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Land Use Effects on Soil Biological Properties in the Hill Ecosystems of Assam, Eastern Himalayan Region of India

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ABSTRACT

This study investigates the impact of land-use changes on soil microbial properties in the hilly terrain of Karbi Anglong district of Assam, Eastern Himalayan region, India. Specifically, it examines the effects of converting natural forest into various agricultural systems, including *jhum* cultivation, rubber plantations, rice/maize croplands, home gardens, and bamboo plantations. Forest soils consistently exhibited the highest MBC, MBN, and DHA levels in all three soil depths (0–15 cm, 15–30 cm, and 30–50 cm), followed by home gardens, while rice/maize croplands recorded the lowest values. Conversion of natural forest into other land uses led to a substantial reduction in surface soil MBC (up to 64.59%) and MBN (up to 38.62%), indicating degradation of soil biological quality. Forest and home garden systems, with richer vegetation and organic inputs, promoted higher microbial activity and nutrient cycling, while cultivated lands and monoculture plantations showed reduced microbial indicators. The decline in microbial attributes with increasing soil depth further highlights the influence of organic carbon availability. Overall, the study emphasizes the sensitivity of microbial indicators to land-use changes and their potential as early markers of soil health and fertility.

Keywords Land use, Microbial biomass carbon, Microbial biomass nitrogen, Dehydrogenase activity.

INTRODUCTION

Biological and ecological indicators are generally more dynamic measures of soil quality than physical or chemical indicators. Growing evidence highlights that microbial attributes serve as sensitive early warning signals of shifts in soil health (Geisseler & Horwath 2009). Alterations in microbial communities can thus indicate early stages of soil degradation or recovery (Vallejo *et al.* 2012). Key microbial indicators such as potentially mineralizable nitrogen (PMN), microbial biomass carbon (MBC), microbial biomass nitrogen (MBN), soil respiration, enzyme activity, and earthworm populations are recognized for their strong predictive value in assessing soil quality (Jenkinson

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& Parry 1989). These indicators are critical because microbial activity directly influence soil fertility and ecosystem resilience (Yang *et al.* 2010).

Among soil enzymes, dehydrogenases stand out as key indicators of overall microbial activity (Quilchano & Marañon 2002). These enzymes are present in all living microbial cells and function intracellularly (Moeskops *et al.* 2010, Yuan & Yue 2012), making them reliable markers of active microbial populations. Dehydrogenases play a vital role in soil respiration and the biological oxidation of organic matter, transferring hydrogen from organic substrates to inorganic electron acceptors (Zhang *et al.* 2010). Dehydrogenase activity (DHA) is highly sensitive to both natural and anthropogenic influences, such as soil aggregation, aeration (Brezezinska *et al.* 2001), organic carbon levels (Gajananda 2007), vegetation cover (Bastida *et al*. 2006), and agricultural practices. As a result, DHA is widely recognized as a valid biomarker for detecting shifts in soil biological activity in response to changes in land management.

The hill zone of Assam, located in the Eastern Himalayan Region of India, is facing increasing ecological stress due to rapid land-use changes, unsustainable agricultural practices, and the persistence of traditional shifting cultivation in fragile hill ecosystems (Shimrah *et al.* 2015). Extensive deforestation caused by slash-and-burn agriculture has led to severe degradation of hilly slopes, as local tribal communities continue this traditional practice. While the adverse effects of forest conversion on soil properties and soil organic carbon (SOC) are well documented (Kalita *et al*. 2023), there remains a significant gap in knowledge regarding soil microbial characteristics under the region's diverse land-use systems. Understanding the relationship between land-use change and microbial indicators is critical for designing sustainable land-use strategies aimed at preserving ecosystem stability in these vulnerable hill regions.

This study aims to assess soil microbial properties across various land-use systems, with a particular focus on the impact of forest conversion to agricultural land in the hilly terrain of Karbi Anglong district, part of the Eastern Himalayan region of Assam, India.

MATERIALS AND METHODS

The study was conducted in the Karbi Anglong district of Assam, Northeast India, located between 25°32′N to 26°36′N latitude and 92°10′E to 93°50′E longitude. Soil samples were collected during November–December 2018 from five land-use systems: Rubber plantation, *jhum* land, crop land, home garden, and bamboo plantation. For each land-use type, samples were collected from three plots in each of three different locations, totaling nine plots per land-use system. Additionally, soil samples were obtained from four randomly selected plots within forested areas adjacent to the study villages.

A purposive sampling approach was used, ensuring that sampled plots across different land-use types were situated in close proximity to minimize variability due to climate, topography, and soil type. In total, 135 soil samples were collected from the five land-use types (5 land uses×3 locations × 3 replications × 3 soil depths), and 27 samples were collected from forest plots (3 locations × 3 replications × 3 depths), resulting in a combined total of 162 soil samples.

The soil samples were maintained at 4^0C and brought to ambient temperature before estimation of each individual biological parameter including Microbial biomass carbon, Microbial biomass nitrogen and Dehydrogenase activity. The soil chemical properties were estimated as per standard procedures and the results are presented in Table 1.

Microbial biomass carbon (MBC) was estimated using the chloroform fumigation-extraction method as described by Vance *et al.* (1987). Fresh soil samples (10 g) were placed in 50 mL glass beakers and positioned inside a desiccator containing a vial of soda lime to absorb released CO_2. A separate beaker containing 100 mL of ethanol-free chloroform ($CHCl_3$) was also placed in the desiccator, which was then sealed with vacuum grease. The desiccator was evacuated until the chloroform boiled vigorously, initiating the fumigation process. It was then incubated in the dark at 25°C for 24 hours.

Following incubation, residual $CHCl_3$ was removed by repeated evacuation. The fumigated soils

were then extracted with 25 mL of 0.5 M K_2SO_4 (soil-to-extractant ratio of 1:5) by shaking for 30 minutes, and the extracts were filtered using Whatman No. 42 filter paper. Unfumigated control samples were prepared in parallel by extracting soil without prior fumigation.

Organic carbon (OC) in the extracts was determined via the dichromate oxidation method (Walkley and Black 1934). MBC was calculated as the difference in extractable OC between the fumigated and non-fumigated soil and expressed in µg g^{-1} on a dry weight basis (Equation-1).

MBC (µg g^{-1}) = Ec/k_{EC} ……(Equation -1)

Where,

EC = (OC extracted from fumigated soil) - (OC extracted from non fumigated soil)

K_{EC} = 0.38 (Vance *et al.* 1987)

The microbial biomass nitrogen was determined from the soil as described by Keeney and Nelson (1982). The fumigated and non-fumigated samples after ten days incubation were extracted for inorganic nitrogen with 2M KCI at 1:5 soil to extractant ratio. The incubated samples were extracted with 100 ml of 2M KCI solution after shaking for one hour. The supernatant was filtered through Whatman No.1 filter paper. Twenty-five milliliter aliquot of the filtrate was analyzed for inorganic nitrogen following Kjeldahal distillation. The amount of nitrogen mineralized by microbial component was determined as a difference of inorganic nitrogen released by fumigated and non-fumigated soil samples.

The microbial biomass nitrogen (MBN) was calculated as given by Jenkinson (1988) and was expressed as mg per kg of dry soil (Equation -2).

MBN (mg kg^{-1})=[Mineral nitrogen in (fumigated soil-non-fumigated soil)]/ 0.57 …Equation -2)

Mineral N (mg kg^{-1}) = [70 * (Vs-Vb)* Ve] / Vf * W

Vs = Volume of 0.005 N H_2SO_4 used for the sample
Vb = Volume of 0.005 N H_2S0_4 used for the blank
Vf = Volume filtrate used for distillation (25 ml)
W = Weight of soil used for incubation (20 g\m)
Ve = Volume of 2N KCl used for extraction (100 ml)
0.57 - Proportion of microbial N mineralized to NH_4^+
1 ml of 0.005 N H_2SO_4 = 70 mg of mineral N.

Dehydrogenase activity (DHA) was determined by the reduction of triphenyl tetrazolium chloride (TTC) to trriphenyl formazan (TPF) as described by Casida *et al.* (1964). Moist soil (10 g) was treated with 10 ml of 3% TTC, and then incubated at 32^0C for 7 days in screw cap test tube (30 ml). After incubation period, the soil was extracted by addition of 10 ml extractant (methanol) following incubation in dark and agitated for 1 hour. The mixture was then filtered using Whatman No. 42 filter paper and 1 ml filtrate was transferred to 1.55 ml micro centrifuge tube and centrifuged at 5000 rpm for 5 min. Absorbance of the supernatant was measured in Nanodrop 1000 Spectrophotometer at $OD_{485\ nm}$. To account for any abiotic TTC reduction, sterile controls consisted of autoclaved soil (121^0C, 20 min. for three consecutive days) to which 10 mL of TTC was added. Spectrophometer blanks consisted of 10 g soil and TTC replaced with 10 mL Millipore water. Except for addition of Millipore water in blank and autoclaving in control, they were treated samples for the rest of the procedure. A calibration curve was constructed by determining $OD_{485\ nm}$ values for the working standard of TPF (209, 40, 80, 120, 200, 300 and 500 µg ml^{-1}). The $Od_{485\ nm}$ was compared to that of TPF standards. DH activity was expressed on dry weight as µg TPF g^{-1} 24 h^{-1} on dry weight basis as calculated by Equation- 3.

$$\text{DH activity (as µg TPF g}^{-1}\text{ 24 h}^{-1}) = \frac{[(\text{TPFs}) - \text{TPFc})] \times 20}{\text{edw}}$$

Where, TPFs = TPF conc. (µg ml^{-1}) in the sample; TPFc = TPF conc. (µg ml^{-1}) in the sterile control; edw is the equivalent dry weight of 1 g soil, 20 is the volume of solution added in the assay (TTC + extractant).

Statistical analysis

Descriptive statistical analysis was carried out using SPSS version 17.0. An analysis of variance

(ANOVA) was done to evaluate if different land uses have significant impact on soil biological properties and significant effect ($p<0.05$) was determined with DMRT post hoc multiple comparisons.

RESULTS AND DISCUSSION

Soil organic carbon and nutrient status under different land uses

Soil organic carbon (SOC) showed a statistically significant change ($p<0.05$) across different land uses and soil depths (Table 1). In forest and other tree-based systems, SOC significantly decreased with increasing depth. In contrast, *jhum* and croplands did not exhibit a significant decline in SOC with depth, likely due to greater SOC loss in the surface layers caused by continuous cropping. Among the various land use types, natural forests recorded the highest SOC levels (1.18%), whereas cultivated lands had the lowest (0.56%) in the surface soil.

As depth increased, the differences in SOC among land uses diminished. At the 30–50 cm depth, SOC levels in forest, jhum land, rubber plantations, and home gardens were comparable, as were those in rice/maize croplands and bamboo plantations. The higher SOC in natural forests and tree-based systems can be attributed to the consistent annual input of organic matter from leaf litter and root biomass, along with minimal soil disturbance, which enhances organic carbon accumulation. Additionally, lower surface soil temperatures and increased biological activity in these systems further contribute to higher SOC in surface layers compared to systems under regular cultivation (Six *et al.* 2002). In cultivated lands, SOC depletion is primarily due to increased aeration and the breakdown of soil aggregates during tillage, which exposes previously protected organic carbon to microbial degradation (Kong *et al.* 2005). Home gardens maintained relatively higher SOC levels, possibly due to the continuous addition of organic matter from diverse crop residues and animal manure. Similar trends have been observed in previous studies (Yao *et al.* 2010, Dutta *et al.* 2015), which reported declines in SOC following the conversion of natural forests to grasslands, fallows, horticultural areas, or agricultural land. The trends observed in soil available nitrogen (N), phosphorus (P), and potassium (K) followed a similar pattern to that of SOC.

The elevated levels of available nitrogen (N) in forest soils are likely driven by the higher soil organic carbon (SOC) content, resulting from greater biomass accumulation through litter fall and root deposition. This relationship is strongly supported by a significant positive correlation between N and SOC ($r = 0.951^{**}$, Table 2). Similarly, the significantly higher availability of phosphorus (P) in forest soils and tree-based land-use systems can be attributed to

Table 1. Organic carbon and nutrient status of soil as affected by different land uses.

Soil attributes	Depth (cm)	Natural forest	*Jhum* land	Rubber plantation	Crop land	Home garden	Bamboo plantation
SOC (%)	0-15	1.18^{aA}	0.58^{bA}	0.82^{cA}	0.56^{bA}	0.94^{dA}	0.94^{dA}
	15-30	0.76^{aB}	0.53^{bA}	0.67^{cB}	0.40^{dB}	0.62^{bcB}	0.54^{bB}
	30-50	0.48^{aC}	0.44^{aB}	0.46^{aC}	0.34^{bB}	0.45^{aC}	0.32^{bC}
Availablr N	0-15	561.68^{aA}	239.96^{bA}	359.53^{cA}	226.21^{bA}	365.06^{cA}	417.96^{cA}
(kg ha^{-1})	15-30	307.38^{aB}	269.46^{bcdB}	290.01^{adAB}	246.28^{bA}	288.26^{acdB}	262.03^{bcB}
	30-50	241.66^{aC}	230.86^{aA}	224.52^{aB}	195.91^{bB}	237.45^{aC}	179.61^{bC}
Available	0-15	26.3^{aA}	13.87^{bcA}	17.89^{cdA}	11.31^{bA}	16.43^{dA}	13.15^{bA}
P_2O_5 (kg ha^{-1})	15-30	21.68^{aAB}	10.80^{bcA}	14.69^{dAB}	8.35^{bB}	14.35^{cdAB}	9.16^{bB}
	30-50	16.77^{aB}	13.74^{bA}	11.73^{bcB}	6.48^{dB}	10.36^{cB}	5.85^{dC}
Available	0-15	312.46^{aA}	185.19^{bcA}	204.83^{cA}	158.10^{bA}	196.38^{bcA}	168.29^{bcA}
K_2O (kg ha^{-1})	15-30	243.12^{aB}	122.18^{bcB}	135.49^{bB}	101.07^{cB}	134.14^{bB}	96.09^{cB}
	30-50	199.75^{aB}	125.68^{bB}	131.73^{bB}	81.86^{cdB}	108.08^{bcC}	70.48^{dC}

* Mean values within a row followed by the same letter (lower case) are not significantly different among the land uses at p = 0.05 by DMRT technique, * Mean values within a column for the individual soil attributes followed by the same letter (upper case) are not significantly different among the depths at p = 0.05 by DMRT technique.

Table 2. Correlation matrix among the chemical and biological parameters of soils under different land uses.

	SOC	N	P	K	MBC	MBN	DHA
SOC	1						
N	0.951**	1					
P	0.807**	0.774**	1				
K	0.825**	0.782**	0.957**	1			
MBC	0.916**	0.843**	0.742**	0.809**	1		
MBN	0.918**	0.831**	0.731**	0.805**	0.997**	1	
DHA	0.883**	0.837**	0.668**	0.779**	0.950**	0.954**	1

efficient nutrient recycling, where deep-rooted tree species extract P from subsurface layers and return it to the surface via litter fall (Panwar *et al*. 2011). Additionally, the increased P availability may also result from the formation of organophosphate complexes facilitated by higher SOC levels. This is corroborated by the significant positive correlation observed between available P and SOC (r = 0.807**, Table 2). Furthermore, the higher surface-level potassium (K) in forest lands can be linked to the continual input of organic matter from litter fall and the minimal soil disturbance from erosive forces such as raindrop impact and surface runoff.

Microbial biomass carbon (MBC)

The results of microbial biomass carbon analysis (Table 3) show that forest land has the highest mean MBC of 259.75 µg C g^{-1} followed by home garden (132.62 µg C g^{-1}), rubber plantation (129.20 µg C g^{-1}), *Jhum* land (118.28 µg C g^{-1}), bamboo plantation (93.09 µg C g^{-1}) and lowest was recorded in rice/maize crop land (91.98 µg C g^{-1}) in the surface soil. The conversion of natural forest into jhum cultivation, rubber plantation, rice/maize crop land, home garden and bamboo plantation resulted in a significant decrease in the MBC contents in the surface soils, with the reduced values being 54.46%, 50.26%, 64.59%, 48.94%, and 64.16%, respectively. In 15-30 cm depth, all the land uses showed similar trend. In 30-50 cm soil depth, although the forest soil (103.20 µg g^{-1}) recorded the highest MBC content but it was observed at par with jhum land (91.55 µg g^{-1}), rubber plantation (98.33 µg g^{-1}) and home garden (92.59 µg g^{-1}) whereas bamboo plantation (72.56 µg g^{-1}) recorded the lowest MBC content which was statistically at par with rice/maize crop land (83.54 µg g^{-1}).

Forested soils, characterized by dense vegetation, abundant litterfall, and fine root biomass, generally support higher microbial populations due to greater availability of organic carbon (C) and nitrogen (N) (Kara & Bolat 2008, Chen *et al.* 2010). In contrast, cultivated lands, rubber, and bamboo plantations typically exhibit lower microbial biomass carbon (MBC) levels, likely due to reduced plant diversity, lower soil organic carbon and nutrient content. The richness of plant species and aboveground biomass strongly influence microbial biomass, as reported by Liu *et al.* (2008). These trends align with findings by Pramod *et al.* (2012), who observed the highest MBC in forest soils (430.7 µg C g^{-1}), followed by horticultural systems (355.5 µg C g^{-1}) and agricultural land (257.8 µg C g^{-1}).

Soil microorganisms drive essential processes that regulate soil quality, and microbial biomass responds rapidly to shifts in soil conditions. An increase in microbial biomass is generally linked to improved soil biological function and elevated organic carbon content, while a decline often signals soil degradation. Because microbial populations are mostly heterotrophic, their activity and distribution are dependent on organic matter availability, making MBC a sensitive and early indicator of changes in SOC and total nitrogen (TN) (Moscatelli *et al.* 2007, Yang *et al.* 2010). The correlation analysis (Table 2) shows that MBC is positively correlated with SOC (r= 0.916**) and available N (r = 0.843**).

The home garden system tends to support higher microbial biomass than other land-use types, likely

Table 3. Microbiological properties of soils under different land uses in hill region of Assam.

Land uses	MBC (µg g^{-1})			MBN (µg g^{-1})			DHA (µg TPF g^{-1} soil 24 h^{-1})		
	0-15 cm	15-30 cm	30-50 cm	0-15 cm	15-30 cm	30-50 cm	0-15 cm	15-30 cm	30-50 cm
Natural forest	259.75^{aA}	136.04^{aB}	103.20^{aC}	32.08^{aA}	18.82^{aB}	14.26^{aC}	56.52^{aA}	16.88^{aB}	9.91^{aC}
Jhum land	175.23^{cA}	118.28^{aB}	91.55^{abB}	22.96^{bA}	15.77^{bB}	12.89^{abB}	37.20^{bA}	13.74^{bB}	8.78^{abC}
Rubber plantation	218.30^{bA}	129.20^{aB}	98.33^{aC}	27.50^{cA}	16.40^{bcB}	13.88^{aC}	28.50^{cA}	11.62^{cB}	7.67^{bcC}
Rice/maize cropland	143.92^{dA}	91.98^{bB}	83.54^{bcB}	19.69^{bA}	12.68^{dB}	11.25^{bcB}	22.12^{dA}	9.62^{cB}	6.43^{cdC}
Home garden	239.29^{abA}	132.62^{aB}	92.59^{abC}	31.38^{aA}	17.69^{acB}	13.50^{aC}	44.17^{eA}	14.47^{bB}	8.22^{bC}
Bamboo plantation	171.99^{cA}	93.00^{bB}	72.56^{cB}	23.15^{bA}	13.75^{dB}	10.65^{cC}	35.99^{bA}	10.46^{cB}	5.43^{dC}

Abbreviation: MBC: Microbial biomass carbon, MBN: Microbial biomass nitrogen, DHA: Dehydrogenase activity,
*Mean values within a column followed by the same letter (lower case) are not significantly different among the land uses at p = 0.05 by DMRT technique,
*Mean values within a row followed by the same letter (upper case) are not significantly different among the depths at p = 0.05 by DMRT technique.

due to improved soil conditions and the greater quantity and diversity of plant litter. Residues from multiple tree species promote a more diverse microbial community, which enhances nutrient cycling and boosts microbial biomass (Yadav *et al.* 2011, Bargali *et al.* 2015).

A significant decline in MBC with increasing soil depth is commonly observed, primarily due to the reduction in SOC content at deeper layers. Microbial biomass is highly responsive to even slight changes in organic matter, which serves as a key energy source for microbial life.

Microbial biomass nitrogen (MBN)

The mean microbial biomass nitrogen (MBN) content (Table 3) in surface soils (0–15 cm) varied across land use types, with the highest value recorded in forest soils (32.08 µg g^{-1}), followed by home gardens (31.38 µg g^{-1}), rubber plantations (27.50 µg g^{-1}), bamboo plantations (23.15 µg g^{-1}), *jhum* lands (22.96 µg g^{-1}), and the lowest in rice/maize croplands (19.69 µg g^{-1}).

At the 15–30 cm depth, MBN levels declined across all land uses but remained highest in forest soils (18.82 µg g^{-1}), followed by home gardens (17.69 µg g^{-1}), rubber plantations (16.40 µg g^{-1}), *jhum* lands (15.77 µg g^{-1}), bamboo plantations (13.70 µg g^{-1}), and lowest again in rice/maize croplands (12.68 µg g^{-1}). In the subsurface layer (30–50 cm), forest soils continued to exhibit the highest MBN (14.26 µg g^{-1}), while bamboo plantations recorded the lowest (10.65 µg g^{-1}). Intermediate values were observed in rubber plantations (13.88 µg g^{-1}), home gardens (13.50 µg g^{-1}), *jhum* lands (12.89 µg g^{-1}), and rice/maize croplands (11.25 µg g^{-1}).

The analysis of soil microbial biomass nitrogen revealed that forest soils exhibited the highest levels of microbial biomass nitrogen (MBN), followed in descending order by home gardens, rubber plantations, *jhum* lands, bamboo plantations, and rice/maize croplands, which had the lowest MBN. A one-way ANOVA confirmed that these differences in MBN among the six land-use types were statistically significant. MBN followed a trend similar to that of microbial biomass carbon (MBC), suggesting a strong linkage between nitrogen and carbon dynamics in mineral soils. This is consistent with the fact that most nitrogen exists in organic forms and within heterotrophic microbial biomass, which relies on organic carbon as an energy source (Padalia *et al.* 2018). Pearson's correlation analysis (Table 2) further supported this relationship, revealing strong positive correlations of MBN with MBC ($r = 0.997^{**}$) and organic carbon ($r = 0.918^{**}$). Forest and home garden soils consistently exhibited significantly higher MBN levels compared to plantations and croplands. This observation aligns with the higher organic carbon and nitrogen content typically found in forest soils. Previous studies have established that soil organic carbon (SOC) plays a pivotal role in determining the size and activity of soil microbial biomass (Diaz-Ravina *et al.* 1988, Jenkinson 1988). The relatively dense vegetation, abundant litter fall, and rich fine root systems in the understory

of forests and home gardens likely create favorable conditions for microbial proliferation. MBN serves as an important indicator of organic matter mineralization, its quality, and the overall fertility status of the soil. The close association between carbon and nitrogen in soil microbial processes is underscored by the strong positive correlation observed between MBN and MBC.

Dehydrogenase activity (DHA)

DHA exhibited significant variation across different land-use types and soil depths (Table 3). In the surface soil (0–15 cm), the highest mean dehydrogenase activity (DHA) was recorded in forest soils (56.52 µg TPF g^{-1} soil 24 h^{-1}), followed by home gardens (44.17 µg TPF g^{-1} soil 24 h^{-1}), *jhum* lands (37.20 µg TPF g^{-1} soil 24 h^{-1}), bamboo plantations (35.99 µg TPF g^{-1} soil 24 h^{-1}), rubber plantations (28.50 µg TPF g^{-1} soil 24 h^{-1}), and the lowest in rice/maize croplands (22.12 µg TPF g^{-1} soil 24 h^{-1}).

At the 30–50 cm depth, forest soils again demonstrated the highest DHA (9.91 µg TPF g^{-1} soil 24 h^{-1}), which was statistically comparable to that of *jhum* land (8.78 µg TPF g^{-1} soil 24 h^{-1}). In contrast, the lowest DHA was observed in bamboo plantations (5.43 µg TPF g^{-1} soil 24 h^{-1}), which did not differ significantly from the rice/maize cropland soils (6.43 µg TPF g^{-1} soil 24 h^{-1}). The higher DHA observed in forest and home garden soils is likely attributable to greater substrate availability and higher levels of soil organic carbon (SOC), which provide essential energy and nutrients for microbial communities. Dehydrogenase activity (DHA) reflects the overall oxidative activity of soil microflora and serves as a reliable indicator of microbial activity (Nannipieri *et al.* 2002). Consequently, higher DHA under forest and home garden land uses suggests enhanced biological activity and stabilization of extracellular enzymes through complexation with humic substances (Colvan *et al*. 2001). In contrast, DHA levels were lower in cultivated lands, *jhum* systems, rubber plantations, and bamboo plantations. This reduction may result from lower plant diversity, reduced SOC and nutrient availability, and the impact of intensive tillage and other cultural practices that increase aeration, accelerating the decomposition of organic matter.

Across all land uses, DHA was significantly higher in the surface soil layer (0–15 cm) compared to sub-surface layers (15–30 cm and 30–50 cm). The decline in DHA with depth can be attributed to reduced nutrient availability, lower aeration, diminished rhizodeposition, and a decrease in easily decomposable organic matter at deeper soil levels (Adak *et al*. 2014, Debnath *et al.* 2015).

DHA showed strong positive correlations with SOC ($r = 0.883^{**}$) and microbial biomass carbon (MBC) ($r = 0.950^{**}$) (Table 2). The significant correlation with SOC is likely due to enhanced microbial activity and enzyme stabilization through the formation of humus-enzyme and clay-enzyme complexes, which protect enzymatic functions in the soil (Klose and Tabatabai 1999). Similarly, the positive correlation between DHA and MBC indicates that active microorganisms—key producers of soil enzymes—are closely linked to DHA levels (Okur *et al.* 2009).

CONCLUSION

The conversion of natural forest into alternative land uses has significantly altered soil microbial properties. Microbial analyses revealed that forest soils exhibited the highest levels of microbial biomass carbon (MBC), microbial biomass nitrogen (MBN), and dehydrogenase activity (DHA), followed in descending order by home gardens, rubber plantations, *jhum* lands, bamboo plantations, and rice/maize croplands. The relatively dense vegetation, higher litterfall, and fine root accumulation in forest ecosystems contribute to increased availability of organic carbon and nitrogen, fostering microbial growth and enhancing carbon sequestration within the microbial biomass. This is supported by the strong positive correlation observed between MBN and organic carbon. The higher DHA levels observed in forest and home garden soils further indicate increased biological activity within these land-use systems. The stabilization of extracellular enzymes through complexation with humic substances likely contributes to this enhanced enzymatic function.

REFERENCES

Adak, T., Singha, A., Kumar, K., Shukla, S. K., Singh, A., & Singh, V. K. (2014). Soil organic carbon, dehydrogenase activity, nutrient availability and leaf nutrient content as affected by organic and inorganic source of nutrient in mango orchard soil. *Journal of Soil Science and Plant Nutrition*, 14, 394-406.

Bargali, S. S., Shukla, K., Singh, L., Ghosh, L., & Lakhera, M. L. (2015). Leaf litter decomposition and nutrient dynamics in four tree species of dry tropical forest. *Tropical Ecology,* 56 (1), 191–200.

Bastida, F., Moreno, J. L., Hernandez, T., & Graccia, C. (2006). Microbiological activity in a soil 15 years after its devegetation. *Soil Biology & Biochemistry,* 38, 2503-2507.

Brezezinska, M., Stepniewska, Z., & Stepniewski, W. (2001). Dehydrogenase and catalase activity of soil irrigated with municipal waste water. *Polish Journal of Environment Studies,* 10, 307-311.

Casida, L. E. Jr., Klein, D. A., & Santoro, R. (1964). Soil dehydrogenase activity. *Soil Science,* 98, 371-378.

Chen, D. D., Zhang, S. H., & Dong, S. K. (2010). Effect of land-use on soil nutrients and microbial biomass of an alpine region on the northeastern Tibetan Plateau, China. *Land Degradation & Development,* 21(5), 446-452.

Colvan, S. R., Syerrs, J. K., & Donnel, A. G. (2001). Effect of long term fertilizer use on acid and alkaline phosphormonoes terase activities in managed grassland. *Biology and Fertility of Soils,* 34, 258-263.

Debnath, S., Patra, A. K., Ahmed, N., Kumar, S., & Dwivedi, B. S. (2015). Assessment of microbial biomass and enzyme activities in soil under temperate fruit crops in north western Himalayan region. *Journal of Soil Science and Plant Nutrition,* 15 (4), 848-866.

Diaz-Ravina, M., Caraballas. T., & Acea, M. J. (1988). Microbial biomass and activity in four acid soils. *Soil Biology and Biochemistry,* 20, 817-823.

Dutta, M., Diengdoh, J., & Ram, S. (2015). Physico-chemical properties of West Khaisi Hills soils of Meghalaya in relation to land uses. *An Asian Journal of Soil Science,* 10(2), 288-294.

Gajananda, K. (2007). Soil organic carbon and microbial activity. *East Antarctica European Journal of Soil Science,* 58,708-713.

Geisseler, D., & Horwath, W. R. (2009). Short-term dynamics of soilcarbon microbial biomass and soil enzyme activities compared to long-term effects of tillage in irrigat edrow-crops. *Biology and Fertility of Soils,* 46, 65-72.

Jenkinson, D. S. (1988). The determination of microbial biomass carbon and nitrogen in soil. In: Advances in Nitrogen Cycling in Agricultural Ecosystems, (Wilson J. R. ed), CAB, Wallingford, England, pp 368-386.

Jenkinson, D. S., & Parry, L. C. (1989). The nitrogen in the Broad balk wheat Experiment: A model for the turn over of nitrogen through the soil microbial biomass. *Soil BioIogy and Biochemistry,* 21(4), 535-541.

Kalita, N., Dutta, S., Das, K. N., Kurmi, K., & Patgiri, D. K. (2023). Impact of different land uses on soil organic carbon stock in Karbi Anglong district of Assam, India. *Journal of Soil and Water Conservation,* 22(1), 15-22.

Kara, O., & Bolat, I. (2008). The effect of different land uses on soil microbial biomass carbon and nitrogen in Bartin province. *Turkish Journal of Agriculture and Forestry,* 32, 281-288.

Keeney, D. R., & Nelson, D. W. (1982). Nitrogen-inorganic forms. In Page A. L. *et al.* (ed.). Methods of soil analysis, Part 2, 2nd ed., Agron. Monogr. 9. ASA and SSSA, Madison, Wis. pp 643-698.

Klose, S., & Tabatabai, M. A. (1999). Arysulfatase activity of microbial biomass in soils as affected by cropping systems. *Biology and Fertility of Soils,* 29(1), 46-54.

Kong, Y. Y. A., Six, J., Bryant, D. C., Denison, R. F., & Kessel, C. V. (2005). The Relationship between Carbon Input, Aggregation, and Soil Organic Carbon Stabilization in Sustainable Cropping Systems. *Soil Science Society of America Journal,* 69(4), 1078-1085.

Liu, Z., Liu, Guo-hua, Fu, Bo-Jie., & Zheng, X. (2008). Relationship between plant species diversity and soil microbial functional diversity along a longitudinal gradient in temperate grasslands of Hulunbeir, Inner Mongolia, China. *Ecological Research,* 23(3), 511-518.

Moeskops, B., Buchan, D., Sleutel, S., Herawaty, L., Husen, E., Saraswati, R., Setyorini, D., & De Neve, S. (2010). Soil microbial communities and activities under intensive organic and conventional vegetable farming in West Java, Indonesia. *Applied Soil Ecology,* 45,112-120.

Moscatelli, M. C., Di Tizio, A., & Marinari, S. (2007). Microbial indicators related to soil carbon in Mediterranean land use systems. *Soil and Tillage Research,* 97(1), 51-59.

Nannipieri, P., Kandeler, E., & Ruggiero, P. (2002). Enzyme activities and microbiological and biochemical processes in soil. In: Burns, R. G., & Dick, R. P. (ed.) *Enzymes in Environment,* pp 133, Marcel Dekker, New York.

Okur, N., Altindisli, A., Cengel, M., Gocmez, S., & Kayikcioglu, H. H. (2009). Microbial biomass and enzyme activity in vineyard soils under organic and conventional farming systems. *Turkish Journal of Agriculture,* 33, 413-423.

Padalia, K., Bargali, S. S., Bargali, K., & Khulbe, K. (2018). Microbial biomass carbon and nitrogen in relation to cropping systems in Central Himalaya, India. *Current Science,* 115(9), 1741-1750.

Panwar, P., Pal, S., Reza, S. K., & Sharma, B. (2011). Soil Fertility Index, Soil Evaluation Factor, and Microbial Indices under Different Land Uses in Acidic Soil of Humid Subtropical India. *Communications in Soil Science and Plant Analysis*, 42(22), 2724-2737.

Pramod, J., Arpan, D., Brij, L. L., Biswas, A. K., Singh, M., Reddy, K. S., & Rao, A. S. (2012). Soil carbon pools, mineralization and fluxes associated with land use change in vertilsols of central India. *National Academy Science Letters,* 35, 475-483.

Quilchano, C., & Maranon, T. (2002). Dehydrogenase Activity in Mediterranean Forest Soil. *Biology and Fertility of Soils,* 35, 102-107.

Shimrah, T., Rao, K. S., & Saxena, K. G. (2015). Soil Property Variations Under Different Land Use/Cover Types In Traditional Agricultural Landscape in Northeast India. *Journal of Chemistry, Environmental Sciences and its Applications*, 2(1), 73-97.

Six, J., Conant, R. T., Paul, E., & Paustian, K. (2002). Stabilization Mechanisms of Soil Organic Matter: Implications for C-Saturation of Soils. *Plant and Soil,* 241(2), 155-176.

Vallejo, V., Arbeli, Z., Terán, W., Lorenz, N., Dick, R. P., & Roldan, F. (2012). Effect of land management and *Prosopis juliflora* (Sw.) DC trees on soil microbial community and enzymatic activities in silvopastoral systems of Colombia. *Agriculture, Ecosystems & Environment*, 150, 139-148.

Vance, E. D., Brookes, P. C., & Jenkinson, D. S. (1987). An extraction method for measuring soil microbial biomass C. *Soil Biology and Biochemistry,* 19, 703-707.

Walkley, A., & Black, I. A. (1934). An examination of the Degtjareff method for determining soil organic matter and a proposed modification of the chromic acid titration method. *Soil Science,* 37, 29-38.

Yadav, R. S., Yadav, B. L., Chhipa, B. R., Dhyani, S. K., & Ram, M. (2011). Soil biological properties under different tree based traditional agroforestry system in a semi-arid region of Rajasthan, India. *Agroforestry Systems,* 81(3), 195-202.

Yang, K., Zhu, J. J., & Zhang, M. (2010). Soil microbial biomass carbon and nitrogen in forest ecosystems of Northeast China: A comparison between natural secondary forest and larch plantation. *Journal of Plant Ecology,* 3(3), 175-182.

Yao, M. K., Angui, P. K. T., Konaté, S., Tondoh, J. E., Yao T., Abbadie, L., & Benest, D. (2010). Effects of Land Use Types on Soil Organic Carbon and Nitrogen Dynamics in Mid-West Côte d'Ivoire. *European Journal of Scientific Research,* 40(2), 211-222.

Yuan, B., & Yue, D. (2012). Soil microbial and enzymatic activities across a chronosequence of Chinese pine plantation development on the loess plateau of China. *Pedosphere,* 22, 1-12.

Zhang, N., He, X., Gao, Y., Li, Y., Wang, H., Ma, D., Zhang, R., & Yang, S. (2010). Pedogenic carbonate and soil dehydrogenase activity in response to soil organic matter in Artemisia ordosica community. *Pedosphere,* 20, 229-235.