

Article

Altering Natural Ecosystems Causes Negative Consequences on the Soil Physical Qualities: An Evidence-Based Study from Nilgiri Hill Region of Western Ghats, India

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Abstract: Land use change (LUC) has direct and indirect consequences on soil quality. To gain insight into how LUC influences the physical properties of soil, it can be advantageous to compare undisturbed ecosystems with those that have naturally evolved over time, as well as to use soil quality indices to pinpoint the sensitivity of each ecosystem and land use change (LUC). A soil survey was carried out in the six major ecosystems of the Nilgiri Hill Region: cropland (CL), deciduous forest (DF), evergreen forest (EF), forest plantation (FP), scrubland (SL), and tea plantation (TP), with those having an establishment for over 50 years being selected and analyzed for soil physical parameters. In addition, soil quality indices were also derived to pinpoint the vulnerability of each ecosystem to LUC. The results reveal that the changes in land use significantly altered the soil physical properties. The content of clay was higher in EF and DF and increased with the soil profile's depth, whereas the sand content was higher in CL and TP and decreased with the depth increment. BD and PD were significantly lower in EF, DF, SL, and FP, whereas they were higher in CL and TP. PS and ASM followed a similar trend to BD and PD. Owing to undisturbed natural settings, an abundance of litter input, and higher carbon concentrations, the HC was higher in EF, DF, SL, and FP, whereas, in the case of anthropogenic-influenced ecosystems such as CL and TP, it was lower. We discovered that LUC has altered Ag S, WSA, and MWD. Due to tillage and other cultural practices, Ag S, WSA, and MWD were significantly lower in CL and TP. However, the results confirm that native ecosystems (EF and DF) with a higher carbon content prevent such degradation, thereby resulting in good Ag S, WSA, and MWD.

Keywords: soil physical properties; land use change; Western Ghats; soil quality index

1. Introduction

Soil quality is defined as “the capacity of a soil to function within an ecosystem and land-use boundaries, to sustain biological (plant and animal) productivity, maintain environmental quality, and support human health and habitation” [1]. One must have a

thorough awareness of soil quality to evaluate sustainable farming practices and provide early warning signals of unfavorable trends [2–5].

Natural resources continue to be fundamental to subsistence [6], yet anthropogenic (farming, grazing, burning) and natural (rainfall, temperature) factors can place a strain on these resources by causing changes in a landscape's both temporal as well as spatial scales. The land use dynamics attributed to human sources are advancing rapidly. The Sustainable Development Goals (SDGs) of the 2030 Agenda raise a strong concern emphasizing tracking shifts in nature's resources and documenting those elements (such as agriculture) triggering these changes. Among them, the impact of land use and land cover and its changes on consequences of soil quality is crucial for comprehending the dynamism of ecosystem functions [7,8].

According to the United Nations Convention to Combat Desertification (UNCCD), erosion destroys 24 billion tons of fertile soil every year, while desertification and drought degrade 12 million hectares of land (or 23 hectares per minute), eradicating the possibility of 20 million tons of grain being grown [9]. Land use change (LUC) thus affects the Earth's climate, biogeochemistry, and hydrology. To ensure any approximations, it is difficult to precisely articulate how LUC modifies the soil quality status at the local or regional scales [9].

Feeding the ever-growing population in this world has remained to be a major challenge that is significantly impacted by land use type [10]. LUC is among the prominent indicators of anthropogenic activity that either causes land degradation (forest clearing, etc.) or helps to restore, maintain, or strengthen land quality (afforestation, conservation agriculture, agricultural practice) [11]. Anthropogenic activity that results in intensive land use can have negative consequences on soil quality, including erosion, a loss of nutrients, and soil structure degradation [12–17]. Deforestation, for instance, not only impacts the structure, species composition, and diversity of forests [18] but also the amount of aboveground vegetation and litter return, both of which have an impact on the soil's physical characteristics [19,20]. LUC can therefore have a variety of effects on soil properties depending on its degree and severity [21,22], and leads to soil quality degradation and a loss of soil organic matter [21]. Even management techniques such as sowing, weeding, and tillage can have an impact on the biomass and soil characteristics, which, in turn, has an impact on the quality and other elements such as flora, water, air, and other land resources [23–26].

LUC has serious repercussions on soil quality at both the regional as well as global scale. Land degradation brought on by LUC occurs frequently in the Nilgiri Hill Region (NHR), which was India's first biosphere reserve, a UNESCO World Heritage Site, and a hotspot for biodiversity [27]. The soils of this region play a vital role in providing ecosystem services such as maintaining the streamflow and fostering a high base flow. The environmental setting in the NHR with an LUC had made this land more vulnerable to erosion. In a high rainfall zone where LUC is prominent, the physical properties of the soils play a critical role in hydrology and water movement and other ecological functions. Since the NHR is highly susceptible to erosion, measuring the physical property can help in implementing strategies to prevent hazards. Assessing the quality of soil is important for crafting suitable management practices that can boost productivity and enhance environmental sustainability [28]. Added to it, this can help decision makers in land use planning by considering the constraints and potentialities of different land uses. Under these circumstances, an assessment of soil physical properties becomes necessary as it can play a critical role in hydrology and water movement in this high-rainfall zone where LUC is prominent. In light of this, the current study put out the hypothesis that different land use types in the NHR would have various physical properties of the soil. In other words, how altering the natural ecosystems causes negative consequences on the soil physical qualities has been hypothesized. Therefore, the goal of this research was to assess the soil quality across various NHR land uses. For sustainable land management and ecosystem preservation, it is critical to comprehend how LUC affects the physical properties of soil. Henceforth, the present study aims to fill this gap by taking a multidisciplinary approach

that incorporates soil science and hydrology aspects. Additionally, it employs advanced statistical modeling to achieve a thorough comprehension of soil physical properties in various land use types.

2. Materials and Methods

2.1. Study Area

The NHR is a landmass encompassing an area of 2551 square kilometers, originating from the tectonic uplift and convergence processes involving the Western and Eastern Ghats ranges, situated at the southern end of the Indian peninsula. Geographically, NHR is located within latitudinal coordinates spanning from 11°30' N to 11°15' N and longitudinally between 76°45' E and 77°00' E. It has a gradient in elevation that ranges from 2000 to 2350 m above mean sea level. The region experiences a distinctive bimodal pattern of precipitation distribution, yielding an average annual rainfall of 1310 mm, while the temperature regime ranges from 1 °C to 25 °C [8].

2.2. Geomorphology and Soils

NHR has charnockite bedrock, which is characterized by its hardness and lower porosity [29]. It falls under the class of deeply weathered due to its extended weathered zone between the parent material and clay [30]. The Ultisol soil order is the dominant soil type in these areas, followed by lateritic, red sandy soil, black soil, red loam, alluvial soil, and colluvial soil [31].

2.3. Field Description and Investigation

The land use history of NHR has been investigated with the help of old records from government departments and local people's traditional wisdom [8]. A field investigation was initially conducted to comprehend and research the various ecosystems in NHR. The field survey confirmed that forestland such as evergreen forest (EF) and deciduous forest (DF) was under protection from any anthropogenic activities by the department of the forest. Other ecosystems such as cropland (CL), forest plantation (FP), scrubland (SL), and tea plantation (TP) were under anthropogenic pressures and are mostly rainfed agroecosystems in NHR. Tillage (both manual and mechanical), pesticides, and fertilizers (both organic and inorganic) applications were commonly noticed among these ecosystems. During selection, care was taken to ensure that these ecosystems were under this type for longer duration.

2.4. Soil Sampling and Analysis

Soil sampling was performed in 2021 from major ecosystems of NHR, viz., cropland (CL), deciduous forest (DF), evergreen forest (EF), forest plantation (FP), scrubland (SL), and tea plantation (TP) as per the standard procedure. A portable global positioning system (GPS) was used to indicate the geographic coordinates of 214 soil profiles in total (47—CL, 39—DF, 27—EF, 23—FP, 24—SL, and 54—TP) (Figure 1, Table S1). Eleven soil series were encountered in the study region and most of them occur in associations. Samplings were performed on the basis of the land use systems that encompassed almost all the soil series associations except for 4. Details are provided in Table S2.

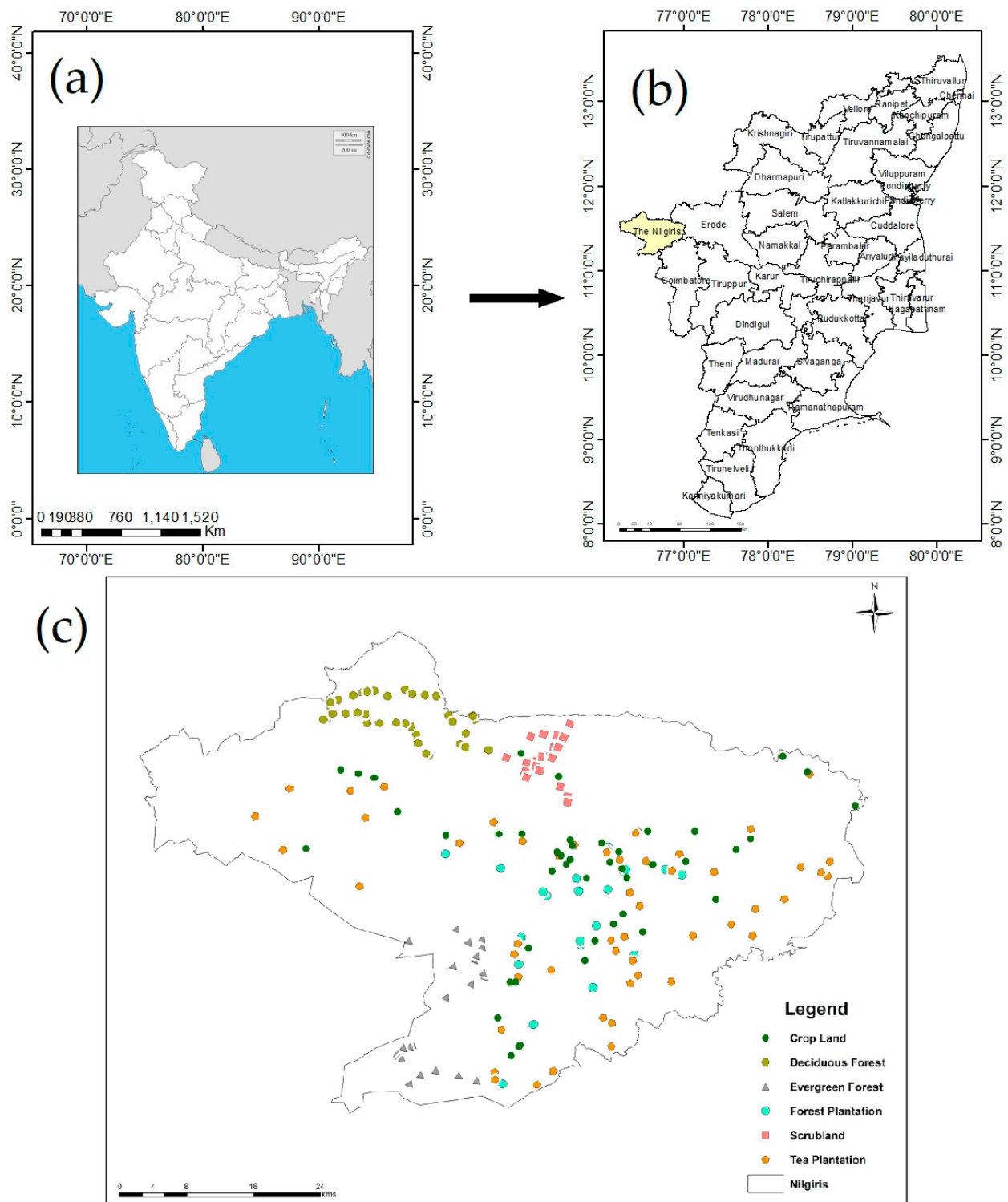


Figure 1. Distribution of the 214 sample sites across the NHR’s various ecosystems: (a) India, (b) Tamil Nadu map showing study region, (c) sampling points. (See Table S1 for a full description of sampling sites in NHR.)

Soil samplings (disturbed soil samples) were performed in three depth classes (“0–15”, “15–30”, and “30–45” cm) and at five quadrats across the soil profile in each location, and, along with it, two undisturbed samples were also collected. Three composite samples per plot and the rocks were obtained by combining the disturbed soil samples from each

depth. Plant debris was taken out using a sieve with a 2.0 mm mesh. Those samples were transported to a laboratory for further analysis by storing them at $-20\text{ }^{\circ}\text{C}$.

2.5. Soil Physical Properties

Physical soil quality indicators such as particle size distribution, bulk density (BD), particle density (PD), total porosity, available soil moisture (ASM), hydraulic conductivity (K_{sat}), and aggregate analysis were estimated in order to determine the physical soil quality. Details about the methodologies are shown in Table 1.

Table 1. Methods of determining the soil physical properties.

Sl. No	Soil Properties	Methodology	Reference
1	Bulk density	The cylinder method	[32]
2	Particle density	The cylinder method	[32]
3	Pore space	The cylinder method	[32]
4	Soil texture	International pipette method	[33]
5	Available soil moisture	Gravimetric method	[34]

2.6. Hydraulic Conductivity

Through constant head arrangements, core samples were saturated in water for 24 h to determine hydraulic conductivity [32]. The values of K were calculated using Darcy's equation—USDA Handbook No. 60.

$$K_s = \left[\frac{Q}{At} \right] \times \frac{L}{L + H} \quad (1)$$

where K_s is saturated hydraulic conductivity (cm min^{-1}), Q is volume of percolated water collected (cm^3), A is cross-sectional area of parameter (cm^2), t is the amount of time needed for collecting percolated water (min), L is length of soil column (cm), and H is how high the water column is above the ground (cm).

2.7. Aggregate Analysis

Before air-drying, the clods of larger size were broken along their natural cleavage with hands and were sieved to receive aggregates that were passed along 8 mm sieves and retained in 5 mm sieves. In terms of soil matrix, aggregates of a size between 5 and 8 mm reflect all soil fractions since the entire soil falls within this range. Using wet sieving [35], 100 g (5 to 8 mm in diameter) of aggregates was placed in a nest of sieves (5 mm, 2 mm, 1 mm, 0.5 mm, 0.25 mm, 0.1 mm, and 0.053 mm) and were sieved at 30 strokes per minute for 30 min. The aggregates were put into pre-weighed beakers after wet sifting and oven-dried for $60\text{ }^{\circ}\text{C}$ until the water evaporated. A temperature of $60\text{ }^{\circ}\text{C}$ was chosen for drying rather than $150\text{ }^{\circ}\text{C}$ to prevent organic carbon losses at higher temperatures. They were weighed and the sand corrections were performed through subtraction of the total sand in each fraction from the sand retained in each fraction. By dispersing aggregates retained in a 0.053 mm sieve using sodium hexametaphosphate (5 g L^{-1}), and weighing the fractions, the entire amount of sand in each fraction was determined. The aggregate stability was estimated as per [32]. The WSAs (water-stable macro aggregates) were calculated by summing the retained material in all sieves and the original amount of unsieved soil [35]. The mean weight diameter (MWD), which was determined in accordance with van Bavel [36], is regarded as an index of aggregation.

Using above observation, following indices were generated:

$$\text{MWD}(\text{mean weight diameter}) = \sum X_i W_i \quad (2)$$

Where X_i is the fraction's mean diameter, I is 0.0765 mm, 0.175 mm, 0.375 mm, 0.75 mm, 1.5 mm, and 3.5 mm, and W_i is the percentage of the sample's overall weight that appears in the fractions.

$$\% \text{Aggregate stability} = \frac{\text{Weight of the soil particle} > 0.25 \text{ mm} - \text{Weight of sand} > 0.25 \text{ mm}}{\text{Oven dry weight} - \text{weight of sand} > 0.25 \text{ mm}} \times 100 \quad (3)$$

$$\% \text{Distribution of WSA in each size group} = \left(\frac{\text{Weight of aggregates in each size group}}{\text{Total weight of soil}} \right) \times 100 \quad (4)$$

2.8. Soil Organic Matter

The samples were analyzed in a TOC analyzer (Elementar) and, using the van Bemmelen factor (1.724), they were converted to soil organic matter (SOM).

2.9. Assessment of Soil Quality

Indexing the quality of soil generally follows three steps, which are as follows: interpretation/scoring and integration into soil quality index value (Table 2).

Table 2. Interpretation of relative soil quality indices.

Class	Value Range	Grade
I	90–100	Best
II	80–90	Very good
III	70–80	Good
IV	60–70	Average
V	<60	Poor

Indicator selection—indicators that reflect land degradation and soil quality were used as minimum data set (MDS) [37,38], and these representative indicators (i.e., MDS) [39] can better represent the total data set (TDS) [40,41]. In addition, necessary expert opinions were also taken into account along with the PCA [42] and included as indicators. In this study, all soil physical soil parameters and SOM were taken into consideration for assessing the soil quality. Data were changed based on each indication's critical levels after the indicator was chosen.

Interpretation of the indicators: Using the linear technique, the aforementioned indications were converted into scores that range from 0 to 1 [43], and they were assigned to indicators applying the function “less is better”, “more is better”, or “optimum is better”. The values of the upper and lower threshold and baseline were established to calculate the indicator score.

Soil quality indexing—the indicator data that were transformed were provided weightage as per the values obtained from PCA. A specific percentage of the variation (in the entire dataset) was described by each main component. By dividing the overall percentage of variation from each PC by the percentage of cumulative variance, the weightage factor (WF) was computed. WFs were assigned in such a manner that the sum of all factors equalled one and this was multiplied with the transformed scores to calculate the SQI. Using SQI, the relative soil quality index (RSQI) was calculated [44]:

$$RSQI = \left(\frac{SQI}{SQI_m} \right) \times 100 \quad (5)$$

where SQI_m is maximum value of SQI. The values RSQI were normalized to 100 and can be classified as per [45].

Additive index—the transformed scores were added to determine the additive index. Weighted index—based on the PCA results, weights were assigned to the converted indicator data. Each PC explained a specific percentage (%) of the variation in the overall dataset. The total percentage of variation from each PC was divided by the cumulative variance percentage to determine the weightage factor [46]. The variables (indicators) from the relevant PCs were combined with the weightage factor that was generated. The index values were then calculated by adding the weighted variables.

2.10. Statistical Analyses

The sampling sites were treated as replicates or random variables in an analysis of variance (ANOVA), whereas the ecosystems were treated as fixed effects. When comparing means amongst ecosystems, the Duncan's multiple range test (DMRT) was utilized to identify significant mean differences. At $p < 0.01$ the statistical significance was determined.

The native function "cor" in the R software version 4.1.1 was used for correlation analysis, the package "qgraph" was used to create network maps, and the PCA (principal component analysis) was computed. R packages like *ggplot*, *Complex Heatmap*, *Factoextra*, *FactoMineR*, and *dendextend* were used [47] for visualization.

3. Results

3.1. Particle Size Distribution (PSD)

3.1.1. Sand Content

The soil texture of the NHR varied significantly ($p < 0.01$) among each ecosystem. The sand content in CL and TP (46.83% and 41.97%) was the highest, whereas SL, FP, EF, and DF recorded the lowest values (25.85%, 19.83%, 19.97%, and 21.50%). The sand content in each ecosystem was decreasing with an increase in depth (Figure 2) and a lower depth showed a decrease in sand content. The sand content in different ecosystems followed the order of CL > TP > SL > DF > EF > FP. This decrease in the sand was rapid between 0–15 cm and 15–30 cm in DF, SL, and FP (10.50%, 9.56%, and 12.64%) when compared with a 15–30 cm and 30–45 cm depth (10.23%, 2.30%, and 12.39%). CL had a higher sand content, i.e., 57.36%, 54.09%, 44.80%, 57.66%, and 10.38%, respectively, as compared to EF, DF, SL, FP, and TP. The sand content of EF at 30–45 cm was statistically significant ($p < 0.01$) in SL, TP, and CL whereas EF, DF, and FP were found to be on par.

3.1.2. Silt Content

The native forest ecosystems (FP, DF, and EF) recorded a higher silt content (15.80%, 14.08%, and 13.79%, respectively, at 0–45 cm depth, whereas the TP, CL, and SL recorded the lowest silt content (11.75%, 11.90%, and 12.24%, respectively, at 0–45 cm). Similar to the trend seen in the sand content, the silt content increased with depth. The silt content recorded in FP was 12.72%, 10.89%, 22.53%, 25.63%, and 24.68% higher than EF, DF, SL, TP, and CL, respectively. At a depth of 0 to 15 cm in the soil profile, LUC has no significant effect on the silt content of the soil. The NHR's silt content varied somewhat between 15 and 30 cm depth (Figure 2). The content of silt recorded in FP was statistically significant ($p < 0.01$) in comparison with other ecosystems of the NHR at 30–45 cm depth. The average silt content of TP was 10.70%, with a range from 6.40 to 14.12%. With the depth increment, the silt content of each ecosystem decreased. The silt content of EF decreased by the maximum amount (14.69%) between 0–15 cm and 15–30 cm depth compared to between 15–30 cm and 30–45 cm depth (7.63%).

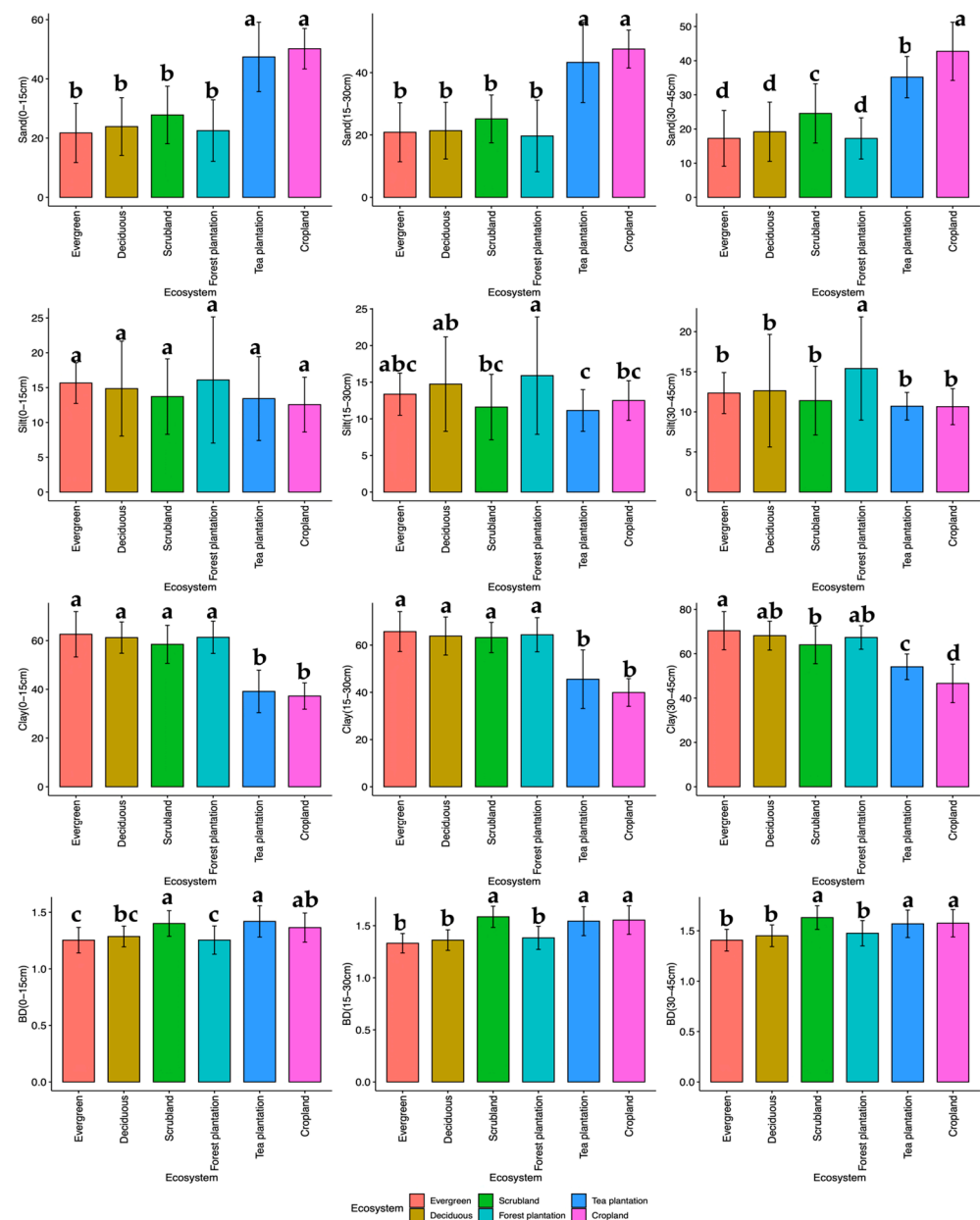


Figure 2. Soil physical properties under different ecosystems of Nilgiri Hill Region (NHR). The figure displays the impact of land use change on the percentages of sand, silt, clay, and bulk density (BD) (Mg m^{-3}) in several NHR ecosystems. According to DMRT, histograms with distinct letters differ significantly ($p < 0.01$).

3.1.3. Clay Content

Compared to DF, SL, FP, TP, and CL, the clay concentration in EF was greater by 2.76%, 6.55%, 2.84%, 30.14%, and 37.70%, respectively (Figure 2). The forest ecosystems (EF, DF, SL, and FP) of the NHR had a higher clay content (0–15 cm), which was statistically significant ($p < 0.01$) when compared to CL and TP. Similar to 0–15 cm, clay content at 15–30 cm depth follows a similar trend. Though the clay content in the upper soil profile (0–30 cm) was similar in forest ecosystems (EF, DF, SL, and FP), the clay content at 30–45 cm depth showed a significant difference. EF and FP were statistically on par, ($p < 0.01$), while the DF was an intermediary between EF, FP, and SL, ($p < 0.01$). The EF at 30–45 cm depth had a clay content ranging from 50.82 to 88.05%, with a recorded average of 70.39%. The soils from DF sampled at 30–45 cm depths had a clay content that ranged from 56.91 to 82.50%, with a mean of 68.16%. The observed average in FP was 64.34%, with a range of 58.93 to 81.00%.

The samples from the SL ecosystem recorded 64.02% as the average at 30–45 cm with a range of 43.00 from 78.44%. In TP, 54.11% was the average clay content noted, with a range between 30.00 and 63.84%. The CL had an average clay content of 46.61%, with a range from 29.48 to 63.84%.

Each ecosystem's clay concentration increased as the depth was increased. The clay content at EF increased least between 0–15 cm and 15–30 cm depths (4.82%) compared to 15–30 cm and 30–45 cm depths (6.58%). Between 0–15 cm and 15–30 cm depths, the increase in clay content at DF was minimal (4.13%) when compared to 15–30 cm and 30–45 cm depths (6.29%). The increase in clay content at SL was more (7.54%) between 0–15 cm and 15–30 cm depths than between 15–30 cm and 30–45 cm depths (1.22%). The clay content at the FP increased to a higher level between 0–15 cm and 15–30 cm depths (4.75%) compared to 15–30 cm and 30–45 cm depths (4.34%). The clay content in TP increased minimally between 0–15 cm and 15–30 cm depths (14.11%) compared to between 15–30 cm and 30–45 cm depths (15.76%). The clay content at CL increased least (6.76%) between 0–15 cm and 15–30 cm depths compared to between 15–30 cm and 30–45 cm depths (14.29%). Conclusively, all ecosystems of the NHR recorded an increase in clay and a decrease in silt and sand with an increased depth. The soils of CL and TP recorded a lower content of fine soil earth (silt and clay) when compared with other ecosystems.

3.2. Bulk Density

There was a 1.04 to 1.85 (Mg m^{-3}) range in the bulk density (BD). CL registered the lowest BD (1.04 Mg m^{-3}) at 0–15 cm depth and the highest of 1.85 Mg m^{-3} at 30–45 cm among all other ecosystems. In all ecosystems except DF, there was a greater increase in density between 0–15 cm and 15–30 cm compared to 15–30 cm and 30–45 cm. Thus, BD is a measure of soil compaction and quality. The BD recorded in SL was 13.64%, 11.04%, 11.04%, 1.95%, and 3.25% higher than EF, DF, FP, TP, and CL. A significant ($p < 0.01$) difference was observed among the ecosystems of the NHR across all depths. The BD of each ecosystem increased with the depth increment (Figure 2). The BD of EF was denser at 0–15 cm and 15–30 cm (6.02%) than at 15–30 cm and 30–45 cm (5.67%). Between 0–15 cm and 15–30 cm depth, the increase in BD at DF was low (5.15%) when compared to 15–30 cm and 30–45 cm (6.21%). The increase in BD at SL was largest at 0–15 cm and 15–30 cm depths (11.95%), compared to 15–30 cm and 30–45 cm depths (2.45%). In comparison to 15–30 cm and 30–45 cm depths (6.76%), the largest increase in BD was seen at FP between 0–15 cm and 15–30 cm depths (9.42%).

3.3. Particle Density

The overall particle density (PD) (0–45 cm) in different ecosystems increased in the following order: CL > TP > SL > FP > DF > EF (Figure 3). It was the highest in CL (2.75 Mg m^{-3} at 0–15 cm depth) and lowest in EF (1.07 Mg m^{-3} at 15–30 cm depth). All ecosystems witnessed an increased PD over depth increment. The PD recorded in CL was 21.34%, 16.74%, 10.46%, 15.06%, and 1.26% higher than EF, DF, SL, FP, and TP, respectively. The conversion of forestland drastically enhanced the density of soil. The native EF was significantly ($p < 0.01$) lower in its PD when compared to other ecosystems studied. The PD of EF varied from 1.09 to $2.62 \text{ (Mg m}^{-3})$, with an average of $1.76 \text{ (Mg m}^{-3})$. The PD for DF from 0 to 15 cm fluctuated from 1.04 to $2.44 \text{ (Mg m}^{-3})$, with an average value of 1.91 Mg m^{-3} being measured. With a range of 1.34 to $2.49 \text{ (Mg m}^{-3})$, the SL ecosystem registered an average of 2.03 Mg m^{-3} at 0–15 cm. The range of the samples from the FP was 1.40 to $2.47 \text{ (Mg m}^{-3})$, with 1.97 Mg m^{-3} serving as the average. In the case of TP, the range was 1.37 to $2.69 \text{ (Mg m}^{-3})$, with 2.30 Mg m^{-3} being the recorded average. The PD of CL ranged from 1.68 to 2.71, with an average of 2.36 Mg m^{-3} . The influence of PD on the surface soils was similar to that in the subsoils. The undisturbed EF was significantly ($p < 0.01$) lower in its PD even in subsoils.

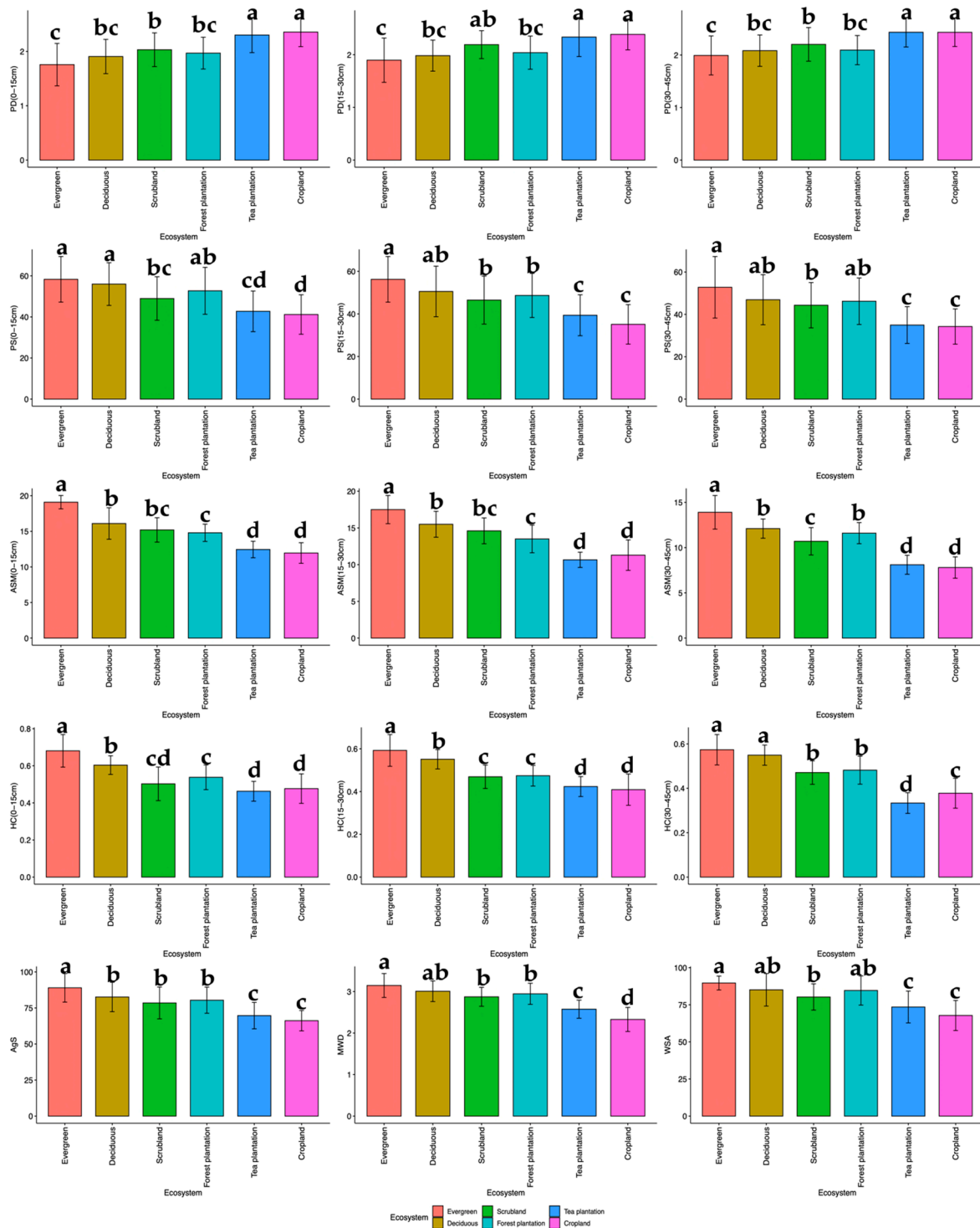


Figure 3. Soil physical properties under different ecosystems of Nilgiri Hill Region (NHR). The figure displays how changing land use has an impact on particle density (PD) (Mg m^{-3}), pore space (PS) (%), available soil moisture (ASM) (%), hydraulic conductivity (HC) (cm hr^{-1}), aggregate stability (Ag S) (%), mean weight diameter (MWD) (mm), and water-stable aggregates (WSA) (%) in different ecosystems of the NHR. According to DMRT, histograms with distinct letters differ significantly ($p < 0.01$).

Each ecosystem’s PD showed a different trend along the depth of the profile. The increase in PD at EF was largest between 0–15 cm and 15–30 cm depths (7.37%), compared to 15–30 cm and 30–45 cm depths (4.52%). In comparison to 15–30 cm and 30–45 cm depths

(4.81%), the increase in PD at DF between 0–15 cm and 15–30 cm depths was minimal (3.54%). Between 0–15 cm and 15–30 cm depths, there was the highest increase in PD at SL (7.30%) compared to 15–30 cm and 30–45 cm depths (0.45%). Conclusively, EF registered the lowest PD values across each depth whereas CL and TP registered the highest PD. The PD recorded at SL, FP, and DF was a mid-value between EF and CL, TP.

3.4. Pore Space

Land use type had a significant ($p < 0.01$) effect on the soil pores space (PS) under different ecosystems of the NHR. The mean PS was the highest in EF (55.78%) followed by DF (51.15%). The overall (0–45 cm) PS in EF was 8.30%, 16.46%, 11.80%, 30.05%, and 33.94% higher than DF, SL, FP, TP, and CL, respectively. Across each ecosystem, it was found that the PS decreased with an increased depth. The PS of EF and DF were significantly ($p < 0.01$) different from other ecosystems studied. The PS in DF decreased drastically when compared to its PS at 0–15 cm. The PS at 15–30 cm depth was significantly lower in CL and TP, whereas it was highest at EF. A drastic reduction in the PS of FP was observed at 30–45 cm depth, whereas other ecosystems were statistically similar ($p < 0.01$) at 15–30 cm (Figure 3). At 30–45 cm depth, the PS of EF varied from 24.24 to 71.86%, with an average value of 52.81%.

With an increasing depth, the PS at each ecosystem followed a declining trend along the profile. Between 0–15 cm and 15–30 cm depths, the reduction in PS at EF was at its least (3.58%) compared to 15–30 cm and 30–45 cm depths (6.06%). The decrease in PS at DF was highest between 0–15 cm and 15–30 cm depths (9.75%) compared to 15–30 cm and 30–45 cm depths (7.12%).

3.5. Available Soil Moisture

The study's measurements of the available soil moisture (ASM) indicated significant ($p < 0.01$) differences between ecosystems. The ASM registered in EF (16.83%) was 13.43%, 19.79%, 20.97%, 38.20%, and 38.50% higher than DF, SL, FP, TP, and CL. In this study, FP recorded 20.97%, 8.71%, 1.48%, 21.80%, and 22.18% decreases in their ASM when compared to EF, DF, SL, TP, and CL respectively. The ASM observed in 0–15 cm of major ecosystems in the NHR was statistically ($p < 0.01$) significant. The ASM in EF at 0–15 cm ranged from 17.59 to 21.25%, with an average of 19.10% (Figure 3).

The ASM at 15–30 cm was statistically similar to 0–15 cm. Each ecosystem's ASM decreased as the depth increased. Interestingly, the EF and DF were able to maintain their moisture significantly ($p < 0.01$) higher across all depths, but SL and FP showed turbulence with the depth increment. At EF, compared to 0–15 cm and 15–30 cm depths (8.38%), the decrease in ASM was greatest between 15–30 cm and 30–45 cm depths (20.57%). In contrast to 0–15 cm and 15–30 cm depths (3.73%), the greatest ASM at DF occurred between 15–30 cm and 30–45 cm depths (21.94%). In TP, the minimum ASM occurred at depths of 0–15 cm and 15–30 cm (14.38%), as opposed to between depths of 15–30 cm and 30–45 cm (24.01%). In comparison to 0–15 cm and 15–30 cm depths (5.44%), the largest increase was seen at CL between 15–30 cm and 30–45 cm depths (30.97%).

3.6. Hydraulic Conductivity

A statistically significant ($p < 0.01$) HC was observed among each ecosystem of the NHR. The hydraulic conductivity (HC) recorded in EF was 8.06%, 22.58%, 19.35%, 33.87%, and 32.26% higher than DF, SL, FP, TP, and CL, respectively. There were distinct trends observed in HC among each ecosystem. In contrast to 15–30 cm and 30–45 cm depths (3.39%), between 0–15 cm and 15–30 cm of depth at EF, the HC decreased most rapidly (13.24%). Because there is no difference between the subsurface, that is, between 15–30 cm and 30–45 cm, the reduction in HC at the surface was greatest between 0–15 cm and 15–30 cm of depth (8.33%). Following a similar trend to DF, the HC of SL decreased by a maximum of 6.0% at 0–15 cm and 15–30 cm depths; thereafter, no change was observed between 15–30 cm and 30–45 cm depths. In FP, the HC decreased at a rate of 12.96%

at 0–15 cm and 15–30 cm depths and then followed an increasing trend of 2.08% at the sub-surface (15–30 cm and 30–45 cm). In comparison to between 15–30 cm and 30–45 cm depths (21.43%), the decrease in HC at the TP was lowest between 0–15 cm and 15–30 cm depths (8.70%). The highest decrease in HC at CL occurred between 0 cm and 15 cm depths (14.58%), followed by 15 cm and 30 cm depths (7.32%) (Figure 3).

3.7. Soil Aggregate Index

3.7.1. Aggregate Stability

The aggregate stability (Ag S) of the EF at 0–45 cm depth ranged between 53.63 and 99.30 (%) and the average value recorded was 88.97 (%). The aggregate stability of the soils collected from DF at 0–45 cm ranged from 57.74 to 100.16 (%), with a mean of 82.61 (%). The samples from the SL ecosystem recorded 78.48 (%) as an average at 0–45 cm with a range of 52.07 to 97.02 (%), whereas in the FP, it ranged between 62.94 and 91.81 (%), with 80.39 (%) being the average value (Figure 3). The aggregate stability of TP ranged from 48.51 to 87.43 (%), and 69.69 (%) was recorded as an average. CL samples had a range between 49.49 and 79.88 (mg kg^{-1}), with 66.16 (mg kg^{-1}) being the average.

The amount of aggregate stability from each ecosystem is as follows: EF (88.97%) > DF (82.61%) > FP (80.39%) > SL (78.48%) > TP (69.69%) > CL (66.16%). EF recorded a 7.15%, 11.79%, 9.64%, 21.67%, and 25.64% higher aggregate stability when compared to DF, SL, FP, TP, and CL, respectively. We observed that LUC has a great impact on aggregate stability. Statistically ($p < 0.01$), the aggregates of EF were most stable than all ecosystems studied. The aggregates stability of DF, SL, and FP was significantly lower. It was observed that the aggregates stability of CL and TP was the lowest among each ecosystem studied.

3.7.2. Mean Weight Diameter (MWD)

At 0–45 cm depth, the MWD of EF ranged from 2.37 to 3.67 (mm) and the average value recorded was 3.15 (mm). The MWD of the DF was 2.00 to 3.80 (mm) at a 0–45 cm range. The samples from the SL ecosystem recorded 2.87 (mm) as an average MWD at 0–45 cm with a range of 2.06 to 3.51 (mm), whereas, in the FP, it ranged between 2.40 and 3.80 (mm), with 2.95 (mm) being the average value.

The MWD of TP ranged between 1.60 and 3.60 (mm), and 2.58 (mm) was recorded as the average. CL samples ranged from 1.46 to 3.33 (mm), with 2.33 (mm) being the average. The MWD from each ecosystem is as follows: EF (3.15 mm) > DF (3.01 mm) > FP (2.95 mm) > SL (2.87 mm) > TP (2.58 mm) > CL (2.33 mm) (Figure 3). EF contains a 4.44%, 8.89%, 6.35%, 18.10%, and 26.03% higher MWD when compared to DF, SL, FP, TP, and CL. Except for SL and FP, each ecosystem was statistically significant.

3.7.3. Water-Stable Aggregates (WSAs)

The WSAs of the EF at 0–45 cm depth ranged between 80.58 and 97.42 (%) and the average value recorded was 89.66 (%). The WSAs of the soils collected from DF at 0–45 cm ranged from 52.03 to 97.67 (%), with a mean of 85.08 (%) (Figure 3). The WSA of SL was 80.25 (%) as an average at 0–45 cm with a range from 63.16 to 96.24 (%), whereas, in the FP, it ranged between 63.56 and 98.21 (%), with 84.63 (%) being the average value. The WSA of TP ranged between 46.80 and 96.23 (%), and 73.50 (%) was recorded as an average, whereas, in CL, it ranged between 48.57 and 88.47 (%), with 67.79 (%) being the average.

The amount of WSA in each ecosystem is as follows: EF (89.66%) > DF (85.08%) > FP (84.63%) > SL (80.25%) > TP (73.50%) > CL (67.79%). EF contains 5.10%, 10.50%, 5.61%, 18.02%, and 24.39% higher WSAs when compared to DF, SL, FP, TP, and CL. A significantly ($p < 0.01$) higher WSA was recorded in EF, whereas the DF, FP, CL, and TP were on par with each other.

3.8. Soil Organic Matter (SOM)

The concentration of SOM across the ecosystem in the NHR is as follows: EF (14.72%) > DF (9.82%) > FP (8.42%) > SL (7.44%) > TP (4.11%) > CL (3.18%) (Figure 4). The SOM was

highest at the surface soil and the concentration was depleted across the depth. Regardless of the ecosystems, 30–45 cm was the minimum and 0–15 cm was the greatest SOM measured. The SOM recorded in EF was 33.32%, 49.45%, 42.78%, 72.08%, and 78.38% higher than DF, SL, FP, TP, and VL.

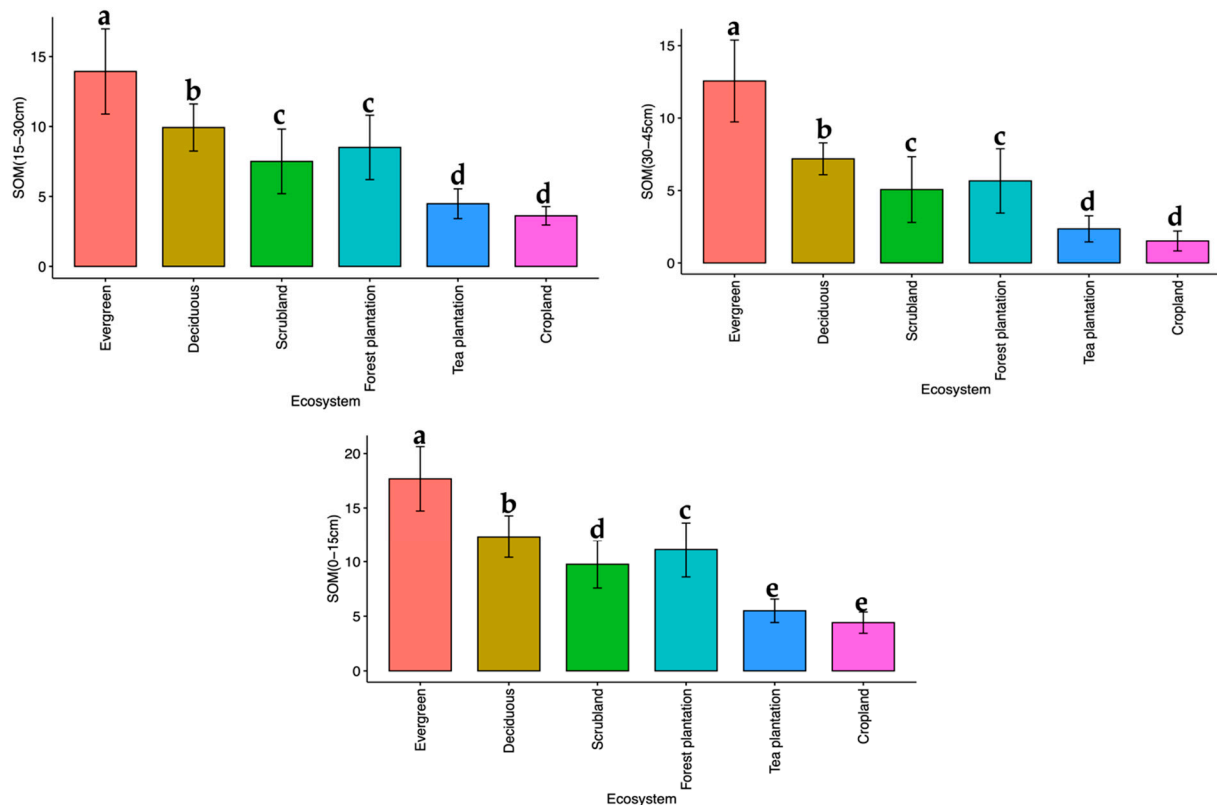


Figure 4. Soil organic matter (%) under different ecosystems of Nilgiri Hill Region (NHR). The figure displays how changing land use has an impact on soil organic matter (%) in different ecosystems of the NHR. According to DMRT, histograms with distinct letters differ significantly ($p < 0.01$).

3.9. Soil Quality Assessment

The SQI and RSQI of the studied physical parameters and SOM were calculated using additive index and weighted index methods (Table 3). Additive index—SQI using AI followed EF > DF > FP > SL > TP > CL. In comparison to DF, SL, FP, TP, and CL, the RSQI recorded in EF was 7.84%, 17.48%, 10.67%, 31.08%, and 33.43% higher. Weighted index—SQI using WI followed EF > DF > FP > SL > TP > CL. EF recorded 100% RSQI and it was greater than DF, SL, FP, TP, and CL by 4.31%, 18.08%, 2.51%, 37.61%, and 41.05%.

Table 3. Soil quality index and relative soil quality index under different ecosystems of NHR.

S. No	Ecosystem	Additive Index (AI)	RSQI (AI) (%)	Weighted Index (WI)	RSQI (WI) (%)
1.	Evergreen	7.32	100.00	0.53	100.00
2.	Deciduous	6.75	92.16	0.51	95.69
3.	Scrubland	6.04	82.52	0.44	81.92
4.	Forest plantation	6.54	89.33	0.52	97.49
5.	Tea plantation	5.04	68.92	0.33	62.39
6.	Cropland	4.87	66.57	0.31	58.95

3.10. Relationship between Soil Physical Properties

The NHR’s soil physical characteristics indicated a wide range of variability (Figure 5). According to the principal component analysis (PCA) findings, ASM, sand, and SOM were mostly responsible for 59% of the variability. The contribution of BD, PS, and PD results in a variable clustering of CL and TP at the extreme right of the biplot and makes it unique when compared to other ecosystems studied.



Figure 5. Principal component analysis of soil physical properties and soil organic matter in different ecosystems. [Available soil moisture (ASM), hydraulic conductivity (HC), pore space (PS), particle density (PD), aggregate stability (Ag S), bulk density (BD)].

4. Discussion

4.1. Particle Size Distribution (PSD)

The process of soil formation has largely controlled the inherent characteristic of soil texture. Within the short term, it may not change, and management practices have a negligible impact on it [48]. However, in the long run, the land use change as well as management strategies used in the ecosystems influence the soil texture. The sand content in CL and TP (46.83% and 41.97%) was the highest, whereas FP, EF, DF, and SL recorded the lowest values (19.83%, 19.97%, 21.50%, and 25.85%). These results indicate that when the natural forest is changed to various land uses, the content of sand is increased (CL and TP). This might be due to the dense canopy cover of natural forests preventing their

soils from getting exposed to natural processes when compared to CL and TP, resulting in rainfall–runoff–soil erosion. These processes remove finer clay particles from their surface and thereby leave behind the heavier sand and silt particles [49,50]. The decrease in the sand across the ecosystems of the NHR was rapid between surface layers (0–15 cm and 15–30 cm) when compared to the sub-surface layers (15–30 cm and 30–45 cm depths). The process of soil erosion had a higher impact on surface soils and therefore the lower depth shows a decrease in sand content. In addition, the findings of Ayele et al. [51] suggested that soil erosion has a significant impact on sand content; hence, this can serve as an indicator for the soil degradation of any ecosystem, if measured in periodical intervals.

The forest ecosystem (EF, DF, SL, and FP) recorded a higher silt content (13.79%, 14.08%, 12.24%, and 15.80% at 0–45 cm depth) whereas the TP and CL recorded the lowest silt content (11.75% and 11.90% at 0–45 cm). This in fact is due to the damage to soil structure due to the trampling effect, and this, in turn, loosens the soil particles and thereby welcomes the erosion. In addition to this, CL and TP had negligible understorey vegetation, thereby suggesting a higher chance for erosion, which fades away the finer clay and silt particles. The content of silt at an increased depth followed a similar trend to sand content, which was congruent with previous studies [52].

The percentage of clay content increased with an increase in depth; thus, the content of clay on surface soils was less than on sub-surface soils. Among the land uses, EF, DF, SL, and FP had the highest content of clay (66.25%, 64.42%, 61.91%, and 64.37% at 0–45 cm) while those under CL and TP recorded a lower percentage of clay (46.28% and 41.27% at 0–45 cm), respectively. The higher clay content in sub-surface soils is because of the translocation of clay due to illuviation from surface soil. Thus, the highest clay content recorded in each ecosystem was at 30–45 cm depth. This is consistent with Alabi et al. [42] and Yeshaneh [53], who noted that sub-surface soils have a larger clay content than surface soils. This confirms that the clay content in the surface soils of CL and TP might have been removed due to erosion, which leaves the coarse sand fragments in this ecosystem [54].

These results provide us opportunities to protect the soils of CL and TP, which are easily prone to natural/anthropogenic-induced erosion, because of their decline in finer particles from the surface, which has the ability to resist the adverse effect. The soil in the study area had a variety of textures, from clay loam to clay. More specifically, these results suggest that the conversion of native forestland to CL and TP has inflicted a variation in texture due to soil pedogenic processes such as aggradation and is enhanced by surficial erosion (degradation). Thus, it is evident that LUC is known to damage the physical properties in the soil such as disturbance in soil macro aggregates (soil structure), and this in turn makes the converted land more susceptible to erosion. Other properties of the soil, such as soil depth, nutrients available to the soil, and soil texture, can also be affected by erosion [55].

EF registered the lowest BD and PD, whereas the CL and TP were denser among all ecosystems studied. These results corroborate the findings of Padbhushan et al. [56] and Getachew et al. [57], who recorded higher density values in croplands. The poor soil aggregation due to a lower clay and organic matter content along with the continuous cropping system and tillage might have resulted in a higher BD and PD in CL and TP, whereas, in the case of EF, DF, SL, and FP, the high organic matter content has resulted in better soil aggregation and structure. This suggests that tillage and deforestation in the NHR can compact the soil and lower the infiltration rate, thus increasing the densities [58,59]. The roots in the surface, along with the higher organic matter, have resulted in a lower BD in surface soils. Each ecosystem's soils became denser as depth was increased. This is similar to the findings of Alabi et al. [42] and Getachew et al. [57]. When compared to 15–30 and 30–45 cm, the increase in density was seen to be greater between 0–15 and 15–30 cm. Thus, the BD is a measure of soil compaction and quality. Higher BD values indicate greater soil compaction, which reduces other soil qualities like water-holding capacity and accessible soil moisture. Conversely, lower BD values indicate less soil compaction, which improves soils' capacity to hold more water [60]. Our findings are similar to Ravina [61],

who reported a lower BD in native forestland when compared with other ecosystems. It can be concluded that upon conversion of native forestland, the soil BD is increased due to compaction. The management of land and history of land uses have brought a significant variation in both BD and PD [62]. The levels of an upper and lower threshold for BD were determined as per [63] and the baseline was 1.5 g cm^{-3} , and the values above this indicate the probability of an increased soil compaction. With this value, the BD of EF, DF, SL, and FP falls within the optimal range of BD. In addition, [64] once proclaimed that fertile soil has a BD of up to 1.5 Mg m^{-3} . As the BD of surface soils (0–15 cm) falls within this range, the productivity in this location (NHR) is considered higher in terms of BD.

The lower PD encountered at EF, DF, FP, and SL could be due to a higher SOM. Our findings are consistent with [65], who observed a lower PD in the soils of forests when compared to the meadow and arable soil. The higher PD in CL and TP corroborated the findings of [66], who reported that the PD of arable (ploughed) soils was significantly greater than in forest and pastureland. Information on PD occurring in the literature varies significantly in relation to land management. All ecosystems witnessed an increased PD over depth increments. The untouched EF and DF along with SL and FP are characterized by dense vegetation; therefore, the dispersion and translocation of clay will be less. This hinders the higher permeability and soil erosion when compared to CL and TP. In the unprotected CL and TP, the removal of finer particles by erosion increases the sand content and thereby enhances the PD [67]. The reduction in organic matter at an increased depth might have led to a higher PD with depth increments [68]. CL and TP upon management and mechanization [69] result in a cohesive sub-surface due to the impact on soil structure [69–71].

The significant variance ($p < 0.01$) in SOM reflects the dynamics of PD among ecosystems. According to [72], the decrease in PD with an increased SOM may be related to the 'dilution impact' of mineral particles in soils. Furthermore, organic matter has a smaller PD than soil mineral components. At 0–45 cm, the correlation data (Figure 6) revealed a negative association ($r = -0.65$) between PD and TOC [73–75].

According to [76], the sensitive and significant reactivity of PD to LUC and SOM content suggested that PD might be used as a measure to assess the influence of soil management on the carbon pool. The loss of OC because of forest conversion to agricultural systems certainly resulted in significant ($p < 0.01$) increases in PD in CL and TP. Furthermore, a reduced soil aggregation after conversion from EF and DF resulted in an increased PD in CL and TP. The continual use of heavy gear for soil tillage may have also hastened this process.

Soil porosity [77] facilitates the movement and availability of air and water [78]. Land use type had a significant effect on the PS ($p < 0.01$). The overall PS was higher in EF, DF, and FP. This suggests that the transition from forest to cultivable land was accompanied by intensive land cultivation without adequate management, resulting in a decrease in the overall PS. This might have a considerable negative impact on the available moisture content and infiltration capability of soils [79]. With more depth came a decrease in PS. The highest PS recorded in EF and DF was due to their higher organic carbon content. Our finding confirms the reports of [48]. The soil compaction in CL and TP directly impels the decreased PS, owing to a higher density. The results of our findings are in line with [72].

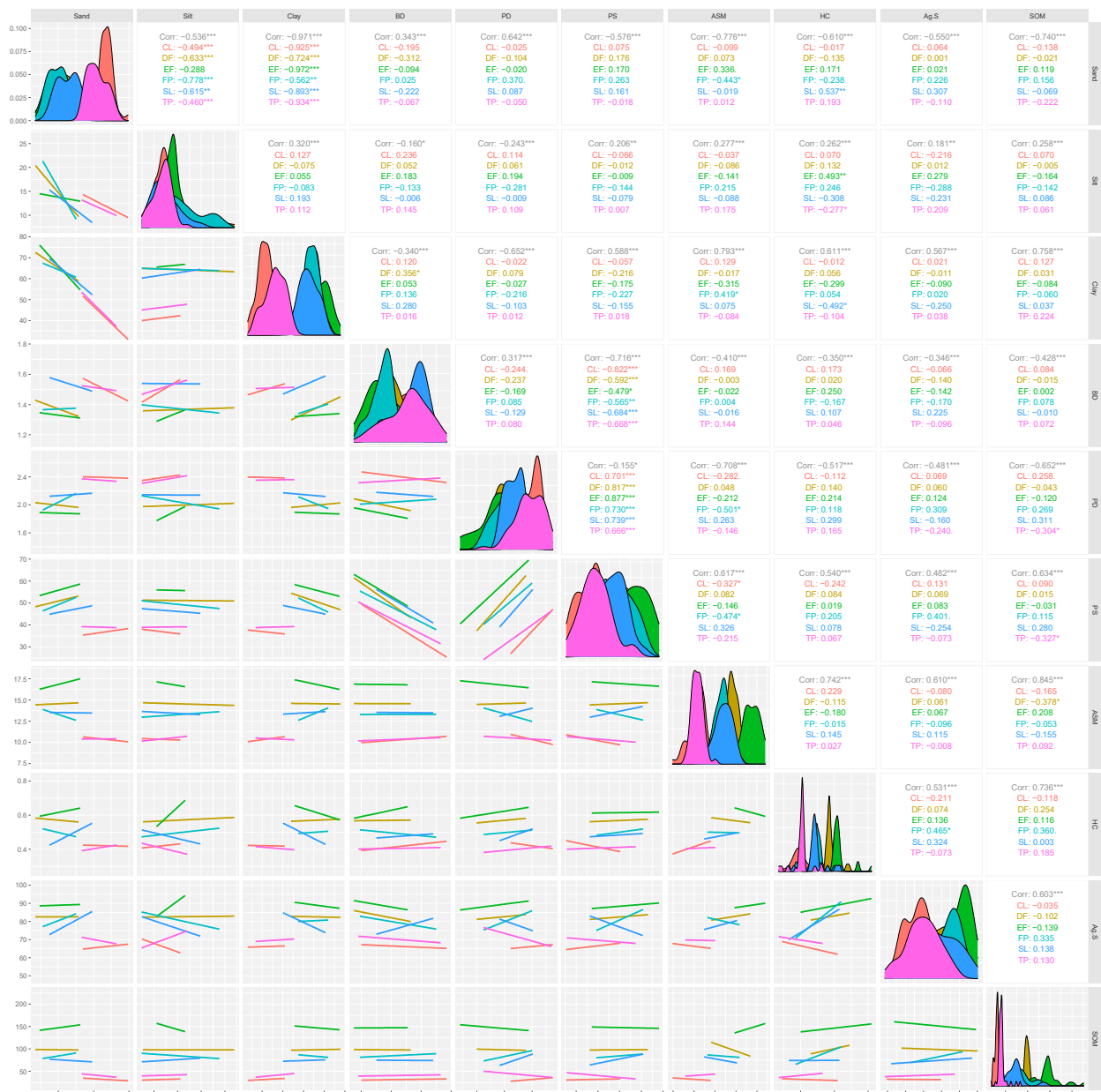


Figure 6. Distribution of soil physical properties and soil organic matter under different ecosystems, and the correlation values with * specify significant correlations (significant levels: 0 *** 0.001 ** 0.01 * 0.05 ' 0.1 ' ' 1).

4.2. Available Soil Moisture (ASM)

The ASM recorded in this study showed a significant variation among ecosystems. It followed EF (16.83%) > DF (14.57%) > SL (13.50%) > FP (13.30%) > TP (10.40%) > CL (10.35%). This is in agreement with Zerga [80,81], who reported a higher soil moisture content in forests than in other land uses. The decreased ASM in CL and TP was attributed to the lack of sparse undergrowth and light canopy, which can cause increased evaporation (Figure 7). Land use is considered a major factor in controlling soil moisture availability [82,83]. By controlling the processes involved in determining the soil’s physical, chemical, and biological properties, soil moisture plays an important part in the soil system [84]. This process, in general, strongly depends upon the soil moisture. For example, in the case of water infiltration, the amount of moisture in the soil determines how much water will flow across the soil profile and percolate to recharge the aquifers [85].

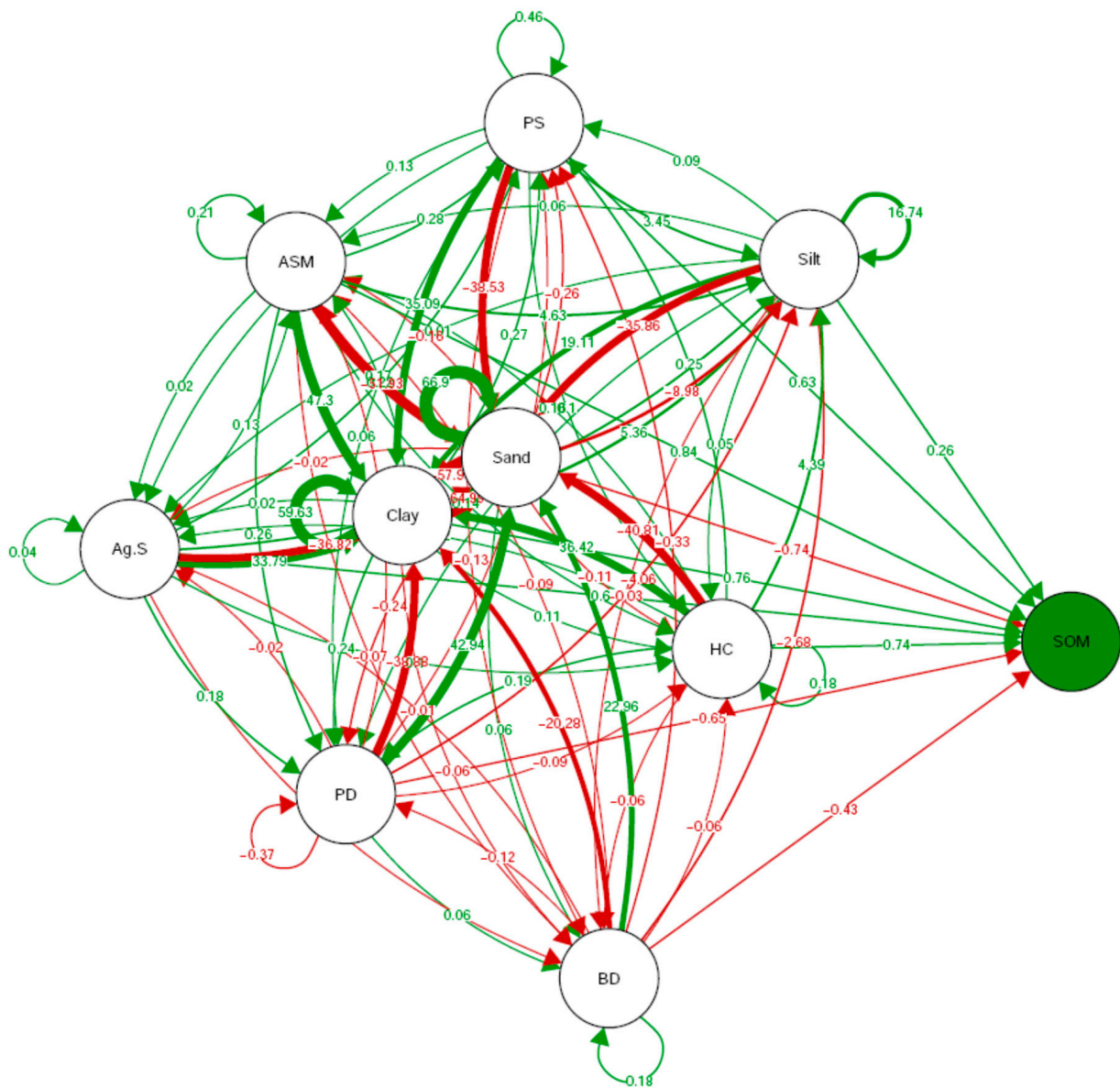


Figure 7. Influence of soil physical properties on soil organic matter. (Values and effects are collinear. Values embedded in each soil properties represent the individual effect, whereas the values directed toward the soil organic matter (SOM) represent the effect of each parameters on soil organic matter.) [Available soil moisture (ASM), hydraulic conductivity (HC), pore space (PS), particle density (PD), aggregate stability (Ag S), bulk density (BD)].

However, in other ecosystems, thick undergrowth can decrease soil temperature, increasing moisture availability through water penetration and decreasing evaporation. This is in agreement with [86,87]. Despite the fact that rainfall and stand density affect the understory [81], in FP, the litterfall (eucalyptus plantations) has an allelopathic effect and reduces the growth of other plants. In addition, the chemicals from the litters of eucalyptus decrease the nutrient availability for the undergrowth and thus pave the way to the degradation of soil quality through the erosion and depletion of soil water and nutrients. Thus, it is clear from this study that FP (eucalyptus), due to its allelopathic effect, suppresses the undergrowth and the plants nearby. In this study, the ASM in FP was lower than in other forest types (EF, DF, and SL). This is in agreement with other report findings by [86]. In all ecosystems, the ASM steadily declined as depth increased. ASM is influenced by a number of variables, including topography, vegetation, and texture [88].

Therefore, the variations in soil texture, topography, and vegetation cover may be the cause of the disparities in ASM. Furthermore, due to the higher SOM content (Figure 4) and high water-holding capacity, the soil surface moisture of all ecosystems is significantly higher ($p < 0.01$) than the subsurface.

4.3. Hydraulic Conductivity (K_{sat})

The ease of water transport via the pore space is shown by soil K_{sat} . It is considered an important factor for the growth of plants through its relation to porosity, temperature, available water to plants, and so on [89]. Thus, it is essential to improve the K_{sat} in order to prevent erosion and avoid runoff [90]. As per our hypothesis, K_{sat} varied significantly among the major ecosystem of the NHR. The result of our findings is similar to other previous reports, which portray that the change in land use type and vegetation showed a variation in K_{sat} [91]. The difference in vegetation, root characteristics such as the biomass, and its distribution among each land use can impact the K_{sat} [83,92–94]. The roots in forest ecosystems like DF and EF are obvious and they extend deeper in the soil profile. In the case of TP and CL, the roots do not extend deeper into the profile when compared to the forest type. In most ecosystems, the vegetation types are characterized by a denser root length and higher surface area in the upper layer of the soil profile and gradually decrease along with the depth. These roots have a direct impact on the soil's physical properties (texture, structure, and so on) through their extrusion and insertion mechanism in the soil [95]. Their distribution affects the texture [96] by means of their adjustments made on litter produced in the soil or on the surfaces, and this, in turn, is related to the organic matter status and other properties of soil [97–99]. The microbial community influences the physi-co-chemical characterisation of the soil by creating gas, gel, and other solid phases as a result of these roots invading the soils and stimulating the microbial biomass and rhizospheres [100]. This in turn controls the overall volume of space that is open to the water flow and, as a result, the K_{sat} . The effect of depth on K_{sat} differed among all the ecosystems. The variation was little in the native forestland (EF and DF) when compared with others. This may be due to the root distribution and physicochemical properties of the soil [101]. These results agree with findings from Celik et al. [102], Fu et al. [103], and Van Lier et al. [104].

The variation recorded in TP and CL was more than in other ecosystems and this may be due to the higher distribution of roots and stems on the surface than underground, which could have affected the values of K_{sat} [83,105]. In the NHR, the decrease in hydraulic conductivity in TP and CL can also be attributed to the implements that usually remove the surface residues, and the management activities along with more than two crops per year can ultimately trigger the sub-surface compaction. This could help to explain some of the upward trend in soil depth from 15 to 30 cm to 30 to 45 cm. The pore characteristics and their dimensions and distribution are found to have an impact on K_{sat} . EF, DF, SL, and FP recorded a stable K_{sat} with an increase in depth. This is in agreement with the findings of Hao et al. [101].

Reports suggest that K_{sat} values are minimal in less aggregated soil than in the larger ones [106]. This could be due to the addition of fresh organic matter produced by various types of vegetation [107], and the litter generated by them simulates aggregation in the soil, [108] which in turn has an influence on BD and PS, [107,109] which are closely related to the adjustments in K_{sat} [110]. This in turn enlightens the complex nature of the hidden treasure below our feet [105].

4.4. Soil Aggregate Stability (SAS)

Soil structure is an important factor in supporting agricultural output, maintaining environmental quality, and delivering ecosystem services [111–113]. This is because soil structure is inextricably linked to the soil's ability to accept, retain, and transfer water, and resist surface crusting, soil erosion, and nutrient cycling [114]. The term "soil structure" refers to the heterogeneous organization of soil solids and void spaces over time, whereas

“soil structural stability” or “soil aggregate stability” refers to the soil’s ability to maintain the physical arrangement of particles (resulting in voids) under various stress levels [115]. Thus, SAS is utilized as an indication of the persistence of soil structure [116–118] and soil condition [119,120]. A soil obtains its structure through soil aggregation in which soil particles are reorganized by cementation and flocculation, which, in turn, are governed by TOC, ion bridging, clay, carbonates, and biota [113,121,122]. Thus, by its interaction with soil mineral particles, TOC plays a crucial role in soil aggregation and leads to a decrease in aggregate wettability and an increase in aggregate mechanical strength (which is a measure of the coherency of inter-particle connections) [123]. These stable aggregates, in turn, physically protect the TOC from microbial degradation and extend its duration in the soil [124–126]. Thus, a higher TOC in EF, DF, SL, and FP results in a greater SAS. The SAS was higher in the forest than in other land uses, according to Shrestha et al. [127], who also found this. Variations in the organic matter input and plant species in an ecosystem can impact the aggregation of soil [128]. In EF, DF, SL, and FP, we noticed a higher proportion of bigger macro aggregates (>2.0 mm), whereas in TP and CL, we found a higher proportion of smaller micro aggregates (0.053 mm) and silt and clay-sized particles (<0.053 mm) in the total aggregated. This might be traced to a reduced soil disturbance coupled with a higher organic input, delayed decomposition, and organo-mineral complexation in EF, DF, SL, and FP as compared to TP and CL. Moreover, owing to the natural settings, the hydrophobicity in forest soils [129,130] could have been preserved, and this, in turn, can increase soil aggregation [112] by increasing cohesiveness and minimising soil aggregate dispersion [131,132]. In line with the findings of [133], we observed that the proportion of aggregate fractions other than >2 mm size was significantly ($p < 0.01$) higher in CL and TP. This may be due to the minimal availability of TOC concentration for binding the aggregates to macro aggregates. In agreement with our observations, various research studies have found that the macro aggregates were higher in the native ecosystem (such as forests), whereas the LUC from native forests to other land uses has increased the proportion of micro aggregates [134,135]. Furthermore, we discovered the abundance of water-stable aggregates in forestland compared to in other land uses, which, in turn, corroborates the findings of [135,136].

In addition, the discrepancies in SAS under different ecosystems might also be attributed to variances in the soil organic carbon. Tillage, according to numerous results, greatly reduces SAS [137,138], whereas non-tillage enhances SAS [137]. Regular tillage has been associated to a drop in the biomass of fungal hyphae that decreases the stability of SOM [139,140]. In this study, ecosystems with plant cover (EF, DF, SL, and FP) exhibited a greater SAS than CL and TP. This emphasizes the significance of adding litter as a possibility to improve soil microbial and fungal biomass for increased SAS [136,141]. Other investigations related to aggregate stability have demonstrated a stable nature of soil properties in the undisturbed native land soil properties due to the slow rate of organic matter decomposition [142,143]. Our results show a positive correlation between aggregate stability and SOM (Figure 5). SAS is a convenient substitute for the key elements, which indicate soil quality due to its relationship with the basic soil quality indicators.

The SOM was substantially lower under CL and TP than under EF, DF, SL, and FP (Figure 6). The concentration of SOM is primarily determined by the dynamics of SOM inputs and microbial breakdown [144]. Leaf litter and root exudates connected to plant biomass contributed to SOM [145], but crop inputs were not returned to CL soils, which in turn reduced the SOM accumulation in CL [146]. Tillage additionally provides oxygen to the soil and exposes organic material from deeper layers, which speeds up the disintegration of SOM. The rate of SOM decomposition in CL normally increases with an increasing time due to an increased soil biological activity [147]. Plant biomass in this study increased from CL to DF and EF due to an increased SOM input. Soil microorganisms readily absorb fresh and labile SOM, resulting in the production of stable organic compounds [148] that, in turn, stabilize the soil carbon pool. The physical separation provided by macro aggregates protects the organic substrate from microbial attack. CL’s

reduced macro aggregate fraction, therefore, limits its physical barrier and ultimately results in a faster microbial breakdown. This implies that any conversion of natural land would cause climate change by destabilizing SAS and releasing enormous amounts of carbon into the atmosphere. This effect would be particularly evident in the NHR as a whole due to its extremely fragile and dynamic landscape.

Conclusively, the SAS, which showed a similar tendency with large macro aggregates under different ecosystems, was higher in EF, DF, SL, and FP than in CL and TP (Figure 3). This is due to a high organic matter input from aboveground biomass beneath these ecosystems [65], which, in turn, offers a mulching effect and a better habitat for soil flora and fauna, and ultimately helps in improving the soil aggregations. The hydrophobic characteristics may also have a role in improving SAS in forest ecosystems [112].

Way Forward

The results of the study confirm that the conversion of natural vegetation or forested areas to agriculture often affects the soil quality. As a result, topsoil and nutrients are lost, the water-holding capacity and compaction of the remaining soil are decreased, and the NHR's soils' physical properties are affected. It is suggested to adopt the following management strategies:

- Implement erosion control measures such as contour plowing, terracing, and buffer strips to reduce runoff and soil loss.
- Adopt conservation tillage practices like no-till or minimum tillage, which help to preserve soil structure and reduce erosion.
- Maintain vegetative cover through cover cropping, crop rotation, or planting grass or trees to stabilize soil and reduce erosion.
- Use regenerative agriculture and agroforestry for diversified products and environment and soil amelioration.
- Avoid excessive machinery traffic or use controlled traffic farming systems to minimize soil compaction.
- Practice timely tillage or subsoiling to alleviate compaction and improve soil structure.
- Incorporate organic matter through composting or cover cropping to enhance soil aggregation and reduce compaction.

In addition, such land use changes involving deforestation or the conversion of grasslands to croplands resulted in a decline in soil organic matter content. Soil organic matter is crucial for nutrient cycling, moisture retention, and soil structure; hence, it needs to be maintained. This can be achieved by adopting sustainable agricultural practices such as conservation agriculture, agroforestry, or organic farming that promote the accumulation of organic matter, applying organic amendments like compost or manure to increase the soil organic matter content, and implementing crop residue management techniques like mulching or cover cropping to enhance organic matter inputs.

5. Conclusions

For sustainable land management and ecosystem conservation, it is essential to understand the effects of various land use practices and how they affect the physical properties of soil. This study confirms that a multidisciplinary approach allows for a more holistic assessment of the complex interactions between land use practices and soil physical qualities and that these properties are significantly impacted by the change in land use. In EF, DF, SL, and FP, there was a noticeably higher concentration of clay, available soil moisture, water-holding capacity, and large macro aggregates (due to a higher soil organic carbon content). However, a decrease in SOC stocks in turn instigates a loss of SOM and aggregate stability. Thus, the response degree of soil physical properties and SOM concentrations varied across land use types. The impact of LUC was more pronounced in available soil moisture, sand, soil organic matter, and clay, which made a significant contribution toward the variations among the parameters studied. The key findings are that altering the natural setting has a major negative impact on the soil physical quality. These findings suggest an explicit consideration of soil physical properties to undermine management strategies'

impact on soil quality and provide a theoretical basis for ecological restoration in the NHR of Western Ghats' global biodiversity hotspot along with the possible management strategies. Our findings emphasize the importance of implementing strategies like carbon management in both croplands and tea plantation ecosystems. However, our study did not specifically focus on improving the soil's capacity to retain more carbon; we suggest this aspect to be explored more thoroughly in future research endeavors.

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