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Biodiversity Characteristics and Carbon Sequestration Potential of Successional Woody Plants versus Tree Plantation under Different Reclamation Treatments on Hard-Coal Mine Heaps—A Case Study from Upper Silesia

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Abstract: In the discussion about sustainable forestry, a key role is played by the development of ecosystem services, including ecological, social, and economic ones, in which biodiversity and carbon (C) sequestration are among the most important. Afforestation of disturbed and post-mining sites is one of the ways to minimize the negative impact of civilization on the environment. Optimizing C sequestration strategies at post-mining sites plays a crucial role in promoting ecosystem recovery, supporting climate change mitigation, and enabling C offsetting. In this study, we compared the C storage in the soil and plant biomass of forest ecosystems developed on coal-mine heaps for different scenarios of reclamation and succession. We tested combinations of sites (i.e., non-reclaimed sites on bare carboniferous rock [BR] and sites reclaimed by applying topsoil [TS]) and successional woodland and tree plantation. The estimated potential for total C storage (in the soil + biomass) for TS sites ranged from 68.13 to 121.08 Mg ha^{−1}, of which 52.20–102.89 Mg ha^{−1} was stored in the soil and 12.09–20.15 Mg ha^{−1} in the biomass. In the non-reclaimed sites on BR, the total C storage was much higher, amounting to 523.14 Mg ha^{−1} (507.66 Mg ha^{−1} being in the soil), which was due to the geogenic coal content in the BR. However, the C storage in the biomass (15.48 Mg ha^{−1}) and litter (5.91 Mg ha^{−1}) was similar to the amounts obtained from the reclaimed sites. The number of species did not differ statistically significantly between the analyzed variants. On average, 14 species were recorded in the plots. The average Shannon–Wiener index (H′) value was higher for sites with BR (1.99) than TS variants on reclaimed plots (1.71). The lowest H′ value was for those plots with *Robinia pseudacacia* in the stand. One of the main implications of the obtained results for sustainable forestry is the perspective of using succession in the recovery of a disturbed ecosystem. We noted that woodlands from succession on BR are highly biodiverse, have high C sequestration potential, and do not require time-consuming reclamation treatments.

Keywords: reclamation; woodland; carbon sequestration; post-mining sites; topsoiling; plant biodiversity

1. Introduction

Despite efforts to transition economies and embrace alternative energy sources, post-mining areas present a significant environmental challenge [1]. The impacts of mining activities, both active and abandoned, have manifested in diverse geomorphological effects, such as waste-rock spoil heaps, contaminated and brownfield sites, and underutilized fallow land. These disturbances extend to all components of the ecosystem due to mining operations affecting both the above- and below ground [2]. The spoil material from coal mining, deposited in heaps, often hinders the establishment of vegetation due to its high skeletal content, low pH from its pyrite content, and relatively high salinity [3,4]. Furthermore, the weathered materials typically exhibit low nutrient availability, especially phosphorus and nitrogen (N) [5], emphasizing the extensive ecological impact of mining activities on post-mining landscapes.

Given the heterogeneity and challenging conditions of post-mining sites, a range of reclamation methods can be utilized to expedite the establishment of vegetation, soil development, and ecosystem growth. These methods include topsoiling, which entails the application of fertile material to spoil heaps, in conjunction with the use of mineral or organic fertilizers, and the introduction of suitable plant species [6,7]. Topsoiling plays a crucial role in enhancing the properties of post-mining soils, leading to improved growth conditions for plants [8]. Despite its benefits, topsoiling is labor-intensive and costly. Topsoil suffers from limited availability and is often of substandard quality, requiring meticulous processing and storage before application [6,9]. Directly introducing suitable vegetation to dumped spoil material can serve as a viable alternative to topsoiling in supporting the establishment of vegetation at restored sites [6]. Another approach involves allowing post-mining heaps or sections to undergo succession to reduce reclamation expenses and foster the development of more resilient, self-sustaining ecosystems over time [10,11]. Succession is a natural process that allows for the spontaneous recolonization of vegetation and the restoration of diverse ecosystems [12]. Studies have shown that post-mining sites can be reclaimed effectively through succession or passive restoration methods, leading to increased biodiversity and ecosystem services [13,14]. By allowing natural processes to unfold and only assisting them when necessary, these sites can recover and continue to support diverse plant communities over time [15,16].

The novel ecosystems that emerge on spoil heaps are considered fragile and require meticulous restoration and management. A comprehensive understanding of the technical aspects involved in handling spoil heaps, and a recognition of their ecological significance in urban and industrial settings, is essential. This knowledge empowers individuals and communities to actively engage in supporting and restoring ecological functions, such as plant community succession, dynamic soil processes, soil microbiological activity, afforestation efficiency, biomass production, carbon (C) sequestration [3], the development of biodiversity [17], as well as the response of the plants to environmental stress factors [18]. A large body of research has shown that biodiversity loss can reduce ecosystem functioning. Novel ecosystems, such as hard-coal mines, increase diversity, allowing the scope and functions of developing ecosystems to expand [19], depending on the type of reclamation. The biodiversity of the plants in post-industrial areas is mainly influenced by substrate parameters and the availability of propagules and is expressed as the average species diversity in individual locations of a post-industrial ecosystem. Moreover, due to the extreme habitat conditions prevailing in such heaps [5,6] and their mosaic arrangement, to assess diversity, it is necessary to compare the vegetation between these post-industrial ecosystems [15]. The use of various types of reclamation leads to the colonization of areas by vegetation within the same dump differently, depending on the type of reclamation method used [6]. This is most often expressed as the value of the Shannon–Wiener index (H') as a way of measuring the diversity of the species in a community and of defining the alpha diversity [20].

Proper reclamation of post-mining sites is of key importance for subsequent forest management, including the performance of basic ecological functions that can increase the resistance of the human environment to climate change. They also provide valuable ecosystem services, including significant C sequestration capabilities [19,21]. This highlights the importance of proactive management and restoration efforts in post-mining landscapes to promote ecological sustainability and environmental resilience.

Vegetation restoration has been recognized as a key and sustainable approach to enhancing post-mining soil quality and functions. It serves as an effective method for increasing the organic matter input to soils and promoting soil C sequestration [22]. The dynamic response of soil organic C varies depending on the type of vegetation restoration, which directly impacts the litter mass [23]. Research has found that the choice of vegetation significantly influences changes in the soil C pool and its dynamics by affecting the physical and chemical protection of the organic matter provided by soil aggregates [24]. The nutrient cycling facilitated by plant–soil interactions and the C sink function of ecosystems may exhibit temporal and spatial variations. Consequently, the timing of introducing different vegetation types, the duration of the vegetation restoration, and the plant combinations can profoundly influence the C sequestration potential in post-mining areas.

Spontaneous succession is a slow process, and it is not always able to restore the forest ecosystem in a comparable time compared to the use of technical reclamation [25]. Although the biodiversity of ecosystems from succession compared to afforestation is often similar [26,27], sites with succession are characterized by a lower C sequestration potential than reclaimed sites [28]. For example, Pietrzykowski and Krzaklewski [28] determined the accumulation rate of C and N to be three times and five higher, respectively, in sand-mine soils under afforestation than under succession communities. A similar tendency has been observed by Frouz et al. [29] in clayey soils in areas after lignite mining. In reclaimed mine soils (under 22- to 32-year-old stands of different tree species), higher C accumulation has been observed [29]. Reclamation leads to increased humus accumulation, humification, and humus transformation rates than succession on spoil heaps after lignite mining. However, the differences between reclaimed and non-reclaimed sites decrease with site age and are very small in 40-year-old sites [30]. The advantages of the reclamation and afforestation of post-mining sites also include both higher biomass and higher C sequestration in the biomass compared to areas left to succession [31,32]. However, Frouz et al. [33] found higher woody biomass in reclaimed sites in younger sites (<20 years) compared to sites with succession. In older sites, this difference disappears [33]. Determining the potential for C sequestration in biomass is particularly important in the first stages of ecosystem development because it is found that, in this period, the impact of different plant communities on the potential for C sequestration is manifested primarily in the biomass and the upper organic horizons of the soils [31,34].

In this work, we aimed to contribute to the discussion on the usefulness of natural regeneration by spontaneous succession versus technical reclamation using topsoiling and planting trees on a post-mining site in the context of biodiversity versus C sequestration potential as the crucial ecosystem services of restored post-mining sites. We hypothesized that tree communities from spontaneous succession on carboniferous substrates in the first phase of ecosystem development would result in a similar soil C storage potential and biodiversity as full reclamation treatments and tree plantation.

2. Materials and Methods

2.1. Study Site

The study was carried out on the “Sośnica” spoil heap after hard-coal mining in Zabrze and Gliwice (50°16′22″ N, 18°44′43″ E), Upper Silesia, Poland. The climate of the area is temperate, with a mean annual precipitation of 737 mm and annual temperature of 8.8 °C (data for 1990–2022 from the Katowice meteorological station, source: www.tutiempo.net, accessed 15 May 2023). The heap had an area of approximately 170 ha and a height of more than 30 m. It comprised carboniferous rocks, which were primarily shale, sandstone, and conglomerates made from these [35].

Our study involved two types of substrates: the bare carboniferous rock (BR) and the bare rock covered by approximately 50 cm of topsoil (TS). Research plots were established with four variants: (i) in communities with succession dominated by *Betula pendula*, *Populus tremula*, and *Pinus sylvestris* on BR (S-BR); (ii) in communities with succession dominated by *Populus tremula* on TS (S-TS); (iii) at sites with *Robinia pseudoacacia* from planting on TS (Rb-TS); and (iv) with a mixture of *Betula pendula* and *Alnus glutinosa* from planting on TS (Re-TS). Soils in the S-BR variant were characterized by a higher sand fraction than the S-TS soils at 0–10 cm depth and a higher clay fraction than all the other variants at 10–30 cm depth. A higher silt fraction was observed in the S-TS than in the S-BR variant at 0–10 cm depth. The S-BR soils had lower pH values than the others at both depths. At 0–10 cm depth, the bulk density (BD) was higher in the S-TS than the S-BR and Re-TS soils, while at 10–30 cm depth, the BD was higher in the Re-TS than the Rb-TS and S-BR soils (Table 1). The stand age was 10–15 years. In total, 16 research plots, with four replications of each variant of 10 × 10 m, were randomly established on the identified experimental patches on the spoil heap.

Table 1. Site characteristics.

Variant		Woodland from Succession on Bare Carboniferous Rock	Woodland from Succession in Site after Topsoil Application	Woodland with <i>Robinia pseudoacacia</i> from Planting in Site after Topsoil Application	Woodland with a Mixture of Different Tree Species on Site after Topsoil Application
		S-BR	S-TS	Rb-TS	Re-TS
Dominant tree species		<i>Betula pendula</i> , <i>Populus tremula</i> , <i>Pinus sylvestris</i>	<i>Populus tremula</i>	<i>Robinia pseudoacacia</i>	<i>Betula pendula</i> , <i>Alnus glutinosa</i>
Admixture		<i>Populus nigra</i> , <i>Populus hybrid</i> ‘Androscoggin’	<i>Populus nigra</i> , <i>Padus serotina</i> , <i>Betula pendula</i>	—	<i>Salix alba</i> , <i>Robinia pseudoacacia</i>
Basic soil parameter	Soil layer	Value (mean ± SE)			
Sand (2–0.05 mm) [%]	0–10 cm	67 ± 3	41 ± 1	55 ± 7	47 ± 8
	10–30 cm	42 ± 4	43 ± 10	44 ± 5	43 ± 5
Silt (0.05–0.002 mm) [%]	0–10 cm	26 ± 2	48 ± 2	38 ± 7	44 ± 7
	10–30 cm	44 ± 4	47 ± 8	48 ± 5	47 ± 5
Clay (<0.002 mm) [%]	0–10 cm	8 ± 1	11 ± 1	7 ± 1	9 ± 1
	10–30 cm	15 ± 0	10 ± 2	9 ± 1	9 ± 0
pH	0–10 cm	5.3 ± 0.2	6.9 ± 0.3	7.2 ± 0.1	6.8 ± 0.1
	10–30 cm	5.6 ± 0.3	7.0 ± 0.4	7.6 ± 0.1	7.6 ± 0.1
BD ¹ [g cm ^{−3}]	0–10 cm	1.20 ± 0.06	1.59 ± 0.06	1.40 ± 0.04	1.22 ± 0.10
	10–30 cm	1.17 ± 0.07	1.51 ± 0.06	1.38 ± 0.05	1.71 ± 0.11

¹ BD = bulk density.

2.2. Soil Sampling and Analysis

Samples from the organic (Oi + Oe) and mineral (0–10 and 10–30 cm depth) horizons were collected from each plot in October 2021. A composite soil sample representing each plot was collected from five subsamples (four from the corners and one from the middle of each plot). For the BD calculation, two independent samples from depths of 0–10 cm and 10–30 cm per plot were taken from the center, keeping their structures intact via collection in 100 cm³ cylinders. The organic horizon (Oi + Oe) samples were collected from five 20 × 20 cm squares in each plot. Each sample of the litter layer in the fresh state was weighed using an laboratory scale, and the mixed samples were considered representative of each test area.

For the mineral layer samples (0–10 cm and 10–30 cm depth), the following parameters were determined: the granulometric composition (texture) was determined using a Fritsch GmbH laser particle sizer ANALYSETTE 22; the pH was determined potentiometrically in water at a 1:2.5 soil–solution ratio; and the total organic C contents were measured using a LECO TruMac[®] CNS. The intact samples collected in cylinders were sieved (2-mm mesh size), weighed, dried at 105 °C, and reweighed. The weight was used to calculate the dry weight of the original sample, which was then divided by the volume to obtain the BD of the fine fraction (<2 mm). The organic horizon samples (Oi + Oe) were oven-dried, and the measured moisture loss was used to calculate the dry mass. The dried samples were grounded, and the C content was measured using a LECO TruMac[®] CNS, with the pH determined potentiometrically in water at a 1:5 ratio.

2.3. Biomass Study

For the aboveground understory vegetation (forest floor herbaceous plants + shrubs) biomass analysis, samples were taken from 1 m² subplots. In the laboratory, the biomass was oven-dried, weighed, and ground. The measured loss of moisture was used to calculate the dry mass.

Tree measurements on each plot were conducted by measuring the tree diameter at 1.3 m (Dbh) and then the height. The most common method of determining the biomass of various tree components is to use empirical allometric equations, in which simple-to-measure tree characteristics, such as Dbh and height, are used as predictors [36,37]. The equations need to be developed for each specific population based on a random sample of the trees destructively separated into individual components, which is very expensive and labor-intensive. Because of these inconveniences, biomass estimates are often obtained from equations developed for similar populations. However, it should be noted that the key to the selection of existing equations is to take into account the similarity of the populations; otherwise, systematic errors in the biomass estimates may occur [38,39].

Here, the biomass of the aboveground tree components (i.e., stems, branches, and leaves) was estimated based on equations already published in the literature [40–47]. To minimize the risk of bias, we used equations built for populations of trees growing under similar growth conditions. Thus, the choice of equations considered various tree and site characteristics, such as tree species, age, Dbh and height range, geographical region, and habitat type, so that they corresponded as closely as possible to those observed in the present study. In addition, we chose equations that were developed using a large sample and preferred those that used two predictors—Dbh and height. The aboveground biomass of the tree components (stems, branches, leaves) for each species was determined based on the equations summarized in Table 2. The belowground (i.e., roots) biomass was estimated as 25% of the total biomass [32].

Table 2. Equations used to determine the tree biomass fractions [40–47].

Species Source	Biomass Fraction ¹	Form of Equation	Parameter				
			b_0	b_1	b_2	b_3	b_4
<i>Robinia pseudoacacia</i> [40]	ST	$y = e^{b_0} \cdot D^{b_1} \cdot CF / 1000$	3.6628	2.3732	1.0733^{*2}	—	—
	B	$y = e^{b_0} \cdot D^{b_1} \cdot CF / 1000$	3.481	2.0551	1.1387^*	—	—
	FL	$y = e^{b_0} \cdot D^{b_1} \cdot CF / 1000$	4.3429	1.1018	1.0814^*	—	—
	AB	$y = e^{b_0} \cdot D^{b_1} \cdot CF / 1000$	4.2568	2.2698	1.0913^*	—	—
<i>Betula pendula</i> [41]	ST	$y = b_0 \cdot D^{b_1} \cdot H^{b_2}$	0.02606	1.70530	1.16391	—	—
	B	$y = b_0 + b_1 D^2 \cdot b_2 H$	0.12882	0.03762	−0.04669	—	—
	FL	$y = b_0 + b_1 D^2 \cdot b_2 H$	0.06611	0.01182	−0.01331	—	—
	AB	$y = \text{Stem} + \text{Branch} + \text{Foliage}$	—	—	—	—	—
<i>Alnus glutinosa</i> [42]	ST	$y = b_0 \cdot (D \cdot 10)^{b_1}$	0.00119	2.17247	—	—	—
	B	$y = b_0 \cdot (D \cdot 10)^{b_1}$	0.0000006	3.28106	—	—	—
	FL	$y = b_0 \cdot (D \cdot 10)^{b_1}$	0.00239	1.32553	—	—	—
	AB	$y = b_0 \cdot (D \cdot 10)^{b_1}$	0.00079	2.28546	—	—	—
<i>Pinus sylvestris</i> [43]	ST	$y = b_0 \cdot (D \cdot 10)^{b_1} \cdot H^{b_2} + b_3 \cdot (D \cdot 10)^{b_4}$	0.00041	1.627725	1.390374	0.000192	2.11719
	B	$y = b_0 \cdot (D \cdot 10)^{b_1} \cdot H^{b_2}$	0.0000038	3.653659	−1.6008	—	—
	FL	$y = b_0 \cdot (D \cdot 10)^{b_1} \cdot H^{b_2}$	0.000212	2.30978	−0.58099	—	—
	AB	$y = \text{Stem} + \text{Branch} + \text{Foliage}$	—	—	—	—	—
<i>Salix alba</i> [44]	ST	$y = e^{b_0} \cdot D^{b_1} \cdot CF / 1000$	2.8992	2.6728	1.2119^*	—	—
	B	$y = e^{b_0} \cdot D^{b_1} \cdot CF / 1000$	2.6787	2.6347	1.3421^*	—	—
	FL	$y = e^{b_0} \cdot D^{b_1} \cdot CF / 1000$	0.3882	3.3431	1.8357^*	—	—
	AB	$y = \text{Stem} + \text{Branch} + \text{Foliage}$	—	—	—	—	—
<i>Populus tremula</i> [45]	ST	$y = b_0 \cdot (D \cdot 10)^{b_1}$	0.000065	2.603533	—	—	—
	B	$y = b_0 \cdot (D \cdot 10)^{b_1}$	0.000515	1.873298	—	—	—
	FL	$y = b_0 \cdot (D \cdot 10)^{b_1}$	0.000847	1.481578	—	—	—
	AB	$y = b_0 \cdot (D \cdot 10)^{b_1}$	0.000146	2.603533	—	—	—
<i>Populus nigra</i> [40]	ST	$y = b_0 \cdot (D \cdot 10)^{b_1}$	0.000065	2.603533	—	—	—
	B	$y = b_0 \cdot (D \cdot 10)^{b_1}$	0.000515	1.873298	—	—	—
	FL	$y = b_0 \cdot (D \cdot 10)^{b_1}$	0.000847	1.481578	—	—	—
	AB	$y = b_0 \cdot (D \cdot 10)^{b_1}$	0.000146	2.603533	—	—	—
<i>Populus hybrid Androscoggin</i> [46]	ST	$y = e^{b_0 + b_1 \cdot \log(D)}$	−2.01	2.14	—	—	—
	B	$y = e^{b_0 + b_1 \cdot \log(D)}$	−5.22	2.76	—	—	—
	FL	—	—	—	—	—	—
	AB	$y = e^{b_0 + b_1 \cdot \log(D)}$	−2.14	2.26	—	—	—

Table 2. Cont.

Species Source	Biomass Fraction ¹	Form of Equation	Parameter				
			b_0	b_1	b_2	b_3	b_4
<i>Padus serotina</i> [47]	ST	$y = b_0 + b_1 \cdot D + b_2 \cdot D^2 + b_3 \cdot D^3$	−95.36	18.78	−0.84	0.02	—
	B	$y = b_0 + b_1 \cdot D^3$	0.04	0.01	—	—	—
	FL	$y = b_0 \cdot D^{b_1}$	0.049727	1.868954	—	—	—
	AB	$y = e^{b_0 + b_1 \cdot \log(D) + b_2 \cdot \log(H)} \cdot CF$	−2.56	2.13	0.5	1.008 *	—

¹ ST = stems; B = branches; FL = foliage; AB = total aboveground biomass; and CF = correction factor value. ² the value marked with “*” equals the correction factor (CF).

2.4. Plant Diversity

Indicators of the plant biodiversity in the study plots with introduced trees and spontaneously encroaching trees were based on the composition of the communities using the number of plant species in the study plot for each taxon separately. The number of species and the Shannon–Wiener index (H' index) were used to represent the alpha diversity [20], the latter calculated using the species abundance data using Kovach Computing Services' MVSP 3.13.p. software.

The H' index was calculated as follows:

$$H = -\sum p_i \times \ln(p_i) \quad (1)$$

where p_i is the proportion of the entire community made up of species i .

2.5. Statistical Analysis

The datasets were statistically analyzed using Statistica 13.1 software. The statistical significance between the tested variants was determined using the non-parametric Kruskal–Wallis test for several groups at $p < 0.05$ [48].

3. Results

3.1. Vegetation Biomass

In the Rb-TS variant, there was a higher herbaceous forest floor plant biomass compared to S-BR and S-TS. Higher shrub biomass was also observed in Re-TS compared to Rb-TS. No differences were found among the variants in tree density, Dbh, height, stem biomass, branch biomass, aboveground or belowground tree biomass, and total biomass (Table 3).

3.2. Plant Biodiversity

The results of the analysis showed no statistically significant differences between the number of species on the study plot variants. The average number of species was between 12.11 (Rb-TS) and 18.11 (S-TS; Figure 1).

The average H' index value was higher for the phytocoenoses on the BR (1.99) than for the TS variants (1.80) and was different and statistically significant. The average H' index value was different in the TS group, where Rb-TS stood out as being statistically significant with dominant *Robinia pseudoacacia* (Figure 2).

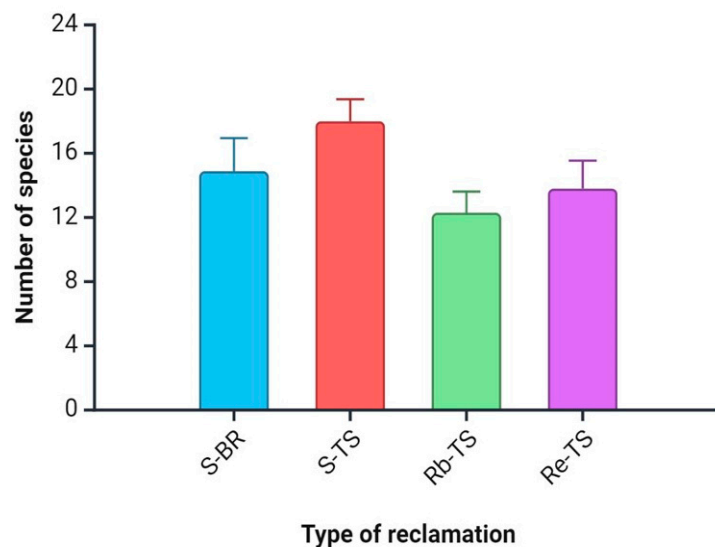


Figure 1. Differences between the number of species on study sites. Explanation: S-BR = succession on bare carboniferous rock; S-TS = succession on topsoil; Rb-TS = *Robinia* from planting on topsoil; and Re = the mixture of tree species from planting on topsoil. Values are means $\pm$ SE for each parameter and each variant.

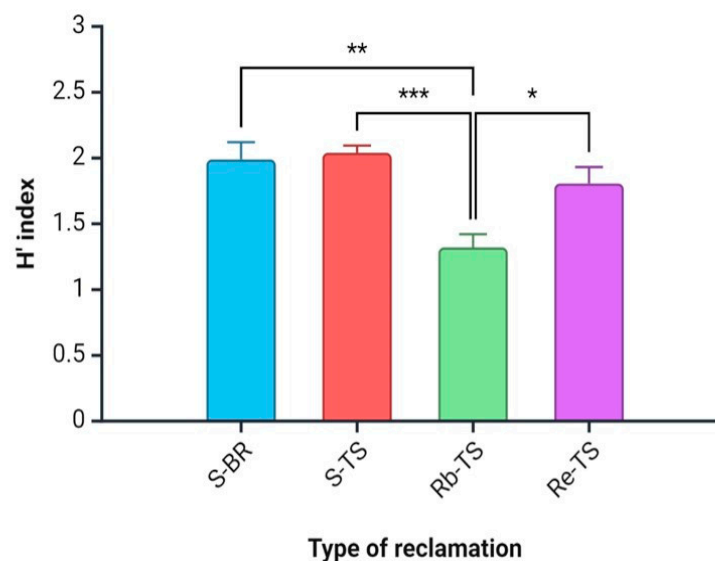


Figure 2. Differences between H' index values of the study sites. Explanation: see Figure 1. Values are means $\pm$ SE for each parameter and each variant. Explanations: *— $p < 0.05$, **— $p < 0.01$, ***— $p < 0.001$.

3.3. Carbon Storage in Ecosystems

The C stock in the Oi + Oe horizon was higher in the S-BR than in the S-TS. The C stock was also higher at both 0–10 cm and 10–30 cm under S-BR compared to all other variants, contributing to a significantly higher total soil C stock in the S-BR variant compared to all other variants. The Rb-TS variant had a higher soil C stock at 0–10 cm and a higher total soil C stock compared to S-TS (Table 4).

Regarding the C stock in the biomass, Rb-TS had a higher forest floor plant C stock than S-BR and S-TS, whereas Re-TS had a significantly higher shrub biomass C stock than Rb-TS. However, there were no differences in the tree aboveground, tree root, tree, or total biomass C stocks among the variants. The ecosystem C stock was the highest under S-BR compared to the other variants, with Rb-TS also exhibiting a higher ecosystem C stock than S-TS (Table 4).

Table 3. Tree parameters and plant biomass on spoil heaps after hard-coal mining.

Variant ¹	Forest Floor Plant Biomass [Mg ha ^{−1}]	Shrub Biomass [Mg ha ^{−1}]	Tree Density [pcs ha ^{−1}]	Dbh [cm]	Height [m]	Stem Biomass [Mg ha ^{−1}]	Branch Biomass [Mg ha ^{−1}]	Foliage Biomass [Mg ha ^{−1}]	Aboveground Biomass [Mg ha ^{−1}]	Belowground Biomass [Mg ha ^{−1}]	Tree Biomass [Mg ha ^{−1}]	Total Biomass [Mg ha ^{−1}] ³
S-BR	0.38 ± 0.14 ^{a2}	0.75 ± 0.50 ^{ab}	800 ± 122 ^a	10.74 ± 0.76 ^a	9.49 ± 0.58 ^a	18.91 ± 5.45 ^a	3.95 ± 1.26 ^a	1.43 ± 0.63 ^a	31.14 ± 7.37 ^a	7.79 ± 1.84 ^a	38.93 ± 9.21 ^a	40.06 ± 9.29 ^a
S-TS	0.33 ± 0.04 ^a	0.68 ± 0.26 ^{ab}	1050 ± 65 ^a	10.21 ± 0.51 ^a	10.63 ± 0.31 ^a	13.58 ± 1.96 ^a	3.38 ± 0.39 ^a	0.91 ± 0.10 ^a	30.18 ± 4.66 ^a	7.55 ± 1.17 ^a	37.73 ± 5.83 ^a	38.74 ± 5.92 ^a
Rb-TS	2.34 ± 0.43 ^b	0.15 ± 0.15 ^a	1250 ± 253 ^a	10.10 ± 0.78 ^a	9.71 ± 0.42 ^a	13.99 ± 3.76 ^a	5.72 ± 1.45 ^a	1.33 ± 0.29 ^a	20.03 ± 5.28 ^a	5.01 ± 1.32 ^a	25.03 ± 6.60 ^a	27.52 ± 6.33 ^a
Re-TS	1.63 ± 0.68 ^{ab}	3.90 ± 1.63 ^b	1450 ± 312 ^a	8.78 ± 0.53 ^a	9.15 ± 0.56 ^a	25.10 ± 4.06 ^a	3.71 ± 0.85 ^a	1.66 ± 0.39 ^a	30.19 ± 5.24 ^a	7.55 ± 1.31 ^a	37.74 ± 6.54 ^a	43.27 ± 8.29 ^a

¹ Variants: S-BR = succession on bare Carboniferous rock; S-TS = succession on topsoil; Rb-TS = *Robinia* from planting on topsoil; and Re = mixture of tree species from planting on topsoil. ² Values are means ± SE for each parameter and each variant. Means with different letters are significantly different at $p < 0.05$. ³ Total biomass included forest floor plant, shrubs, and tree biomass.

Table 4. C storage in ecosystem components (soil and biomass) on studied spoil heap after coal mining.

Variant ¹	C Stock in Soil [Mg ha ^{−1}]				C Stock in Biomass [Mg ha ^{−1}]						Ecosystem C Stock [Mg ha ^{−1}]
	Litter	0–10 cm	10–30 cm	Total Soil	Forest Floor Plants	Shrubs	Tree Aboveground	Tree Roots	Trees	Total Biomass ²	
S-BR	5.91 ± 0.76 ^{b2}	149.97 ± 17.29 ^b	351.77 ± 50.74 ^b	507.66 ± 60.32 ^b	0.16 ± 0.06 ^a	0.35 ± 0.24 ^{ab}	11.31 ± 3.31 ^a	3.65 ± 0.88 ^a	14.97 ± 4.14 ^a	15.48 ± 4.24 ^a	523.14 ± 62.75 ^c
S-TS	3.07 ± 0.38 ^a	26.10 ± 1.64 ^a	26.88 ± 7.02 ^a	56.05 ± 8.40 ^a	0.13 ± 0.02 ^a	0.33 ± 0.13 ^{ab}	8.21 ± 1.13 ^a	3.42 ± 0.54 ^a	11.63 ± 1.67 ^a	12.09 ± 1.72 ^a	68.13 ± 9.07 ^a
Rb-TS	5.31 ± 0.55 ^{ab}	58.76 ± 17.79 ^a	44.13 ± 12.54 ^a	108.19 ± 17.29 ^a	0.97 ± 0.18 ^b	0.07 ± 0.07 ^a	9.66 ± 2.56 ^a	2.20 ± 0.60 ^a	11.86 ± 3.15 ^a	12.89 ± 3.05 ^a	121.08 ± 16.15 ^b
Re-TS	4.59 ± 0.52 ^{ab}	35.39 ± 4.10 ^a	34.78 ± 2.25 ^a	74.76 ± 3.75 ^a	0.68 ± 0.28 ^{ab}	1.80 ± 0.76 ^b	14.17 ± 2.39 ^a	3.50 ± 0.60 ^a	17.67 ± 2.99 ^a	20.15 ± 3.80 ^a	94.90 ± 2.58 ^{ab}

¹ Explanation: see Table 3. ² Total biomass included forest floor plants, shrubs, and tree biomass. Values are means ± SE for each parameter and each variant. Means with different letters are significantly different at $p < 0.05$.

4. Discussion

The much higher C storage value in the soil in the S-BR variant compared to the variants with topsoiling resulted from the presence of geogenic (hard-coal) C in the carboniferous rocks [3]. Geogenic C affects the overall soil C balance in the initial ecosystem and is also important for soil water retention [49,50]. However, the process of its incorporation into the biogeochemical cycle has not been thoroughly studied, although it is known that soil microorganisms play a key role in the process, especially in the case of lignite [49–51]. Determining the amount of geogenic C in the total organic C pool in post-mining soils is difficult from a methodological point of view [52]. The content of fossil C has been determined based on the degree of ^{14}C radioactive isotope decay [51] and via formulas that take into account specific post-mining site indicators [53]. Despite the higher C stock in the BR, the literature indicates that soils reclaimed through topsoil application are often characterized by a higher rate of C accumulation (per year) in the soils [54]. This is because the geogenic C may be subject to gradual decomposition, which may be reflected in a decrease in C storage in the soil [55]. Another factor contributing to lower C accumulation in soils developing from carboniferous rock BR may be the so-called soil C saturation [56]. Due to the geogenic C content, it was difficult to compare the C storage in the mineral layers between the BR and TS variants. Reliable results have been obtained by comparing the C stocks in the organic horizons [3], and the BR and TS variants did indeed have similar values in these horizons. However, the soil was tested down to 30 cm, and under the topsoiling, there was a layer of carboniferous waste. Hence, if we had analyzed deeper layers, the amount of C storage might have been similar in both habitat variants.

In sites with topsoiling, no significant differences were found between different types of vegetation. A period of only 10 to 15 years since reclamation seems to be too short to reveal the impact of the various vegetation on C storage in the mineral layers. However, when comparing the values obtained from the shallower (0–10 cm) to deeper (10–30 cm) layers, it can be concluded that the Rb-TS variant is characterized by a higher potential for soil C accumulation compared to the other variants. Nitrogen-fixing species, such as *Robinia pseudoacacia*, in addition to increasing the N content, also increase the C content in soils [57,58]. This is due to the high decomposition rate of N-fixing species in the litter, which results in low-mass organic horizons and an increase in the amount of organic C reaching the mineral soil [59]. The high N content in the litter increases the decomposition rate during the initial phase while inhibiting the decomposition of the humified soil C [60].

The soil C storage for the sites with topsoiling was within the range given for natural soils under a temperate climate, as well as for post-mining soils. Data from the National Forest Soil Inventory indicate that the C storage at locations throughout Poland has been, on average, 26 Mg ha^{-1} in the organic horizons and 66 Mg ha^{-1} in the mineral horizons of the soils (up to 1 m deep) [61]. These values are similar in the organic horizons to those reported in the applicable European forest literature but lower in the mineral horizons (22.1 and 108 Mg C ha^{-1} , respectively) [60,62]. Vinduřková and Frouz [63] reported that the C stock in post-mining soils in northern temperate climate zones can range from 4.49 to 93.20 Mg ha^{-1} . Pietrzykowski and Daniels [3] found soil C storage values of 16.8 to 65.0 Mg ha^{-1} at various post-mining sites in Poland (i.e., sites after lignite, sulfur, and sand mining). Similarly to our findings, much higher values (from 1959.1 to $2975.2 \text{ Mg ha}^{-1}$) were obtained from sites after hard-coal mining [3].

Our results do not support statements concerning the benefits of reclamation and the afforestation of post-mining sites, including higher biomass production and C storage in the tree biomass being greater in reclaimed sites than in those left to succession [64,65]. However, Frouz et al. [33] reported higher woody biomass only in younger stands (<20 years) on reclaimed sites compared to sites left to natural regeneration, although the difference disappeared in older stands [33]. Plant communities from succession are often heterogeneous and significantly different from each other, even over small areas [31]. This observation may explain the differences reported in the literature regarding the growth parameters and biomass achieved by stands from succession and afforestation [66]. For example, in

North America, when succession occurs under favorable conditions, some fast-growing timber trees may grow to harvestable size as early as 30 to 40 years after disturbance, while slower-growing hardwoods may require 50 to 60 years or longer. Other sites may remain in the grass–herb–shrub stage, with only scattered trees, for several decades after a disturbance because the soil conditions are unsuitable or the understory vegetation is too competitive for tree growth [66].

Tree stands arising from the process of spontaneous succession (BR) or reclamation through topsoiling (TS) are a priority option for the sustainable forest management of mine spoil heaps because they can provide a wide range of ecosystem services, such as the control of soil erosion, increased biomass and C pool, and restored biodiversity [67]. Our obtained results indicate that biodiversity does not depend on the method of reclamation, which translates into key ecosystem functions, such as CO₂ storage and tree productivity [68]. However, this may be due to having access to similar propagules and the extent of the environmental adaptation of the plants to the environmental conditions of spoil heaps [18].

5. Conclusions

Our study has demonstrated the high biodiversity and C sequestration potential of vegetation from succession on a BR substrate. Thus, the involvement of this process in the ecosystem restoration of post-mining sites after hard-coal mining is crucial for subsequent forest management. The alpha and beta biodiversities of the plant communities were similar for the reclaimed and non-reclaimed sites. Unpredictable woodlands from succession and woodlands from planting were characterized by similar C storage values in the total tree biomass. The estimated potential for total C storage (soil + biomass) at the sites with topsoiling ranged from 68.13 to 121.08 Mg ha^{−1}, of which 56.05–108.19 Mg ha^{−1} was stored in the soil and 12.09–20.15 Mg ha^{−1} in the biomass. In non-reclaimed sites on carboniferous rock BR, the total C storage was much higher, amounting to 523.14 Mg ha^{−1} (of which 507.66 Mg ha^{−1} was in the soil), which was due to the geogenic coal content in the carboniferous rock BR. However, the C storage in the biomass (15.48 Mg ha^{−1}) and litter (5.91 Mg ha^{−1}) was similar to the results obtained from the reclaimed sites.

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