ecosystem: a multidisciplinary approach. J. Nat. Conserv. 18, 258–267. https://doi.org/10.1016/j.jnc.2009.11.003.

Fois, M., Fenu, G., Bacchetta, G., 2016. Global analyses underrate part of the story: finding applicable results for the conservation planning of small Sardinian islets' flora. Biodivers. Conserv. 25, 1091–1106. https://doi.org/10.1007/s10531-016-1110-1.

Fois, M., Fenu, G., Cañadas, E.M., Bacchetta, G., 2017. Disentangling the influence of environmental and anthropogenic factors on the distribution of endemic vascular plants in Sardinia. PLoS One 12, e0182539. https://doi.org/10.1371/journal.pone.0182539.

Fois, M., Bacchetta, G., Cogoni, D., Fenu, G., 2018. Current and future effectiveness of the Natura 2000 network for protecting plant species in Sardinia: a nice and complex strategy in its raw state? J. Environ. Plan. Manag. 61, 332–347. https://doi.org/10.1080/09640568.2017.1306496.

Fois, M., Podda, L., Médail, F., Bacchetta, G., 2020. Endemic and alien vascular plant diversity in the small Mediterranean islands of Sardinia: Drivers and implications for their conservation. Biol. Conserv. 244, 108519 https://doi.org/10.1016/j.biocon.2020.108519.

Fois, M., Bacchetta, G., Caria, M.C., Cogoni, D., Farris, E., Fenu, G., Manca, M., Pinna, M. S., Pisanu, S., Rivieccio, G., Bagella, S., 2021. Proposals for improvement of Annex I of Directive 92/43/EEC: Sardinia. Plant Sociol. 58, 65–76. https://doi.org/10.3897/pls2021582/06.

Fois, M., Farris, E., Calvia, G., Campus, G., Fenu, G., Porceddu, M., Bacchetta, G., 2022. The endemic vascular flora of Sardinia: a dynamic checklist with an overview of biogeography and conservation status. Plants 11, 601. https://doi.org/10.3390/plants11050601.

Gentili, R., Sgorbati, S., Baroni, C., 2011. Plant species patterns and restoration perspectives in the highly disturbed environment of the Carrara marble quarries (Apuan Alps, Italy). Restor. Ecol. 19, 32–42. https://doi.org/10.1111/j.1526-100X.2010.00712.x.

Gentili, R., Armiraglio, S., Sgorbati, S., Baroni, C., 2013. Geomorphological disturbance affects ecological driving forces and plant turnover along an altitudinal stress gradient on alpine slopes. Plant Ecol. 214, 571–586. https://doi.org/10.1007/s11258-013-0190-1.

Giaccone, E., Luoto, M., Vittoz, P., Guisan, A., Mari´ethoz, G., Lambiel, C., 2019. Influence of microclimate and geomorphological factors on alpine vegetation in the Western Swiss Alps. Earth Surf. Process. Landf. 44, 3093–3107. https://doi.org/10.1002/esp.4715.

Gossner, M.M., Fonseca, C.R., Paˇsali´c, E., Türke, M., Lange, M., Weisser, W.W., 2014. Limitations to the use of arthropods as temperate forests indicators. Biodivers. Conserv. 23, 945–962. https://doi.org/10.1007/s10531-014-0644-3.

Jim´enez, M.N., Bacchetta, G., Casti, M., Navarro, F.B., Lallena, A.M., Ferna´ndez-Ondon˜o, E., 2011. Potential use in phytoremediation of three plant species growing on contaminated mine-tailing soils in Sardinia. Ecol. Eng. 37, 392–398. https://doi.org/10.1016/j.ecoleng.2010.11.030.

Jim´enez, M.N., Bacchetta, G., Casti, M., Navarro, F.B., Lallena, A.M., Fernand`ez-Ondono, E., 2014. Study of Zn, Cu and Pb content in plants and contaminated soils in Sardinia. Plant Biosyst. 148, 419–428. https://doi.org/10.1080/11263504.2013.776651.

Jost, L., 2010. The relation between evenness and diversity. Diversity 2, 207–232. https://doi.org/10.3390/d2020207.

Khater, C., Martin, A., Maillet, J., 2003. Spontaneous vegetation dynamics and restoration prospects for limestone quarries in Lebanon. Appl. Veg. Sci. 6, 199–204. https://doi.org/10.1111/j.1654-109X.2003.tb00580.x.

Kontopanou, A., Panitsa, M., 2020. Habitat islands on the Aegean islands (Greece): elevational gradient of chasmophytic diversity, endemism, phytogeographical patterns and need for monitoring and conservation. Diversity 12, 33. https://doi.org/10.3390/d12010033.

Li, Q., Zhou, D., Jin, Y., Wang, M., Song, Y., Li, G., 2014. Effects of fencing on vegetation and soil restoration in a degraded alkaline grassland in Northeast China. J. Arid. Land 6, 478–487. https://doi.org/10.1007/s40333-013-0207-6.

Lorite, J., Salazar-Mendías, C., Pawlak, R., Can˜adas, E.M., 2021. Assessing effectiveness of exclusion fences in protecting threatened plants. Sci. Rep. 11, 16124. https://doi.org/10.1038/s41598-021-95739-4.

Lozano, V., Vacca, G., Camarda, I., Brundu, G., Sanna, F., Caredda, S., Re, G.A., 2021. Plant diversity in Sardinian mountain rangelands: analysis of its relationships with grazing, land management, and pastoral value. Ecologies 2, 164–174. https://doi.org/10.3390/ecologies2010009.

Macdonald, S.E., Landh¨ausser, S.M., Skousen, J., Franklin, J., Frouz, J., Hall, S., Jacobs, D.F., Quideau, S., 2015. Forest restoration following surface mining disturbance: challenges and solutions. New For. 46, 703–732. https://doi.org/10.1007/s11056-015-9506-4.

Magiera, A., Feilhauer, H., Waldhardt, R., Wiesmair, M., Otte, A., 2018. Mapping plant functional groups in subalpine grassland of the greater caucasus. Mt. Res. Dev. 38, 63–72. https://doi.org/10.1659/MRD-JOURNAL-D-17-00082.1.

Mannu, R., Pantini, P., Sassu, A., Verdinelli, M., 2020. A multidiversity approach to investigate the impact of mining exploitation on spider diversity in the abandoned mine district of Montevecchio-Ingurtosu (Sardinia, Italy). Environ. Sci. Pollut. Res. 27, 32615–32627. https://doi.org/10.1007/s11356-020-09553-y.

Martínez-Ruiz, C., Fern´andez-Santos, B., 2005. Natural revegetation on topsoiled mining-spoils according to the exposure. Acta Oecol. 28, 231–238. https://doi.org/10.1016/j.actao.2005.05.001.

Martínez-Ruiz, C., Fern´andez-Santos, B., Putwain, P.D., Fern´andez-Go´mez, M.J., 2007. Natural and man-induced revegetation on mining wastes: changes in the floristic composition at early succession. Ecol. Eng. 30, 286–294. https://doi.org/10.1016/j.ecoleng.2007.01.014.

Mazerolle, M.J., 2019. Package AICmodavg in R. R package version 2.2.0. http://CRAN.R-project.org/package=AICcmodavg (accessed 14 July 2019).

Mota, J.F., Sola, A.J., Jim´enez-S´anchez, M.L., P´erez-García, F., Merlo, M.E., 2004. Gypsicolous flora, conservation and restoration of quarries in the southeast of the Iberian Peninsula. Biodivers. Conserv. 13, 1797–1808. https://doi.org/10.1023/B:BIOC.0000035866.59091.e5.

Müllner, D., 2013. Fastcluster: fast hierarchical, agglomerative clustering routines for R and Python. J. Stat. Softw. 53, 1–18. https://doi.org/10.18637/jss.v053.i09.

Muntoni, F., Balvis, T., Rizzo, R., Loru, P., 2020. Territorial planning of geological mining historical and environmental park of Sardinia. Geoheritage 12, 1–11. https://doi.org/10.1007/s12371-020-00423-0.

Oksanen, J., Blanchet, F., Friendly, M., Wagner, H., Roeland, K., Pierre, L., et al., 2017. Vegan: Community Ecology Package version 2.4-4. https://CRAN.R-project.org/package=vegan (accessed 4 August 2017).

Pallavicini, Y., Alday, J.G., Martínez-Ruiz, C., 2015. Factors affecting herbaceous richness and biomass accumulation patterns of reclaimed coal mines. Land. Degr. Dev. 26, 211–217. https://doi.org/10.1002/ldr.2198.

Pielou, E., 1966. The measurement of diversity in different types of biological collections. J. Theor. Biol. 13, 131–144. https://doi.org/10.1016/0022-5193(66)90013-0.

Quantum GIS Development Team, 2014. Quantum GIS geographic information system. In: Open Source Geospatial Foundation Project.

R Development Core Team, 2015. R: A Language and Environment for Statistical Computing. R Foundation for Statistical Computing, Vienna, Austria. http://www.R-project.org/ (accessed 3 November 2016).

Rehounkova´, K., Prach, K., 2006. Spontaneous vegetation succession in disused gravel-sand pits: Role of local site and landscape factors. J. Veg. Sci. 17, 583–590. https://doi.org/10.1111/j.1654-1103.2006.tb02482.x.

Rˇehounkova´, K., Vítovcov´a, K., Prach, K., 2020. Threatened vascular plant species in spontaneously revegetated post-mining sites. Restor. Ecol. 28, 679–686. https://doi.org/10.1111/rec.13027.

Rivas-Martinez, S., 2005. Notions on dynamic-catenal phytosociology as a basis of landscape science. Plant Biosyst. 139, 135–144. https://doi.org/10.1080/11263500500193790.

Roz´e, F., Lemauviel, S., 2004. Sand dune restoration in North Brittany, France: a 10-year monitoring study. Restor. Ecol. 12, 29–35. https://doi.org/10.1111/j.1061-2971.2004.00264.x.

Salgueiro, P.A., Prach, K., Branquinho, C., Mira, A., 2020. Enhancing biodiversity and ecosystem services in quarry restoration-challenges, strategies, and practice. Restor. Ecol. 28, 655–660. https://doi.org/10.1111/rec.13160.

Santoro, R., Jucker, T., Prisco, I., Carboni, M., Battisti, C., Acosta, A.T., 2012. Effects of trampling limitation on coastal dune plant communities. Environ. Manag. 49, 534–542. https://doi.org/10.1007/s00267-012-9809-6.

SardegnaARPA, 2020. WebGIS. http://www.sar.sardegna.it/servizi/sit/webgis.asp (accessed 30 August 2021).

Sciandrello, S., Minissale, P., Del Galdo, G.G., 2020. Vascular plant species diversity of Mt. Etna (Sicily): endemicity, insularity and spatial patterns along the altitudinal gradient of the highest active volcano in Europe. PeerJ 8, e9875. https://doi.org/10.7717/peerj.9875.

Wei, T., Simko, V., Levy, M., Xie, Y., Jin, Y., Zemla, J., 2017. Package 'corrplot'. Statistician 56, 316–324.

Wildi, O., 2017. Data Analysis in Vegetation Ecology. Wiley-Blackwell, Chichester, England. https://doi.org/10.1079/9781786394224.0000.