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Effects of Sand Mining on Species Diversity, Stand Structure and Carbon Stocks in a Tropical Forest in Ghana: Implications for Biodiversity Conservation and Ecosystem Services

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ABSTRACT

The extractive industry, particularly sand mining, underpins infrastructure and socioeconomic development in many tropical countries but poses severe ecological risks when poorly regulated. This study quantified the long-term effects of sand extraction on plant diversity and conservation status, stand structure and carbon stocks at a 30-year abandoned sand mine site within a tropical forest in Ghana. The site was stratified by extraction intensity into sand mine core (SMC), sand mine buffer (SMB) and non sand mine (NSM) areas. Nested plots (400 m^2) were randomly established across the sites to collect data on plant species and tree biomass. The results showed that species richness declined significantly from 81 species in the NSM to 45 in the SMC. Stand-structural attributes (height, basal area and density) collapsed to zero with a corresponding carbon stock of 00 ± 00 (tC/ha) in the SMC compared with 69.89 ± 16.06 (tC/ha) in the NSM. Multivariate analysis revealed the near-total reassembly of plant community in mined areas. Nonmetric multidimensional scaling (nMDS) of species importance value index (IVI) clearly separated the NSM from mined sites, with Bray–Curtis dissimilarity highest between NSM and SMC (94.07%), but relatively similar between SMB and SMC (50.99%). SIMPER analysis identified the replacement of sensitive native species at the NSM (e.g., *Mallotus oppositifolius,* IVI: 54.09) by resilient grasses at the SMC (e.g., *Imperata cylindrica,* IVI: 58.26). Conservation status assessments using both localised star rating and IUCN categorisation confirmed significant losses of high-priority conservation species in mined areas. This study demonstrates that unregulated sand mining drives irreversible biodiversity loss and profound shifts in the community structure and severely impairs carbon sequestration. We argue that integrating sustainable sand mining practices and restoration targets into national land-use policies is critical to safeguard tropical forests' biodiversity and their ecosystem services.

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1 | Introduction

Tropical forests constitute one of the most important repositories of biodiversity and are vital for maintaining global ecological equilibrium, regulating climate and supporting human well-being and livelihoods [1, 2], but they are continually threatened by unsustainable human activities. In Ghana, forests occupy approximately 35% of the land area and are home to distinct biodiversity, including rare, endemic or threatened species of both flora and fauna [3]. They also provide ecosystem services such as carbon sequestration, hydrological regulation and soil stabilisation [4]. Nevertheless, these ecosystems face growing threats from anthropogenic pressures, including land-use changes, deforestation, agricultural expansion and mining [5–7]. Among these threats, sand mining has emerged as a prevalent [8] driver of environmental degradation, particularly in forests and arable lands.

Sand mining is the process of extracting sand and gravel from riverbeds, floodplains, coastal areas and other places where these materials naturally accumulate [9, 10]. This activity has grown quickly over the past few decades, mainly due to increased global demand for construction materials. According to the United Nations Environment Programme (UNEP), globally, about 50 billion tons of sand is extracted each year [5], making sand the second most heavily exploited natural resource after water [11]. The construction sector accounts for much of this demand since sand is essential for making concrete and supporting large infrastructure projects [12]. Besides construction, sand is used in various industrial processes, such as glass production and other specialised uses [13, 14]. The industry also brings economic benefits by generating both direct and indirect jobs in many areas [15].

Despite the socioeconomic importance of sand mining, there are significant environmental impacts [16, 17], particularly where the degree of exploitation far exceeds the rate at which sand is replenished [5]. The negative environmental effects of sand extraction have become a global concern, especially in tropical developing regions where regulatory frameworks are often weak or poorly enforced [11, 18]. Sand mining has been linked to riverbank erosion, vegetation destruction, habitat degradation and biodiversity loss, all of which have serious consequences for ecosystem health [8, 18]. The effects of sand mining can be particularly severe in tropical forests, which are both highly biodiverse and sensitive to disturbances. For instance, studies in Brazil and India have documented significant losses of plant species and shifts in community composition due to mining activities encroaching on forested areas [19, 20].

In Africa, sand mining is increasingly recognised as an environmental challenge, driven by rapid urbanisation and infrastructure expansion [16]. Due to unregulated sand mining activities, significant ecological damage has been reported in South Africa, Kenya and Nigeria. The environmental repercussions include impacts on land, air quality, water quality and availability, as well as a reduction in arable land [21–23]. In many cases, sand mining occurs informally, showing little regard for environmental legislation or sustainable practices, leading to conflicts and resource overexploitation [24].

In Ghana, sand mining is prevalent in regions like Greater Accra and Ashanti, where it has been associated with reduced soil fertility, extensive loss of agricultural land and deforestation [25]. In particular, sand mining negatively impacts forest ecosystems due to their complex structure and reliance on stable soil and hydrological conditions [26]. The removal of tree cover, destruction of vegetation structure and topsoil affect the abundance and tree species composition [27] and also disrupt microclimates, compact the soil structure, deplete nutrients, disrupt plant community dynamics and adversely affect species diversity [28]. Such changes can disrupt the balance of plant communities by decreasing species abundance and changing structural attributes, such as tree density, canopy cover and understory composition [29]. Furthermore, the disturbances can selectively eliminate sensitive species while promoting disturbance-tolerant or invasive ones, thereby reducing overall species richness and altering forest dynamics [30, 31]. Concurrently, the removal of mature trees, which serve as primary carbon sinks, diminishes the carbon storage potential of these ecosystems [32].

Despite these evident threats posed by sand mining, there is a paucity of research on its impacts on species composition, diversity stand structure and ecosystem services of forest ecosystems [5, 33]. Locally, existing studies such as Mensah [34], Jonah et al. [35], Nasare et al. [26] and Amoah et al. [36] have primarily focused on sand extraction in riverbeds and coastal areas and its impacts on water quality and riparian vegetation, while the effects on forest ecosystems remain largely overlooked. Understanding the effects of sand mining on plant species diversity and structure is critical for biodiversity conservation and environmental sustainability. This is particularly relevant in light of the nation's commitments to climate change mitigation and biodiversity conservation under frameworks such as the Rio Conventions on the interconnected challenges of climate change, desertification and biodiversity loss, the Paris Agreement and the United Nations Sustainable Development Goals (e.g., SDGs 11, 13 and 15). This knowledge is also essential for policy development for an integrated management of natural resources that balances economic growth with ecological integrity, ensuring sustainability and long-term health of forests in off-reserve areas.

In light of these concerns, this study aims to address this gap by investigating the ecological impact of sand mining at varying intensities on plant communities in a tropical semideciduous forest ecosystem in Ghana. By comparing sand-mined areas with unmined adjacent forest site, the research specifically (i) assessed changes in plant species composition, richness and diversity; (ii) analysed patterns in community structure and spatial distribution of regenerating species, and identified key species sensitive or resilient to sand mining disturbance; (iii) evaluated the effects of sand mining on stand-structural attributes (height, basal area and density) and carbon stocks; and (iv) assessed the implications of sand mining on the conservation status of plant species. The study findings will contribute to a deeper understanding of the ecological consequences of sand mining and inform land-use policy and practices in off-reserves.

2 | Materials and Methods

2.1 | Study Area Description

The study was undertaken at a 30-year-old abandoned sand mine site (which was a site of prevalent sand mining activities) proximate to the Kwame Nkrumah University of Science and Technology (KNUST) Research station at Anwomaso, Ejisu-

Juaben Municipality, in the Ashanti Region of Ghana (Figure 1). The Ejisu-Juaben Municipality lies within latitudes 1° 15′ N and 1° 45′ N and longitudes 6° 15′ W and 7° 00′ W, occupying a land area of 582.5 km^2 [37]. The municipality is located in the central part of the Ashanti Region and shares boundaries with six districts: Sekyere East and Afigya Kwabre to the northeast and northwest, respectively, the Bosumtwi and Asante Akim South districts to the South, Asante Akim to the northeast and Kumasi Metropolis to the west [37].

The municipality is situated in the semideciduous forest zone of Ghana, characterised by a bimodal rainfall pattern. The major rainfall period begins from March to July, with an average annual rainfall of 1200–1500 mm per year.

The minor rainfall period also begins in September and tapers off in November, with an average minor annual rainfall of 900 – 1120 mm per year. December to February is usually a dry, hot and dusty period. Mean annual temperatures in the municipality are lowest around 25°C in August and highest around 32°C in March [37]. Relative humidity averages around 65% in the dry season, but it can be quite high (averaging about 85%) during the rainy season. The municipality has diverse flora, including *Terminalia superba, Nesogordonia papaverifera, Triplochiton scleroxylon* and a host of *Entandrophragma* species, as well as diverse fauna species. The topography of the area is generally undulating, dissected by plains and slopes with heights ranging between 240 and 300 m above sea level. The geology of the Municipality consists of Precambrian rocks of the Birimian and Tarkwaian formations, which are generally suitable for agriculture [37].

2.2 | Research Design

This study implemented a Geographic Information System (GIS)–based stratified random sampling design to obtain a spatially balanced, flexible and efficient approach for the inventory of floral species as described by Theobald et al. [38]. The approach involves generating a grid (fishnet) over the study area and selecting random points within these grids to ensure a statistically robust survey design. A preliminary field reconnaissance survey was conducted using a Garmin GPS 76csx to identify and map the different land-use/land cover types in the abandoned sand mine and adjacent nonmined areas (i.e., bare ground/settlements, open forest and agricultural land areas), as identified on the topographic map and to delineate the extent of the sand-mined area. The points were subsequently plotted in ArcGIS (ver. 10.5.1) to establish the boundaries of the sand-mined area.

The sand-mined area, measuring 33.51 ha, was stratified into buffer (21.37 ha) and core zones (12.14 ha) based on sand mining intensity (Table 1). A 30-m sampling grid (fishnet) was overlaid over the sand mine area to divide the area into a series of cells and create a regular pattern that facilitates spatial analysis. Using the 'Create Random Points' tool in Data Management Toolbox, random points were generated within these grids in ArcGIS (Version 10.5.1). Points located along the grid boundaries were excluded, and the remaining random points served as locations for the sampling plots in the sand-mined areas (buffer and core) for the floral species survey.

The procedure was repeated in the adjoining open forest at the nonsand-mined area. Thus, using this approach, the locations of the sampling plots in the three distinct zones—the sand mine core (SMC), the sand mine buffer (SMB) (Figure 2) and the non sand mine (NSM) areas—were determined. The method is particularly useful for vegetation and other natural resource inventories. The integration of GIS tools enhances the spatial balance and representativeness of the sampling plots to maximise species heterogeneity, leading to more accurate and efficient inventories [38].

2.3 | Data Collection Methods

A three-nest plot measuring 20 × 20 m, 5 × 5 m, and 1 × 1 m was employed to assess the floral structure at multiple scales. The 20 × 20 m main plots were established at the exact locations of the random points to enumerate tree species (individuals with a diameter at breast height (DBH) ≥ 10 cm). Four 5 × 5 m subplots were nested within each main plot to assess saplings (individuals with a DBH ≥ 2 cm and < 10 cm), shrubs and climbers. Furthermore, two 1 × 1 m quadrats were laid diagonally within the subplot to evaluate herbaceous vegetation, including individuals < 1 m in height or girth < 2 cm and grasses (Figure 3).

In both the SMC and SMB, there were 20 main plots, 80 subplots and 40 quadrats each, while in the control site (NSM), 10 main plots, 40 subplots and 20 quadrats were laid out due to the relatively high homogeneity of the site. In total, 50 sampling plots (400 m^2), 200 subplots (25 m^2) and 100 quadrats (1 m^2) were distributed across the study area for species enumeration. The plots were appropriately laid out with the help of a compass and measuring tape to eliminate the confounding effects of slope on floristic diversity and composition responses [39].

All vascular plants within each plot were identified to the species level using field guides from the Photo Guide for the Forest Trees of Ghana [40] and with the assistance of a botanist from the Forest Research Institute of Ghana (FORIG). Dead tree species were not considered due to difficulties in identifying them. Plant specimens that could not be identified in the field were collected, labelled, pressed and dried for subsequent identification through comparison with preserved specimens at the herbarium of the Resource Management Support Centre (RMSC), Forestry Commission, Kumasi, Ghana. The DBH of the trees and saplings, including seedlings, was measured using a diameter tape and tree callipers, while the height was determined with a Vertex IV hypsometer and Transponder T3 (Haglöf Sweden AB). The number of individuals per plot was counted, and the ground cover was estimated using the Domin-Krajina cover-abundance scale [41].

2.4 | Conservation Status Assessment of Species

We assessed the conservation status of all recorded plant species using a two-step approach.

Firstly, the species were assessed using the local star rating by Hawthorne and Abu-Juan [42]. This framework ranks species according to their rarity, ecological importance and sensitivity to disturbance and provides an objective approach for establishing priorities for conservation. We assigned each species a star rating according to the published criteria, which range from common to rare or those considered resilient to vulnerable taxa. The definitions of these categories are listed in Table 2 in increasing order of conservation concern.

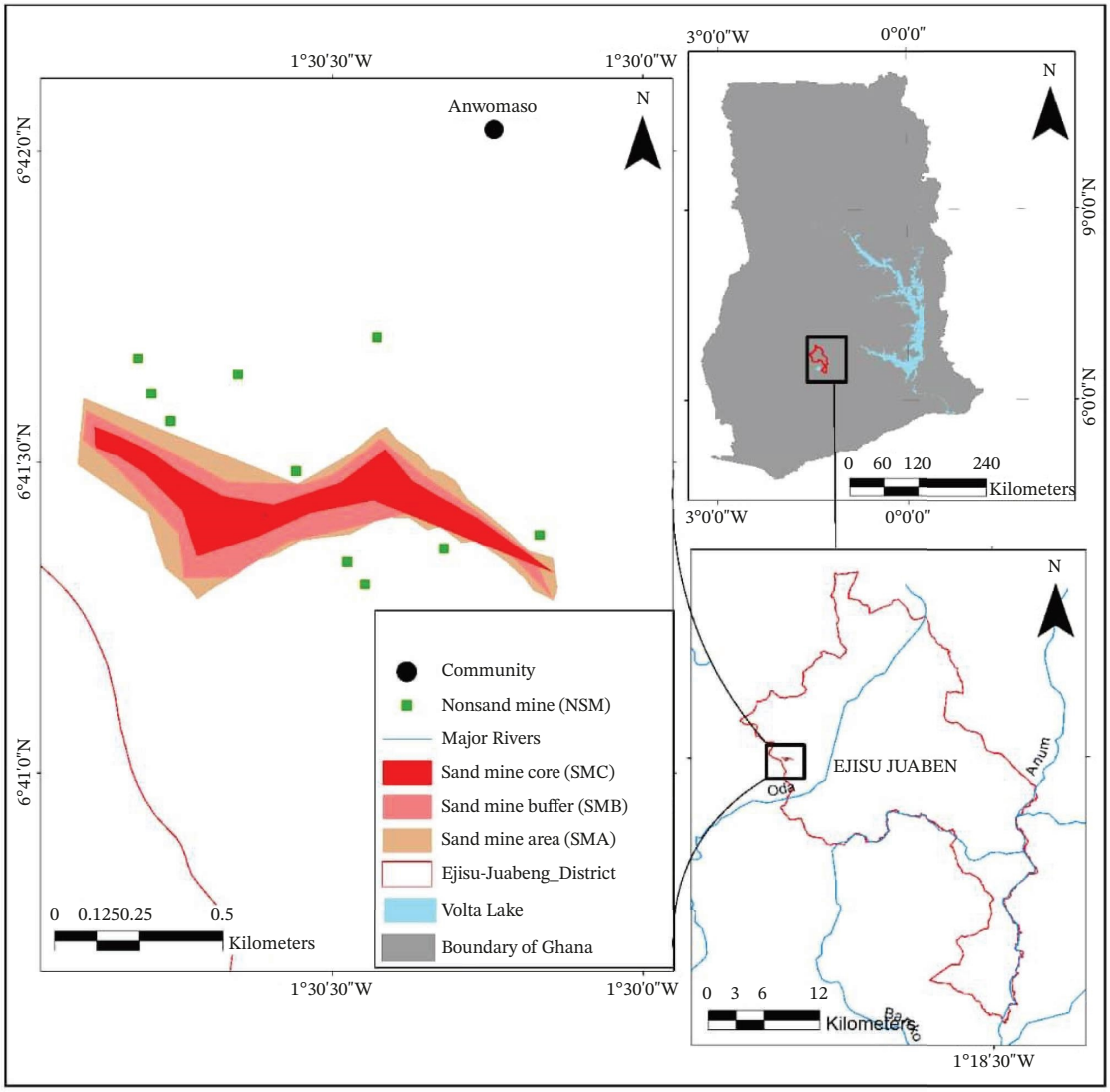


FIGURE 1 | Map of Ghana showing the study area at Anwomaso, Ejisu-Juaben Municipality.

Secondly, after establishing local star ratings by Hawthorne and Abu-Juan [42] for all the taxa recorded, we reassessed their conservation status according to the International Union for Conservation of Nature (IUCN) Red List categories [43] to provide a universal standardised perspective. Each species was classified according to the IUCN categories: least concern (LC) (i.e., species that are widespread and abundant); near threatened (NT) (i.e., species likely to become endangered in the near future); vulnerable (VU) species (i.e., species facing a high risk of endangerment in the wild); endangered (EN) (i.e., species facing a very high risk of extinction in the wild); critically endangered (CR) (i.e., species facing an extremely high risk of extinction in the wild); extinct in the wild (EW) (i.e., species known to only survive in captivity); and not evaluated (NE) (i.e., species that has not yet been evaluated against any criteria) [43]. This dual method approach provided a structured framework for local and internationally recognised conservation priorities.

TABLE 1 | Delineated sand mining sites and their corresponding definitions.

Sand mining sites	Description
Sand mine core (SMC)	Area with the greatest sand mining impact (12.14 ha), the depth of sand removal ranges from 80 to 120 cm, with sparse vegetation and compact soils
Sand mine buffer (SMB)	Area with minimal disturbance from the impact of sand mining activities (21.37 ha). Approximately 30 m from the boundary of the sand mine core area, the depth of sand removal is less than 80 cm, with substantial vegetation cover and less compact soils
Nonsand mine (NSM)	Adjacent open forest with no sand mining activity, but interspersed with annual crop farms to serve as a control experimental site

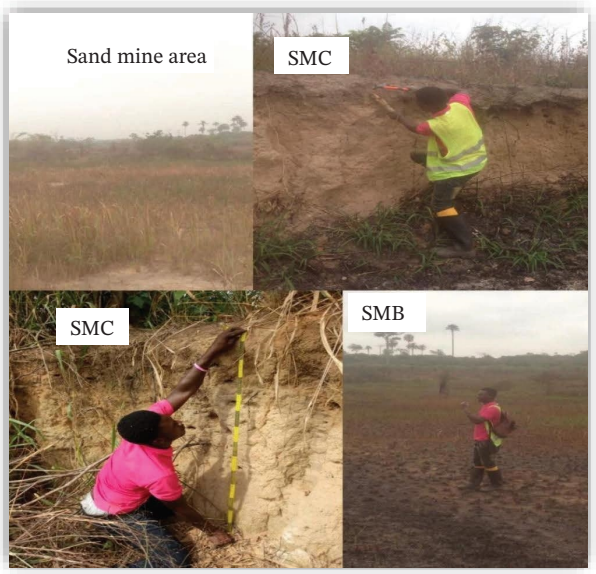


FIGURE 2 | Sand mining area showing sand mine core (SMC) and sand mine buffer (SMB) areas.

2.5 | Data Analysis

An initial Shapiro–Wilk test was carried out to determine the normality or otherwise of the dataset distribution to inform the selection between parametric and nonparametric statistical techniques [44]. Given the non-normal distribution of the dataset ($W = 0.31$, $p < 0.05$, Shapiro–Wilk test), we employed the Kruskal–Wallis *test* to assess significant differences among the study sites. Dunn's *post hoc* multiple comparison test was subsequently used to identify the sites responsible for these significant differences.

2.5.1 | Species Richness and Diversity

Individual-based rarefaction curves were used to evaluate plant species richness. This approach standardises species richness estimations by accounting for different sample sizes or the number of individuals collected per site. In contrast to raw species counts, rarefaction provides a statistically sound method for assessing species richness across sites, particularly when sampling effort or total abundance varies [45, 46]. The rarefaction method estimates the expected number of species $E(S_n)$ in a random subsample of n individuals selected without replacement from a larger sample of N individuals [45]. Species diversity within each site was calculated using the Shannon–Weiner diversity index, which considers species richness and evenness in each habitat type (Table 3).

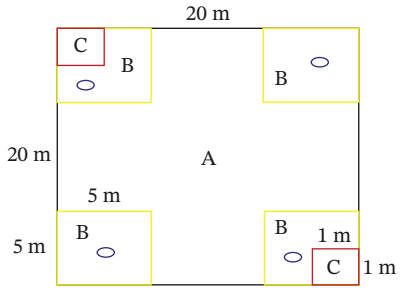


FIGURE 3 | Experimental plot layout using a nested plot design: 20 × 20 m main plot (A) for sampling tree life forms (DBH ≥ 10 cm); 5 × 5 m subplot (B) for saplings, climbers and shrubs; 1 × 1 m quadrat (C) for herbs and grasses.

2.5.2 | Patterns in Community Structure and Spatial Distribution of Regenerating Species and Identification of Key Species Sensitive or Resilient to Sand Mining Disturbance

The importance value index (IVI) for each species was calculated following Curtis and McIntosh [47] to identify ecologically important, resilient and sensitive species across various levels of mining disturbance. The formulae for the respective attributes are presented in Table 3. To test for significant differences in the overall IVI in species distributions among the sites, we first applied the Kruskal–Wallis *test*. However, no *post hoc* pairwise test was performed as the Kruskal–Wallis *test* revealed no significant difference ($p > 0.05$). This nonparametric test was selected because of the violation of the assumptions of normality in the IVI data.

In addition, multivariate analysis was performed to explore community structure reorganisation from the IVI data that the initial univariate test could not capture. Thus, we performed the nonmetric multidimensional scaling (nMDS) ordination based on Bray–Curtis dissimilarity, a robust method to visualise compositional separation among sites [48]. We further applied the similarity percentage (SIMPER) analysis to decompose Bray–Curtis dissimilarity and identify species contributing most strongly to community differences among sites. Heatmaps and hierarchical clustering dendrograms (average linkage Bray–Curtis distance) were generated to visualise species-level patterns of IVI turnover with disturbance. Because our dataset comprises site-level IVI summaries with limited within-group replication, permutation-based multivariate hypothesis tests (e.g., PERMANOVA) were not applied as these required independent replicates per group to produce valid p values.

2.5.3 | Carbon Stock Estimation

The nondestructive method was employed to estimate the aboveground biomass (AGB) and belowground biomass (BGB) of individual tree species listed in the study plots. This technique enables the estimation of tree biomass without felling the tree. It typically involves taking dendrometric measurements of the trees, such as DBH, tree height (H) and wood density (ρ). The dendrometric measurements are then fitted to allometric equations for biomass estimation [49]. The study used the well-developed and widely applied pantropic allometric equation of Chave et al. [50] for moist tropical forest ecosystems to estimate the tree AGB, while the BGB biomass was derived from the AGB using the IPCC [51] equation. This is because the characteristics of the study area are similar to those of the moist tropical woodlands from which this model was developed. Biomass was converted to carbon stock using a carbon fraction coefficient of 0.47 [51] (Table 3).

Although a localised allometric equation developed by Aabeyir et al. [52] exists, this allometric model is best suited for the savannah ecological zones, which was inappropriate for our study site within the moist semideciduous forest zone with mixed

TABLE 2 | Star ratings of tree species with their respective meanings.

Star ratings	Definition
Green ★	Species are of little conservation concern
Pink ★	Species are common and moderately exploited, possessing potential economic value
Red ★	Species are common but exploited in Ghana due to pressure, necessitating strict supervision and preservation
Scarlet ★	Species are common but under severe threat from overexploitation, requiring a comprehensive approach to protection and sustainable use
Blue ★	Species are globally prevalent but rare in Ghana, highlighting the need for their protection
Gold ★	Species are globally restricted and relatively rare both worldwide and locally; therefore, Ghana has an indisputable concern for their preservation
Black ★	Species are rare worldwide and require careful management, especially in Ghana, where urgent conservation measures are needed for their survival
NE (not evaluated)	Species whose conservation status remains uncertain due to insufficient data or those that are eliminated or nonforest species

Note: Hawthorne and Abu-Juan [42].

tree species. The carbon dioxide equivalent (CO_2e) was calculated by multiplying the total carbon (TC) by (44/12) (Table 3). All analyses were performed using Microsoft Excel (Version 2021) and PAST statistical software (Version 4.03) application packages.

3 | Results

3.1 | Species Composition, Richness and Diversity

3.1.1 | Floristic Composition

A total of 4385 individuals from 37 families, 105 genera and 122 plant species were sampled across all growth forms at the three sand mine sites (Table 4, Supporting information 1). Trees were the dominant growth form observed in the study area, accounting for 42.62% of the total species sampled, followed by herbs (18.03%), grasses (12.30%), sedge (9.02%), lianas (7.34%), climbers (5.74%) and shrubs (4.92%). The majority of the species belonged to the Poaceae ($n = 15$), Cyperaceae ($n = 11$), Apocynaceae, Euphorbiaceae and Rubiaceae families ($n = 8$ each), and Papilionaceae and Sapindaceae ($n = 7$ each) (Supporting information 1). The highest mean number of individuals was recorded at the SMC ($n = 1936$, 15.87 ± 5.37 per plot), while the lowest was at the NSM ($n = 1146$, 9.39 ± 2.40 per plot) (Table 4). Species variances (s^2) were highest at the SMC ($s^2 = 3520.10$), followed by the SMB ($s^2 = 1396.14$) and the lowest at the NSM ($s^2 = 702.24$) (Table 4).

3.1.2 | Species Richness and Diversity

Species richness of all growth forms showed significant differences ($Hc = 11.53$, $p < 0.05$, *Kruskal–Wallis test*) among the three sites, with the NSM site showing the highest richness ($n = 81$). The SMC had the lowest species richness ($n = 45$) (Figure 4). Dunn's *post hoc* test indicated that this significant difference was observed between the NSM and SMB ($p < 0.05$) and the NSM and SMC ($p < 0.05$).

Analysis of diversity indices for all species growth forms showed no significant difference ($Hc = 0.80$, $p > 0.05$, *Kruskal–Wallis test*) across the sites. The Shannon–Weiner diversity index indicated high species diversity at the NSM ($H' = 3.33$), followed by the SMB ($H' = 2.77$) and the SMC ($H' = 2.74$), which recorded the least diversity (Figure 5). Similarly, the Simpson index and Margalef index ($D_{Margalef}$) showed the same trend, with the highest species diversity at the NSM and the least at the SMC (Table 5).

3.2 | Patterns in Community Structure and Spatial Distribution of Regenerating Species, and Identification of Key Species Sensitive or Resilient to Sand Mining Disturbance

We first assessed whether the overall IVI distributions differed across the three sites. We found that the IVI values did not differ significantly across NSM, SMB and SMC (Kruskal–Wallis *test*: $Hc = 1.41$, $p > 0.05$). Although the univariate tests revealed no statistically significant difference in the overall IVI distributions among sites, we found that the identity of the species holding the importance has been fundamentally altered. This prompted us to use multivariate community analyses to detect structural patterns not evident from IVI distributions alone.

However, the multivariate analysis demonstrated a strong site-level compositional shift. We found that the nMDS ordination based on the Bray–Curtis dissimilarity of IVI composition produced a perfect model fit (stress = 0.00) and clearly separated all three sites in ordination space (Figure 6(a)). NSM was positioned on the far left of the NMDS1 and formed a clearly isolated point relative to the SMB and SMC, which reflects its greater compositional dissimilarity from the mined sites. In contrast, the SMB and SMC were positioned on the positive side of NMDS1 ordination but were vertically separated on NMDS2. This pattern indicates that although the SMB and SMC share more species in common compared to the NSM, their relative abundance still differs sufficiently (Figure 6(a)).

TABLE 3 | Description of indices and models.

Index	Formula	Description
Individual-based rarefaction	$E(S_n) = \sum_{i=1}^{k}\left[1 - \binom{N - N_i}{n} / \binom{N}{n}\right]$	$E(S_n)$: expected number of species in a rarefied sample of n individuals, S: total number of species in the original sample, N: total number of individuals sampled and N_i: abundance of species i [45]
Shannon–Wiener diversity (H')	$-\Sigma\,(p_i \ln (p_i))$	p_i is the proportion of individuals that belong to the ith species
Density (D)	$N_A / T_{A\text{samped}}$	N_A is the total number of individuals of a particular species 'A', and $T_{A\text{samped}}$ is the total area sampled
Relative density (RD)	$(\text{AD}/\text{TD}) \times 100$ ℘	AD is the absolute density of a particular species, and TD is the total density of flora species
Frequency (F)	$\text{NP}_a / \text{TP}_{\text{samped}}$	NP_a is the number of plots in which species 'A' occurs, and $\text{TP}_{\text{samped}}$ is the total number of plots sampled
Relative frequency (RF)	$(\text{AF}/\text{TF}) \times 100$ ℘	AF is the absolute frequency of a species, and TF is the sum of the absolute frequencies of all species
Basal area	$(\pi * (\text{DBH})^2)/(40,000)$	DBH is the diameter at breast height (1.3 m); $\pi = 3.14$
Relative cover (RC)	$(\text{AC}/\text{TC}) \times 100$℘	AC is the absolute cover (or basal area) of a particular species, and TC is the total cover (or basal area) of all species
Importance value index (IVI)	RD + RF + RC	
Aboveground biomass (AGB)	$0.0673 \times (\rho \times \text{DBH}^2 H)^{0.976}$	AGB is in kg; ρ is wood-specific density (g/cm^3); DBH is the diameter in cm (1.3 m aboveground), and H is the total tree height (m) [50]
Belowground biomass (BGB)	$0.20 \times \text{AGB}$	BGB is in kg; DBH is the diameter in cm (1.3 m aboveground) [51]
Total biomass (TB)	AGB + BGB + DWB	DWB is deadwood biomass
Total carbon (TC)	$0.47 \times \text{TB}$	[51]
CO_2e	$\text{TC} \times M_w\ (CO_2)$	TC is the total carbon; M_w is the molecular weight of carbon (IV) dioxide (44/12)

TABLE 4 | Mean number of species recorded at each growth form/sampling size across the three study sites.

	NSM	SMB	SMC
Overall growth forms (combined plots)			
Taxa_S	81	50	45
Individuals	1146	1303	1936
Mean ± S.E.	9.39 ± 2.40	10.68 ± 3.38	15.87 ± 5.37
Variance (S^2)	702.24	1396.14	3520.10
Tree layer (main plot level)			
Taxa_S	45	5	0
Individuals	296	9	0
Mean ± S.E.	6.58 ± 1.24	0.20 ± 0.10	0
Variance (S^2)	69.29	0.48	0
Saplings layer (subplot level)			
Taxa_S	39	11	0
Individuals	423	39	0
Mean ± S.E.	10.32 ± 2.45	0.95 ± 0.39	0
Variance (S^2)	245.92	6.20	0
Herbs/Grasses layer (quadrat level)			
Taxa_S	33	43	45
Individuals	427	1255	1936
Mean ± S.E.	5.55 ± 2.19	16.30 ± 5.25	25.14 ± 8.35
Variance (S^2)	368.57	2125.71	5368.12

Abbreviations: NSM = Nonsand Mine, SMB = Sand Mine Buffer, SMC = Sand Mine Core.

These spatial arrangements of points in the NMDS ordination corresponded closely to the pairwise Bray–Curtis dissimilarity distances. The greatest dissimilarity was found between NSM and SMC (94.07%), followed by NSM and SMB (77.85%), whereas SMB and SMC were comparatively similar (50.99%), suggesting partial overlap in species assemblages. The hierarchical clustering dendrogram (Figure 6(b)) and heatmap (Figure 7) further supported these trends. We observed that the NSM formed a distinct branch, which was characterised by the dominance of forest-associated species. In contrast, the SMB and SMC clustered closely, with both being characterised by the dominant representation of disturbance-tolerant grasses and early successional colonisers (Figures 6(a), 6(b)).

3.2.1 | Species Contributing to Shifts in Community Structure

The SIMPER analysis decomposed the overall Bray–Curtis dissimilarity between site pairs into contributions from individual species. The analysis revealed that a relatively small group of taxa accounted for the most differences observed across all the pairwise site comparisons (Tables 6, 7, 8).

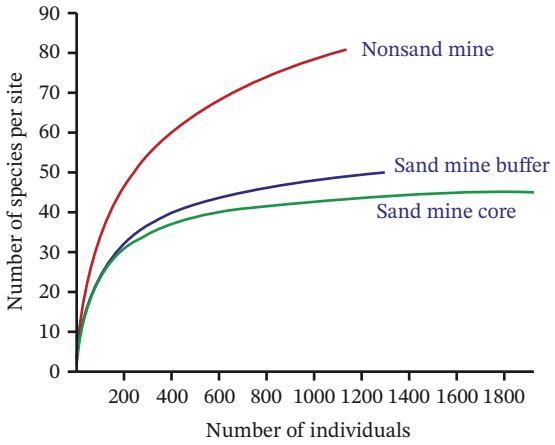


FIGURE 4 | Individual-based rarefaction curve displaying expected species richness as a function of number of individuals sampled from each site (NSM, SMB and SMC). The curve's position illustrates species richness, with upper curves (in red) indicating high species richness and lower curves (in green) signifying lower richness.

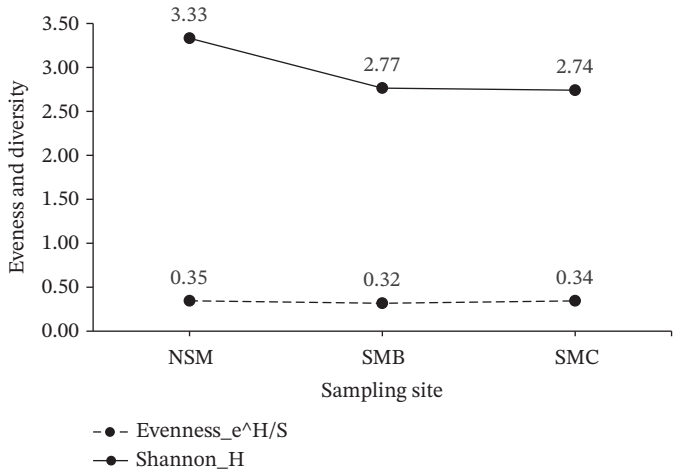


FIGURE 5 | Shannon–Weiner diversity (H') and evenness index values comparing species diversity and evenness, respectively, across the three sites (NSM = Nonsand Mine, SMB = Sand Mine Buffer and SMC = Sand Mine Core).

TABLE 5 | Diversity indices of species across the three sites.

Diversity index	NSM	SMB	SMC
Evenness_e^H/S	0.35	0.32	0.34
Shannon_H	3.33	2.77	2.74
Simpson_1-D	0.93	0.89	0.88
$D_{margalef}$	11.36	6.83	5.81

Abbreviations: NSM = Nonsand Mine, SMB = Sand Mine Buffer and SMC = Sand Mine Core.

3.2.1.1 | NSM vs SMB. We found that the shift between NSM and SMB was primarily a result of the differences in IVI values between woody understorey species in NSM and disturbance-tolerant grasses in SMB. For example, *Sorghum arundinaceum* had an IVI of 60.00 in SMB but 0.00 in NSM, contributing the highest absolute difference (60.00) and explaining 12.84% of the overall dissimilarity. Similarly, the native shrubs *Mallotus oppositifolius* (IVI = 54.09 in NSM vs IVI = 3.60 in SMB; AbsDiff 50.49, 10.81%) and *Lecaniodiscus cupanioides* (IVI = 45.71 in NSM vs IVI = 0.69 in SMB; AbsDiff 45.02, 9.64%) were severely reduced in SMB. The reduction of these species was in contrast with the introduction of disturbance-tolerant species like *Imperata cylindrica* that increased from 0.00 IVI in NSM to IVI of 36.25 in SMB (AbsDiff 36.25; 7.76%). Overall, these few species accounted for more than 40% of the total average dissimilarity between the NSM and SMB (Table 6).

3.2.1.2 | NSM vs SMC. For the comparison between the NSM and SMC, where the largest dissimilarity (94.07%) was recorded, we found that the continuous sand mining activities intensified the patterns we observed in the SMB. *I. cylindrica* explained the single largest contribution to the dissimilarity, with an IVI of 58.26 in SMC but 0.00 in NSM (AbsDiff 58.26; 10.32%). Native species that were typical of the NSM showed complete collapse in SMC: *M. oppositifolius* reduced from an IVI of 54.09 in NSM to 0.00 in SMC (AbsDiff 54.09; 9.58%) and *L. cupanioides* from 45.71 to 0.00 (AbsDiff 45.71; 8.10%). We found that disturbance-resilient grasses such as *Sporobolus pyramidalis* (0.00 IVI in NSM to 31.47 in SMC; AbsDiff 31.47; 5.58%) further contributed to the compositional shift (Table 7). These species explained 38.32% of the overall dissimilarity between the NSM and SMC.

3.2.1.3 | SMB vs SMC. Although the SMB and SMC sites were more compositionally similar (50.99%) to each other than to NSM, we still observed ecologically meaningful differences. Our results indicated that the IVI of *S. arundinaceum* dropped from 60.00 in SMB to 21.36 in SMC (AbsDiff 38.64; 12.63%), indicating partial displacement as disturbance persisted. Likewise, *I. cylindrica* increased from IVI = 36.25 in SMB to IVI = 58.26 in SMC (AbsDiff 22.01; 7.19%), indicating its role as a dominant grass in intensely disturbed areas. Other contributors included *Centrosema pubescens* (IVI = 24.69 in SMB vs 4.67 in SMC; AbsDiff 20.02; 6.54%), *Chromolaena odorata* (IVI = 22.51 in SMB vs 3.29 in SMC; AbsDiff 19.22; 6.28%) and *Saccharum spontaneum* (IVI = 17.50 in SMB vs 1.72 in SMC; AbsDiff 15.78; 5.16%). These species explained 37.80% of the overall dissimilarity between the SMB and SMC (Table 8).

3.2.2 | Identification of Disturbance-Sensitive and Disturbance-Resilient Taxa

By integrating IVI patterns (Figure 7) with SIMPER contributions (Tables 6, 7, 8), we identified taxa that serve as strong ecological indicators of disturbance intensity. We found that a distinct group of forest-associated species, such as *M. oppositifolius*, *L. cupanioides*, *Griffonia simplicifolia*, *Salacia elegans* and *Dioscorea praehensilis* are sensitive to disturbance as they showed high IVI values in NSM but were absent or nearly absent in both SMB and SMC (Tables 6 and 7, Figure 7, Supporting information 2).

In contrast, a group of grass species, such as *I. cylindrica*, *S. arundinaceum*, *Sp. pyramidalis*, *Brachiaria deflexa* and *C. odorata,* were also found to be disturbance-resilient as they showed high IVI values in mined sites (SMB and SMC) and were ranked among the largest contributors to dissimilarity across comparisons (Tables 5a–c, Figure 7, Supporting information 2). Their dominance highlights their capacity to colonise and persist in heavily disturbed areas.

In summary, we found that species from the Poaceae family demonstrated high resilience to disturbances, while certain species from the Euphorbiaceae and Sapindaceae families exhibited high sensitivity to disturbances from the sand mining activities (Tables 6, 7, 8, Figure 7, Supporting information 1 and 2).

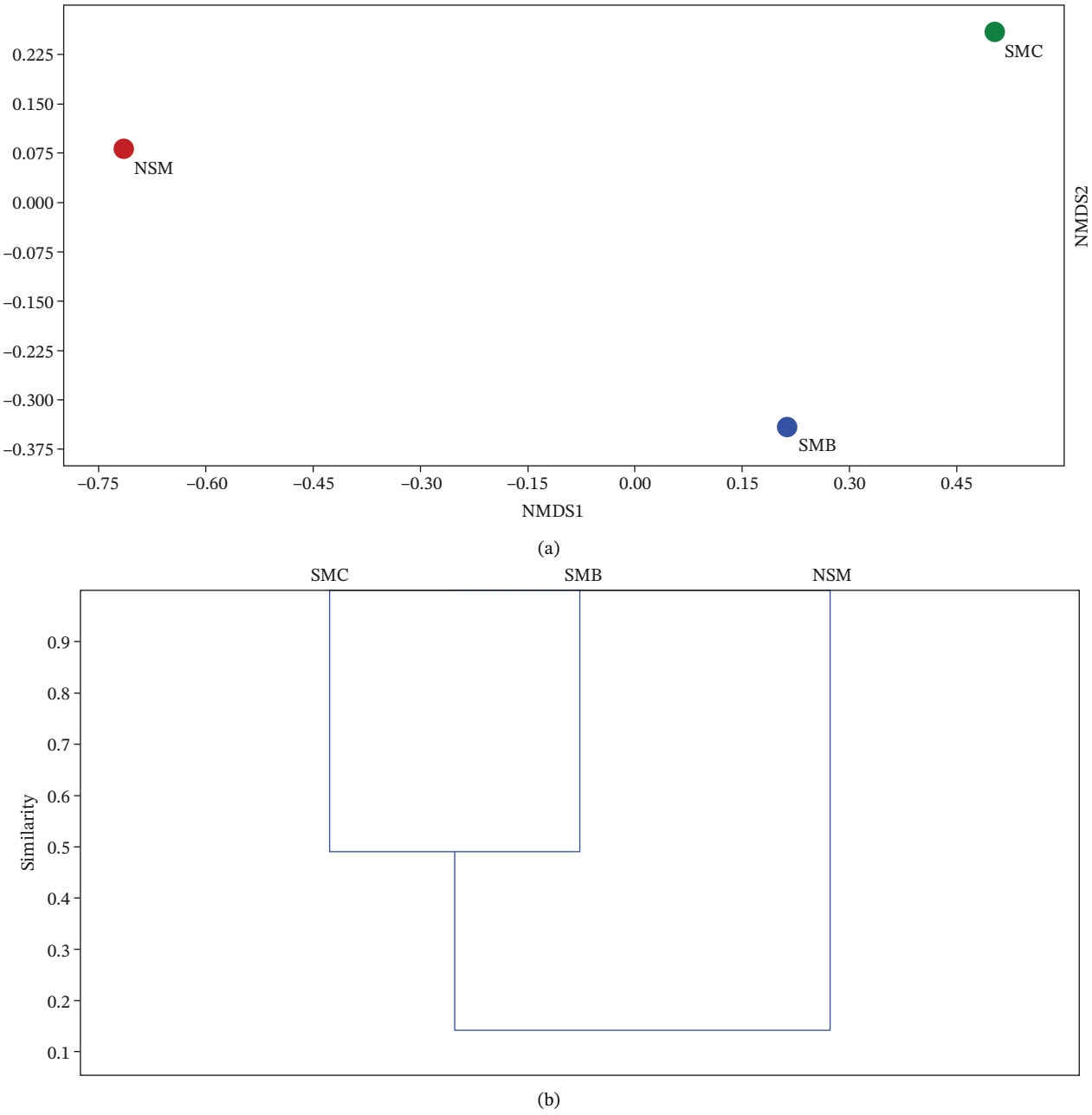


FIGURE 6 | Multivariate analysis of plant community distribution. (a) = nMDS ordination of sites based on the Bray–Curtis dissimilarity of IVI data, showing a clear separation of the NSM (red) from the SMB (blue) and SMC (green). Stress = 0.00. (b) = Hierarchical clustering dendrogram based on Bray–Curtis dissimilarity (average linkage), confirming the similarity between the two mined sites (SMB and SMC) and their distinctness from NSM. The short linkage between SMB and SMC indicates low dissimilarity. The longer branch to NSM indicates strong divergence from the mined sites (SMB and SMC). NSM = Nonsand Mine, SMB = Sand Mine Buffer and SMC = Sand Mine Core.

3.3 | Structural Attributes and Carbon Stock Potential of the Study Sites

3.3.1 | Structural Attributes

At the tree layer analysis, we found significant differences in all stand-structural attributes across the sites (NSM, SMB and SMC). Diameter, height, basal area, tree density and carbon stock all differed significantly among sites (Kruskal–Wallis test: $p < 0.001$). The Dunn's *post hoc* test showed that NSM differed significantly from both SMB and SMC for every stand-structural attribute ($p < 0.01$), but the SMB and SMC did not differ significantly ($p > 0.05$). NSM recorded the highest stand-structure values, including stem diameter (9.20 ± 0.41 cm), height (10.20 ± 0.41 m), basal area (0.011 ± 0.0015 m^2) and estimated tree density (925 ± 181.33 stems ha^{-1}). The SMB recorded substantially reduced values, while the SMC plots recorded 0.00 ± 0.00 for all attributes (Table 9).

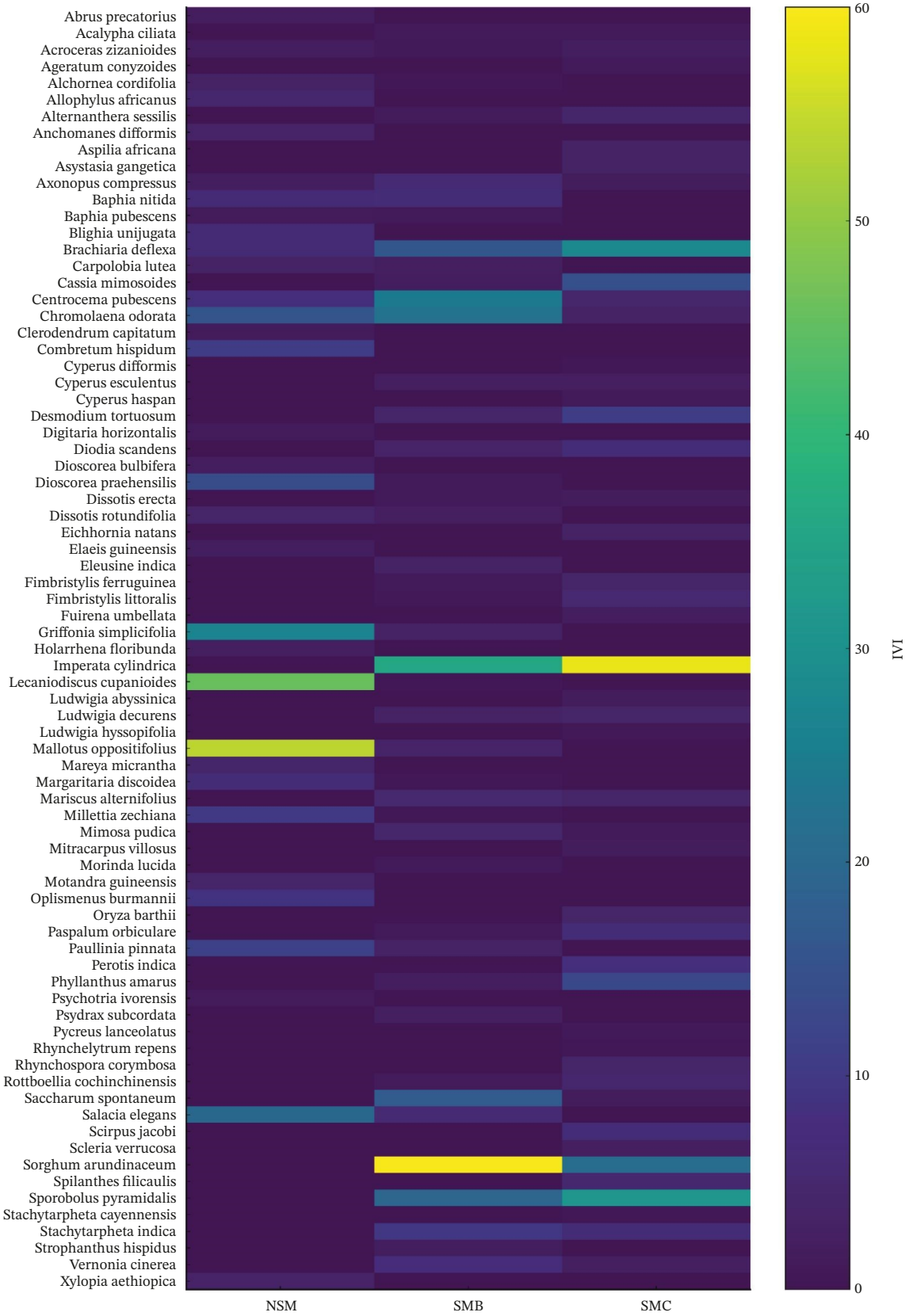


FIGURE 7 | Heatmap of species IVI across the sites illustrating the shift in community dominance. Rows represent species arranged in alphabetical order, and the columns represent the three sites. High IVI values (warmer colours) in NSM represent the forest-associated species, whereas warmer colours (in yellow) in SMB and SMC indicate the colonisation by grasses. NSM = Non Sand Mine, SMB = Sand Mine Buffer, SMC = Sand Mine Core.

TABLE 6 | Top 10 dominant species based on SIMPER analysis contributing to dissimilarity between NSM and SMB sites.

Rank	Species	IVI (NSM)	IVI (SMB)	AbsDiff	Average dissimilarity	%contribution	Cumulative (%)
1	*Sorghum arundinaceum*	0.00	60.00	60.00	10.00	12.84	12.84
2	*Mallotus oppositifolius*	54.09	3.60	50.49	8.41	10.81	23.65
3	*Lecaniodiscus cupanioides*	45.71	0.69	45.02	7.50	9.64	33.29
4	*Imperata cylindrica*	0.00	36.25	36.25	6.04	7.76	41.05
5	*Griffonia simplicifolia*	26.75	3.43	23.32	3.89	4.99	46.04
6	*Sporobolus pyramidalis*	0.00	19.77	19.77	3.30	4.23	50.27
7	*Saccharum spontaneum*	0.00	17.50	17.50	2.92	3.75	54.02
8	*Centrosema pubescens*	8.00	24.69	16.69	2.78	3.57	57.59
9	*Salacia elegans*	20.11	5.83	14.28	2.38	3.06	60.65
10	*Dioscorea praehensilis*	13.84	1.22	12.62	2.10	2.70	63.35

Note: Overall average dissimilarity = 77.85%, AbsDiff = absolute difference.
Abbreviations: IVI = Importance Value Index, NSM = Nonsand Mine, SMB = Sand Mine Buffer.

TABLE 7 | Top 10 dominant species based on SIMPER analysis contributing to dissimilarity between NSM and SMC sites.

Rank	Species	IVI (NSM)	IVI (SMC)	AbsDiff	Average dissimilarity	%contribution	Cumulative (%)
1	*Imperata cylindrica*	0.00	58.26	58.26	9.71	10.32	10.32
2	*Mallotus oppositifolius*	54.09	0.00	54.09	9.02	9.58	19.90
3	*Lecaniodiscus cupanioides*	45.71	0.00	45.71	7.62	8.10	28.00
4	*Sporobolus pyramidalis*	0.00	31.47	31.47	5.25	5.58	33.58
5	*Griffonia simplicifolia*	26.75	0.00	26.75	4.46	4.74	38.32
6	*Brachiaria deflexa*	6.10	27.92	21.82	3.64	3.87	42.19
7	*Sorghum arundinaceum*	0.00	21.36	21.36	3.56	3.78	45.97
8	*Salacia elegans*	20.11	0.00	20.11	3.35	3.56	49.53
9	*Cassia mimosoides*	0.00	14.68	14.68	2.45	2.60	52.13
10	*Dioscorea praehensilis*	13.84	0.00	13.84	2.31	2.45	54.58

Note: Overall average dissimilarity = 94.07%, AbsDiff = absolute difference.
Abbreviations: IVI = Importance Value Index, NSM = Nonsand Mine, SMC = Sand Mine Core.

TABLE 8 | Top 10 dominant species based on SIMPER analysis contributing to dissimilarity between SMB and SMC sites.

Rank	Species	IVI (SMB)	IVI (SMC)	AbsDiff	Average dissimilarity	%contribution	Cumulative (%)
1	*Sorghum arundinaceum*	60.00	21.36	38.64	6.44	12.63	12.63
2	*Imperata cylindrica*	36.25	58.26	22.01	3.67	7.19	19.82
3	*Centrosema pubescens*	24.69	4.67	20.02	3.34	6.54	26.36
4	*Chromolaena odorata*	22.51	3.29	19.22	3.20	6.28	32.64
5	*Saccharum spontaneum*	17.50	1.72	15.78	2.63	5.16	37.80
6	*Cassia mimosoides*	1.99	14.68	12.69	2.12	4.15	41.95
7	*Sporobolus pyramidalis*	19.77	31.47	11.7	1.95	3.82	45.77
8	*Brachiaria deflexa*	16.36	27.92	11.56	1.93	3.78	49.55
9	*Phyllanthus amarus*	2.02	13.01	10.99	1.83	3.59	53.14
10	*Perotis indica*	0.00	8.16	8.16	1.36	2.67	55.81

Note: Overall average dissimilarity = 50.99%, AbsDiff = absolute difference.
Abbreviations: IVI = Importance Value Index, SMB = Sand Mine Buffer and SMC = Sand Mine Core.

TABLE 9 | Structural attributes and carbon stock potential metrics across the sites.

Structural attribute	NSM	SMB	SMC	*p* -value
	Trees (Main plots)			
DBH (cm)	9.20 ± 0.41^{a}	2.32 ± 0.63^{b}	0.00 ± 0.00^{b}	$p < 0.001$
Height (m)	10.20 ± 0.41^{a}	2.84 ± 0.71^{b}	0.00 ± 0.00^{b}	$p < 0.001$
Basal area (m^2/ha)	0.011 ± 0.0015^{a}	0.0011 ± 0.0003^{b}	0.00 ± 0.00^{b}	$p < 0.001$
Tree density (no/ha)	925 ± 181.33^{a}	30.26 ± 2.40^{b}	00.00 ± 0.00^{b}	$p < 0.001$
Carbon stocks (tC/ha)	69.89 ± 16.06^{a}	2.22 ± 0.67^{b}	00.00 ± 0.00^{b}	$p < 0.001$
CO_2e (tCO_2e/ha)	256.28 ± 58.89^{a}	8.13 ± 2.45^{b}	0.00 ± 0.00^{b}	$p < 0.001$
	Saplings (Subplots)			
DBH (cm)	2.23 ± 0.06^{a}	0.96 ± 0.12^{b}	0.00 ± 0.00^{c}	$p < 0.001$
Height (m)	3.34 ± 0.06^{a}	1.04 ± 0.11^{b}	0.00 ± 0.00^{c}	$p < 0.001$
Basal area (m^2/ha)	0.0005 ± 0.0000^{a}	0.0002 ± 0.0000^{b}	0.00 ± 0.00^{c}	$p < 0.001$
Tree density (no/ha)	$21{,}200 \pm 2371^{a}$	2360 ± 494^{b}	00.00 ± 0.00^{c}	$p < 0.001$
Carbon stocks (tC/ha)	0.59 ± 0.03^{a}	0.016 ± 0.003^{b}	0.00 ± 0.00^{b}	$p < 0.001$
CO_2e (tCO_2e/ha)	2.15 ± 0.12^{a}	0.059 ± 0.011^{b}	0.00 ± 0.00^{b}	$p < 0.001$

Note: The letter superscript denotes a significant difference between any paired sites. Groups with different letters are statistically significant for any given pairwise plots. Groups with the same letters are not statistically different from each other, Mean ± Std. Error.
Abbreviations: DBH = Diameter at Breast Height, NSM = Nonsand Mine, SMB = Sand Mine Buffer and SMC = Sand Mine Core.

At the subplot layer, we found similar patterns, where all stand structures differed significantly across sites ($p < 0.001$, *Kruskal–Wallis test*). This significant difference was observed between the NSM and both SMB and SMC for all stand-structural attributes ($p < 0.01$, Dunn's *post hoc test*). SMB differed significantly from SMC for diameter, height, basal area and tree density ($p < 0.05$), but not for carbon stock ($p = 0.055$). NSM again recorded the highest subplot-level stand-structure values, including diameter (2.23 ± 0.06 cm), height (3.34 ± 0.06 m), basal area (0.0005 ± 0.0000 m^2) and estimated tree density ($21{,}200 \pm 2371$ stems ha^{-1}). SMB values were significantly lower, while SMC subplots recorded 0.00 ± 0.00 for all attributes (Table 9).

3.3.2 | Carbon Stock Potential

At the tree layer analysis, we found significant differences in carbon stocks among the sites ($p < 0.001$, *Kruskal–Wallis test*). NSM stored significantly more carbon than both SMB and SMC ($p < 0.01$), while SMB and SMC did not differ significantly ($p > 0.05$) in terms of carbon stock potential. Our results indicated that the NSM stored the highest carbon of 69.89 ± 16.06

(tC/ha), followed by SMB (2.22 ± 0.67 tC/ha), with SMC recording 0.00 ± 0.00 tC/ha (Table 9).

Subplot-level carbon stocks analysis also differed significantly across sites ($p < 0.001$, *Kruskal–Wallis test*). NSM had significantly higher carbon stocks than both SMB and SMC ($p < 0.01$). SMB and SMC did not differ significantly ($p > 0.05$). NSM recorded the highest subplot-level carbon storage (0.59 ± 0.03 tC/ha), followed by SMB (0.016 ± 0.003 tC/ha) and SMC with 0.00 ± 0.00 tC/ha (Table 9).

3.4 | Sand Mining Implications on the Conservation Status of Priority Species

3.4.1 | Assessment of Star-Rating Conservation Status

At the tree layer (20 × 20 m; see Figure 8(a)), we found a statistically significant difference in conservation-priority ratings among sites (Kruskal–Wallis: Hc = 10.56, $p < 0.05$). The SMB ($n = 5$) recorded the highest proportion of green species with all the evaluated species rated green (100.00%, $n = 5$). In the NSM ($n = 45$ species), although most species were classified as green (71.11%, $n = 32$), a non-negligible portion were assigned threat categories (pink: 6.67%, $n = 3$; red: 8.89%, $n = 4$; scarlet: 8.89%, $n = 4$), rare (blue: 2.22%, $n = 1$) and not evaluated (NE: 2.22%, $n = 1$). No species was recorded at SMC at this scale ($n = 0$) (Table 10, Figure 8(a), Supporting information 1). Dunn's post hoc comparisons indicated that the overall significant effect was driven by differences between NSM and SMB ($p = 0.014$) and between NSM and SMC ($p = 0.0020$).

At the subplot layer (5 × 5 m), we similarly found significant differences among sites (Kruskal–Wallis: Hc = 8.13, $p < 0.05$). Dunn's *post hoc* tests showed that NSM differed from SMB ($p = 0.036$) and from SMC ($p = 0.0065$). At this scale, the SMB ($n = 11$) again showed the highest proportion of green star species with all evaluated species rated green (100.0%, $n = 11$). The NSM ($n = 39$), however, was dominated by green species (82.05%, $n = 32$), but included species in threat categories (pink: 5.13%, $n = 2$; red: 5.13%, $n = 2$) and one globally restricted (gold: 2.56%, $n = 1$). No species was recorded at SMC ($n = 0$) (Table 10, Figure 8(b), Supporting information 1).

At the quadrat scale (1 × 1 m), we found no significant difference in star-rating distribution among sites (Kruskal–Wallis: Hc = 0.94, $p > 0.05$). Species were comparatively evenly distributed across sites at this scale. NSM ($n = 33$) recorded the highest proportion of green species (81.82%, $n = 27$) and low NE representation (3.03%, $n = 1$); it also contained species in higher-concern categories (pink: 9.09%, $n = 3$; blue: 3.03%, $n = 1$; gold: 3.03%, $n = 1$). SMB ($n = 43$) similarly showed a high proportion of green species (72.09%, $n = 31$) and a substantive pink star (4.65%, $n = 2$) and NE component (23.26%, $n = 10$). At SMC ($n = 45$), the distribution was dominated by green species (53.33%, $n = 24$) and had the highest proportion of NE species (46.67%, $n = 21$) (Table 10, Figure 8(c), Supporting information 1).

3.4.2 | Assessment of IUCN Conservation Status

Analysis of IUCN conservation categories across all sites showed no statistically significant differences at any plot size ($p > 0.05$ for all scales, *Kruskal–Wallis test*).

At the main plot scale (20 × 20 m), the SMB site ($n = 5$), recorded the highest proportion of LC species with all evaluated species classified as LC (100%, $n = 5$). The NSM site ($n = 45$) contained a high proportion of LC species (77.78%, $n = 35$), along with small proportions of NT (6.67%, $n = 3$), VU(6.67%, $n = 3$) and NE(8.89%, $n = 4$) taxa. No species was recorded at the SMC site ($n = 0$) (Table 11, Figure 9(a), Supporting information 1).

At the subplot scale (5 × 5 m), LC species dominated across sites. SMB ($n = 11$) had the highest proportion of LC species (90.91%, $n = 10$) with one NE species (9.09%, $n = 1$). NSM ($n = 39$) contained 66.67% ($n = 26$) LC species alongside an increase in the NE category (33.33%, $n = 13$). No species was recorded at SMC at this scale ($n = 0$) (Table 11, Figure 9(b), Supporting information 1).

At the quadrat scale (1 × 1 m), all sites were again dominated by LC species (NSM: 63.64%; SMB: 60.47%; SMC: 48.89%). We found that the NE category increased progressively along the disturbance gradient, from 36.36% at NSM to 39.53% at SMB and to 51.11% at SMC. No species from the IUCN threat categories were recorded at both the subplot and quadrat scales (Table 11, Figure 9(c), Supporting information 1).

4 | Discussion

4.1 | Species Composition, Richness and Diversity

4.1.1 | Floristic Composition

In this study, trees accounted for about 42.62% of all recorded plant species, making them the dominant growth form. However, this proportion is comparatively lower than values reported in other tropical forests in Ghana, where trees accounted for more than 70% of all individuals recorded [53, 54]. This observed trend in our study is a result of the effects of the sand mining disturbances. These activities removed the canopy trees and created the conditions that favour herbaceous and grass-dominated communities (which cumulatively recorded about 57.35% of growth forms).

The species composition dominated by Poaceae and Cyperaceae reflects a strong response to soil disturbance and canopy removal in the mined sites (SMB and SMC) [15, 55]. These families are symbolic of ruderal life-history strategies, possessing traits such as high seed output, rapid germination and vegetative reproduction that allow them to colonise bare ground rapidly [56]. Their dominance in the SMB and SMC sites directly explains the compositional shift observed in our multivariate analysis and the collapse of the tree layer in the SMC. However, the diverse woody families that form the architectural backbone of the intact forest, such as Apocynaceae, Euphorbiaceae and Rubiaceae, were gradually eliminated because they do not possess characteristics to resist the physical disturbances and altered microclimate of the mined sites.

The highest mean number of individuals and variance recorded at the SMC, despite its degraded condition, can be attributed to the proliferation of herbaceous and fast-growing pioneer species, particularly grasses [57, 58]. These species often dominate open and disturbed habitats due to their capacity for rapid germination, growth in nutrient-deficient soils and adaptation to exposed microclimates [56, 59]. Furthermore, the near-total absence of trees at the SMC

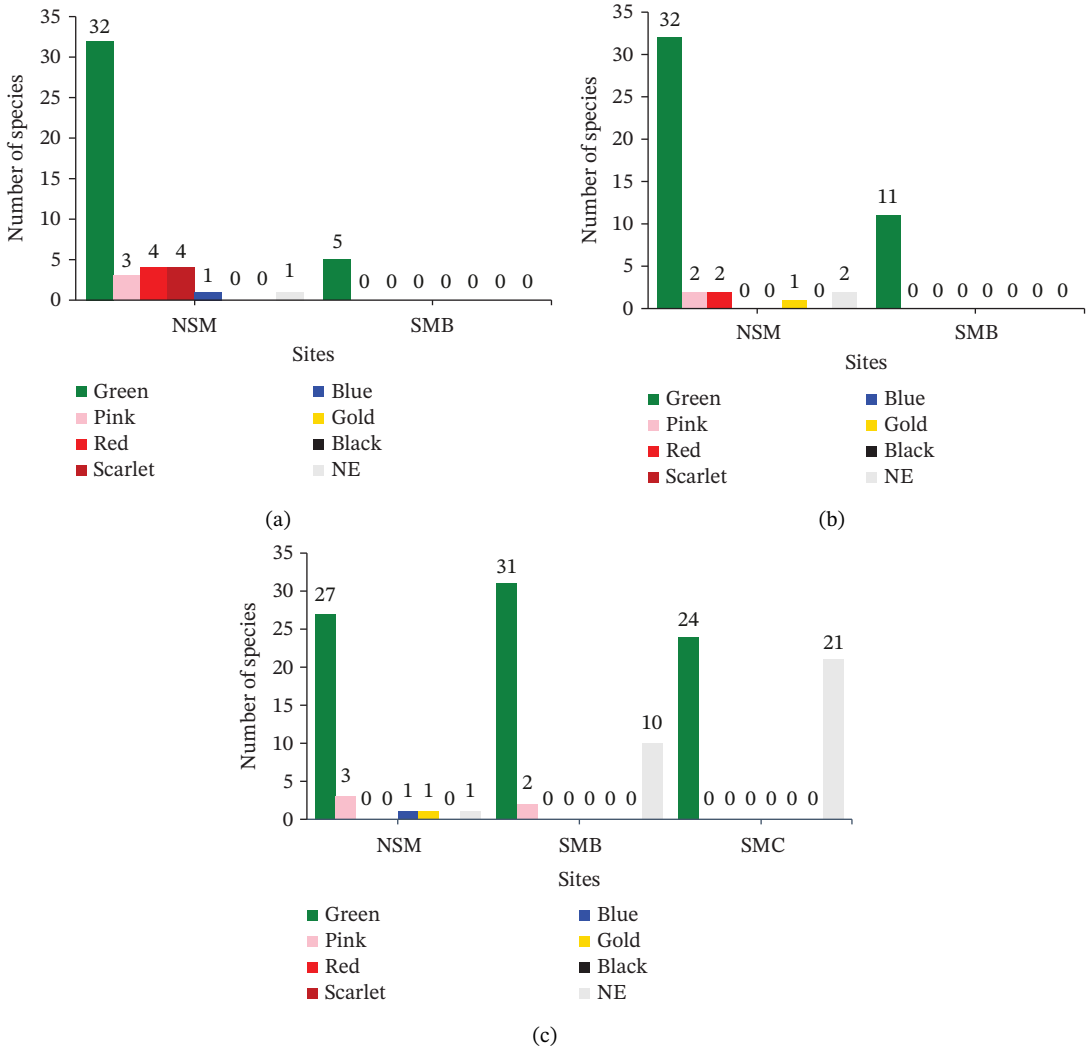


FIGURE 8 | Summary charts of the star ratings of species arranged in ascending order of conservation concern (from green to black) across different sand mine sites and sampling sizes. Each star rating is represented by its corresponding colour, with species that are not evaluated shown in grey. (a) = main plot, (b) = subplot and (c) = quadrats (see Figure 3). NSM = Nonsand Mine, SMB = Sand Mine Buffer, SMC = Sand Mine Core. Note that the SMC recorded no tree species in the main (a) and the subplots (b).

creates a simplified vertical vegetation structure [60], allowing pioneer species to colonise the ground layer in high densities rapidly. Again, the destruction of canopy cover, removal of vegetation and soil profile and loss of seed banks resulting from sand mining severely hinder the establishment of late-successional or shade-tolerant species [23, 61].

TABLE 10 | Summary counts (n) of star ratings of species and percentages (%) by site and plot size.

Plot size	Site	n	Green	Pink	Red	Scarlet	Blue	Gold	Black	NE	p value
20 × 20	NSM[a]	45	32 (71.11%)	3 (6.67%)	4 (8.89%)	4 (8.89%)	1 (2.22%)	0 (0.00%)	0 (0.00%)	1 (2.22%)	
(A)	SMB[b]	5	5 (100.00%)	0 (0.00%)	0 (0.00%)	0 (0.00%)	0 (0.00%)	0 (0.00%)	0 (0.00%)	0 (0.00%)	= 0.0049**
	SMC[b]	0	—	—	—	—	—	—	—	—	
5 × 5	NSM[a]	39	32 (82.05%)	2 (5.13%)	2 (5.13%)	0 (0.00%)	0 (0.00%)	1 (2.56%)	0 (0.00%)	2 (5.13%)	
(B)	SMB[b]	11	11 (100.00%)	0 (0.00%)	0 (0.00%)	0 (0.00%)	0 (0.00%)	0 (0.00%)	0 (0.00%)	0 (0.00%)	= 0.017*
	SMC[b]	0	—	—	—	—	—	—	—	—	
1 × 1	NSM[a]	33	27 (81.82%)	3 (9.09%)	0 (0.00%)	0 (0.00%)	1 (3.03%)	1 (3.00%)	0 (0.00%)	1 (3.03%)	
(C)	SMB[a]	43	31 (72.09%)	2 (4.65%)	0 (0.00%)	0 (0.00%)	0 (0.00%)	0 (0.00%)	0 (0.00%)	10 (23.26%)	= 0.63
	SMC[a]	45	24 (53.33%)	0 (0.00%)	0 (0.00%)	0 (0.00%)	0 (0.00%)	0 (0.00%)	0 (0.00%)	21 (46.67%)	

Note: The letter superscript denotes a significant difference between any paired sites. Groups with different letters are statistically significant for any given pairwise plots. Groups with the same letters are not statistically different from each other, n = total number of species count per site.
Abbreviations: NE = Not Evaluated, NSM = Nonsand Mine, SMB = Sand Mine Buffer, SMC = Sand Mine Core.

TABLE 11 | Summary counts of IUCN conservation status and percentages (%) by site and plot level.

Plot size	Site	*n*	LC	NT	VU	NE	*p* value
20 × 20	NSM	45	35 (77.78%)	3 (6.67%)	3 (6.67%)	4 (8.89%)	
(A)	SMB	5	5 (100.00%)	0 (0.00%)	0 (0.00%)	0 (0.00%)	> 0.05
	SMC	0	—	—	—	—	
5 × 5	NSM	39	26 (66.67%)	0 (0.00%)	0 (0.00%)	13 (33.33%)	
(B)	SMB	11	10 (90.91%)	0 (0.00%)	0 (0.00%)	1 (9.09%)	> 0.05
	SMC	0	—	—	—	—	
1 × 1	NSM	33	21 (63.64%)	0 (0.00%)	0 (0.00%)	12 (36.36%)	
(C)	SMB	43	26 (60.47%)	0 (0.00%)	0 (0.00%)	17 (39.53%)	> 0.05
	SMC	45	22 (48.89%)	0 (0.00%)	0 (0.00%)	25 (51.11%)	

Note: n = total number of species count per site. IUCN categories with ($n = 0$) across all sites [endangered (EN); critically endangered (CR); extinct in the wild (EW)] were omitted from the table for clarity.
Abbreviations: LC = Least Concern, NE = Not Evaluated, NSM = Nonsand Mine, NT = Near Threatened, SMB = Sand Mine Buffer, SMC = Sand Mine Core, VU = Vulnerable.

4.1.2 | Species Richness and Diversity

The significant reduction of species richness from the NSM to the severely mined site (SMC) is a clear indication of biotic impoverishment due to physical disturbances. This observation is consistent with the proven paradigm that severe disturbances reduce the number of species by creating environmental conditions that exceed the tolerance threshold of most native forest-associated species [62]. In contrast, the NSM, which showed the highest species richness despite having the lowest raw species counts, suggests that the NSM may contain more stable microhabitats with less anthropogenic impact and greater niche heterogeneity [63]. This supports species coexistence and a wider range of growth forms and taxa [64].

However, the lack of statistically significant differences in the Shannon–Weiner and Simpson diversity indices across the disturbance gradient is an important finding. This suggests that while mining significantly reduces the total number of species, the remaining community in the SMB and SMC is restructured to maintain a relatively even distribution of individuals among the surviving disturbance-tolerant species. The initial dominance of

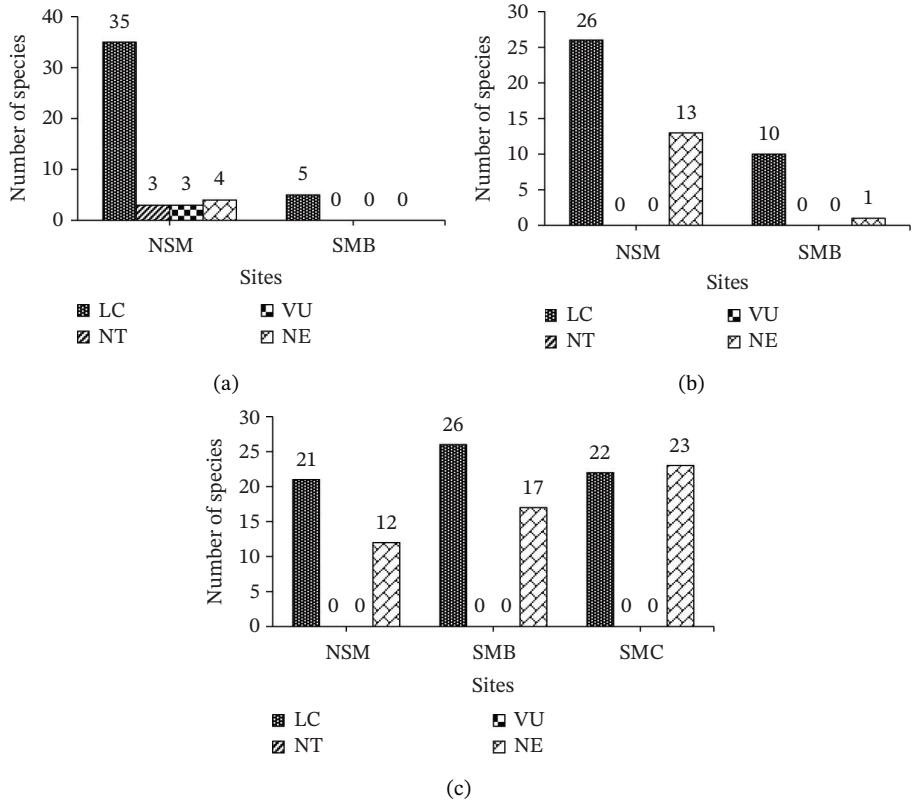


FIGURE 9 | Summary chart of the IUCN conservation status of species obtained across the three different intensities of sand mine sites (NSM, SMB and SMC) at various sampling sizes. LC = least concern, NT = near threatened, VU = vulnerable and NE = not evaluated. (a) = main plot, (b) = subplot and (c) = quadrats (Figure 3). NSM = nonsand mine, SMB = sand mine buffer, SMC = sand mine core. Note that the SMC recorded no species in the main (a) and the subplots (b).

a few disturbance-adapted species in the SMB, such as *I. cylindrica,* led to a marginally more complex assemblage of various grasses and herbs in the SMC. This increased the evenness component of the Shannon index, offsetting the overall diversity index against the significant loss in richness. This pattern has been reported in other postextraction or heavily disturbed sites where a few colonising species dominate numerically, but evenness across remaining species is similar [55, 65].

The higher mean number of individuals but lower taxonomic richness at the SMC highlights an important ecological paradox; a greater number of individuals does not necessarily indicate ecological health. It might rather signify imbalance, competitive exclusion and community simplification, characteristics of degraded systems [66].

4.2 | Patterns in Community Structure and Spatial Distribution of Regenerating Species, and Identification of Key Species Sensitive or Resilient to Sand Mining Disturbance

This study demonstrates that forest ecosystems undergo significant structural reorganisation in response to sand mining disturbance, even in the absence of significant univariate changes in community importance values. Our key finding is a clear taxonomic and functional shift from a diverse forest community dominated by native woody species toward a species-poor, grass-dominated state characterised by disturbance-resilient taxa. This transition underscores that the impact of mining is not merely a reduction in vegetation cover but a fundamental restructuring of the plant community, with significant implications for ecosystem function and recovery [67].

The absence of statistically significant differences in univariate IVI metrics across the disturbance gradient (NSM, SMB and SMC) could misleadingly suggest minimal mining impact. This is because the multivariate techniques revealed a distinct and more ecologically insightful pattern. The clear separation of the NSM from the SMB and SMC in the nMDS ordination and the high Bray–Curtis dissimilarities between the NSM and mined sites provide robust evidence of severe community reassemblage as a result of sand mining activities. This mismatch between the univariate and multivariate suggests a functional replacement process in which reductions of disturbance-sensitive species in the NSM are effectively neutralised by the numerical rise of disturbance-tolerant taxa in the SMB and SMC, thereby producing similar univariate summaries despite substantial shifts in species identity. This suggests that univariate measures can underestimate underlying compositional changes by averaging out species-specific responses. This finding corroborates prior studies that indicated that careful consideration is needed when relying solely on univariate biodiversity metrics as they may fail to capture major ecological changes when species identity, dominance and functional roles are reshaped [68–70].

The SIMPER analysis effectively identified the species driving this community reorganisation, revealing a pattern consistent with ecological theory on disturbance and succession. The loss of sensitive native species was the primary driver of dissimilarity between NSM and mined sites (SMB and SMC). For example, *M. oppositifolius* and *L. cupanioides* alone contributed to nearly 20% dissimilarity between the NSM and SMC. Their decline from high IVI values in NSM to near-zero in mined sites signifies a severe loss of forest structure and function. This indicates that the species that are typically associated with stable forest understories and rely on specific soil conditions, mycorrhizal associations and seed dispersal mechanisms [71] have been severely affected by sand mining. Their near-complete absence in SMC suggests a local ecological extinction and a breakdown in recruitment processes, a phenomenon documented in other ecosystems subjected to intense soil disturbance [72].

Concurrently, the increase in the dominance of a small taxa of disturbance-resilient species (e.g., *I. cylindrica* and *Sp. pyramidalis*) observed in SMB and SMC created a new community assemblage. The ecological success of these grasses is not a coincidence but a direct consequence of their ruderal life strategy. They possess traits such as rhizomatous growth, high seed dispersal and efficient nutrient uptake in impoverished soils [73]. These traits allow them to capitalise on the open, well-lit and physically unstable conditions created by sand mining. The significant increase in *I. cylindrica* from SMB to SMC further positions it as a clear indicator of disturbance intensity. This species is known to form single-species-dominant stands that can suppress succession, potentially creating a stable, alternative state that prevents the recovery of the native forest community [74].

Again, the transition from NSM to SMC, as shown by pairwise comparisons (Table 6) and nMDS ordination (Figure 6(a)), indicates a clear pathway of ecological simplification and homogenisation. Our pairwise comparisons and the behaviour of the key sensitive species indicate that the initial mining disturbance (SMB) causes a major shift in community composition. As disturbance continues at the SMC, it does not create a new direction of change, but rather increases the shift that has already begun. This explains the high dissimilarity between NSM and the mined sites, alongside the relative similarity between SMB and SMC. The observed pattern supports the finding that a disturbed ecosystem could shift toward a different and more stable state dominated by a group of disturbance-adapted grasses rather than native forest species [58].

The replacement of deep-rooted woody perennials with shallow-rooted grasses has significant implications for ecosystem services. It is likely to alter hydrological processes, reduce carbon sequestration and diminish habitat complexity for dependent fauna [75]. The resilience of this new grass-dominated state could pose a major challenge for forest restoration.

The feedback mechanisms, such as changes in soil microbial communities, litter composition and soil moisture retention, effectively work together, making it unlikely that forest species will return naturally without deliberate and sustained restoration efforts [76]. Thus, future restoration efforts should employ activities such as assisted natural regeneration and enrichment planting.

4.3 | Structural Attributes and Carbon Stock Potential

4.3.1 | Structural Attributes

Our findings demonstrate that forest ecosystems experience significant stand structure and compositional reorganisation in response to sand mining disturbance. The significant differences in all stand-structural attributes (diameter, height, basal area and

density) between NSM and mined sites (SMB and SMC) suggest that mature forest structures are vulnerable to disturbance. The presence of a more developed stand structure at the NSM site indicates a relatively intact and mature forest ecosystem, which is similar to other tropical forest landscapes where anthropogenic pressure is minimal [66, 77, 78]. On the other hand, research shows that increasing disturbances reduces stand structure, which supports the association that well-developed structure reflects undisturbed conditions [79]. However, some studies indicate that forest structural attributes can recover after moderate disturbances, and in some cases, disturbed sites may recover structural attributes similar to less-impacted stands. This highlights that structural complexity may not always depend on minimal disturbances but on complex successional dynamics [80].

The stand-structure degradation in the SMB, although moderately disturbed, supports the concept of ecosystem legacy effect, where initial disturbances continuously affect structural recovery [81]. The complete absence of all stand-structural attributes in the SMC plots indicates that continuous mining degrades the forest, leading to a total loss of the tree component and its associated ecological functions. This complete structural collapse in the SMC aligns with studies on mining impacts, which report the homogenisation of habitat complexity, leading to irreversible losses in biodiversity and ecosystem services [20].

Furthermore, although there were no statistical differences between SMB and SMC at the tree layer, significant differences were obtained at the sapling stage. This suggests that the initial mining impact (SMB) severely reduces the mature tree stand, while continued mining (SMC) eradicates the regenerative capacity of the understory, which effectively eliminates any potential for natural forest recovery. This may further imply that the SMC might be colonised by pioneer species, which lack structural-architectural development [82].

4.3.2 | Carbon Stock Potential

The findings from the study indicate that the stand-structural degradation was directly transmitted to a decline in carbon storage. Therefore, the smaller the structural parameters in the mined plots (SMB and SMC), the lower the sites' capacity to act as carbon sinks. The NSM obtained significantly higher carbon stocks compared to the SMB and SMC sites. The high carbon storage potential at NSM is directly reinforced by the higher number of mature and diverse woody species. This creates an association between the primary stand-structural attributes of the physical forests and the functional ecosystem services. The observed trend of high carbon storage capability of the NSM aligns with studies that report intact tropical forests store significant quantities of carbon in both the biomass and the soil components [83, 84].

The SMB and SMC sites showed no significant difference in carbon stocks in the tree and sapling stages, although there was a complete absence of woody species at the SMC site. This indicates that there is a functional threshold beyond which a site's capacity to sequester carbon is impaired to a great extent, even if there is remnant woody vegetation. This trend in the SMB is consistent with the existing studies that show that extreme disruption has a lasting effect on the carbon interactions of an ecosystem, despite the presence of vegetation [77, 85].

The reduction in carbon stock at the SMB and the total collapse in the SMC cannot be viewed in isolation, but rather, it is the ultimate functional outcome of the compositional and structural shifts documented in our study. The sensitive native species we identified as drivers of dissimilarity, such as *G. simplicifolia* and *L. cupanioides,* are precisely woody, long-lived perennials that build and maintain stable, long-term carbon pools. Their replacement by ruderal grasses like *I. cylindrica* represents a shift from a high-biomass, stable carbon system to a low-biomass system where carbon is cycled rapidly with minimal long-term storage [86].

Carbon stock assessments are an important consideration for informing national climate policy, understanding land-use impacts and supporting national inventories for global climate commitments. The magnitude of carbon loss reported in this study has direct implications for climate change mitigation strategies, sustainable development (SDGs 13 and 15) and the REDD+ (reducing emissions from deforestation and forest degradation) initiative.

Our study, therefore, provides a quantifiable baseline for the degradation component of REDD+, demonstrating that small-scale localised extractive activities can lead to disproportionately large carbon emissions at the ecosystem level. The net carbon emission of about 248.15 tCO_2e/ha from the mined sites represents a significant contribution to the atmospheric greenhouse gas budget and loss of local ecosystem services. In a similar study conducted by Abugre et al. [87] in the moist semideciduous forest in Ghana, consistent patterns were observed where the intact unmined forest site maintains significantly higher carbon stocks compared to mined or disturbed areas, which showed a complete loss of carbon stock, resulting in higher carbon emissions.

This highlights the significant effects of local biodiversity loss on the global carbon cycle and the integration of biodiversity conservation with climate action. Thus, the reconstruction of the structure of the degraded sites (SMB and SMC) is a requirement not only for vegetation restoration but also for climate regulation and ecological stability in the long term [88].

4.4 | Sand Mining Implications on the Conservation Status of Priority Species

4.4.1 | Assessment of Star Ratings of Species

The significant differences in star rating at the main plot and subplot scales provide the most direct evidence of how sand mining filters out species of local conservation concern. The intact forest (NSM) was found to be not just a repository of common species; it served as a refugium for different taxa under various degrees of local pressure, including pink (moderately exploited), red (exploited, needing supervision) and even scarlet (severely threatened) species. Their collective presence indicates a healthy, functionally intact NSM ecosystem that can support species with narrow ecological requirements and limited disturbance tolerance. However, the presence of a single 'gold'-rated species (globally restricted and rare) in the NSM subplot underscores that these forests may harbour rare taxa [64] for which Ghana has a disproportionate responsibility to protect from the destruction by sand mining.

The absence of these threat categories in the SMB and SMC sites, and the dominance of the green star species at the SMB, indicates

a functional and taxonomic simplification of the forest ecosystem. This pattern is consistent with previous studies that habitat degradation and mining-related disturbances remove specialist species and favour generalist species [20, 64]. The findings further indicate that mining activities are not random in their effect; they selectively eliminate the species already under socioeconomic stress, thus homogenising the plant community to taxa of low conservation value. This finding aligns with the biotic homogenisation phenomenon, where anthropogenic disturbance reduces the uniqueness of local species by selectively removing endemic or vulnerable species and favouring widespread generalists [89].

Furthermore, the progressively high proportion of not evaluated (NE) species along the disturbance gradient (from 3.03% at NSM to 46.67% at SMC) reflects not only the loss of known species of conservation concern but also the emergence of species that are taxonomically underassessed. These may potentially include invasive species and opportunistic grasses that are typical of degraded tropical areas [90, 91]. Their presence underscores a worrying shift from biodiversity loss to biodiversity replacement, where new assemblages form but with reduced ecological function.

Moreover, the absence of higher star taxa at the SMB and SMC subplot and quadrat scales suggests that the recruitment potential of conservation-priority species may be severely constrained by extreme microclimatic conditions (higher surface temperature, low soil moisture retention and exposed soil minerals) that characterise mined sites [92].

The findings where green star species were prevalent at all sites may be attributed to its extensive distribution throughout the country, while the lack of black star species at all sites may be attributed to its global rarity [42, 93].

4.4.2 | Assessment of the IUCN Conservation Status

The multiscale assessment of conservation status provided a robust analysis of the impact of sand mining on the species conservation status. The application of both localised, threat-based star rating and the global IUCN Red List provides complementary information, which indicates that the loss of species conservation value is both scale-dependent and context-specific. The NSM reported the dominance of LC species alongside a substantial proportion of NT and VU taxa, suggesting that the NSM continues to harbour species of global conservation concern. The coexistence of LC, NT and VU indicates that the NSM still functions as a genetic and ecological refuge, capable of sustaining rare species that may be destroyed at the SMB and SMC.

In contrast, the SMB and SMC sites demonstrated what can only be described as a 'conservation vacuum'. The total absence of NT or VU taxa in these sites supports our observation of the absence of higher threat taxa in the local star-rating assessment and further reflects the structural degradation observed in the field. These were mainly bare or sparsely vegetated plots dominated by disturbance-tolerant grasses. These results align with existing studies, which reported that mining and quarrying rapidly remove ecosystem species listed under IUCN threat categories, and replace them with common widespread taxa of minimal conservation value [20, 64].

The increase in NE species along the disturbance gradient (i.e., from 36.36% in the NSM to over 50% in the SMC), particularly at the quadrat scale, raises both conservation and management concerns. On one hand, it points to the emergence of poorly known species that may represent taxonomically neglected species. On the other hand, it could imply the colonisation by non-native or opportunistic species whose ecological roles remain poorly studied [90]. In both cases, the study underscores the urgent need for systematic taxonomic revision and continuous monitoring in postmining landscapes.

The outcome of our work indicates that a species can be considered least concern on a global scale (IUCN category) but faces serious and imminent extinction in the local context. Our findings indicated the presence of threat species, including 'blue' (rare in Ghana) and 'gold' star (relatively rare globally), but the absence of threatened species on the IUCN category at the NSM site. This highlights a significant limitation of global assessment as the basis of local or national conservation planning and justifies the importance of context-focused assessment systems like the star rating to measure local exploitation pressures and ecosystem-specific threats [94]. This finding underscores that the star-rating system does not replace the IUCN Red List, but a powerful complementary tool.

From a conservation standpoint, we can say that the local star-rating system is an effective early warning indicator, which allows species under local pressures to be identified before they might qualify for a threatened IUCN category. Therefore, the conservation of the NSM is important for biodiversity conservation and safeguarding the ecosystem of genetic resources and species with specific socioeconomic value.

5 | Conclusions and Policy Recommendation

This study provided clear evidence of the effects of sand mining on species composition and diversity, stand structure and carbon stocks in a tropical forest in Ghana, focusing on biodiversity conservation and ecosystem services. The sand-mined sites (SMB and SMC) showed a significant reduction across all the measured ecological indicators (species composition and diversity, structural attributes, carbon storage and conservation value). The undisturbed site (NSM) supported richer and more diverse plant communities, complex stand structure and significantly higher carbon stocks. In contrast, the mined sites were dominated by grasses and early successional species, lacking tree cover, structural complexity, regeneration capacity and conservation-priority species. This suggests that sand mining severely altered the plant community and stand structure compared to the NSM. Although the number of individuals enumerated in the mined sites was higher, it reflected ecological imbalance rather than true biodiversity, as the species conservation status and importance value evaluation revealed the absence of threatened and disturbance-sensitive species, indicating a risk of local species loss.

Furthermore, the strong association between stand-structural degradation and declining carbon stock has important implications for carbon sequestration and ecosystem services. The findings highlight the importance of protecting the unmined site, which acts as a refugium for biodiversity, restoring the degraded

mined sites through assisted natural regeneration, and controlling sand mining to prevent long-term ecological harm while preserving unique forest ecosystem services. Overall, with increasing demand for sand as a construction material, there is a need to integrate sustainable sand mining practices into national land-use policies to safeguard forest sustainability, ecological integrity and human well-being and prevent further loss of biodiversity and ecosystem services.

Author Contributions

Each author has made significant contributions to the development of this manuscript. Nat Owusu-Prempeh led the research conception and design, conducted analyses, interpreted results and participated in preparing and revising the manuscript. Acheampong Boakye Kesse was actively involved in data organisation and analysis, result interpretation and manuscript drafting and revision. Peter Kwasi Akomatey contributed to field data collection, organised the field data and helped draft and revise the manuscript. Collins Ayine Nsor played a key role in data analysis, drafting and critically revising the manuscript. Bright Danso Appiagyei interpreted the study results, drafted and reviewed the manuscript.

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All authors reviewed and approved the final version of the manuscript before submission.

Ethics Statement

The authors confirm that written informed consent for the publication of the image (seen in Figure 2) has been obtained from the participant/co-author.

Conflicts of Interest

The authors declare no conflicts of interest.

Data Availability Statement

Data supporting the study will be made available upon request from the corresponding author.

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Supporting Information

Additional supporting information can be found online in the Supporting Information section. *(Supporting Information)*

Supporting information provides additional data that support the results presented in the manuscript. Supporting information 1 presents a summary of plant species recorded across the study plots (i.e., main plot, subplot and quadrat), organised by family, scientific and local names, growth habit and conservation status assessment (local star ratings and IUCN Red List category) used for the conservation priority of species locally and internationally. Supporting information 2 presents a summary of the IVI distribution of the vegetation structure of regenerating species across the study sites.