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Ecosystem restoration can lead to carbon recovery in semi-arid savanna grasslands in India

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Author contributions

MB, CM and ATV conceived and designed the research; MB, CM and AK collected the data; MB, CM and AJH analyzed the data; MB, CM, AK, AJH and ATV wrote and edited the manuscript.

Abstract

Semi-arid savanna grasslands (SG) in India deliver enormous benefits to people and nature but are currently undergoing large-scale degradation. Soil carbon stocks in degraded SGs vary in response to a host of anthropogenic driving factors including agricultural expansion and industrial development. Although there is increasing support for restoring grasslands by planting native grass species, its impact on soil carbon recovery is largely unknown. In this study, we undertake a plot-level investigation of soil and above-ground biomass carbon stocks to provide robust estimates of carbon densities across sites which have undergone restoration over the last 3 years and compare that with a no-intervention control using a space for time substitution framework. We find that SGs store significant amounts of carbon (12.74 - 22.11 tC/ha across 1-year to 3-year restoration sites respectively), with most of the carbon stored in soils (8.72 -12.54 tC/ha across 1-year to 3-year restoration sites respectively). The carbon stored progressively increases with the age of grass plantation. The 3-year site shows an increase of 34% carbon stock compared to the no-intervention control, and an increase of 30% and 21% in comparison to the 1-year and 2-year sites respectively. Our study demonstrates a robust approach to estimate soil carbon stocks in these ecosystems and highlights that effective conservation and restoration can enable SGs in India to act as natural carbon sinks at scale.

Key words

savanna grasslands, carbon stocks, soil organic carbon, restoration monitoring, grassland restoration, carbon recovery

Implications for Practice

- Semi-arid savanna grasslands are important carbon sinks because they store significant carbon stocks in their soils relative to local water availability.
- The planting of indigenous grass species in semi-arid savanna grasslands can boost carbon take-up in the soil.
- A robust grassland-specific monitoring framework is required to assess carbon fluxes in semi-arid savanna grasslands.
- Long-term monitoring of soils in semi-arid savanna grasslands can reveal if soil carbon stocks remain stable and progress towards attaining old-growth characteristics.

Introduction

Grassy biomes, comprising open grassland, grassy shrublands and savannas, cover about 40% of the Earth's surface (Bardgett et al., 2021) and play a critical role in climate change mitigation by acting as natural carbon sinks (Bai and Cotrufo, 2022; Strömberg and Staver, 2022). Current estimates suggest these biomes store more than a third of the global terrestrial carbon stocks, with about 90% of it stored belowground in root biomass and as soil organic carbon (SOC). In contrast, carbon stocks in forests are concentrated in above-ground biomass (Brown et al., 1993; Anderson-Teixeira et al., 2016), where environmental risks like fire and pest attacks as well as climate change are ever-present threats (Reddy et al., 2017; Dass et al., 2018). Effectively, this means that grassy biomes act as a more stable and relatively more permanent form of carbon storage than forests (Dass et al., 2018).

Semi-arid savanna grasslands (SGs) in India have existed for millions of years, formed by the complex ecological and evolutionary interactions between herbaceous plants (grasses and forbs with extensive root networks), environmental change (cooling, heating, changes in atmospheric CO_2), fire and herbivory (Ratnam et al., 2016; Buisson et al., 2020). Currently, SGs and other open natural ecosystems (together denoted as Open Natural Ecosystems, or ONEs) are spread over almost 32 Mha across 15 states, with Rajasthan, Madhya Pradesh and Maharashtra as the 3 states with maximum coverage (Madhusudan and Vanak, 2023).

In India, as in other countries across the world, ONEs are under severe threats from agricultural expansion and industrial development. These pressures have been exacerbated by the unforeseen consequences of global and national environmental policies. One of the principal threats to these grasslands today is indiscriminate tree-planting efforts to meet global and national carbon sequestration targets (Bastin et al., 2019). The logic of such proposals for climate mitigation, however, is increasingly being challenged because of their damaging impact on natural grasslands and because their carbon sequestration potential is considered inflated

(Bardgett et al., 2021). Such efforts, rather than contributing to effective land-based climate action, endanger the long-term integrity and viability of these grasslands, and of the people and biodiversity that depend on them. Unfortunately, a lack of data on the extent of carbon stocks in SGs hinders meaningful policy and decision making on the potential contribution of these ecosystems to land-based climate action.

Plot-level carbon inventories are considered the building-blocks of assessments of carbon storage in such and semi-natural ecosystems (Malhi et al., 2021). These inventories include the measurement of biomass in 5 major carbon pools - grass and woody vegetation (above-ground biomass), roots (below-ground biomass), litter, deadwood, and soil (soil organic carbon) (Marthews et al., 2014). These estimates are then often used as a reference for demonstrating regional and national carbon storage potentials. While plot-level carbon inventories in forest ecosystems in India are relatively widespread, SGs in India have been particularly neglected in such initiatives (Bhadwal and Singh, 2002; Wani et al., 2012; Salunkhe et al., 2018; Brahma et al., 2021).

Conducting and updating such inventories assumes increased significance in the light of the recent global recognition of these ecosystems as effective carbon sinks and the attention that is being given to their restoration (Brancalion et al., 2019; Elias et al., 2021). Reliable estimates of carbon stocks and sequestration potential are a prerequisite to measure progress towards these initiatives.

In India, state-level forest departments are mandated to contribute to ecosystem restoration activities through ecosystem protection, plantation, and management. In the state of Maharashtra, the plantation and management of indigenous grass species has also been initiated in degraded SG patches over the last few years. However, evaluating progress in terms of the build-up of soil organic carbon (SOC) remains a key gap, and is fundamental to assess if interventions are leading to desired results.

In this study, our objectives are twofold. First, we estimate carbon stocks in degraded SG patches and compare it with recently restored areas to assess the impacts of the intervention on SOC. Second, we describe the build-up of SOC and the relative contribution of the restoration activity to the observed changes. To do this, we employ a space-for-time substitution approach combined with a paired sampling procedure. While we focus on SOC dynamics, we also measure above-ground biomass (in grass and woody vegetation) on our sampling sites to quantify the distribution of carbon stocks among two different carbon pools.

In this way, we provide some of the first estimates of the average carbon stock densities in SG patches and the contribution of restoration in increasing SOC, thus providing the platform for initiatives which aim to account for the carbon stored, and potential for further carbon sequestration, in these ecosystems at the regional, state or national levels.

Materials and Methods

Study site and context

The study was conducted in the Malshiras Taluka of Solapur district in the state of Maharashtra *(Figure 1)*. The area is part of the Malshiras Range of the Solapur Forest Division of the Maharashtra Forest Department (MFD).

The area falls under the semi-arid biogeographic region of India. It receives a mean annual rainfall of ~ 500mm, concentrated during the Indian monsoon season (June to September). The land cover of the area is dominated by a mosaic of scattered grassland patches and irrigated fields. The main agricultural crops in the area are sugarcane, maize and sorghum.

Administratively, a total of 9,250 ha of SGs in the area falls under the Malshiras Range. This area includes 8,234 ha classified as Reserved Forests (with restricted access to local communities) and 1,016 ha of community grazing lands locally known as '*Gairan'* (Maharashtra Forest Department, pers. comm.).

Certain land management practices can be commonly found in these Reserved Forests. The MFD has undertaken occasional tree plantation drives, including the planting of trees like *Azadirachta indica*, with the age of some plantations even dating back to the 1970s. These plantation drives also involve occasional gap-filling to compensate for tree mortality. Since the area is drought-prone, trenches, contours and bunds are also commonly created as a soil and water conservation measure (Maharashtra Forest Department, pers. comm.).

Since 2019, the MFD has undertaken the restoration of this ecosystem by planting local indigenous grasses such as *Dicanthium annulatum, Chrysopogon fulvus* and *Cenchrus setigerus* under the Compensatory Afforestation Fund Management and Planning Authority (CAMPA) Act 2016. These grasses are typically germinated in a local nursery for 6 months (January to June) and planted in-situ after the first monsoon showers in the region in the month of July. Standard operating procedures involve protection and regular monitoring in the 1st year and gap-filling to make up for above-average mortality in the 2nd year. From the 3rd year onwards, the grass plantations see no further interference (Maharashtra Forest Department, pers. comm.).

The age of these planted grasses is locally determined by the number of monsoon seasons that the landscape has witnessed. So, grasses planted in 2021 would be considered one-year old till April 2023 (Maharashtra Forest Department, pers. comm.). Following this convention, we base our study on sites with an age of 1, 2 and 3 years respectively. We pair these plots with a site where no grass planting had been done as control, while other forms of management conducted at irregular intervals (for example, building trenches and bunds) is similar to the treatment sites.

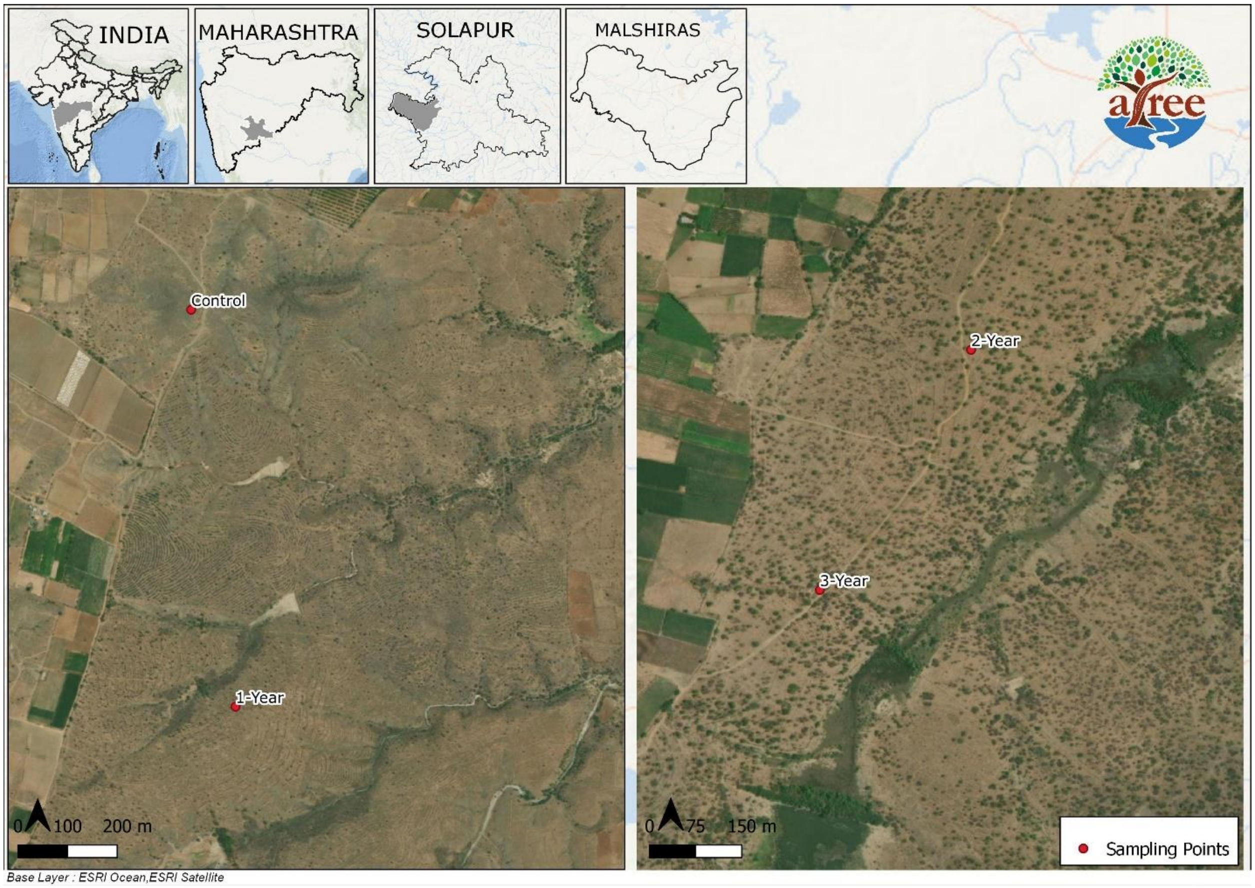

Figure 1: Map showing sampling sites in the Malshiras taluka in the Solapur district of Maharashtra. The Control and 1-Year Restoration sites are together denoted as Sulki Aai, while the 2-Year and 3-Year Restoration sites are together denoted as Motewadi (see text for further details).

Sampling design

The field activity was conducted in March 2023 at the peak of the dry season. Our sampling design was divided into treatment and control plots. To determine carbon stocks under different levels of vegetation cover, we stratified areas by relying on the grass plantation done by the MFD in preceding years. We identified a total of three treatment types based on the age of the grass plantations of 1, 2 and 3 years. We denoted these treatment types as *1-year, 2-year* and *3-year* respectively. These sites were chosen at random by the MFD to undertake the planting activity. Our control plots were those where no grasses had been planted. Both treatment and control sites shared similar land use histories (see *Study Site and context*).

For each identified age class, we identified sites within the Malshiras Range. Restored sites in the Malshiras Range exist in a mosaic of land cover and land uses, and typically span over 10-20 ha.

It was not possible to find all three types (ages) of treatment sites in the same area to control for site-specific variations. Thus, we paired the 1-year site with the control site in its proximity *(hereafter named Sulki Aai to denote the nearby village name)*. We paired the 2-year with the 3-year sites in its proximity *(hereafter named Motewadi to denote the nearby village name)*. In addition to similar land use management histories, both sites display similar biophysical characteristics because the linear distance between Sulki Aai and Motewadi is about 8 kms.

At each site, we placed multiple sampling plots by identifying the boundaries of these sites on Google Earth and placing random points within those sites using QGIS. The number of plots to be laid on each site were determined accordingly to achieve a sampling coverage of 3-5% of total area of each site *(Table 1)*.

We laid square plots covering a total area of 0.1ha (31.6 x 31.6 m) in a nested manner *(Figure 2)*, adapted from similar plot design protocols from the World Agroforestry Centre and the RAINFOR network (Kurniatun et al., 2010; Marthews et al., 2014). Within each 0.1 ha plot, two sub-plots were laid at diagonal ends to collect multiple grass and soil samples.

Table 1: Details of restoration sites and total number of plots laid in each site.

Restoration year	Total area of site (ha)	Number of plots laid (n = 23)	Sampling area (ha)	Sampling coverage (%)
Control	10.6	5	0.5	4.72
1 Year	12.7	6	0.6	4.72
2 Year	16.6	5	0.5	3.01
3 Year	15.5	7	0.7	4.52

A total of 8 soil samples (4 from each sub-plot) were collected from the top 0-30 cm of the soil layer (or the most depth possible before hitting bedrock) using a standard soil corer. IPCC guidelines state that it is good practice to measure the SOC pool to a depth of at least 30 cm (Eggleston et al., 2006). This is the depth where the changes in the soil carbon pool are likely to be fast enough to be detected with monitoring at realistic time intervals. Obtaining a core of more than 30 cm was not logistically possible because of the lack of topsoil depth in the areas we sampled. It is important to note that in these landscapes, there is limited data on the depth at which SOC responds to changes in ecosystem types, management practices and disturbance regimes.

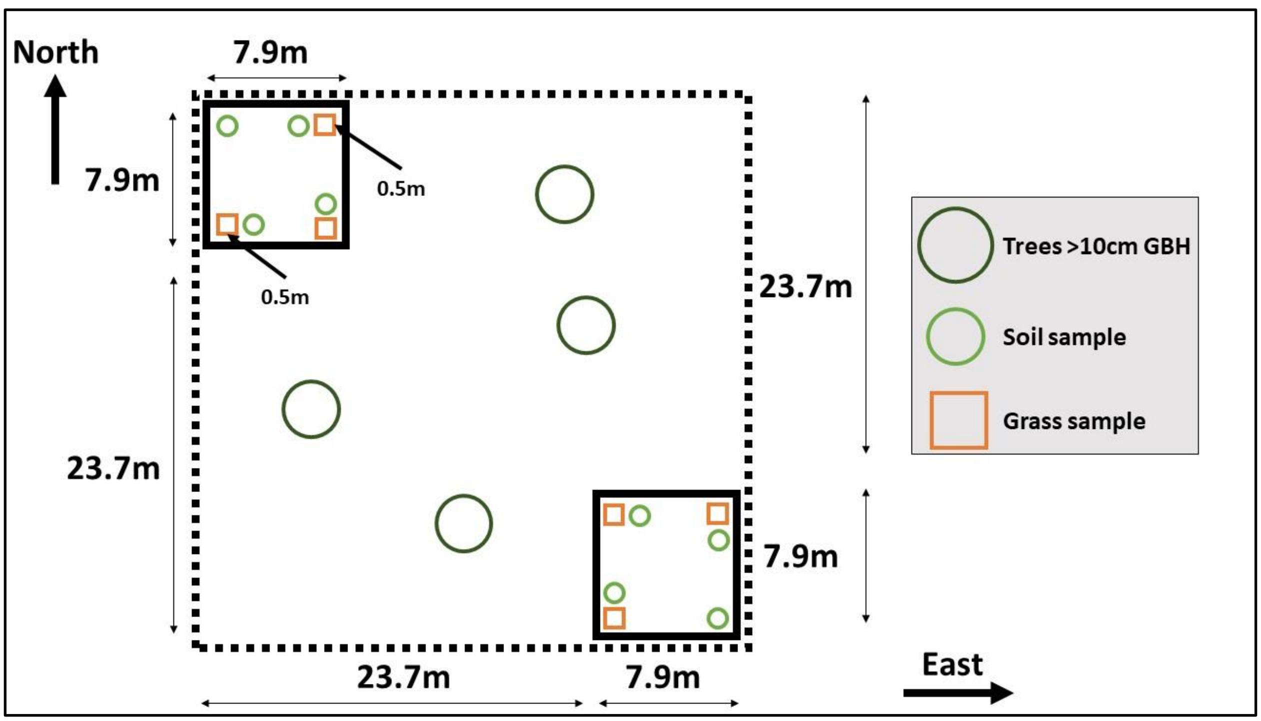

Figure 2: Stylized description of the nested plot design employed for sampling. Within the 0.1ha plot, two sub-plots were laid on diagonal ends to collect grass and soil samples. In each plot, a total of 8 soil samples and 6 grass samples were collected and a census of woody vegetation (GBH > 10cm) was conducted.

Sample processing and data extraction:

Above-ground tree biomass

A tree census was conducted in each plot. Discussions with local MFD officials revealed that trees found in each site were planted as part of occasional plantation and gap-filling activities done by the MFD stretching back several decades. Due to local environmental conditions, tree densities are extremely sparse. Common trees found in our study sites included *Senegalia catechu, Azadirachta indica* and *Zizyphus mauritiana*.

Considering the prevalence of stunted tree growth due to local climatic and environmental conditions, we modified standard tree census techniques of the minimum size for the recording of individual stems. Tree girth at breast height (GBH) and height was recorded for each tree >10cm GBH within each plot. To determine the volume of each stem, GBH was converted to diameter at breast height (DBH) and species-specific allometric equations were applied (Forest Survey of India, 1996, 2021). Species-specific wood density values were taken from literature (Zanne et al., 2009). We used a carbon fraction of 0.5 (Eggleston et al., 2006).

Above-ground grass biomass

A total of 6 grass samples (3 from each sub-plot) were harvested using scissors within an area of 0.5m x 0.5m marked by a steel quadrat. Their wet weight was recorded at the site using a weighing balance. The samples were then brought back to the ATREE campus in Bengaluru for further analysis. They were oven-dried at 70ºC for 24 hours to achieve a constant weight, and then weighed again using a weighing balance. We used a carbon fraction of 0.5.

Soil analysis

We estimated both the SOC% and bulk density of the soil samples to calculate per-hectare SOC values. We collected soil samples by using a soil core of fixed volume (height, h = 10cm; diameter, d = 5cm). We cored the soil thrice to reach the required depth of 30 cm. We excavated around the core without disturbing or loosening the soil that it contained and carefully removed it with the soil intact. We removed any excess soil from the outside of the soil core and cut any plants or roots off at the soil surface with scissors. We placed the collected soil samples into plastic zip-lock bags, emptied out the excess air from the bag and sealed it.

Each sample was well-mixed in the bag and clumps were broken down. The moist weight was noted using a standard weighing scale. The average weight of each soil sample was found to be ~700 gms. We took a sub-sample weighing ~100-150gms for further processing. This sub-sample was oven-dried at 70ºC for 72 hours and its dry weight was recorded.

We sieved the sample using a 2mm sieve to separate fine earth particles from the coarse mineral fraction. Evidence suggests that the coarse mineral fraction has a negligible capacity to store carbon, therefore it was removed before analysis and SOC content was measured for the fine earth fraction (FAO, 2019). At this stage, the sub-sample was further sub-divided into 2 sections - one to measure bulk density (BD) and one to measure SOC% *(See Soil Section 1 and Soil Section 2 in Figure 3)*.

SOC % analysis: We determined the SOC% of each soil section using the combustion gas chromatography method in a CHNS analyzer. Approx. ~2 gm of the section was dried for 1 hour at 105ºC and a small proportion of soil (0.110 - 0.111 mg) was weighed and packed into a small tin foil to be inserted into the CNHS analyzer to get the SOC % value *(Figure 3)*.

Bulk density: BD is the mass per unit volume of the soil. Here, we estimated $BDfine_2$, denoted as the mass of fine earth per total volume of the soil sample. To estimate the mass of fine earth particles in each soil core, we oven-dried the soil section again at 105ºC for 24 hours to ensure complete loss of moisture. Comparison of the final weight of the section with the wet weight of the sample allowed us to estimate the proportion of moisture content in the soil section. As we had taken a random sample from the original soil core, we assumed that the original soil core collected on site would have the same moisture content as the sub-sample. In this way, we could calculate the dry weight of the fine earth particles of the original soil core. We combined that with the volume of the soil core (known already) to estimate $BDfine_2$ (FAO, 2019) *(Figure 3)*.

Finally, SOC stock for each sample was determined by the following equation:

$$\textbf{SOC stock (tC/ha)} = OC_i \times BDfine_2 \times T_i \times 0.1$$

where,

SOC stock (tC/ha) is the soil organic carbon stock of the sampled depth increment;

OC_i *(mgC/g of fine earth)* is the organic carbon content of the fine earth fraction (< 2 mm) in the sampled depth increment;

$BDfine_2$ *(g fine earth per* cm^3 *of soil)* is the mass of fine earth per total volume of the soil sample (equivalent to the mass (g) of fine earth/total volume of soil sample (cm^3) in the given depth increment;

T is the thickness (depth, in cm) of the depth increment;

0.1 is a factor for converting mgC/cm^2 to tC/ha.

Statistical analysis

We used a linear mixed modeling approach to compare the total SOC content between the different control and treatment plots. As our sampling sites were spread out, we used individual plot ID as a random effect variable and restoration type as fixed effect variable. We used the *'lme'* function from the R package *'nlme'* to run this model. We also performed *Tukey's post-hoc* analysis to compare the difference in SOC between the two pairs of treatments. The *ggplot2* package was used for the graphic representation of our results (R Core Team, 2023). Results at the core level were aggregated to the sub-plot, plot and site-type levels to discuss our observations.

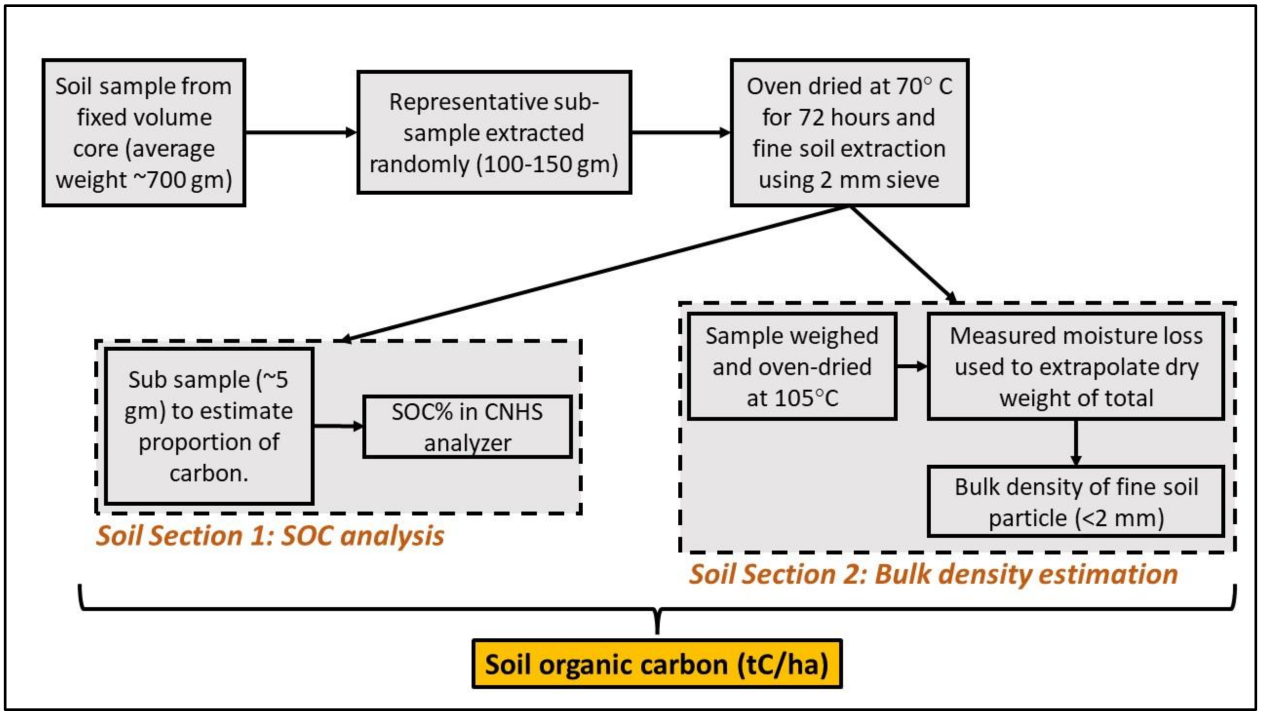

Figure 3: The steps outlining the procedure for SOC analysis employed in the study.

Results

A total of 178 soil cores were sampled to estimate soil organic carbon in 23 different plots categorized into three restoration treatments and one control. Due to shallow topsoil, we were not able to core to the mandated depth of 30cm depths to collect soil samples (for 72% samples). However, % SOC values for each soil sample did not show a significant relationship with soil depth (R^2 = 0.035) *(Figure 4)*, indicating that variation in sampling depths in the 0-30cm stratum did not affect the capturing of SOC% values. So, we report SOC estimates up to the average soil depth found in sampling sites (22 cm).

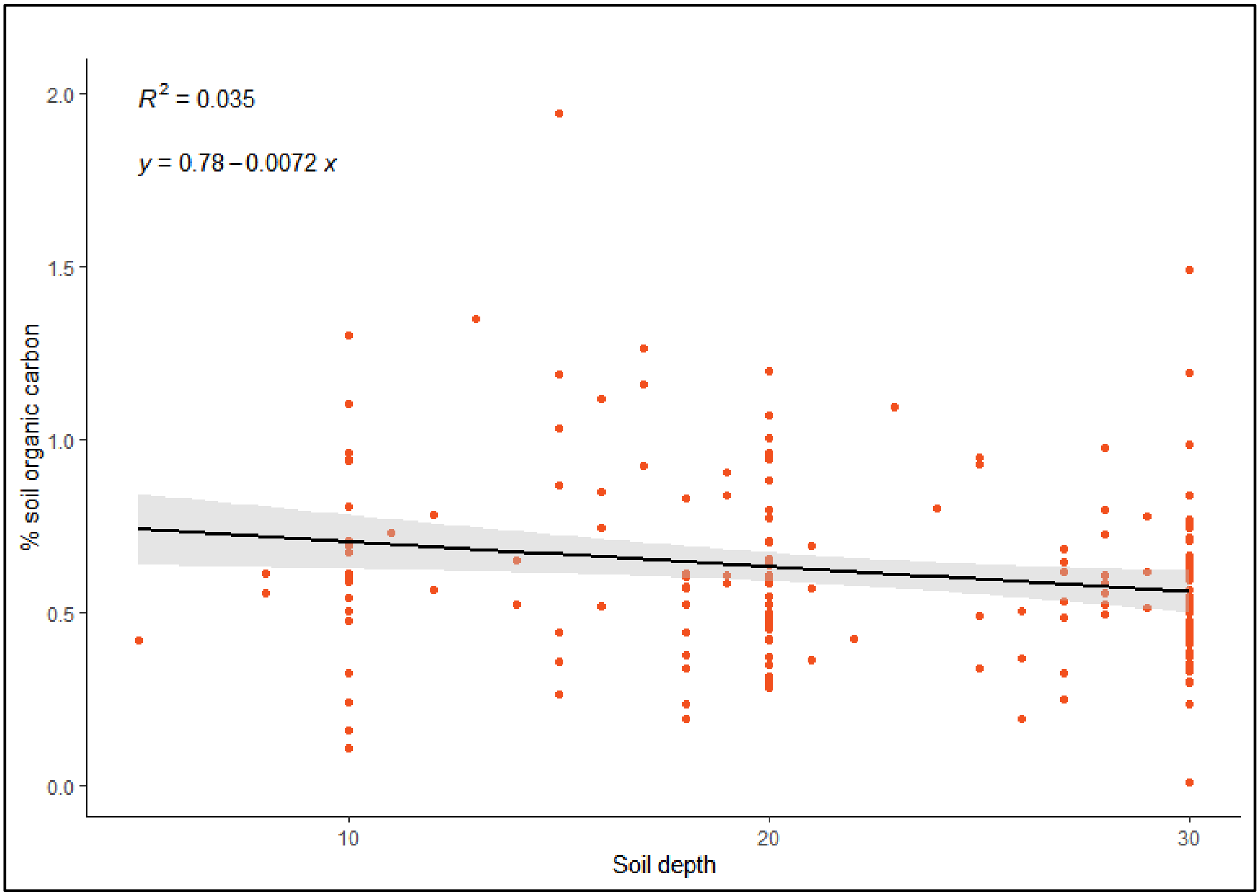

Figure 4: The relationship between sampling depth and soil organic carbon % (SOC %) for each soil sample. The lack of a strong relationship indicates that SOC% did not vary significantly even in cases of shallow topsoil depths (<30cm) (see text for further explanation).

Highest average total carbon stocks were found in the 3-year sites (22.11 ± 1.86 tC/ha; Mean ± Standard Error), compared to the 2-year (18.08 ± 1.40 tC/ha) and the 1-year sites (12.74 ± 0.37 tC/ha). The control plots reported the lowest average total carbon stocks (9.10 ± 1.19 tC/ha) *(Table 2)*. See *SI Table 1* for a plot-level breakup of soil and above-ground woody vegetation and grass carbon stocks and SI Figure 1 for SOC stocks in each site.

SOC contributed the highest proportion to the total carbon pool in all treatment and control sites *(Figure 5)*. SOC estimates in 3-year sites ranged from 6.16 to 16.42 tC/ha, with a mean of 10.0 tC/ha, with the minimum in control sites and the highest found in 3-year sites. SOC formed 89% of the total carbon pool of control plots, 68% of the 1-year plots, 54% of the 2-

year plots and 57% of the 3-year restoration plots. Woody biomass contributed the second highest proportion, while grass biomass accounted for the lowest proportion of the total carbon stock in each plot *(Figure 5)*. This was expected as grass biomass is transient in these ecosystems based on moisture availability and time since grazing and varies widely between dry and wet seasons. It is also to be noted that sampling plots were chosen at random within the sites without controlling for tree densities.

Table 2: Estimated mean tC/ha for each sampling plot and associated contribution of each carbon pool in each type of plot with standard error and % share.

Restoration type	Mean carbon tC/ha (± S.E.)	Carbon pool	Mean tC/ha (S.E.)	% share
Control (N=5)	9.10 (± 1.19)	Grass	0.24 (± 0.04)	2.66
		Soil	8.14 (± 1.05)	89.43
		Woody	0.72 (± 0.20)	7.91
1 Year (N=6)	12.74 (± 0.37)	Grass	0.82 (± 0.14)	6.40
		Soil	8.72 (± 0.32)	68.43
		Woody	3.21 (± 0.47)	25.18
2 Year (N=5)	18.08 (± 1.4)	Grass	0.94 (± 0.11)	5.21
		Soil	9.85 (± 1.00)	54.47
		Woody	7.29 (± 0.95)	40.32
3 Year (N=7)	22.11 (± 1.86)	Grass	0.59 (±0.05)	2.65
		Soil	12.54 (±0.70)	56.72
		Woody	8.98 (± 1.78)	40.63

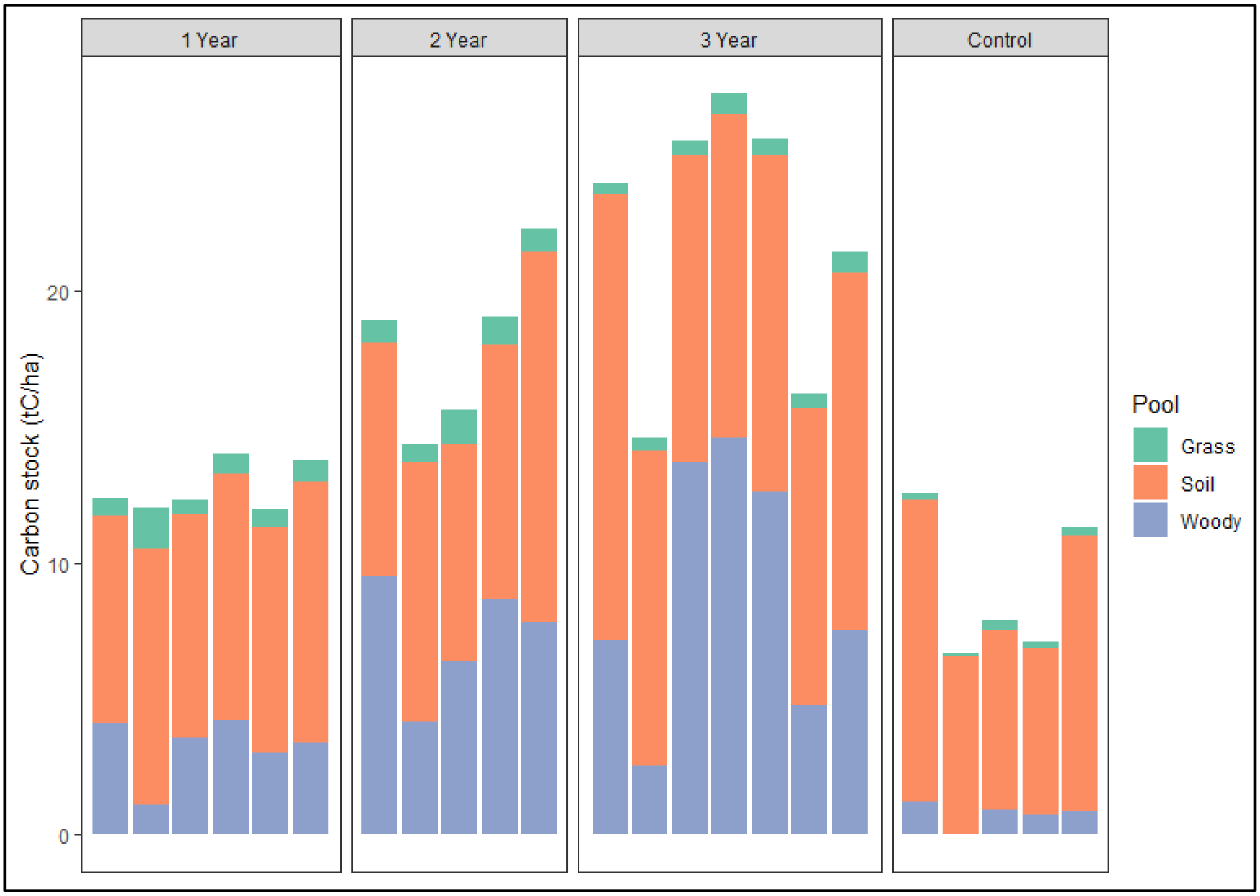

Figure 5: Contribution of each carbon pool in total carbon stock (tC/ha) in each sampling plot across different restoration sites. Each bar represents one sample plot in the respective treatment.

The linear mixed effect model showed that 87% of the residual variation (Intercept, Ψ = 3.67, Residual σ = 1.37) *(Table 3)* was explained by the random intercept term (Plot ID), indicating that the intervention has contributed significantly to observed SOC values for 3-year sites.

Tukey's post-hoc pairwise comparison shows that total SOC stock in the 3-year sites was significantly higher than all other treatments *(Table 4)*. There was a 34% increase in SOC in the 3-year site compared to the no-restoration treatment (β = 4.27, z = 5.02, $p < 0.001$) followed by 30% from the 1-year site (β = 3.82, z = 4.98, $p<0.001$), and 21% increase compared to the 2-year site (β = 2.69, z = 3.33, p = 0.005) *(Figure 6)*.

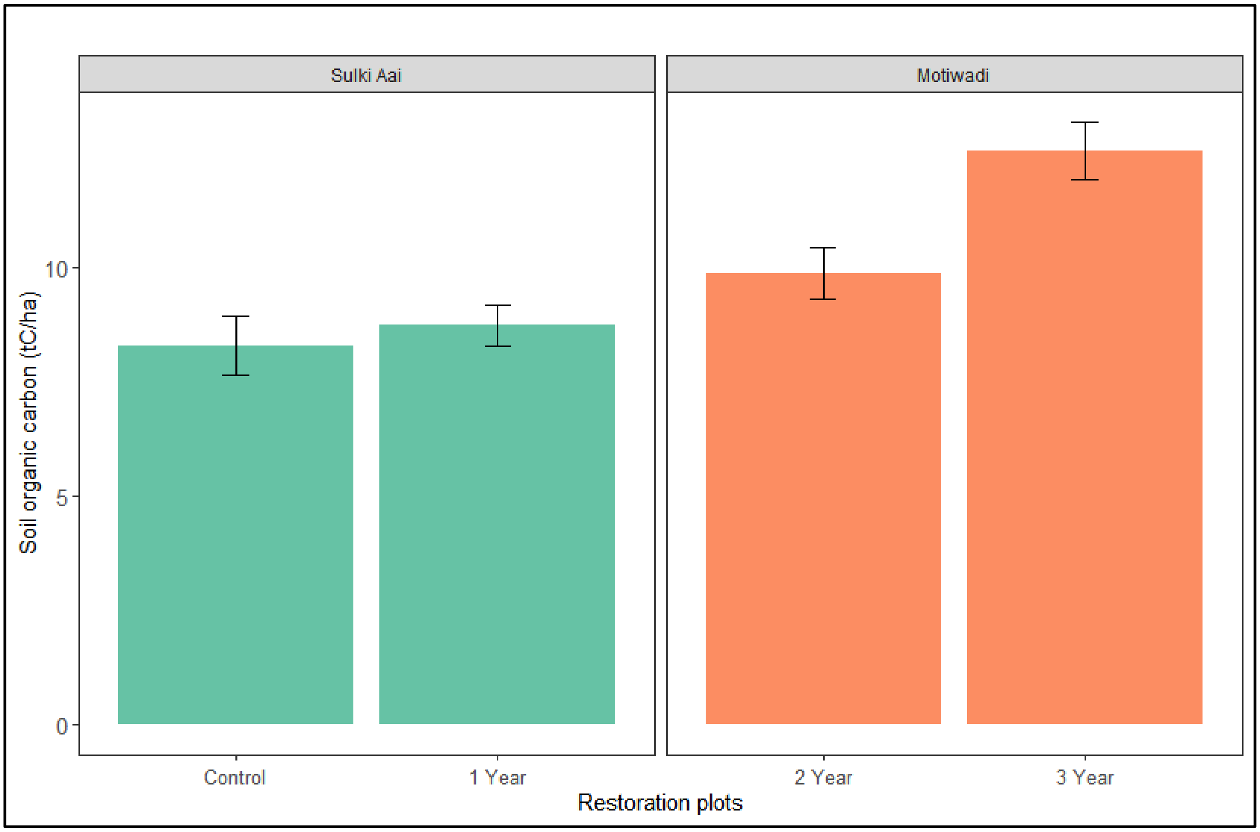

Figure 6: Soil organic carbon (tC/ha) at each restoration treatment paired with sampling sites. Error bars denote standard error in estimated carbon stock (tC/ha).

Table 3: Results of a linear mixed-effect model comparing variation in estimated soil organic carbon across different restoration treatments. Associated β estimates, standard error (S.E.), degrees of freedom (DF), z-statistics, and p-values are shown.

Random effects: ~1 | Plot					
	(Intercept)	Residual			
Std. Dev:	3.658	1.372			
Fixed effects: Carbon tC/ha ~ Treatment					
	Value	Std. Error	DF	t-value	p-value
(Intercept)	8.278	0.670	174.000	12.356	0.000
Treatment 1 Year	0.442	0.876	174.000	0.504	0.615
Treatment 2 year	1.573	0.911	174.000	1.727	0.086
Treatment 3 Year	4.266	0.849	174.000	5.023	0.000

Table 4: Results of Tukey- pairwise comparisons for variation in soil organic carbon. The estimates are differences between the means of two groups along with associated standard error (S.E.), Z-statistic, and p-value. Values in bold show statistically significant differences.

Pair	Estimate	Std. Error	z value	Pr(>|z|)
1-0 year	0.4417	0.8757	0.504	0.95789
2-0 year	1.5733	0.9113	1.727	0.30882
3-0 year	***4.2664***	***0.8493***	***5.023***	***<0.001***
2-1 year	1.1316	0.8363	1.353	0.52793
3-1 year	***3.8247***	***0.7684***	***4.977***	***<0.001***
3-2 year	***2.6931***	***0.8087***	***3.33***	***0.00502***

Discussion

Carbon stock dynamics in semi-arid savanna grasslands

We provide an estimate for biomass in grass and woody vegetation as well as soil carbon stocks in SGs which are currently part of restoration activities initiated by the MFD in Maharashtra. We show that SGs in the region store significant amounts of carbon in their soils despite the semi-arid nature of the study site and the existence of a pronounced and prolonged dry season as compared to other relatively humid ecosystems (Grace et al., 2006; Wang et al., 2010).

Our observed range of SOC stocks of 8.28-12.54 tC/ha fall on the lower end of SOC values found for other SGs in Sub-Saharan Africa, Australia as well as South America observed through field-based observation as well as modeling efforts *(Table 5)*. These differences may be due to soil degradation over time, including the combined impacts of soil disturbances, increased mineralization, leaching losses and variation in shrub/grass species composition, which are known to change the rate of inflows and outflows of SOC as well as other nutrients (Thokchom et al., 2016). Conversely, given the relatively lower mean annual precipitation in our sample sites as compared to similar ecosystems globally, these ecosystems may be considered as highly water-efficient in soil carbon storage.

These differences also point towards the vast carbon storage and sequestration potential that exists if these ecosystems are protected and managed effectively (Griscom et al., 2017; Buisson et al., 2022). However, there remains a dearth of reliable estimates of soil carbon sequestration potentials in such ecosystems. In this context, continuous monitoring over the next several years across both wet and dry seasons would be able to reveal estimates of the long-term carbon sequestration potential in these ecosystems, strengthening ecosystem conservation efforts.

Table 5: Comparison of carbon stocks in biomass and soils from other ecosystems globally.

Carbon stocks (tC/ha)	Carbon pool	Depth (m)	Ecosystem	Region	Mean Annual Precipitation (mm)	Source
Region: Global						
1.8-34	Above-ground biomass (Leaf + wood)	-	Tropical humid + dry savannas	Global	-	(Grace et al., 2006)
18-373	Soil	NA	Tropical humid + dry savannas	Global	-	(Grace et al., 2006)
12.61-17.92	Soil + litter	0.2	Cerrado + shrub savannas	Brazil	1100-1300	(Abreu et al., 2017)
0.79-22.08	Woody vegetation	-	Cerrado + shrub savannas	Brazil	1100-1300	(Abreu et al., 2017)
49-79	Soil	0.35-0.38	Tropical grasslands	Pantropic	474-5100	(Don et al., 2011)
36*	Soil	0.3	Tropical savanna	Nigeria	250-2000	(Akpa et al., 2016)
23.2*	Soil	0.3	Tropical grassland	Nigeria	250-2000	(Akpa et al., 2016)
112.69	Soil	0.2	Tropical savanna	North Australia	1700	(Chen et al., 2005)
79.23	Soil	0.2	Tropical grassland	North Australia	1700	(Chen et al., 2005)
30.02	Soil	0.3	Savanna	Ghana	1400-1800	(Bessah et al., 2016)
22.01	Soil	0.3	Cashew plantations	Ghana	1400-1800	(Bessah et al., 2016)

16.02	Soil	0.2	Woodland savanna	Senegal	350-580	(Elberling et al., 2003)
9.83	Soil	0.2	Grassy savanna	Senegal	350-580	(Elberling et al., 2003)
28.94	Above-ground biomass	-	Humid Guinean savanna	Northern Sierra Leone	2244	(Amara et al., 2019)
41.53	Soil	0.2	Humid Guinean savanna	Northern Sierra Leone	2244	(Amara et al., 2019)
27.26	Soil	0.3	Grassland fallow	Southeastern Nigeria	1550	(Obalum et al., 2012)
10.32	Soil	0.2	Dry grasslands	Senegal	280-400	(Woomer et al., 2004)
Region: India						
8.28-12.54	***Soil***	***0.22***	***Semi-arid savanna grasslands***	***Peninsular India***	***~500***	***This study***
0.24-0.94	***Above-ground biomass (Grass)***	***-***	***Semi-arid savanna grasslands***	***Peninsular India***	***~500***	***This study***
0.72-8.98	***Above-ground biomass (Woody)***	***-***	***Semi-arid savanna grasslands***	***Peninsular India***	***~500***	***This study***
49.2	Soil	0.3	Semi-arid grasslands	Karnataka and Telangana	~900-1150[#]	(Mitran et al., 2018)
1.1-4.1	Above-ground biomass (grass)	-	Humid grasslands	Manipur	1408	(Thokchom et al., 2016)
0.36-8	Above-ground biomass (grass)	-	Semi-arid grasslands	Bundelkhand	~834	(Gupta and Ratan, 2005)
0.16-3.73	Above-ground biomass (grass)	-	Alpine grasslands	Garhwal Himalayas	-	(Dhaulakhandi et al., 2000)

** Adding up SOC values for 0-5, 5-15 and 15-30 cm soil depths.*

[#] *MAP of both states, since the study is state-wide.*

Monitoring carbon recovery in study sites

While we attempted to sample soils up to the standard depth of 30cm, shallow topsoil depths restricted sampling in many cases. We found a weak inverse relationship between SOC and the depth to which we could sample *(Figure 4)*.

This could be due to two factors. First, because of the relatively recent nature of the restoration activity, carbon cycling may be currently occurring at shallower depths, which may eventually percolate deeper. Evidence from managed grassland experiments suggest this to be the case – researchers have previously found that carbon storage is limited to the top 5 cm of soil post a change in land use and land cover for the first 2 years after disturbance, while deeper depths might even witness a loss (Steinbeiss et al., 2008). Further, SOC changes are likely to affect shallow soil layers faster because that is where most of the root production occurs, in line with our observations. These changes based on depth recede with age because soil strata approach equilibrium after 25-40 years (McSherry and Ritchie, 2013).

Two, sampling sites in our analysis have been occasionally plowed by the MFD to lay the groundwork for the digging of bunds and trenches as a water conservation measure. This could lead to the overturning of the soil, leading to the carbon stock-depth relationship we observe. Such management may even lead to subpar annual soil carbon growth rates. In fact, previous evidence from agricultural ecosystems suggest that no-till practices lead to higher SOC growth rates and more stable SOC stocks (World Bank, 2012; Pandey et al., 2014; Modak et al., 2019; Yadav et al., 2019). Adopting similar soil management regimes, including a focus on no-tilling and a complete absence of mechanical methods, can contribute significantly to boosting SOC growth and stability in these SGs as well.

The analysis revealed significant variations in SOC stocks with chrono-sequence. In our paired plots, we found a slow build-up of carbon stocks when comparing the control site with the 1-year restoration site. However, subsequent years demonstrate a relatively substantial increase in carbon stocks. Taken together, there is a 34% increase in carbon stocks from the control sites to the 3-year sites, which may be the combined effects of the intervention, site-specific variability, the variability in grazing access and intensity (if any), among other drivers. While we could not isolate the relative contribution of each, it is likely that restoration has contributed a significant amount to observed SOC changes because of the relatively uniform nature of the other drivers at all sample sites. With increasing time since restoration, other soil properties like the density of organic matter, changes in soil structure and aeration are also likely to occur. All this would determine soil bulk density, and thereby affect SOC. Future studies in SGs in the region can aim to systematically control for all these variables when attributing SOC changes to respective drivers. Observed increments in SOC may only be permanent in case of grazing management and protection from woody encroachment (for example, due to tree-planting programmes) since evidence suggests that SOC, although resilient to fire, disease and droughts, can be lost due to such changes due to nutrient depletion and shifts in vegetation composition (Buisson et al., 2022).

Above-ground litter is one of the most important determinants of SOC in SG since it impacts carbon accumulation below-ground (Wang et al., 2013). Recent evidence suggests that the upper layer of the soils (which we assessed) are mostly influenced by the aboveground and belowground litter productions and its subsequent decomposition, while carbon at deeper depths is influenced by land use legacies (Nath et al., 2018). In our sample plots, above-ground grass biomass stocks ranged from 0.24-0.94 tC/ha. These stocks are transient in nature. This is lower than the estimates for humid *Imperata* grasslands in Northeast India (approx.1.1-4.1 tC/ha) (Thokchom et al., 2016), but comparable to estimates in the Bundelkhand region (0.36-8 tC/ha) (Gupta and Ratan, 2005) and in the alpine grasslands of the Western Garhwal Himalayas (0.16-3.73 tC/ha) (Dhaulakhandi et al., 2000). Observed values are also on the lower end of values from corresponding values for litter in global savanna ecosystems, which can range from as low as 0.2 tC/ha to as high as 22.5 tC/ha (Grace et al., 2006). However, it is important to note that our study was conducted in the peak dry season (March 2023), where environmental conditions are hot and dry and coincide with minimum observed grass biomass stocks.

On the other hand, the range of above-ground carbon in woody vegetation was 0.72-8.98 tC/ha, which falls on the lower end of observed woody biomass estimates for global savanna ecosystems (Grace et al., 2006). This is not surprising, given the aridity and poor soil quality in our sample sites. Furthermore, these low values provide further grounds to question the suitability of planting trees in SGs as a carbon sequestration tool - recent evidence suggests that grasses contribute the highest to increasing SOC in savanna ecosystems and increasing tree cover has little, if any, impacts on increasing SOC (Zhou and Staver, 2022).

Strategies for grassland restoration

SGs have existed across the Cenozoic Era globally (from 66 million years ago) and for at least 1 million years, going up to even 10 million years in India (Ratnam et al., 2016). There is now a global and national recognition of the importance of these unique ecosystems, which has been provided further impetus by the declaration of the UN Decade of Ecosystem Restoration (2021-2030). At the national level, however, there are mixed signals. While India has also signed up for ambitious Land Degradation Neutrality targets, SGs do not feature in India's NDC despite their vast expanse, partly because of colonial legacies of being termed as 'wastelands' (Madhusudan and Vanak, 2023).

Concurrently, soil carbon sequestration is also now recognized as a natural climate solution. In international climate governance, the need to enhance SOC across land uses and across ecosystems is now a common denominator, highlighted by the adoption of the *4 per Mille* Initiative at COP21 in Paris in 2015 (Minasny et al., 2017) and the formal recognition of SOC sequestration at COP23 in 2017 (COP23 decision 4/CP.23) (Bossio et al., 2020).

Grassland restoration can deliver on a two-pronged strategy – conserving existing stocks (avoiding losses) and restoring stocks in carbon-depleted soils. This could be done through employing proper management strategies including rotational grazing and reducing ecosystem conversion (Padbhushan et al., 2020; Bai and Cotrufo, 2022). Meeting these objectives can deliver additional co-benefits including, but not limited to, (1) increasing soil fertility and reducing soil erosion, (2) maintaining or increasing resilience to climate change for communities who derive livelihood benefits from these ecosystems and (3) providing habitat to endemic species. All these actions are in line with the UN Sustainable Development Goals, the UN Convention on Combating Desertification (UNCCD) and the Global Biodiversity Framework.

However, a question remains, as to what reference should SGs be restored to deliver these soil carbon sequestration benefits.

From an ecological perspective, attaining old-growth characteristics is the ultimate objective of any restoration initiative. Restoration in SGs (like our study site) should aim to have long-lived perennial plants; a complex diversity of below-ground structures that enable re-sprouting after above-ground disturbances such as fire and grazing occur; and substantial below-ground carbon stores, which are characteristics typical to old-growth SGs (Buisson et al., 2022).

Attaining these characteristics is not straightforward. Whereas the destruction and degradation of SGs can occur rapidly, recent work indicates that complete recovery of carbon storage potential and essential structure, composition, and functions occurs slowly (Buisson et al., 2022), often in the order of decades or even centuries (Nerlekar and Veldman, 2020).

Increasing carbon storage in SGs would also have to account for saturation and non-permanence in soils. SOC saturation refers to a maximum capacity of the soil to retain organic carbon, meaning that SOC does not increase indefinitely. Soils saturate at timescales of a few decades and reach a new steady state. The time when saturation is achieved is also determined by the soil type, management intervention, climate regime and pre-existing SOC depletion. The comparison of our results with national and global averages *(Table 5)* reveals that a saturation point may still be sufficiently far, implying that SGs in the region can keep delivering carbon benefits for realistic future timeframes at the very least.

With respect to non-permanence, maintaining high SOC stocks requires some form of protection and management, even after saturation is achieved and no further mitigation benefits accrue. Since SG sites in the region are under the management of the MFD and protected from conversion under law, it can be expected that it remains stable at realistic multi-decadal timescales if protection is encouraged and sustained (Bossio et al., 2020). Our efforts in this study are also just a sample of the time and rigor required to conclusively attribute the changes in carbon stocks to the restoration activity. This is especially so because short-term monitoring may lead to different conclusions about the efficacy of restoration itself (Török et al., 2021).

The need for a flexible carbon measurement protocol for SGs

A considerable effort has been made by the global scientific community to measure the amount of SOC using a variety of techniques including both ex-situ as well as in-situ methods (Eggleston et al., 2006; Kurniatun et al., 2010; Marthews et al., 2014; FAO, 2019; Vagen and Winowiecki, 2023). These approaches have attempted to come up with standardized approaches to monitor and evaluate carbon stocks as well as fluxes in diverse ecosystems (Stockmann et al., 2013). However, these approaches, while tailored for a global audience, are often found to be disproportionately focused on forest ecosystems for carbon measurements. This makes them unfit for use in many tropical non-forest ecosystems – like SGs – where there are unique logistical and protocol-based constraints.

For example, guidelines often state that soil sampling should ideally be performed up to a soil depth of 1m, and at least up to a depth of 30 cm. However, soils in SGs, especially in degraded SGs, are often extremely shallow and may not allow soil cores to be collected to prescribed depths.

Therefore, we adapted existing field protocols to come up with a flexible approach developed specifically for SOC measurements for SGs in this study. It is designed to provide carbon measurements at plot-level, which if repeated at regular intervals in wet and dry seasons, can demonstrate changes in carbon fluxes in these ecosystems. Going forward, it is crucial that region and ecosystem-specific tools, approaches and guidelines are developed which can take contextual challenges into account as well as are flexible and cost-effective to verify and potentially monitor carbon fluxes in ecosystems like SGs.

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Disclosure statement

The authors report no conflict of interest.

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