

**ASSESSMENT OF SOIL ECOLOGICAL FUNCTIONS AND PROCESSES
ALONG RURAL-URBAN GRADIENT IN SHIVAMOGGA DISTRICT**

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***DEPARTMENT OF SOIL SCIENCE AND AGRICULTURAL CHEMISTRY
COLLEGE OF AGRICULTURE, SHIVAMOGGA***

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UNIVERSITY OF AGRICULTURAL AND HORTICULTURAL SCIENCES
SHIVAMOGGA**

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**KELADI SHIVAPPA NAYAKA
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CERTIFICATE

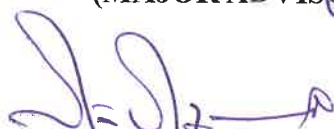
This is to certify that the thesis entitled 'ASSESSMENT OF SOIL ECOLOGICAL FUNCTIONS AND PROCESSES ALONG RURAL-URBAN GRADIENT IN SHIVAMOGGA DISTRICT' submitted in partial fulfillment of the requirements for the award of the degree of **DOCTOR OF PHILOSOPHY in SOIL SCIENCE AND AGRICULTURAL CHEMISTRY** to the College of Agriculture, Shivamogga. Keladi Shivappa Nayaka University of Agricultural and Horticultural Sciences, Shivamogga is a bonafide record of research work carried out by **Sudarshan Varma, ID NO. PA1TBZ129** (sudarshanvarma95@gmail.com) during the period of study in this university under my guidance and supervision and no part of this thesis has previously formed the basis for the award of any other degree, diploma, associateship, fellowship or any other similar titles.

Shivamogga
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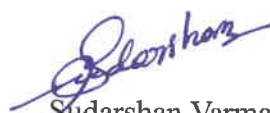
(Sudarshan Varma)

**Assessment of Soil Ecological Functions and Processes along Rural-Urban Gradient
in Shivamogga District
(Sudarshan Varma)**

ABSTRACT

A study was undertaken to evaluate the effects of urbanization on land-use patterns, biomass and nutrient turnovers, soil ecological functions and processes. Land-use and land-cover (LULC) analysis across urban peri-urban and rural zones revealed significant soil sealing variations. Urban zones recorded the highest proportion of sealed areas (65.59 %), while peri-urban zones showed moderate sealing (19.26 %) and rural zones recorded lowest soil sealing (5.76 %). Biomass and nutrient turnover differed across land-use types. Among the land use systems, organic matter recycling was significantly higher in arecanut plantations (10.50-10.65 Mg ha⁻¹ yr⁻¹) followed by Forest/forest plantations (9.01-9.38 Mg ha⁻¹ yr⁻¹). Across the gradient, urban zone exhibited lesser biomass and nutrient turnovers, while rural zone recorded higher turnovers. Soil ecological functions in terms of physical, chemical and biological properties varied significantly along the gradient as influenced by the extent of soil sealing. Urban soils exhibited higher bulk density (1.42 Mg m⁻³) and lower porosity (35.75 %). Soil organic carbon (SOC) was highest in forest/forest plantations (6.83 g kg⁻¹) and lowest in sealed areas (2.72 g kg⁻¹). Soil nutrient status, carbon and nitrogen fractions were also greatly altered by the intensity of urbanization. Microbial populations and enzymatic activities were higher in rural soils than urban soils indicating greater biological activity. Soil Quality Index (SQI) was lowest in sealed areas of urban zone (0.20) and highest in forest and agricultural soils of rural zone (0.64). The study highlights the adverse impacts of urbanization on soil ecological functions and underscores the need for sustainable land management to mitigate soil degradation, enhance productivity and promote ecological resilience in urbanizing landscapes.

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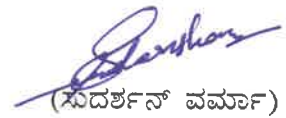

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ಶಿವಮೊಗ್ಗ ಜಿಲ್ಲೆಯಲ್ಲಿನ ಗ್ರಾಮೀಣ-ನಗರ ಅನಂತರದ ಮಣ್ಣಿನ ಪರಿಸರ ಕ್ರಿಯೆ ಮತ್ತು ಪ್ರಕ್ರಿಯೆಗಳ ಮೌಲ್ಯಮಾಪನ
(ಸುದರ್ಶನ್ ವರ್ಮಾ)

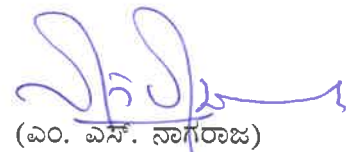
ಸಾರಾಂಶ

ಇಂದಿನ ನಗರೀಕರಣವು ಭೂ-ಬಳಕೆ ಮಾದರಿಗಳು ಹಾಗೂ ಅವುಗಳಲ್ಲಿನ ಮಣ್ಣಿನ ಗುಣಗಳನ್ನು ಬಹಳಷ್ಟು ಬದಲಾಯಿಸಿದೆ. ಜೈವಿಕರಾಶಿ, ಪೋಷಕಾಂಶ ಚಕ್ರಣ ಹಾಗೂ ಮಣ್ಣಿನ ಪರಿಸರ ಕ್ರಿಯೆ-ಪ್ರಕ್ರಿಯೆಗಳನ್ನು ಶಿವಮೊಗ್ಗ ಜಿಲ್ಲೆಯ ನಗರ, ಅರೆ-ನಗರ ಮತ್ತು ಗ್ರಾಮೀಣ ವಲಯಗಳಲ್ಲಿಯ ಮಣ್ಣಿನ ಗುಣಗಳನ್ನು ಆಧರಿಸಿದ ಅಧ್ಯಯನವೊಂದನ್ನು ಕೈಗೊಳ್ಳಲಾಗಿತ್ತು. ಮೇಲಿನ ವಲಯಗಳ ಭೂ-ಬಳಕೆಯ ವಿಶ್ಲೇಷಣೆ ಹಾಗೂ ಮಣ್ಣು ಪದರಗಟ್ಟಿಗೊಳ್ಳುವಿಕೆಯಲ್ಲಿ ಗಮನಾರ್ಹವಾದ ವ್ಯತ್ಯಾಸಗಳು ಕಂಡುಬಂದಿತ್ತು. ನಗರ ವಲಯದಲ್ಲಿನ ಮಣ್ಣು ಪದರಗಟ್ಟಿಗೊಂಡ ಪ್ರಮಾಣವು ಅತ್ಯಧಿಕವಾದರೆ (೬೫.೫೯ %), ಅರೆ-ನಗರ ವಲಯದಲ್ಲಿ ಪದರಗಟ್ಟಿಗೊಳ್ಳುವಿಕೆ ಮಧ್ಯಮ ಪ್ರಮಾಣದಲ್ಲಿ (೧೯.೨೬ %) ದಾಖಲಾಯಿತು. ಗ್ರಾಮೀಣ ವಲಯದಲ್ಲಿ ಹೆಚ್ಚು ಕೃಷಿ-ಭೂಬಳಕೆಯನ್ನು ಹೊಂದಿದ್ದರಿಂದ, ಪದರಗಟ್ಟಿಗೊಳ್ಳುವಿಕೆಯ ಪ್ರಮಾಣ ಕನಿಷ್ಠವಾಗಿತ್ತು (೫.೭೬ %). ಈ ಅಧ್ಯಯನದಲ್ಲಿ ಜೈವಿಕರಾಶಿ ಮತ್ತು ಪೋಷಕಾಂಶ ಚಕ್ರಣಗಳು ಕೂಡಾ ಭೂ-ಬಳಕೆ ಪ್ರಕಾರಗಳಿಗೆ ಅನುಗುಣವಾಗಿ ಬದಲಾಗುವುದು ಕಂಡುಬಂದಿತ್ತು. ಗ್ರಾಮೀಣ-ನಗರ ಅನಂತರದುದ್ದಕ್ಕೂ ಪ್ರತಿ ಹೆಕ್ಟೇರ್‌ನಲ್ಲಿನ ಸಾವಯವ ಪದಾರ್ಥದ ವಾರ್ಷಿಕ ಪುನಃಚಕ್ರಣದ ಪ್ರಮಾಣವು ಅತಿಹೆಚ್ಚಾಗಿ ಅಡಿಕೆ ತೋಟಗಳಲ್ಲಿ (೧೦.೫೦-೧೦.೬೫ ಮೆಗಾ ಗ್ರಾಂ) ಹಾಗೂ ಅರಣ್ಯ/ಅರಣ್ಯ ತೋಟಗಳಲ್ಲಿ (೯.೦೧-೯.೩೮ ಮೆಗಾ ಗ್ರಾಂ) ಕಂಡುಬಂದಿತ್ತು. ನಗರ ವಲಯದ ಪೋಷಕಾಂಶ ಚಕ್ರಣಗಳಲ್ಲಿ ಇಳಿಕೆ ಕಂಡುಬಂದರೆ, ಗ್ರಾಮೀಣ ವಲಯದಲ್ಲಿ ಹೆಚ್ಚಿನ ಪೋಷಕಾಂಶ ಚಕ್ರಣವು ಕಂಡುಬಂದಿತ್ತು. ಮಣ್ಣಿನ ಭೌತಿಕ, ರಾಸಾಯನಿಕ ಮತ್ತು ಜೈವಿಕ ಗುಣಲಕ್ಷಣಗಳ ಆಧಾರದ ಮೇಲೆ ಮಣ್ಣಿನ ಪರಿಸರ ಕ್ರಿಯೆಗಳ ಅನಂತರದುದ್ದಕ್ಕೂ ಮಣ್ಣಿನ ಪದರಗಟ್ಟಿಗೊಳ್ಳುವಿಕೆಯಿಂದ ಗಮನಾರ್ಹವಾಗಿ ಬದಲಾಗಿತ್ತು. ನಗರ ವಲಯದಲ್ಲಿನ ಮಣ್ಣು ಹೆಚ್ಚಿನ ಪ್ರಮಾಣದ ಸಾಂಧ್ರತೆಯನ್ನು (೧.೪೨ ಕೆ.ಗ್ರಾಂ ಪ್ರತಿ ಘನ ಮೀ) ಮತ್ತು ಕಡಿಮೆ ಸರಂಧ್ರತೆಯನ್ನು (೩೫.೭೫ %) ತೋರ್ಪಡಿಸಿತ್ತು. ಮಣ್ಣಿನ ಇಂಗಾಲದ ಪ್ರಮಾಣವು (SOC) ಅರಣ್ಯ/ಅರಣ್ಯ ತೋಟಗಳಲ್ಲಿ ಗರಿಷ್ಠವಾಗಿಯೂ (೬.೮೩ ಗ್ರಾಂ ಪ್ರತಿ ಕೆ.ಗ್ರಾಂ) ಹಾಗೂ ಪದರುಗಟ್ಟಿದ ಪ್ರದೇಶಗಳಲ್ಲಿ ಕನಿಷ್ಠವಾಗಿಯೂ (೨.೭೨ ಗ್ರಾಂ ಪ್ರತಿ ಕೆ.ಗ್ರಾಂ) ಕಂಡುಬಂದಿತ್ತು. ನಗರೀಕರಣದ ತೀವ್ರತೆಯಿಂದಾಗಿ ಮಣ್ಣಿನ ಪೋಷಕಾಂಶ ಸ್ಥಿತಿಗತಿ, ಇಂಗಾಲ ಮತ್ತು ಸಾರಜನಕ ಅಂಶಗಳು ಸಹ ಗಣನೀಯವಾಗಿ ಬದಲಾಗಿದ್ದಿತು. ಹೆಚ್ಚಿನ ಜೈವಿಕ ಚಟುವಟಿಕೆಗಳಿಂದರೆ ಸೂಕ್ಷ್ಮಾಣುಜೀವಿಗಳ ಸಂಖ್ಯೆ ಮತ್ತು ಕೀಣ್ವಗಳ ಚಟುವಟಿಕೆಗಳು ಗ್ರಾಮೀಣ ಪ್ರದೇಶದ ಮಣ್ಣಿನಲ್ಲಿ ಹೆಚ್ಚಾಗಿ ಕಂಡುಬಂದಿತ್ತು. ಮಣ್ಣಿನ ಗುಣಮಟ್ಟ ಸೂಚ್ಯಂಕವು (SQI) ನಗರ ವಲಯದ ಪದರುಗಟ್ಟಿದ ಪ್ರದೇಶಗಳಲ್ಲಿ ಕನಿಷ್ಠವಾಗಿದ್ದರೆ (೦.೨೦), ಗ್ರಾಮೀಣ ವಲಯದ ಅರಣ್ಯ ಮತ್ತು ಕೃಷಿ ಭೂಬಳಕೆಯ ಮಣ್ಣಿನಲ್ಲಿ ಗರಿಷ್ಠವಾಗಿ (೦.೬೪) ಕಂಡುಬಂದಿತ್ತು. ಈ ಅಧ್ಯಯನವು ಮಣ್ಣಿನ ಪರಿಸರೀಯ ಕ್ರಿಯೆಗಳ ಮೇಲಿನ ನಗರೀಕರಣದ ಪ್ರತಿಕೂಲ ಪರಿಣಾಮಗಳನ್ನು ಎತ್ತಿ ತೋರಿಸುತ್ತದೆ ಹಾಗೂ ಮಣ್ಣಿನ ಹದಗೆಡುವಿಕೆಯನ್ನು ತಡೆಯಲು, ಉತ್ಪಾದಕತೆಯನ್ನು ಹೆಚ್ಚಿಸಲು ಮತ್ತು ನಗರೀಕರಣಗೊಳ್ಳುವ ಭೂಪ್ರದೇಶಗಳಲ್ಲಿ ಪರಿಸರೀಯ ಸ್ಥಿತಿ ಸ್ಥಾಪಕತ್ವವನ್ನು ಉತ್ತೇಜಿಸಲು ಸುಸ್ಥಿರ ಭೂ-ನಿರ್ವಹಣೆಯ ಅವಶ್ಯಕತೆಯನ್ನು ಒತ್ತಿಹೇಳುತ್ತದೆ.

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INTRODUCTION

I INTRODUCTION

Soil is a vital component of the natural environment. Across the globe, all economies rely upon the offerings and services furnished by the natural environment. Soil ecological function can be defined as the capacity of a specific kind of soil to function within natural or managed ecosystem boundaries, to sustain plant and animal productivity, maintain or enhance water and air quality and support human health and habitation. As a natural resource, soil undertakes numerous pivotal ecological functions and processes (Muñoz-Rojas, 2018). Among these, noteworthy functions of soil encompass biomass generation, the prevention of noxious pollutants from infiltrating food chains through its filtration, buffering, transformation capabilities, providing habitat for important microorganisms, nutrient cycling, water retention and sequestration of carbon (C). The qualities of the soil govern its functions, which in turn regulate ecosystem services and human well-being (Liu *et al.*, 2022).

The significance of soil extends to multiple Sustainable Development Goals (SDGs), contributing directly to food security (SDG 2: Zero Hunger) by supporting agricultural productivity. It also plays a crucial role in water purification and conservation (SDG 6: Clean Water and Sanitation) by filtering contaminants and improving water quality. Furthermore, soil acts as a carbon sink, aiding in climate change mitigation (SDG 13: Climate Action) by storing organic carbon. Additionally, sustainable soil management helps by combating land degradation and desertification (SDG 15: Life on Land), preserving terrestrial ecosystems (Nagaraja *et al.*, 2018). Ensuring soil health is, therefore, integral to achieving these global objectives for sustained development.

Urbanization is an integral part of nation's development. Urbanization refers to the process by which an increasing proportion of a population migrates from rural areas to urban centres, leading to the growth and expansion of cities (Anestis and Stathakis, 2024). This phenomenon is often driven by factors such as industrialization, economic opportunities, improved infrastructure and better access to services. Thus, the future growth of the world population will almost entirely occur in urban areas of developing countries (Mahtta *et al.*, 2022). As of 2023, around 56 per cent of the global population resides in urban areas. The progress of urbanization is very rapid in in developing nations. Projections indicate that by 2050, approximately 68 per cent of the world's population will live in cities. In India, urban areas are home to about 36 per cent of the population (Census, 2011). Although India's urbanization rate is slower compared to some other developing countries, its urban population is expected to rise significantly, reaching 40 per cent by 2030 and 50 per cent by 2050. Karnataka, one of India's more urbanized states, had approximately 38.6 per cent of its population living in urban areas (Census, 2011).

As urban areas continue to develop, they tend to encompass more land and other natural resources, altering the landscape and environment (Awuah and Abdulai, 2022). Soil sealing is the most significant land cover change caused by urbanization. Soil sealing refers to the process of covering natural soil surfaces with impermeable materials, such as asphalt, concrete, or buildings, which prevent the natural exchange of water, air and nutrients between the soil and the atmosphere (Vieillard *et al.*, 2024). This phenomenon is a direct result of urbanization and construction activities. When soil is sealed, it loses its ability to perform essential ecological functions and processes, leading to various environmental and ecological impacts.

Rapid urbanization presents a significant challenge to sustainable development (O'Riordan *et al.*, 2021). The conversion of land for urban infrastructure reduces habitat availability and disrupts plant growth and natural processes. Impermeable surfaces in urban areas lead to water-related issues like flooding, decreased groundwater recharge and altered nutrient cycling. Soils are natural filters that help to remove pollutants and impurities from the water as it percolates through. Urbanization with its hard surfaces increases the volume and speed of runoff, bypassing this natural purification process and leading to increased pollution in water bodies. Furthermore, urbanization hinders C sequestration and diminishes biodiversity as species adapted to specific soil conditions are lost (Nagaraja *et al.*, 2016). The heat-retaining nature of urban landscapes creates heat islands, affecting soil organisms and processes. Construction-related soil disturbance harms essential microorganisms, while vegetation removal and soil disruption contribute to erosion and sedimentation in water bodies, negatively impacting aquatic ecosystems and water quality. In essence, urbanization's detrimental effects encompass various aspects of soil health, ecosystem function and environmental balance (Tombolini *et al.*, 2022).

The metropolitan regions grow by altering rural landscapes. A series of land uses evolves from agricultural and natural areas to built-up urban zones. Such land-use changes can have a significant impact on soil ecosystems, possibly affecting their physical, chemical and biological properties (Chen *et al.*, 2022). The rural-urban gradient, characterized by a spatial continuum ranging from undisturbed rural environments to densely developed urban areas, provides a unique opportunity to study the complex interactions between human activities and ecological processes (Pandey and Ghosh, 2023). Understanding how these changes influence the activities and processes that occur within the soil matrix is vital because they underlie critical ecosystem services such as nutrient cycles, water purification and C sequestration.

The study of soil ecology along the rural-urban gradient holds both ecological and societal significance. Ecologically, soil serves as a dynamic interface where the interactions between living organisms and abiotic components shape the overall health

and resilience of ecosystems. Changes in land use and urban infrastructure can disrupt these interactions, leading to shifts in microbial communities, nutrient availability and soil structure. Additionally, the ability of soil to support plant growth and sustain biodiversity is a pivotal determinant of ecosystem stability (Muñoz-Rojas, 2018).

From a societal perspective, the provision of ecosystem services is of paramount importance. Urban areas heavily depend on rural landscapes for resources such as food, clean water and recreational spaces, making the health of soils in rural areas crucial to overall urban well-being. Conversely, the expansion of urban areas places pressure on nearby rural environments, potentially altering soil properties and functions in these areas (Imbrenda *et al.*, 2021). Therefore, a comprehensive understanding of the soil's ecological functions and processes across the rural-urban gradient is essential for informed land-use planning, sustainable development and effective management of natural resources. This study focuses on assessing soil ecological functions and processes along the rural-urban gradient of Shivamogga.

Considering the facts mentioned above, a study entitled “Assessment of Soil Ecological Functions and Processes along Rural-Urban gradient in Shivamogga district” was conducted at the Department of Soil Science, College of Agriculture, Shivamogga, Keladi Shivappa Nayaka University of Agricultural and Horticultural Sciences, Shivamogga, Karnataka, with the following objectives

1. To assess the magnitude of different land use systems and extent of land sealing along rural-urban gradient using satellite data;
2. To quantify biomass and nutrient turnovers in different land use systems along rural-urban gradient;
3. To assess the impacts of land sealing on soil ecological functions along rural-urban gradient; and
4. To identify key indices and develop management strategies to maintain soil ecological functions.

REVIEW OF LITERATURE

II REVIEW OF LITERATURE

The role of soil in supporting ecological functions and processes is vital. It contributes to nutrient cycling, water filtration and storage, carbon sequestration, biodiversity habitat and organic matter decomposition. Unfortunately, the rapid growth of urbanization has a profound impact on these soil functions and processes. It leads to soil compaction, alterations in hydrological characteristics, decreased availability of organic matter, soil contamination and changes in soil biodiversity.

The term soil sealing describes the condition of imperviousness of the soil. Among the various degradation processes resulting from urbanization, soil sealing significantly impacts soil ecological functions and processes. In this particular context, the current study was undertaken to examine the impact of urbanization on soil ecological functions and processes. This chapter endeavors to provide a pertinent literature review of the research conducted, organized under the subsequent headings.

2.1 Assessment of the magnitude of different land use systems and extent of land sealing along rural-urban gradient using satellite data

2.1.1 Assessment of the magnitude of different land use systems along the rural-urban gradient

Assessment of land use systems along the rural-urban gradient is crucial for understanding the spatial dynamics that shape landscapes and ecosystems (Warren *et al.*, 2018). This gradient represents a continuum of land use intensification, where land is typically more intensely developed and utilized in urban areas, while rural areas are characterized by lower population density and more agricultural or natural land uses. As cities expand and human populations grow, land use patterns shift, leading to significant environmental, economic and social impacts (Mahtta *et al.*, 2022). Examining these varying land uses along the rural-urban continuum helps identify the interactions between urbanization, agricultural practices, conservation efforts and sustainability challenges. This assessment is vital for urban planning, biodiversity conservation and resource management strategies.

Mapping of LULC and change detection using remote sensing and GIS techniques is a cost-effective method of obtaining a clear understanding of the land cover alteration processes due to land use change and their consequences (Twisa and Buchroithner, 2019).

Chandra (2011) emphasized the significance of vegetation in ecosystems and the importance of mapping it to understand ecological and geomorphological changes. Remote sensing technology, with its varying spatial, spectral and temporal resolutions, plays a crucial role in vegetation mapping. The study highlighted the utility of

vegetation indices, especially the NDVI, for mapping vegetation and other land cover types. It concludes that NDVI is more effective for assessing vegetation, offering better accuracy.

Ghorbani *et al.* (2012) investigated the effectiveness of NDVI for mapping land and canopy cover in Khalkhal County, Iran, using satellite imagery from Landsat TM, ETM⁺ and IRS P6LISSIII. Despite thorough preprocessing and accuracy assessment, the results show significant discrepancies between NDVI-derived data and field-collected information. Correlation analysis revealed no significant relationships between the two data sources, indicating that NDVI may not be a reliable tool for canopy or land cover mapping in heterogeneous landscapes like Khalkhal County.

Chaturvedi (2014) evaluated the land use and land cover in Allahabad district using remote sensing and GIS techniques, highlighting spatial variations caused by geomorphic processes like erosion and sedimentation. The research identified the dominant floodplain features in the northern region formed during the Quaternary period, while the southern part showed characteristics of the Vindhyan plateau from the Pre-Cambrian to Recent periods. These variations influenced the socio-cultural development. The study also mapped the land use patterns, noting cultivated land in floodplains, rocky wastelands in pediment zones and forest areas near denudation hills.

Olokeogun *et al.* (2014) assessed the landscape changes in Shasha Forest Reserve over an 18-year span using LANDSAT satellite imagery. The study classified the land into five categories: water body, forest reserve, built-up area, vegetation and farmland, utilizing a maximum likelihood algorithm for classification. Results indicated significant forest degradation, with a 46.12 per cent reduction in forest cover, while vegetation, farmland and built-up areas increased by 30.96 per cent, 22.82 per cent and 3.09 per cent, respectively.

Karakacan Kuzucu and Bektas Balcik (2017) explored the effectiveness of combining spectral vegetation indices (NDVI, RATIO, SAVI) with supervised classification methods, specifically Maximum Likelihood Classifier (MLC) and Support Vector Machine (SVM), to improve land use/land cover (LULC) classification using SPOT 7 multispectral imagery. Focusing on a heterogeneous landscape in Catalca, Turkey, the results showed that adding vegetation indices reduced the classification accuracy for both methods. However, MLC slightly outperformed SVM in overall accuracy and kappa coefficient.

Shishir and Tsuyuzaki (2018) focused on detecting fine-scale spatiotemporal land use changes using vegetation indices (VIs) in an urbanizing area, Purbachal New Town, Bangladesh. By comparing four popular VIs—NDVI, GRVI, EVI and EVI2—across two high-resolution satellite data sets (IKONOS and WorldView-2), the

research highlighted the effectiveness of a decision tree (DT) classifier in distinguishing land use types with high accuracy. The DT model outperformed individual VIs, particularly in classifying native ecosystems.

da Silva *et al.* (2020) investigated the vegetation cover types in Campo Belo do Sul, Brazil, using satellite-derived vegetation indices. Five indices, including NDVI, SAVI, LAI, EVI and NDWI, were calculated using Quantum GIS software. The resulting maps helped identify various vegetation cover classes. The findings revealed that no single index was optimal for representing all vegetation classes; however, NDVI, EVI and SAVI performed well in most thematic classes.

Madasa *et al.* (2021) employed remote sensing to assess land-use changes in the Welkom–Virginia Goldfields, focusing on a 30-year period from 1988 to 2018. By analyzing Landsat images with indices such as GEMI, NDBI, NDVI, NDSI and NDWI, it effectively differentiates between various land cover types. The use of supervised classification through the maximum likelihood method yielded accurate results, with classification accuracies between 88 per cent and 96 per cent.

Somayajula *et al.* (2020) highlighted how changes in land use/land cover (LULC) due to population growth are affecting the environment. It points out the importance of knowing land cover info for policies and agriculture and emphasizes the power of remote sensing for monitoring Earth's surface. NDVI, a go-to tool for checking vegetation health, helps analyze remote sensing images to spot land cover changes like vegetation, construction and water content.

Idrees *et al.* (2022) discussed the use of Landsat 8-derived NDVI to simplify land cover classification for urban areas. NDVI data from three distinct seasons in 2018 (dry, wet and harmattan) were analyzed to extract thresholds for various land cover types, such as water, built-up areas, soil, grassland and shrubland. The study achieved classification accuracies of 71.3 per cent, 46.4 per cent and 75.6 per cent for January, May and December, respectively, using an unbiased matrix coefficient.

2.1.2 Estimation of the extent of soil sealing along the rural-urban gradient

Soil sealing refers to the process of covering natural soil surfaces with impermeable materials, such as concrete, asphalt or buildings. This process is typically associated with urbanization and the expansion of human infrastructure.

Remote sensing has been widely used for ISA mapping and dynamics monitoring at multiple spatial resolutions with different satellite data. In the last decade, the increasing availability of data from aerial platforms made remote sensing together with Geographic Information Systems (GIS) the most widely adopted technology to map and assess soil sealing, at different geographic scales of analysis. In fact, satellites,

aeroplanes and fixed-wing UAV platforms, equipped with a wide range of sensors can increase spatial, temporal and radiometric resolution. The spread of big open data from satellites combined with the development of AI algorithms is making geodata mining and mapping techniques essential to quantify soil sealing at different spatial and temporal resolutions. Indeed, the use of GIS and geodata allows to analyze and model spatial data at a different scale, from a small portion of the land surface, up to the national or global scale (Zhang *et al.*, 2019).

Other technologies such as field survey, census data or ground-based data collection are generally costly and time-consuming. However, they have been frequently used in recent past years for monitoring soil sealing (Lu and Weng, 2006). An important example is represented by LUCAS Program (Land Use/Land Cover Area Frame Survey) which was managed by EU Member States to measure the nature of land cover and its use (Ballin *et al.*, 2018).

Two significant studies highlighted the effectiveness of integrating open satellite data with GIS modeling involving the utilization of imagery from the Landsat satellite constellations and the Sentinel missions. The former study employed Landsat image series to map urban expansion across 50 global cities (Bagan and Yamagata, 2014). Meanwhile, the latter study utilized the ‘Urban Atlas’ generated by the EU monitoring service, which leveraged Copernicus Sentinel Satellites to map soil sealing in the EU27 and 11 additional European countries (European Commission, 2020).

In fact, since the launch of the Sentinel missions in 2015, various satellites have become readily available, offering open spatial data for monitoring global land use and land cover changes, as well as for detecting emerging urbanization trends. A noteworthy example is the Italian National Institute for Environmental Protection and Research (ISPRA), which has been utilizing open Sentinel satellite data from the Copernicus Program since 2015 to monitor soil sealing throughout Italy.

The Index-Based Built-Up Index (IBI) was introduced as a novel approach to swiftly extract built-up land information from satellite imagery (Xu, 2008). Unlike conventional methods, it employs thematic index-derived bands-Soil Adjusted Vegetation Index (SAVI), Modified Normalized Difference Water Index (MNDWI) and Normalized Difference Built-Up Index (NDBI). These indices represent urban components-vegetation, water and built-up land. Validated using Fuzhou City’s Landsat ETM+ image, IBI effectively enhanced built-up features while minimizing background noise.

To address the limited methods for estimating impervious surfaces, a study was undertaken, that employed the Normalized Difference Impervious Surface Index (NDISI) to detect and quantify the impervious surfaces of Fuzhou City and Xiamen

City, China. NDISI effectively detected and quantified the extent of impervious surfaces. The index was also used to study the relationship between impervious surfaces and land surface temperature (LST), revealing a positive exponential connection (Xu, 2010).

A study was conducted to address soil sealing, a significant environmental concern impacting land productivity (Munafò *et al.*, 2013). Traditional data sources offered limited insight, prompting the development of a method to quantify soil sealing's progression over time at a national scale. Italy serves as the study area due to complex urbanization. Aerial photos and topographic maps from 1956, 1994, 1999 and 2006 were visually interpreted for this analysis. Outcomes revealed a continuous increase in soil sealing across Italy, with the most pronounced growth rates found in northern and southern regions. Per-capita sealing escalated notably from 1999 to 2006, reflecting worsening conditions.

Between 1989 and 2010, Landsat TM images were used to assess soil sealing in the Madrid Autonomous Region, Spain (Garcia *et al.*, 2014). The analysis revealed a substantial loss of 42,800 hectares of agricultural land during this period. Soil sealing was primarily concentrated in metropolitan areas and regions to the east, southwest and northwest of the capital city. By overlaying Landsat images with agrological and soil association maps, it was observed that the most significant losses occurred in agricultural classifications B and C, as well as in luvisols associated with cambisols, regosols, calcisols and fluvisols (WRBSR, 2007).

Smiraglia *et al.* (2014) developed a cost-effective method for detecting changes in soil sealing by combining radiometric analysis, multi-resolution segmentation and object-based classifiers in two regions of Italy: Campania and Veneto. They utilized multi-temporal satellite images and CORINE Land Cover maps (CLC), resulting in accurate soil sealing maps. The method enhanced the mapping unit and changed object size compared to CLC, proving its cost-effectiveness due to freely available data and high automation.

Bouzekri *et al.* (2015) introduced a novel spectral index for extracting built-up areas from medium-resolution satellite imagery, specifically Landsat OLI/TIRS data. This index, derived from Band ratios, outperforms four established indices; Band Ratio for Built-up Area (BRBA), the Normalized Built-up Area Index (NBAI), the New Built-up Index (NBI) and the Normalized Built-up Index (NDBI) in terms of overall accuracy and kappa coefficient. Applied to Djelfa city, South Algiers, the new index demonstrates superior performance in built-up area extraction. This advancement holds promise for more accurate urban analysis using satellite data.

Sertel and Akay (2015) employed an innovative approach for precise urban land cover mapping using SPOT-5 satellite images. By integrating diverse data sources such

as NDVI, NDWI, cadastral maps and Open-Street Map data, the study developed comprehensive rule sets for object-based classification. Applied to Gaziantep, Turkey, the method leveraged spectral, spatial and geometric attributes for automatic urban mapping. The incorporation of multi-source data significantly enhanced the accuracy. Evaluation indicated that the proposed rule sets achieve over 94 per cent overall accuracy in mapping 17 urban sub-classes, showcasing their effectiveness in high-resolution urban mapping.

Vanderhaegen *et al.* (2015) addressed the pressing environmental concerns posed by urban expansion and heightened surface sealing in Flanders, Belgium. They introduced a spatial model to accurately predict soil sealing density at a regional level, complementing the Flanders land-use change model. By factoring in elements like land-use composition, urban positioning and transport connectivity, the model outperformed simpler alternatives, enhancing estimation precision. Future projections using this model anticipate substantial growth in sealed surfaces within residential and mixed-use zones by 2030 and 2050, highlighting the urgency of implementing effective mitigation strategies for sustainable urban development.

Akay and Sertel (2016) utilized high-resolution SPOT 5 and SPOT 6 satellite imagery from 2010 and 2015, alongside ancillary data such as NDVI, NDWI, cadastral maps and more, to analyze urban land cover/use changes in Gaziantep city centre over a five-year period. Geometrically corrected to a 1:10,000 scale, the images underwent object-oriented classification, identifying twenty urban land cover/use classes from the European Urban Atlas project. Decision tree analysis incorporated spectral, spatial and geometric parameters, resulted in accurate urban maps.

Garcia and Perez (2016) examined methods for detecting soil sealing changes due to urban growth in Madrid, Spain, using remote sensing and GIS. This study employed multi-temporal data, including cartography, photogrammetry and satellite databases. The most effective detection method was found to be the combination of principal component analysis and the SAVI index. Supervised classifications revealed that, soil sealing increased from 15 to 24 per cent in the municipality of Madrid between 1984 and 2013.

Sinha *et al.* (2016) investigated a technique for extracting urban built-up areas in Adama Municipal Area, Ethiopia, from Landsat TM and ETM+ imagery spanning from 1984 to 2015. They utilized three indices (EBBI, SAVI and MNDWI) to distinguish between built-up, water and vegetation. By computing the difference between these indices, vegetation and water interference were minimized. The resulting index images were spectrally segmented to isolate built-up areas, revealing a significant 293 per cent expansion of urban areas from 1984 to 2015.

Sun *et al.* (2016) introduced a novel impervious surface extraction index called the Combinational Build-Up Index (CBI). CBI combines Principal Component Analysis (PC1), Normalized Difference Water Index (NDWI) and Soil-Adjusted Vegetation Index (SAVI) to create three thematic-oriented features representing high albedo, low albedo and vegetation. The index was tested with various remote-sensing images, evaluated qualitatively and quantitatively for accuracy and separability and compared to existing indices. The result of this study indicated that the proposed method was able to serve as an effective impervious index and can be applied widely.

Alikhanov *et al.* (2019) analyzed the transformation of the built-up area in Tashkent and its surrounding areas spanning from 1992 to 2018. This investigation utilized Landsat 5 TM and Landsat 8 OLI satellite images to assess urban changes through diverse indices. The focal point was on Tashkent city and the study period spanned 26 years. The findings of the research indicated a 1.6 per cent growth in urban areas during this period. However, this urban expansion was counter balanced by a decrease of 5.8 per cent in regions covered by greenery, agricultural activities and urban parks. This underlines the significant impact of urbanization on these essential land uses over the study's timeline.

Ukhnaa *et al.* (2019) developed faster and more accurate spectral signature analysis tools for urban area delineation and change detection. This study enhanced accuracy by merging multiple indices, including Enhanced Built-Up and Bareness Index (EBBI), Soil Adjusted Vegetation Index (SAVI), Modified Normalized Difference Water Index (MNDWI), New Water Index (NWI) and Normalized Different Pond Index (NDPI). The most accurate combination (98.21 %) included EBBI, SAVI, MNDWI and NDPI, while EBBI, SAVI and NWI also provided the good results (94.64 %) for urban built-up area extraction.

Pristeri *et al.* (2020) assessed the soil sealing in Padua's municipal territory using the Biotope Area Factor (BAF) index, utilizing digital land cover maps and aerial images. The results revealed that Padua has an average BAF index of 0.6, with 59.5 per cent of the area being permeable ($BAF = 1$) and 40.3 per cent sealed ($BAF = 0$). Most sealed areas are concentrated in the eastern sector and the historical core, with BAF values between 0 and 0.2. Notably, the new industrial area exhibited significant soil sealing, indicating a critical environmental concern in that region.

Zheng *et al.* (2021) evaluated the accurate urban built-up area extraction using Suomi National Polar-orbiting Partnership Visible Infrared Imaging Radiometer Suite (NPP-VIIRS) nighttime light (NTL) data. An Enhanced Nighttime Light Urban Index (ENUI) approach was introduced to mitigate blooming effects, incorporating NTL data with NDVI, NDWI and NDBI. Tested in the Guangdong-Hong Kong-Macao Greater Bay Area for 2012, 2015 and 2018, the ENUI approach demonstrated superior

extraction accuracy compared to Local-Optimized Thresholding (LOT), with 93.56 per cent overall accuracy and 0.77 MICE.

2.2 Quantification of litter biomass and nutrient turnovers in different land use systems along rural-urban gradient

Agricultural systems differ significantly from natural ecosystems. Intensive practices such as cultivation and the addition of nutrients via fertilizers and manure to boost productivity and quality disrupt natural processes (Scotti *et al.*, 2015). These frequent disturbances, along with increased nutrient inputs, lead to smaller detritus, flora and fauna pools, but larger nutrient pools, making the system more vulnerable to nutrient loss—an “open” nutrient cycle (Wolf *et al.*, 2022). The high fertility of these “young systems”—agricultural systems with open nutrient cycles—cannot be sustained without continuous nutrient inputs.

Many agricultural systems rely on monoculture practices, leaving little room for species diversity, as even weeds are eliminated to maximize crop yields. This creates a favourable environment for the dominance of certain organisms, such as insects and microbes, while significantly limiting biodiversity (Isbell *et al.*, 2017).

A significant amount of nutrients is utilized in food production, with fertilizer application accounting for almost 30 per cent of global nitrogen fixation and 10 per cent of phosphorus uptake by terrestrial organisms (Khan *et al.*, 2019). Consequently, current research is now prioritizing low-input, high-efficiency agricultural practices over simply maximizing yields.

In many agricultural systems, crop residues are often either removed or burned and they serve as a primary fuel source in much of the tropics and subtropics. However, recycling organic matter over a long period, instead of burning it, can enhance SOM-C and SOM-N levels (Wutzler *et al.*, 2017). Additionally, this practice leads to an increase in SMB-C and SMB-N. SOM turnover plays a crucial role in the cycles of nitrogen, phosphorus and sulfur, as well as cations in soils with limited mineral nutrient reserves. Residues tend to immobilize nutrients, making them less available until released by microbial biomass for future crops (Medina *et al.*, 2015).

Fertilizer sources are a key factor influencing nutrient cycling and the timing of application plays a significant role as well (Hendrix *et al.*, 2018). Crop rotation also impacts the amount and quality of organic matter, which affects soil organic matter (SOM) and nutrient turnover (Rutkowska and Piłkuła, 2013). The rate of complex compound breakdown is influenced by temperature and moisture levels (Follstad Shah *et al.*, 2017). In humid tropical regions, organic matter can decompose rapidly, though this process varies across landscapes (Cusack *et al.*, 2016).

Frequent agricultural activities such as land preparation, fertilization, irrigation and crop harvesting are essential for achieving desired yields and they significantly affect soil microbial biomass (SMB), organic matter turnover and nutrient cycling (Naresh *et al.*, 2013). Decomposition and nutrient cycling are governed by input rates, resource quality, environmental conditions and soil organisms (Hobbie, 2015). Decomposition releases substrates that support other organisms and contribute to food web formation.

Chomel *et al.* (2016) highlighted the importance of aligning nutrient release from decomposing plant matter with plant nutrient uptake. Improving nutrient and resource conservation through practices like green manuring, fertilization and incorporating crop residues can aid in this synchronization. Long-term productivity and environmental sustainability are tied to maintaining SOM, which is highly dependent on effective management practices (Bai *et al.*, 2018).

Horticultural systems involve both intensive and extensive management practices. Intensive systems, such as those for flowers, vegetables and some perennial crops use large amounts of fertilizers, farmyard manure (FYM), irrigation and pesticides to maximize crop yield and quality (Kayesh *et al.*, 2023). In less intensively managed systems, such as guava and mango orchards, natural litter recycling occurs instead of FYM application, but fertilizers are still used to meet nutrient demands, along with pesticides for pest control. Nutrient cycling openness varies based on management practices (Rastetter *et al.*, 2021).

Similar to agriculture, horticulture maintains large nutrient pools with smaller detritus, flora and fauna pools. Monoculture systems dominate, leading to limited species diversity and the prevalence of certain organisms. High matter and energy fluxes are also present, emphasizing the need for efficient nutrient management to maintain production without compromising quality, especially as energy and nutrient costs rise (Sutton *et al.*, 2013).

Compared to other terrestrial ecosystems, grasslands are structurally straightforward and seem relatively uniform. In India, most grasslands have originated from human activities (Box and Fujiwara, 2013). Human-made grasslands differ from natural ones due to practices like cultivation and the use of fertilizers and manure. As a result, the inorganic nutrient pool in human-altered pastoral grasslands is typically larger than in natural grasslands.

McGrath *et al.* (2001) reviewed over 100 studies on nutrient dynamics in Amazonian forests and land-use changes over the past 40 years. It examined the effects of converting forests to agriculture, including changes in soil pH, C, N, P and other nutrients. Key findings showed that forest-to-pasture conversion significantly alters soil

properties, with higher pH and Ca levels but reduced P availability over time. Soil C and N levels remain stable or increase in pastures compared to cropland, while secondary forests experience slow nutrient recovery. The review also highlighted the need for further research on nutrient cycles in diverse land-use systems.

Guo *et al.* (2006) examined litter fall and nutrient return in three Eucalyptus species (*E. botryoides*, *E. globulus* and *E. ovata*) over a 3-year period, comparing the effects of irrigation with meat works effluent versus no irrigation. The highest litter fall recorded was $13.4 \text{ t ha}^{-1} \text{ yr}^{-1}$, with nutrient returns of up to $159 \text{ kg N ha}^{-1} \text{ yr}^{-1}$. Effluent irrigation increased both litter fall and nutrient return, particularly in *E. globulus*. The results showed that up to 20 per cent of total biomass was returned as litter, contributing to 24 per cent of nitrogen uptake being recycled to the soil.

Quaye and Volk (2013) investigated the impact of organic and synthetic fertilizers on the biomass production of two willow varieties (*Salix miyabeana*-SX64 and *Salix purpurea*-9882-34) across three locations in the Northeastern U.S. The treatments included urea, biosolid compost, digested dairy manure and a control. Results showed no significant difference in biomass production across fertilization treatments, although SX64 produced more biomass than 9882-34. Organic amendments increased soil nutrient levels at some sites, but overall, the study suggested that fertilization may not be necessary for willow biomass production due to existing soil nutrient cycling.

Sileshi *et al.* (2020) highlighted the role of agroforestry in addressing land degradation by enhancing nutrient cycling and soil fertility. Agroforestry systems, which integrate trees with crops and livestock, show improved nutrient capture, nitrogen fixation and increased soil biological activity compared to annual cropping systems. Deep-rooted trees help recycle nutrients, improving efficiency and reducing environmental impacts. However, phosphorus deficiencies and water stress can challenge tree establishment on degraded lands.

2.3 Assessment of the impacts of land sealing on soil ecological functions along rural-urban gradient

The increasing evidence of improvement in ecosystem services can increase the quality of human life is gaining lot of attention in the recent past. There are three main ecosystem services that are linked in to soil *i.e.*, global climate regulation via carbon storage, biomass provisioning and regulation of health and environmental risks linked to anthropic stresses in urban context. Soils provide multiple functions that support the delivery of key ecosystem services. However, to achieve the above-mentioned ecosystem services, four key soil functions can be taken in consideration (Sere *et al.*, 2024). The four key soil functions that are linked to main ecosystem services are nutrient cycle, organic matter cycle, water cycle and biodiversity support

(Sere *et al.*, 2024). These functions can be seriously affected in degraded ecosystem created due to urbanization.

Land sealing, also known as soil sealing, refers to the covering of the soil surface with impermeable materials such as concrete, asphalt and buildings, which typically occurs during urbanization. This phenomenon is particularly prevalent along rural-urban gradients, where landscapes transition from natural or semi-natural environments to densely built-up areas. The impact of land sealing on soil ecological functions is profound, as it disrupts the natural processes that occur in soil, leading to a cascade of environmental consequences. This section provides a detailed review that assesses the impacts of land sealing on soil ecological functions along the rural-urban gradient, highlighting the changes in soil physical, chemical and biological properties.

2.3.1 Effect of soil sealing on soil physical properties

Soil sealing significantly alters the physical properties of soil. In natural or rural areas, soil is typically porous, allowing for the infiltration of water, exchange of gases and root penetration. However, when soil is sealed, these functions are inhibited (Burghardt *et al.*, 2015).

In the urban context, soil beneath sealed surfaces often becomes compacted due to the weight of buildings and infrastructure. Compaction reduces soil porosity, which in turn limits the movement of air and water through the soil profile. Sealed surfaces prevent water infiltration, leading to increased surface runoff. This not only reduces the soil's ability to recharge groundwater but also increases the risk of flooding in urban areas (Han *et al.*, 2017). Sealed soils tend to have altered thermal properties. The lack of vegetation and soil cover in urban areas leads to higher soil temperatures, which can affect the soil's microclimate and exacerbate the urban heat island effect (Fokaides *et al.*, 2016).

The variations in physical properties of urban and non-urban soils were outlined by Craul (1999). They proposed that urban soils would be distinguished by reduced soil structure, compaction, surface crusting, aeration and drainage limits and adjusted pH and temperature regimes.

Scharenbroch *et al.* (2005) reported that the bulk densities of newer urban soils (1.73 g cm^{-3}) were significantly greater than the older urban soils (1.41 g cm^{-3}). Doichinova *et al.* (2006) reported that the six forest locations were chosen from urban and semi-urban zones of Sofia, Bulgaria. Soil profiles were characterized for their particle size distribution. Significant differences were observed for the Ah horizon in the urban and semi-urban locations with respect to silt/clay ratio. Clay formation appeared to be reduced in this horizon in the urban soils.

Vania *et al.* (2006) reported that urban soils were characterized by higher content of the coarse silt (0.05–0.01 mm) and fine silt (0.005–0.001 mm) fractions; higher silt/clay ratio; higher C:N ratio and reduced intensity of mineralization processes of the organic matter.

Gregory *et al.* (2006) assessed soil compaction's effect on infiltration rates in urban North Central Florida, where sandy soils prevail. The study found that construction activities significantly reduced infiltration rates, leading to increased stormwater runoff. Non-compacted soils exhibited much higher infiltration rates, while compacted soils saw a reduction of 70 to 99 per cent. Maximum compaction occurred at depths of 20 to 30 cm. The findings highlighted the importance of addressing soil compaction in low-impact development strategies, as reduced infiltration necessitates larger stormwater management systems in urban areas.

Weng and Lu (2008) used a spectral unmixing model to analyze urban landscape changes in Indianapolis, Indiana and their impact on land surface temperatures (LST). They examined Landsat TM/ETM+ images from 1991, 1995 and 2000, using an unconstrained least-squares method to extract impervious surfaces, green vegetation and shade fractions. A hybrid classification approach combined maximum likelihood and decision-tree algorithms to classify these fractions into land use categories. The study found that urbanization significantly altered LST, with an inverse relationship between impervious surfaces and vegetation. Future research should enhance spectral mixture modelling by incorporating stratification and hyperspectral data.

Tang *et al.* (2011) studied the influence of urbanization on soil temperature in Nanjing, China, monitoring changes in urban and rural areas over a year. Soil temperatures were measured continuously at depths of 10 to 150 cm and spot-checked at 600 locations. Results showed urban soils were on average 1.21 °C warmer than rural soils, with differences varying by time scale: daily (0.29–3.03 °C), monthly (0.62–2.0 °C) and seasonal (0.84–1.51 °C). Temperature variations decreased with depth but increased with urbanization, emphasizing the significant influence of urban environments on soil thermal dynamics.

Yang and Zhang (2011) explored the water infiltration in urban soils with varying compaction levels, focusing on factors affecting infiltration rates. Using a dual-ring method, they examined ten land use patterns in Nanjing, China. They found that higher bulk density and reduced air-filled porosity decreased infiltration rates, while macropores increased infiltration and reduced surface runoff. Compacted soils, especially those highly compacted, had low infiltration rates and high runoff coefficients, leading to a greater risk of flooding and decreased surface water quality during flood events.

Aswathy and Sindhu (2013) reported that the rate of soil erosion due to urbanization is rising at a faster rate. Urbanization includes the rapid expansion of the associated urban area and the conversion of natural physical scenery into more productive land use. Because of urbanization, the soil's water holding capacity decreases and the rate of runoff increases which in turn results in soil erosion.

Scott and McGrane (2016) stated that the expansion of global urban areas has led to significant changes in natural processes, environmental quality and the use of natural resources. The urban landscape affects infiltration and evapotranspiration, complicating our ability to measure their dynamics across a heterogeneous landscape at contrasting scales. Impervious surfaces intensify runoff processes, whereas runoff from pervious areas remains uncertain owing to variable infiltration dynamics.

Shi *et al.* (2016) examined urbanization's impact on water retention in artificial disturbed landform units (DLUs) versus original landform units (OLUs). Using field studies, soil analysis and the analytic hierarchy process (AHP), they found DLUs had higher particle content, natural repose angles and soil bulk density, leading to lower soil infiltration rates. Converting natural landscapes like forests into DLUs, particularly construction roads, severely reduced soil infiltration and storage capacity.

Demina *et al.* (2018) conducted a study in which soil samples were collected from depths of 0–10 cm (topsoil) and 10–30 cm (subsoil) in 11 urbanized and 11 non-urbanized sites. These sites included areas that were once forests, pastures, or croplands. The study measured various physical properties (bulk density and texture). The average pH of topsoil was found to be 4.7 in forested areas, 6.8 in natural pastures and 6.3 in croplands. Urban soils were more compacted compared to non-urban soils, with forested soils experiencing a bulk density rise from 900 kg m^{-3} to $1,300 \text{ kg m}^{-3}$ after urbanization.

Li *et al.* (2018) assessed the effect of urbanization on direct runoff in Shenyang using a modified Soil Conservation Service Curve Number model with remote sensing data, focusing on Urban Functional Zones (UFZs). The study found a 68.02 mm increase in runoff depth due to urbanization, with high-density residential, business and industrial zones being major contributors. It revealed that 6.53 per cent of the area faces significant flooding risk, especially in industrial zones. Spatial analysis identified runoff hotspots in the city center and Boosted Regression Trees (BRT) highlighted rainfall, impervious surfaces and vegetation as key factors.

Wang *et al.* (2018) reported that steady infiltration rates of urban forest soil were highly variable. The variations in the final infiltration rates were observed for different vegetation patterns and compaction degrees. Trees with shrubs and grasses had the highest infiltration rate and the lowest infiltration rate for trees with bare land. The soil

infiltration rate is inversely proportional to the bulk density and directly proportional to the soil organic matter content and the non-capillary porosity. The steady infiltration rates of urban forest soils varied widely and ranged from 0.23 to 25.50 cm h⁻¹. Urban soil compaction contributed to low soil infiltration rates.

Herrmann *et al.* (2020) explained the influence of urban development on soil properties, focusing on changes in mineral-fraction particle size across various cities. They found that urbanization leads to a convergence of finer, more homogenous soil textures, which, while beneficial for urban infrastructure, diminishes the soil's ability to support essential ecosystem services. Their findings suggest that these altered soil profiles can serve as a predictive model for assessing urbanization's effects on soil properties and guiding urban management strategies to better protect soil resources and ecological sustainability.

Corrêa *et al.* (2021) investigated the impact of urbanization on water infiltration in São Paulo's Corujas Watershed. Human activities have increased flood risks by creating non-permeable surfaces and compacted soils. Their analysis showed that only 29.55 per cent of the area remains permeable, with soil compaction highest in grassy areas, moderate in riparian forests and lowest in the Corujas urban garden. The study highlighted urban gardening and green infrastructure as effective strategies to enhance soil permeability and mitigate flooding. Future research will assess whether these benefits are applicable to other urban gardens in different watersheds.

Mónok *et al.* (2021) examined the physical properties of soils from urban, green-urban and non-urban grasslands near Budapest, assessing the impact of human activities. There were significantly greater amounts of artefacts in urban soils than in the other two types of soil. Particle size distribution results showed sand > silt > clay in all types of sites. Sand content of soils was larger for non-urban sites, whereas silt and clay contents were smaller in these areas.

Enescu *et al.* (2022) conducted a study to assess the effect of rapid urbanization on soil physical properties of urban and peri-urban forests. The results revealed that the soil from the peri-urban plots was sandier and other ones were more clayey. The edaphic volume (V_e) was higher in the peri-urban forest plots (0.50 vs. 0.35 m³ m⁻²) than in the urban forest plots, which means the peri-urban soils were deeper and had a lower skeleton proportion. This study throws light on the changing soil properties of urban forests with regards to ecosystem services.

Wang *et al.* (2023) investigated the soil particle size distribution across an urban–suburban–rural–desert gradient in Urumqi, China, using laser diffraction. Sand dominated in urban, sub-urban and rural soils, while desert soil was mainly dominated by silt. Fractal dimension (D_v) values decreased from desert to rural soils, with rural

soils having the highest mean particle size (M_G). Sorting was poor across all gradients, with significant skewness in desert and urban soils. The study found that D_v is strongly influenced by particle size, especially clay content and suggests its use as an indicator of fine particles in soil structure.

2.3.2. Effect of soil sealing on soil chemical properties

Sealing of soil significantly disrupts biogeochemical cycles, particularly those of carbon, nitrogen and phosphorus, which are crucial for maintaining soil fertility and ecosystem productivity (O'Riordan *et al.*, 2021).

Sealed soils are often deprived of organic matter inputs due to the absence of vegetation. This can lead to a decline in soil organic carbon, which is a critical component of soil structure and fertility. Moreover, the decomposition of organic matter is reduced in sealed soils, impacting carbon sequestration potential. The nitrogen cycle is also disrupted by land sealing. The reduction in microbial activity, due to a lack of organic matter and oxygen, leads to decreased nitrogen fixation and nitrification processes (Pulikova and Gorovtsov, 2022). This results in lower soil nitrogen availability, which can affect plant growth and soil fertility in the surrounding areas.

Scharenbroch *et al.* (2005) revealed that older urban soils had slightly more extractable phosphorus, low Bray P (24 %), solid Bray P (51 %) and K (45 %) than newer urban soils. The microbial biomass N (71 %), potential C mineralization (20 %) and potential N mineralization (83 %) were high in old urban soils. Total soil organic matter was significantly greater (35 %) in old urban soils and fine Particulate organic matter (POM) was a larger contributor to total soil organic carbon in old compared to new urban soils.

Doichinova *et al.* (2006) reported that the six forest locations were chosen from urban (U) and semi-urban (SU) zones of Sofia, Bulgaria. Soil profiles were characterized for their generic physico-chemical properties such as pH, C content (%) and N (%). Their results revealed that C content was higher in semi-urban soils in comparison with the urban soils. With respect to C:N ratio, significant differences were observed for the Ah horizon in the urban and semi-urban locations. Based on the results obtained, a reduction in the buffering capability was noted for the topsoil of the urban sites compared to semi urban sites.

Pouyat *et al.* (2007) investigated how land use, cover and surface geology affect soil properties in Baltimore, MD, revealing significant variability, especially in chemical attributes. Key soil indicators like potassium, phosphorus and bulk density were distinct between forested and turfgrass areas. Soil pH varied notably in residential zones, reflecting different fertilizer practices. The study highlighted the ability of soil

properties to distinguish between remnant and disturbed urban soils. Interestingly, surface geology had a significant impact on certain properties, such as aluminium and heavy metals, showing that geological factors often outweighed urban influences in shaping soil characteristics.

Hagan *et al.* (2012) explored how urbanization affects soil properties in Tampa, FL, examining the roles of land use, land cover, population density and socioeconomic factors (*e.g.*, income, property values) on soil bulk density (BD) and chemical traits. The study revealed substantial variability in soil chemical properties, such as Mehlich-1 (M1) extractable phosphorus (P), calcium (Ca) and sodium (Na), across different land uses, with trends in pH and M1-copper (Cu) also observed. Time since urbanization impacted M1-P and Na, while socioeconomic factors influenced M1-potassium (K), magnesium (Mg), pH, P, Ca and Na, highlighting the need for tailored soil management in urban environments.

Raciti *et al.* (2012) highlighted the significant impact of impervious surfaces on soil carbon (C) and nitrogen (N) pools in urban areas. By comparing soils in open areas and those beneath impervious surfaces in New York City, the researchers found a substantial reduction in C and N content, with 66 per cent and 95 per cent lower levels, respectively, under impervious surfaces. The study also indicates that recalcitrant compounds dominate the organic matter in these soils. These findings suggest that urbanization could result in considerable losses of C and N.

Li *et al.* (2013) characterized urban soils from nine cities in Hubei Province, China, examining the impact of land use and topography on soil nutrients. They found that urban soils had higher pH but lower levels of SOC, available N, P and B compared to natural soils. Nutrient levels varied by land use, with roadside and residential areas showing higher concentrations of Ca, S, Cu, Mn and Zn, all within recommended ranges. Topography significantly influenced eight of the 11 chemical variables, with higher concentrations of N, Ca, Mg, S, Cu and Mn in plain cities, negatively correlated with elevation.

Mao *et al.* (2014) examined the spatial variation of soil pH and soil organic matter (SOM) in Xuzhou, China, using conventional statistics, geo-statistics and GIS with 172 soil samples. Soil pH ranged from 6.47 to 8.48 (average 7.62) and SOM from 3.51 g kg⁻¹ to 17.12 g kg⁻¹ (average 8.26 g kg⁻¹). Soil pH followed a normal distribution, while SOM was log-normally distributed, reflecting different spatial patterns. Kriging interpolation showed higher pH and SOM in southwestern mixed forest areas and lower levels in the east and north, likely due to industrial and human activities. This study highlighted the urbanization's impact on soil properties, aiding urban planning and environmental management.

Oktaba *et al.* (2014) investigated how human activities influence carbon and nitrogen levels in surface soils (0-20 cm) in Pruszków, Poland. They analyzed soil samples from 36 sites, including lawns, allotment gardens (high anthropogenic impact), fields and fallow lands (low impact). Lawns and gardens had higher organic carbon (20.5 and 21.7 g kg⁻¹) compared to fallow lands (10.4 g kg⁻¹) and fields (1.27 g kg⁻¹). Allotment gardens also showed higher total nitrogen (1.1 g kg⁻¹). Elevated organic carbon and high C:N ratios, especially in lawns (26.2), indicated anthropogenic carbon sources and minimal humus transformation. Urban areas also had increased hydrolytic acidity, cation exchange capacity and exchangeable base cations.

Wei *et al.* (2014) studied the influence of impervious surfaces on soil organic carbon (SOC) in Yixing City, China, finding SOC density under these surfaces to be about 68 per cent lower than in open areas. The research highlighted the disruption of natural carbon and nitrogen cycles in urban settings, with no correlation between SOC and total nitrogen beneath impervious surfaces. A 28-day incubation experiment showed lower carbon decomposition rates under these surfaces, indicating reduced SOC turnover and stability. The study underscored the need to consider urban carbon stocks in regional assessments and suggested that impervious surfaces may diminish urban soil's carbon sequestration potential.

Miljan *et al.* (2017) conducted research in the Carnok Nature Reserve (as a control) during July 2016, as well as in urban settlements such as Zmajev, Vrbas and Kula, 12 km from each other. Urban sites were lawns, chosen according to the owner's knowledge (no documented ploughing, no introduction of sandy or clay material over the last 70 years, no sowing of grass and only direct human activity was trimming of grass). The results showed significant reduction of humus content in urban ecosystems: Carnok (control, natural reserve) [U+0336] humus 5.33 per cent, organic C 3.488 per cent; Zmajev [U+0336] humus 2.51 per cent, organic C 1.963 per cent; Vrbas [U+0336] humus 3.81 per cent, organic C 4.216 per cent; Kula [U+0336] humus 1.95 per cent, organic C 1.51 per cent. The differences in organic carbon also showed the same trend with notable exception of Vrbas.

Asabere *et al.* (2018) demonstrated that soil organic matter (SOM) contents and stocks of long-term urban arable soils in Kumasi have been maintained and even increased, compared to short-term urban arable soils. At the same time, coarse fragments (including various kinds of artificial materials) that are related to rapid urbanization have considerably accumulated in the long-term urban arable soils.

Demina *et al.* (2018) conducted a study in which soil samples were collected from depths of 0–10 cm (topsoil) and 10–30 cm (subsoil) in 11 urbanized and 11 non-urbanized sites. These sites included areas that were once forests, pastures or croplands. The study measured various chemical properties (pH, total carbon and

nitrogen). The average pH of topsoil was found to be 4.7 in forested areas, 6.8 in natural pastures and 6.3 in croplands. Urbanization significantly impacted forest soils, where the pH increased from 4.7 to 7.6. Compared to non-urban soils, urban soils recorded significantly lower soil C and N contents.

Qin *et al.* (2019) investigated the impact of rapid urbanization on soil P fractions. Thirty sites were selected along the rural-urban gradient in Nanchang, China for the study. Findings revealed that urban soils had higher levels of total P and P fractions, especially in the 0-30 cm layer, compared to non-urban soils. This phosphorus enrichment, linked with increased soil pH and calcium/magnesium content, could lead to aquatic pollution due to the replenishment of labile P from other P pools.

Lu *et al.* (2020) studied the significant impact of impervious surfaces on C and N storage in urban soils, specifically in cold climates like the Boreal zone. By analyzing soils from 13 sites in Lahti, Finland, the research revealed that soils under sealed surfaces contain substantially lower amounts of C and N compared to those in warmer regions. The findings suggest that the deep construction layers and limited C and N accumulation under these surfaces contribute to this deficit. The study emphasized how soil sealing, combined with substantial topsoil removal in colder climates, leads to significant losses of C and N in urban areas.

Kuntoji *et al.* (2021) assessed the soil chemical properties along rural-urban interface of southern transect of Bengaluru. The findings of the study revealed that, rural soils recorded significantly higher values of SOC (0.70 %), N (323.01 kg ha⁻¹), P₂O₅ (28.98 kg ha⁻¹), K₂O (246.16 kg ha⁻¹), S (19.32 kg ha⁻¹), Zn (0.73 ppm) and B (0.84 ppm). Whereas the urban soils were characterized by higher values of soil pH (7.93), EC (1.33 dS m⁻¹), exch. Ca [12.03 cmol (p+) kg⁻¹] and Mg [7.70 c mol (p+) kg⁻¹] and heavy metals such as Cd (0.088 ppm), Cr (0.086 ppm), Pb (0.065 ppm) and Ni (0.056 ppm). This study concluded that, rural soils recorded higher nutrient levels, followed by peri-urban and lowest were recorded in urban soils.

Mónok *et al.* (2021) examined the chemical properties of soils from urban, green-urban and non-urban grasslands near Budapest, assessing the impact of human activities. Analysis of 72 topsoil samples revealed significant variations. Urban soils exhibited higher pH, carbonate and salt content but lower organic carbon compared to others, largely due to artefacts linked to anthropogenic influence. Heavy metal levels (e.g., Cu, Cr, Ni, Pb) exceeded natural background concentrations in urban and green-urban soils.

O'Riordan *et al.* (2021) studied soil sealing effects on soil properties, carbon storage and nutrient cycling in Manchester, UK. They found that sealing generally reduces soil carbon and nutrient stocks compared to unsealed greenspaces. However,

anthropogenic inputs, such as organic and mineral additions, can increase soil carbon and nitrate levels, sometimes surpassing those in greenspaces. This suggests that urban soils, especially in historically industrial areas, may hold significant, stable carbon reserves due to human activities. The study underscores the complexity of urban soils and the need for further research on human impacts on soil functions and ecosystem services.

Enescu *et al.* (2022) conducted a study to assess the effect of rapid urbanization on soil chemical properties of urban and peri-urban forests. The results revealed that soils from urban forests had an alkaline reaction and the ones from peri-urban forests had a moderately acid reaction. Urban forests recorded higher amount of exchangeable cations and base saturation than the peri-urban forests. This study is more relevant in light of the EU Soil Strategy, which sets out a framework and concrete measures to protect and restore soils.

Su *et al.* (2022) evaluated the soil nutrient status along the urban-rural gradient under the Chinese pine (*Pinus tabulaeformis* Carr.) in parks, neighborhoods and roadsides in Beijing, China. The findings of the study showed SOC, Ca, Mg and nitrate-N were higher in neighborhoods and parks than in roadsides. Interestingly, nitrate-N, Ca, Mg and available P and K all decreased in parks along the urban-rural gradient. Soil available P also declined along urban-rural gradients in neighborhoods. Thus, land use plays an important role in modifying urban-rural gradients of soil nutrients.

Cambou *et al.* (2023) studied soil organic carbon (SOC) stocks in parks and sealed areas in Marseille, Nancy and Nantes, France. They found SOC stocks in parks were 7 to 10 times higher than in sealed areas but still lower than natural soils. The top 30 cm of soil held about 40 per cent of the total SOC, with notable variability in sealed soils. Parks had 25-48 times more SOC in particulate organic matter and 4-6 times more in mineral-associated fractions. Sealed soils had a significant amount of SOC in the heavy fraction linked to sand and bitumen. The study underscored the need for more research on urban SOC.

2.3.3. Effect of soil sealing on soil biological properties

Soil biodiversity, including microorganisms, fungi, invertebrates and plant roots, plays a critical role in maintaining soil health and ecosystem functions (Havlicek and Mitchell, 2014). Land sealing drastically reduces soil biodiversity. The microbial communities in soil are highly sensitive to changes in soil structure, moisture and organic matter content. In sealed soils, microbial diversity and activity are significantly reduced, leading to a decline in essential soil processes such as decomposition, nutrient cycling and disease suppression (Sahu *et al.*, 2017).

Soil-dwelling invertebrates, such as earthworms and insects, are also negatively affected by land sealing. The loss of habitat and food sources, as well as changes in soil

temperature and moisture, result in decreased invertebrate populations, which further impacts soil structure and nutrient cycling (Louwagie *et al.*, 2017). The absence of vegetation in sealed areas leads to a loss of root systems that contribute to soil stability and organic matter inputs. This not only affects soil structure but also reduces the availability of niches for soil organisms.

Urbanization has led to notable changes in soil properties, necessitating a comprehensive assessment for effective soil management. A study in Shenzhen city examined the impact of urbanization on soil enzymes and physical-chemical properties under two land uses (Shi *et al.*, 2008). Findings revealed that roadside soils had higher invertase activity, while urban park soils showed greater catalase activity. Soil organic matter positively correlated with several enzyme activities, except catalase. Soil pH negatively influenced urease and acid phosphatase activities, counteracted by organic matter. Electrical conductivity was positively linked to catalase activity. Overall, soil organic matter, pH and EC were key factors affecting enzyme activities.

Zhao *et al.* (2012) assessed the effect of soil sealing on N transformation and microbial activity in urban areas. Conducted in Beijing, China, the research involved analyzing soil samples from different depths across three land cover types: impervious surfaces, forested areas and bare land. The study found that soil sealing significantly altered the microbial biomass and enzyme activities. The lowest levels of microbial biomass were observed in sealed soils. The study concluded that soil sealing negatively impacts N cycling and microbial functions in urban environments. The findings suggest that mitigating the effects of soil sealing could enhance urban soil functions and help remediate the adverse effects of urban nitrogen deposition.

Wang *et al.* (2017) studied microbial communities in turf grass-grown soils from urban and suburban areas around Xiamen City and compared them to microbial communities in the soils of rural farmlands. The potential N₂O emissions, potential denitrification activity and abundances of denitrifiers were higher in the rural farmland soils compared to turf grass soils. Ammonia oxidizing archaea (AOA) were more abundant than ammonia-oxidizing bacteria (AOB) in turf grass soils. The soil moisture, soil organic matter (SOM), total carbon (TC), total nitrogen (TN) and nitrate concentration were significantly higher in farmland soils compared to turf grass soils, while pH, C:N ratio, nitrite and ammonium concentrations were higher in the turf grass.

Demina *et al.* (2018) conducted a study in which soil samples were collected from depths of 0–10 cm (topsoil) and 10–30 cm (subsoil) in 11 urbanized and 11 non-urbanized sites. These sites included areas that were once forests, pastures or croplands. The study measured various soil characteristics, including microbial biomass carbon and respiration rates. Urbanization had negative effects on microbial

properties in topsoil from areas previously forested or used as pastures, while the conversion of cropland to urban soil resulted in increased microbial biomass carbon and respiration rates.

Pradeep *et al.* (2018) analyzed the effect of urbanization along a rural–urban gradient on soil microbial biomass and physico-chemical properties of the soil. The mean microbial biomass carbon (MBC) values were 107.4, 121.3 and 134.2 $\mu\text{g g}^{-1}$ of soil, respectively, for urban, sub-urban and rural sections of the gradient. Whereas, the mean microbial biomass nitrogen (MBN) was 10.2, 11.5 and 12.5 $\mu\text{g g}^{-1}$ of soil for urban, sub-urban and rural gradient. Similarly, the mean values of microbial biomass phosphorus (MBP) were 5.1, 5.8 and 6.3 $\mu\text{g g}^{-1}$ of soil, for urban, sub-urban and rural gradient, respectively.

Rai *et al.* (2018) evaluated how urbanization affects cyanobacterial diversity in soil along a rural-urban gradient. Using a culture-based approach, they identified 22 morphotypes, including 14 genera and 21 species, predominantly from *Nostocales*. The study found a decrease in diversity from rural to urban areas, with nitrogen-fixing heterocystous forms becoming more common in urban environments. Diversity indices showed higher cyanobacterial diversity in rural regions (Shannon-Weaver: 2.56; Simpson's: 2.32) compared to sub-urban and urban areas. Soil factors like pH, organic carbon, nitrogen and bulk density influenced community composition, highlighting urbanization's impact on soil microbial diversity.

Rai *et al.* (2018) examined how urbanization affects soil microbial biomass and related properties along a rural-urban gradient. Their study found that microbial biomass carbon (MBC), nitrogen (MBN) and phosphorus (MBP) decreased from rural to urban areas, with the highest levels in rural soils: 134.2 $\mu\text{g g}^{-1}$ for MBC in rural soils compared to 107.4 $\mu\text{g g}^{-1}$ in urban soils. Statistical analyses, including ANOVA and Tukey's HSD test, highlighted significant differences in microbial biomass due to changes in soil texture, bulk density and moisture content. The research suggested urbanization impairs soil health by reducing microbial biomass.

Braschler *et al.* (2020) examined the role of private domestic gardens in urban biodiversity, focusing on less mobile, ground-dwelling invertebrates like gastropods, spiders, millipedes, woodlice, ants, ground beetles and rove beetles. Conducted in 35 gardens across Basel, Switzerland, the study found these gardens significantly support regional biodiversity, with notable species richness. Woodlice were particularly abundant and the proximity to the city center influenced species richness and composition. This research highlighted the crucial role of domestic gardens in urban green infrastructure and underscores the need to incorporate them into urban planning to boost regional biodiversity.

Gorbov *et al.* (2021) explored the distribution of enzymatic activity in urban soils within the Chernozem zone, extending to a depth of 200 cm. By examining various soil types, including *Urbic Technosols* and *Technic Chernozems*, the research revealed that urban soils exhibited lower enzymatic activity compared to natural soils. Their study identified a negative correlation between oxidative and hydrolytic enzymes in natural soils, while urbanized soils show a positive correlation between oxidases and hydrolases. The findings suggested that prolonged sealing of soils impairs enzymatic activity. The study advocated for using enzymatic activity as a tool for urban soil bio-diagnostics and monitoring.

Kuntoji *et al.* (2021) studied the impact of the rapid increase in urbanization and industrialization on soil dehydrogenase activity (DHA) along the rural-urban interface of the southern transect of Bengaluru. The results indicated that rural soils recorded significantly higher DHA activity ($14.59 \mu\text{g TPF g}^{-1} \text{ soil } 24 \text{ h}^{-1}$) followed by peri-urban soils and the lowest DHA activity was recorded in urban soils, this clearly suggests that the process of urbanization has adversely affected the soil quality along the rural-urban interface.

Mónok *et al.* (2021) examined the microbiological properties of soils from urban, green-urban and non-urban grasslands near Budapest, assessing the impact of human activities. Analysis of 72 topsoil samples revealed significant variations. Microbial activity varied, with non-urban soils showing higher dehydrogenase activity and bacterial counts. Soil properties such as organic matter, nutrients and metal content influenced microbial parameters.

The detrimental effects of urbanization-induced soil sealing on soil properties and microbial communities was investigated by Pereira *et al.* (2021). The study revealed that the microbial community, including fungal and bacterial populations were suppressed in sealed soils, which is linked to the altered soil properties. The decline in fungal:bacterial and gram-positive:gram-negative bacterial ratios further suggests a degradation in soil carbon sequestration and storage potential. Overall, the findings emphasized that soil sealing due to urbanization led to significant negative consequences for soil microbial health and ecosystem services, particularly in relation to carbon cycling and storage.

Yu *et al.* (2021) examined how urbanization affects soil mesofauna in Guangzhou by comparing 47 sites across rural forests, urban forests, urban woodlands and urban parks. Their study revealed that increased urban soil pH, total nitrogen and heavy metals like Pb negatively impacted mesofauna abundance and diversity. While the impervious surface ratio didn't directly affect mesofauna, it was associated with altered soil properties. Urban forests, despite urban disturbances, supported higher

mesofauna abundance, aligning with the intermediate disturbance hypothesis. The research underscored the complex interplay between urban soil characteristics, landscape features and mesofauna communities.

Huang *et al.* (2022) examined urbanization's impact on soil fauna in riparian zones along an urban-rural gradient, focusing on macro- and mesofauna richness, abundance and diversity. Their findings revealed that natural riparian areas with permeable revetments host the most diverse and abundant soil fauna, due to minimal disturbance and favorable conditions. Hygrophilous mesofauna, which prefer humid environments, were more abundant during wetter seasons and were influenced by their proximity to water. The study suggested these mesofauna are good indicators of hydrological conditions and emphasizes the importance of preserving natural riverbanks and native vegetation in riparian management.

Yu *et al.* (2022) studied how urbanization affects soil physicochemical properties and soil fauna biodiversity in Guangzhou, examining four sites with varying urbanization levels. They found significant variation in soil properties across the gradient, but soil fauna biodiversity peaked in suburban areas rather than urban or rural zones. Although soil properties impacted biodiversity and community structure to some extent, they explained only a small portion of the variance in soil fauna metrics. The study highlighted that other urbanization factors beyond soil changes might more significantly influence soil fauna biodiversity, suggesting a need for further research into these additional factors.

Juceviča *et al.* (2023) evaluated how urbanization affects *Collembola* communities in Rīga, Latvia, by analyzing soil samples from various urban habitats using a Berlese-Tullgren extractor. Their Nonmetric Multidimensional Scaling (NMS) analysis, which explained 49 per cent of the variation on the first axis, revealed significant changes along an urban-rural gradient. Results showed a decline in species richness and an increase in Simpson's Dominance Index towards the city center. *Mesaphorura macrochaeta* dominated central urban areas, while *Parisotoma notabilis* was more common in suburban forests.

Rusterholz and Baur (2023) studied the impact of urbanization on soil fungal communities in 20 beech forests along a rural-urban gradient in Basel, Switzerland, using DNA metabarcoding of the rDNA ITS2 region. Their findings revealed that soil fungal richness was highest in less urbanized forests and lowest in rural forests. Urbanization was associated with increased fungal diversity and affected fungal evenness. Symbiotrophic fungi declined, while saprotrophic and pathotrophic fungi increased with urbanization. These results indicated that urbanization significantly alters soil fungal communities, which could have broader effects on forest ecosystems.

Sathish and Devakumar (2024) assessed soil mesofauna in Bengaluru during the rabi season of 2020–2021 across different cropping systems: pulses, vegetables, horticulture and agri-horticulture. They found 714 individuals from 16 taxa, with Collembolans being the most common. December 2020 had the peak count of 248 individuals, while February 2021 had the lowest at 104. Horticulture had the highest mesofauna population at 277, followed by agri-horticulture with 158. Urban areas had the highest count of 270 individuals. The study suggested that less disturbed soils, post-monsoon conditions and urban heat boost mesofauna populations in urban settings.

2.4 Identification of key indices and development of management strategies to maintain soil ecological functions

Land sealing along the rural-urban gradient has significant and far-reaching impacts on soil ecological functions (Massad *et al.*, 2019). The alteration of soil physical properties, disruption of biogeochemical cycles, loss of soil biodiversity and degradation of ecosystem services all contribute to environmental challenges that require urgent attention. By implementing sustainable urban planning practices and adopting green infrastructure solutions, it is possible to mitigate the negative impacts of land sealing and promote soil health in urban environments (Sturiale and Scuderi, 2019). This is essential for maintaining the ecological balance and ensuring the long-term sustainability of both rural and urban ecosystems.

To mitigate the impacts of land sealing on soil ecological functions, several strategies can be implemented, particularly in urban planning and development (Vieillard *et al.*, 2024). Incorporating green infrastructure, such as green roofs, permeable pavements and urban green spaces, can help reduce the extent of land sealing and promote soil health in urban areas. These practices allow for water infiltration, support vegetation and enhance biodiversity. In areas where land has already been sealed, soil restoration efforts, such as decompaction, re-vegetation and the addition of organic matter, can help restore some of the lost soil functions (Vieillard *et al.*, 2024).

Integrating soil conservation principles into urban planning can reduce the negative impacts of land sealing. This includes minimizing the footprint of built environments, protecting existing green spaces and promoting the use of permeable surfaces.

Soil quality indicators (SQI) include a range of soil physical, chemical and biological characteristics and are connected to key soil ecosystem functions and processes such as climate and water regulation, nutrient cycling, land structure and stability and microbial and plant biodiversity. SQI can help in guiding restoration, predominantly with respect to understanding the role of soil properties and plant-soil relationships that promote revegetation and enhance soil ecosystem functions and processes (Muñoz-Rojas, 2018).

A 4-year study evaluated how three types of pavements, varying in water and gas permeability, impacted soil properties and the growth and physiology of newly planted *Celtis australis* and *Fraxinus ornus* (Fini *et al.*, 2017). The treatments included: 1) impermeable pavement (asphalt), 2) permeable pavement (pavers), 3) porous pavement and 4) an unpaved control. Porous pavements allowed greater water infiltration and evaporation, reducing soil temperature compared to other pavement types, which caused significant soil warming. CO₂ accumulation was notable under impermeable and permeable pavements but not under porous ones. Tree growth was largely unaffected by pavement type, though *F. ornus* showed slight stress under impermeable pavement. The research suggested that porous pavements may mitigate urban heat islands, but further studies are required to understand their impact on mature trees.

Piero *et al.* (2017) introduced a Spatial Decision Support System (SDSS) based on a Geospatial Cyber-Infrastructure designed to mitigate soil sealing. Targeting spatial planners rather than agricultural experts, the web-based tool enables users to generate ‘what-if’ land planning scenarios through real-time modelling. The system integrates geospatial data, facilitating free access for planners. It has been tested in southern Italy at both municipal and detailed spatial levels, providing insights into rural fragmentation, loss of ecosystem services and soil sealing trends. Developed in collaboration with end-users, the tool promotes a new paradigm, fostering collaboration between soil science and urban planning for proactive solutions to soil sealing.

Nutrients exported from rural and peri-urban areas, like those in food, are often not returned to these environments to support primary production. To address these imbalances, (Zhu *et al.*, 2017) introduced the Peri-Urban Ecosystem (PURE) concept. PURE integrates principles from Critical Zone science and successional dynamics to function at the interface between rural and urban ecosystems, facilitating the re-coupling of resource flows. This framework aims to tackle key societal challenges and advance global sustainability objectives.

Mamehpour *et al.* (2021) assessed 24 variables related to fertility, salinity, sodicity and heavy metal content in surface soils (0-50 cm depth) across 14 profiles from urban (10) and non-urban (4 control) areas. Using the Integrated Quality Index (IQI) and Nemoro Quality Index (NQI) models, along with different data selection and scoring methods, the study found urban soils experienced both negative (e.g., increased salinity, heavy metals) and positive (e.g., higher organic matter, total N) changes. Overall, urban soils exhibited a 12.5–22 per cent reduction in SQI compared to controls. Fluvisols had the highest SQI, followed by Cambisols, Calcisols and Regosols. IQI-MDS with non-linear scoring proved the most effective in evaluating soil quality, showing strong correlations with maize yield.

Bonilla-Bedoya *et al.* (2022) highlighted the importance of incorporating urban land management as a key adaptation measure in cities of the Tropical Andes, focusing on Quito. Predictors included geographic, spectral and human activity variables. The study compared two spatial modeling approaches—simple multiple regression (SMRM) and random forest (RFM)—finding that SMRM performed better (OC: $r^2 = 20\%$, BD: $r^2 = 16\%$) than RFM (OC: $r^2 = 8\%$, BD: $r^2 = 5\%$). Results showed soil carbon stock variations ranging from 9.89 to 21.48 kg m⁻². The findings emphasized that urban land management is essential for implementing effective climate adaptation and mitigation strategies in the Tropical Andes.

Chen *et al.* (2022) investigated soil nutrient distribution and multifunctionality across different depths in Kyrgyzstan using field sampling data, factor analysis and structural equation modeling (SEM). Results showed that the topsoil stores significant quantities of SOC, nitrogen, phosphorus and potassium, with grasslands holding the largest share. A strong relationship exists between soil nutrients and environmental factors. Soil multifunctionality varies widely across regions and SEM analysis revealed that NPP, wind speed, MAP, longitude, SWC and pH have direct impacts, while temperature, latitude and other factors have indirect effects. These findings provide insights into Kyrgyzstan's soil ecosystem, helping to predict environmental change impacts and guide resource management.

Hyun *et al.* (2022) proposed an urban Soil Quality Index (uSQI) to assess soil conditions across various urban greenery types. Using a DPSC (drivers, pressures, state and changes) framework, six key soil functions were evaluated: physical stability, water infiltration, habitat provision, organic matter stabilization, nutrient cycling and pollutant breakdown. These functions were quantified using ten indicators, such as bulk density and pH. The uSQI, calculated as the average of the soil function scores through fuzzy logic, successfully identified sites with lower soil quality and highlighted areas for management improvements.

Kuntoji and Subbarayappa (2022) assessed the soil quality in rural and peri-urban areas along the southern transect of Bengaluru. Principal component analysis (PCA) identified organic carbon, pH, sulfur, phosphorus and clay as key indicators influencing soil quality. Soil quality index (SQI) scores were 0.61 for rural and 0.54 for peri-urban areas, placing both in the medium category (0.50-0.75) for soil health.

Wang *et al.* (2022) used the urban-rural gradient analysis method (UGAM) to examine the relationship between urbanization and ESs in the Central Yunnan City Cluster (CYCC). By simulating spatio-temporal changes along urban-rural gradients, the findings revealed that rapid urban expansion is the main driver of ESs degradation. The study highlighted the need for a sustainable, eco-friendly urbanization approach.

that incorporates ecological principles into top-level planning, ensuring balanced land use and development policies. Establishing ecological buffers around areas with significant land use shifts can mitigate negative impacts on regional ESs.

Sathish *et al.* (2024) examined how urbanization affects soil quality by analyzing soil samples from Bengaluru, India, across rural, peri-urban and urban zones during October to February 2020-2021. Using PCA to establish a minimum data set (MDS), the researchers identified critical soil properties, including microbial biomass carbon (MBC), SOC, nitrogen, manganese and urease, to calculate the SQI. Results revealed that rural areas had the highest SQI (0.57), followed by peri-urban (0.47–0.48) and urban zones (0.45–0.47). The findings highlighted the need for sustainable land management to counteract urbanization's adverse effects on soil health.

Séré *et al.* (2024) identified key soil indicators through field observations, measurements and lab analyses. Nine soil functions were selected based on their relevance to urban soil health. Suitable indicators for each function were identified, leading to the creation of reference frameworks with four classes, scored from 0 to 3. These frameworks were designed using scientific standards or expert input. The model was tested on a dataset of 109 urban soils across seven Western European cities under varying land uses. Results showed a Gaussian distribution in scores for eight soil functions, with the lowest scores in sealed or brownfield soils and the highest in park or agricultural soils. This practical approach could help optimize urban soil management, promoting healthier urban environments.

Zhang and Yuan (2024) integrated environmental remote sensing technology with urban planning. Focusing on Langfang, the research utilized remote sensing to monitor vegetation, water, soil and landscape data, identifying key issues and offering solutions to improve ecological urban design. The study highlighted the benefits of remote sensing for more accurate urban analysis compared to traditional methods, reducing errors in data collection. Findings showed significant improvements in environmental metrics post-planning, including reductions in total suspended solids in water bodies. The proposed strategies, such as ecological embankment restoration and green belt protection, provide a scientifically grounded approach to more sustainable urban planning.

MATERIAL AND METHODS

III MATERIAL AND METHODS

The present investigation on assessment of soil ecological functions and processes along the rural-urban gradient of Shivamogga district was carried out during 2021-2023. The details of the methodology adopted for the selection of the study area, assessment of different land uses, biomass and nutrient turnovers, soil ecological functions and statistical analysis are presented in this chapter.

3.1 Location

Shivamogga, one of the major tier-II cities of Karnataka, is situated on the banks of river Tunga in southern Karnataka (Fig. 1). The city is present on the eastern edge of the Western Ghats and is one of the gateways to this biodiversity hotspots (Ramachandra *et al.*, 2020). Being situated in the southern transition zone, the district exhibits diverse climate with rainfall ranging from 2,091.33 mm in south western parts to 714.75 mm in north eastern parts. The temperature is known to vary from 20 °C (in December) to 36 °C (in May).

Shivamogga is the 4th populated city of Karnataka with a population of 3.75 lakhs. Despite the absence of major industries, the city is a major business hub for agricultural commodities namely, paddy, maize, ginger, arecanut, cocoa, pepper and vegetables. The city's population has increased from 1.48 lakhs in 1980-81 to 3.75 lakhs in 2020-21. Consequently, the per capita land availability has decreased from 0.05 to 0.02 ha per person for the respective period. Similarly, area under Municipal Corporation has increased from 48.7 sq. km to 70.01 sq. km for the corresponding years. Rapid urbanization is visible in terms of expansion of urban infrastructure, community amenities and transport infrastructure to cater to the needs of the growing population. Considering this rapid growth, Shivamogga city was chosen under the Smart Cities Project-2016 by the Government of India under Ministry of Urban Development (Sharma *et al.*, 2024).

3.2 Identification and selection of the study area

The ecological parameters of a region vary spatially as influenced by climate and soil factors. Additionally, human activities play a significant role in shaping ecological dynamics. To understand these variations, an area with similar parent material and uniform rainfall was identified for the study. The geological setting and rainfall distribution pattern for Shivamogga district and the study area is depicted in Figures 2 and 3 respectively.

3.3 Delineation of urban, peri-urban and rural areas

The study area was comprehensively studied for its land use pattern and population density. The population density of the study area is presented in Fig. 4.

Urban zone was characterized by larger population density (> 1000 persons per sq. km), higher built-up areas and lower open lands. Peri-urban zone consisted of slightly lower population density (500-1000 persons per sq.km), built-up area and slightly higher agricultural activities compared to urban zone. The rural area was characterized with lower population density (0-100 persons per sq.km), smaller built-up areas, large unsealed open lands namely croplands, forests, different plantations, fallow lands, etc. The land use pattern in terms of total area under vegetation and barren area was extracted by using satellite imagery and google maps. The same area was overlaid with sealed areas *i.e.*, under roads, buildings, pavements, *etc.* Based on the proportion of open lands and sealed areas, the study area was categorized into urban, peri-urban and rural zones (Fig. 5).

3.4 Identification and assessment of different land uses and extent of soil sealing in the study area

Assessment of land use and soil sealing are critical for understanding the impact of urbanization and human activities on soil ecological functions. A reconnaissance survey was conducted to identify and select the prominent land uses across the study area. In order to achieve higher accuracy in assessing area under different land use systems and extent of soil sealing in urban, peri-urban and rural areas, five circular plots of 1 km radius were demarcated in each zone.

The land uses identified were grouped under three broad categories namely, non-sealed areas, compacted areas and sealed areas. The non-sealed areas comprised of open soil surface with no restriction for air and water exchange. These included forest/forest plantation, horticultural orchards, paddy, dryland crops and waste lands. The sealed areas included areas with impervious layer on the land surface that prevents air and water entry into the soil. It included, buildings, asphalt/concrete roads, concrete surfaces and tile/stone slab pavements. Compacted surfaces are the areas with significant restrictions for air-water exchange. This included playgrounds, unpaved pathways and un-metalled roads.

The methodology adopted for identifying and assessing the selected land uses using geospatial technique is described in the following sections.

3.4.1 Image processing and classification

The land uses in the study area were identified using the worldview-II satellite imagery (0.5 m resolution) obtained from the Karnataka State Remote Sensing Application Centre (KSRSAC), Government of Karnataka. Initially, False Colour Composite (FCC), Normalized Difference Vegetation Index (NDVI), Normalized Difference Water Index (NDWI) and Normalized Difference Built-up Index (NDBI)

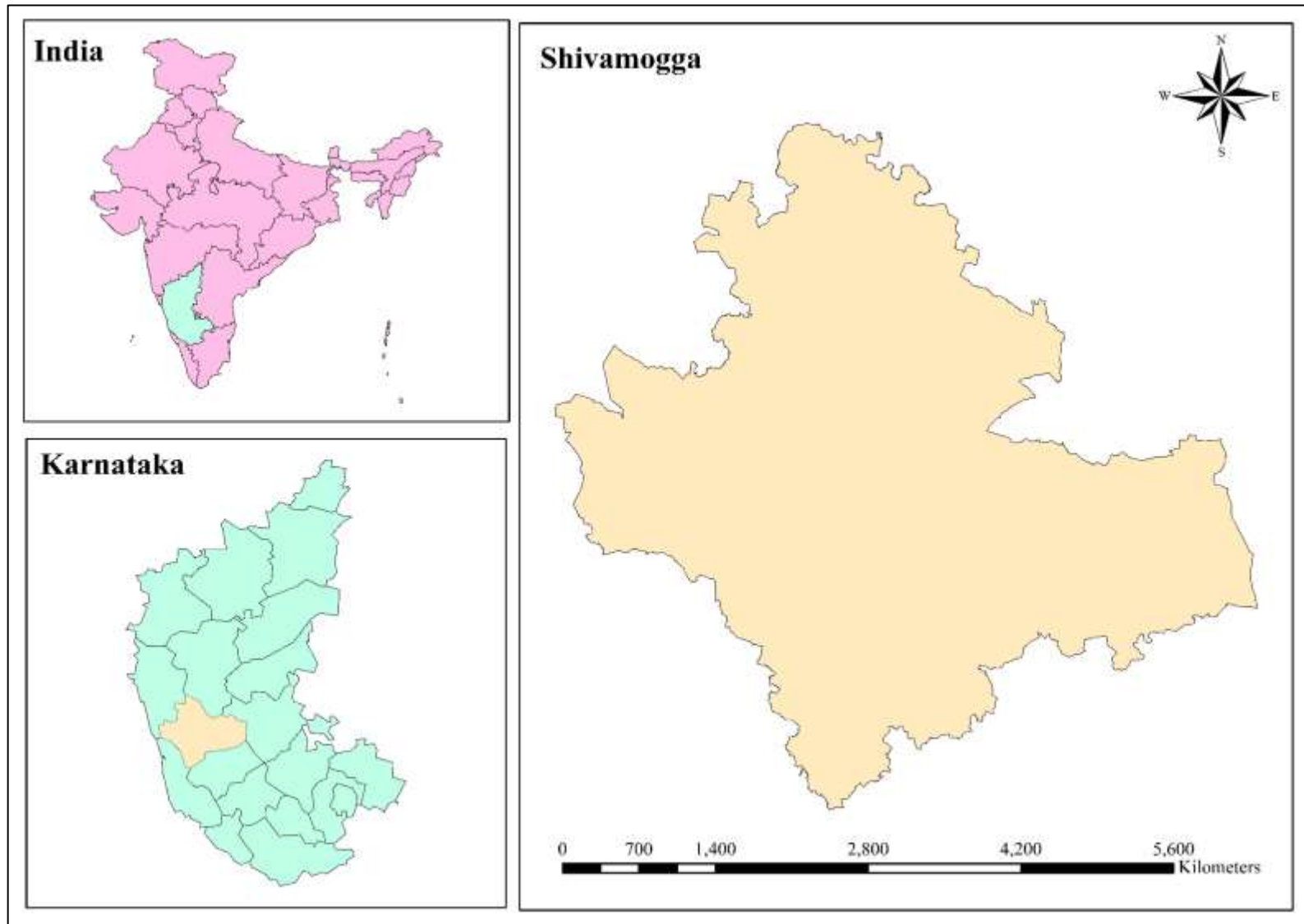


Fig. 1. Map showing the Shivamogga district in Karnataka, India

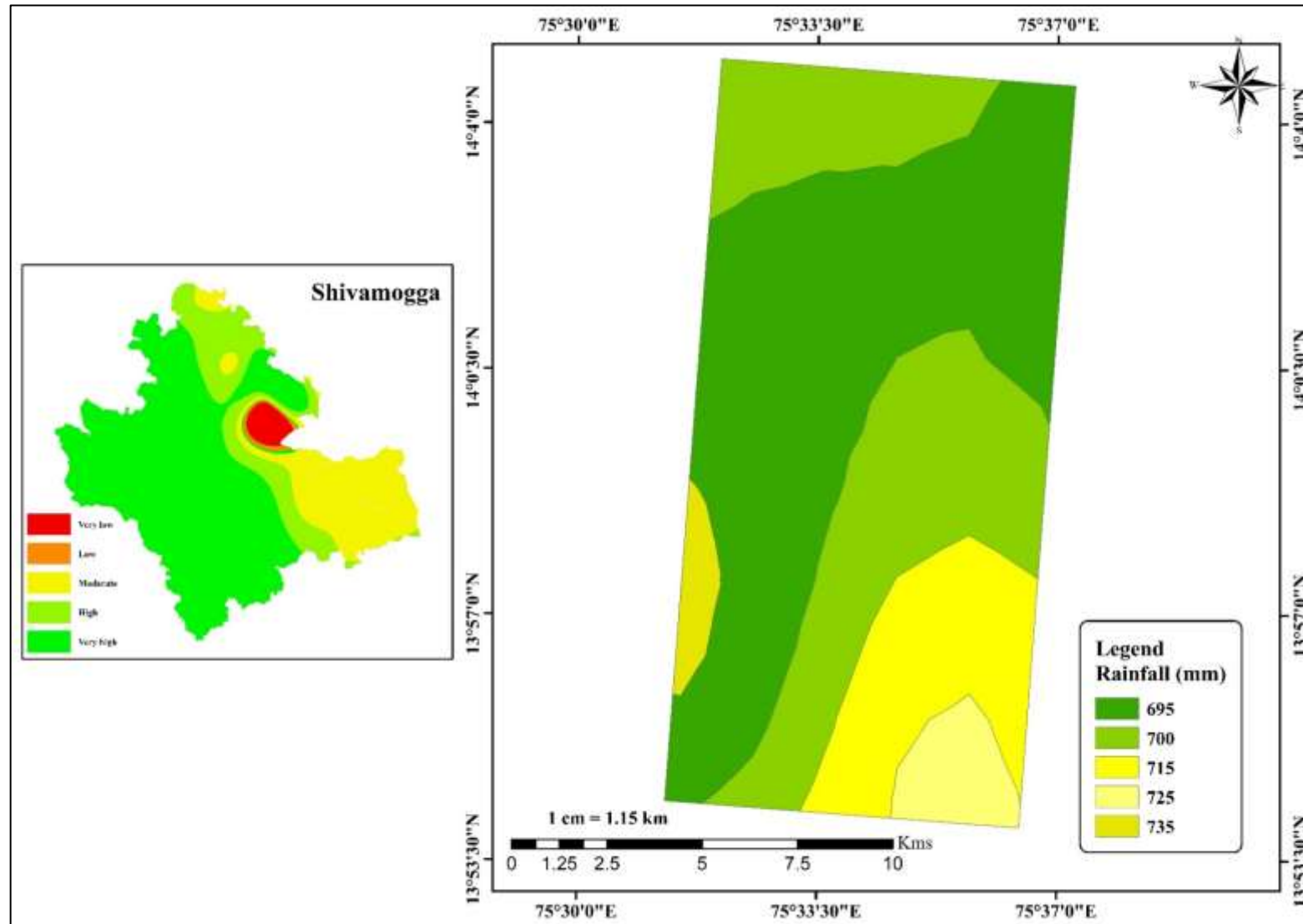


Fig. 2. Map showing the uniform rainfall pattern in the study area
(Source: Karnataka State Remote Sensing Application Centre, Bengaluru)

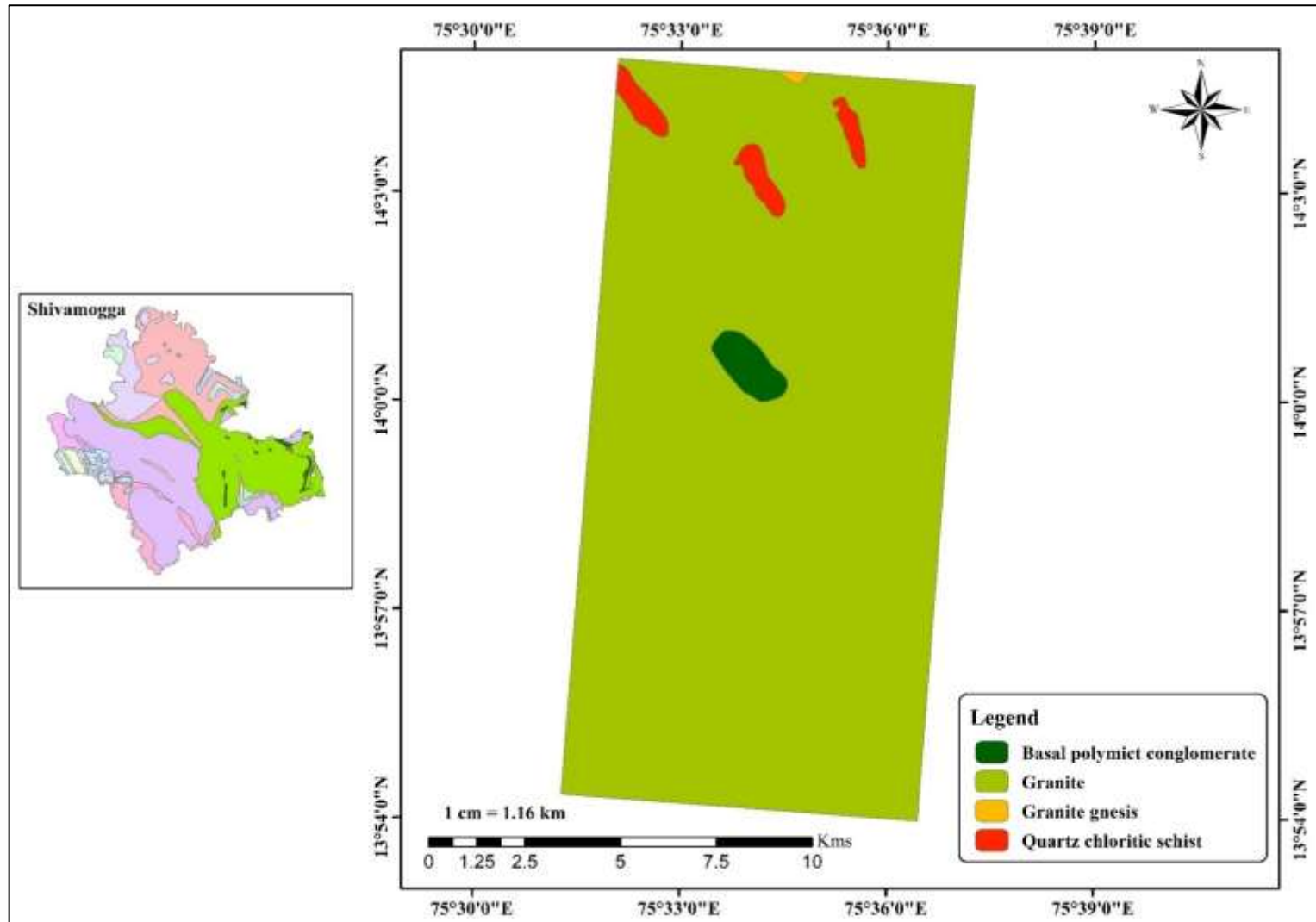


Fig. 3. Map showing the lithology of the study area
(Source: Karnataka State Remote Sensing Application Centre, Bengaluru)

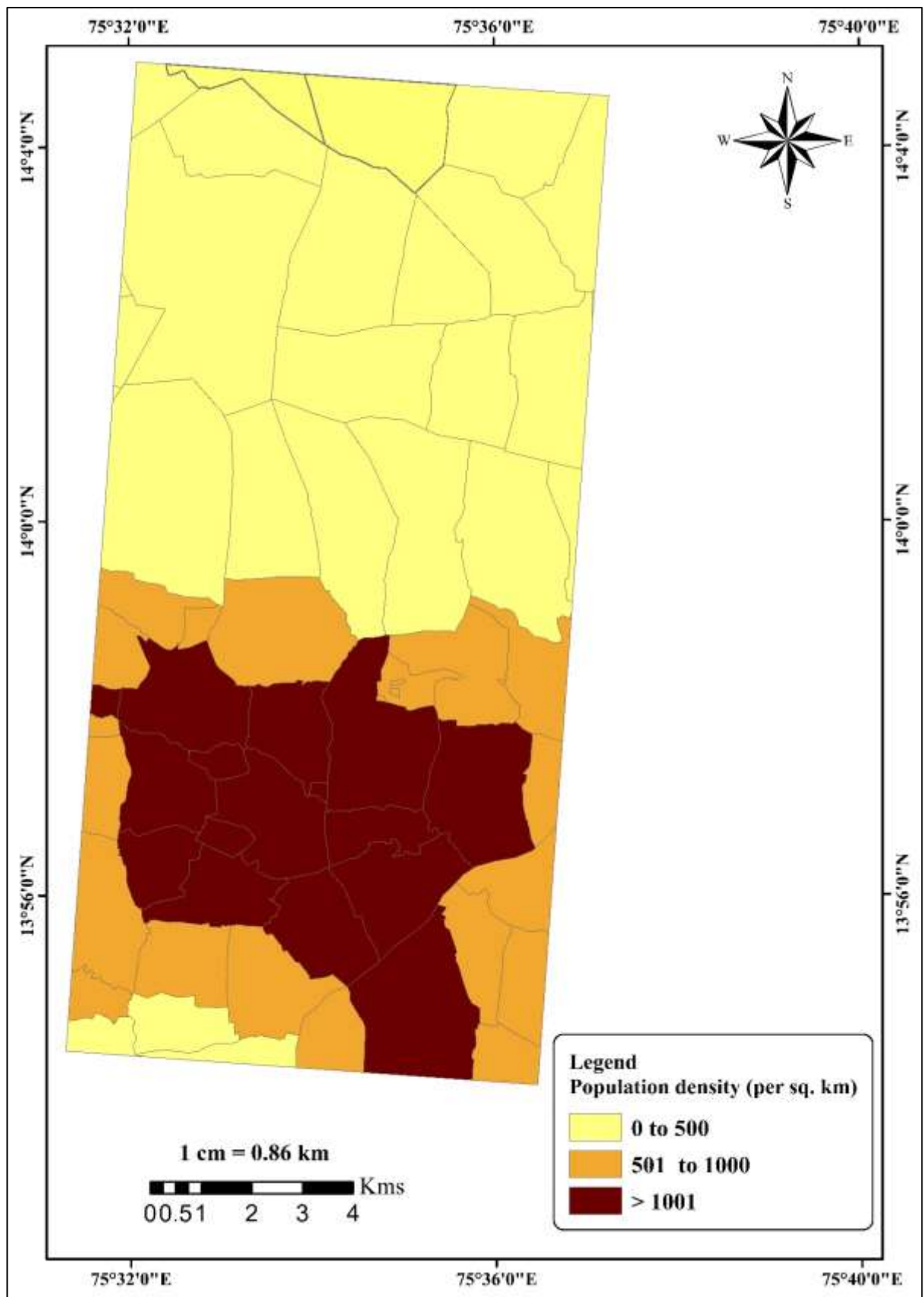


Fig. 4. Map showing the population density of the study area
(Source: 2011 Census data)

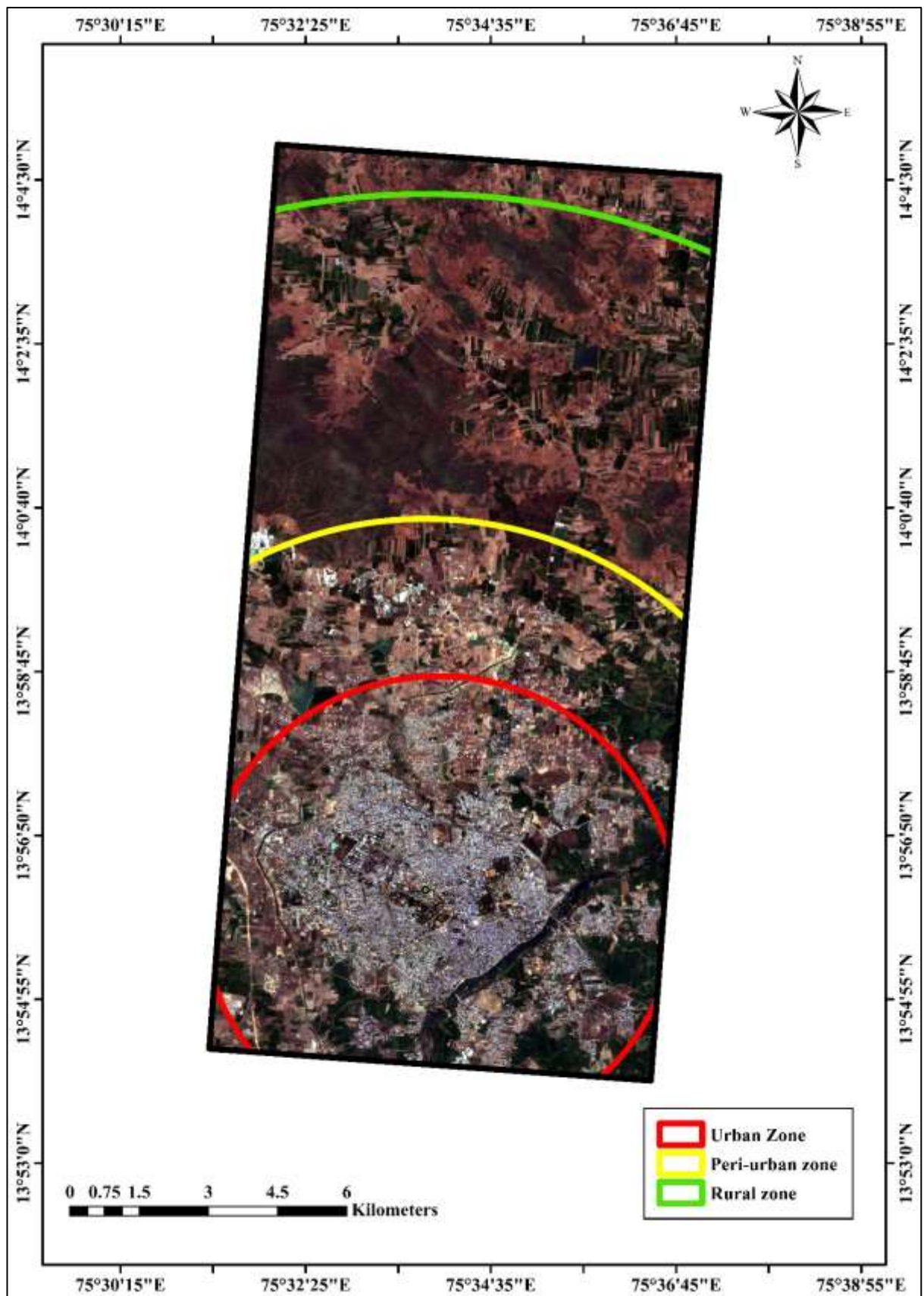


Fig. 5. Map showing the rural-urban gradient of the study area

were generated using ArcGIS 10.8 software by using the following equations (Zheng *et al.*, 2021).

$$NDVI = \frac{NIR - RED}{NIR + RED}$$

$$NDWI = \frac{GREEN - NIR}{GREEN + NIR}$$

$$NDBI = \frac{SWIR - NIR}{SWIR + NIR}$$

where, *NIR*, *RED*, *GREEN* and *SWIR* are the spectral reflectance in the near infrared band, red band, green band and short wavelength infrared band, respectively.

All these images were subjected to supervised classification using 150 training sites (Meti *et al.*, 2019). All the outputs were vectorized and stacked to obtain the final land use map of the study area. The flow methodology adopted is depicted in the Fig. 6.

3.4.2 Accuracy assessment

To evaluate the accuracy quantitatively, stratified random sampling method was applied and Google Earth data was used as reference for accuracy assessment. For this purpose, 50 pixel points were randomly collected for each class. Kappa co-efficient was used to evaluate the accuracy of the classification model used in this study. Kappa co-efficient was computed by using the below mentioned formula (Zheng *et al.*, 2021).

$$\text{Kappa Coefficient } (k) = \frac{p_o - p_e}{1 - p_e}$$

Where, p_o is the relative observed agreement (accuracy) and p_e is the hypothetical probability of chance agreement. Again, in order to compute p_o and p_e , the following equations were employed.

$$p_o = \frac{\text{Total number of correctly classified pixels}}{\text{total number of reference pixels}} \times 100$$

$$p_e = \frac{\sum_{i=1}^n (\text{Total Actual for Class } i \times \text{Total Predicted for Class } i)}{(\text{Total number of predictions})^2}$$

3.4.3 Quantification of different land uses and extent of soil sealing in urban, peri-urban and rural areas

The FCC, NDVI, NDWI and NDBI of the delineated circular blocks were subjected to maximum likelihood classification as described in the previous section. The resultant imagery obtained was reclassified as forest/forest plantations, water

bodies, horticultural orchards, arecanut, coconut, paddy, rainfed crops, waste lands, compacted areas and sealed areas. The extent of soil sealing and the magnitude of different land uses in each block was derived. The mean values were used to develop area under different land use systems and cumulative sealed areas in urban, peri-urban and rural areas separately.

3.5 Litter biomass turnover

In order to assess biomass turnovers, five locations for each land use systems were randomly selected along the rural-urban gradient and the methodology adopted are detailed.

The quantity of biomass added in a perennial system namely, forest/forest plantations and mango plantations were estimated by sweeping method (Nagaraja, 1997). An area of about 2 m x 2 m was demarcated and kept clean by sweeping the entire area. A smaller area of 1.5 m x 1.5 m was marked by pegs and the litter biomass was collected at three months interval. The biomass produced from annuals was also collected by harvesting (cutting) at the end of their peak growth (during December). In case of arecanut and coconut plantations, the number of fronds fallen in an area of 3 m x 3 m was counted and their weights were recorded. The quantity of organic manures added in plantations, in the form of FYM and composts was collected through interviews using structured questionnaires. In agricultural systems, the biomass left in the field after crop harvest was collected and weighed.

In case of compacted areas and sealed areas, the litter biomass added was estimated by sweeping method as described earlier. However, the biomass turnover was derived based on the proportion of canopy cover to the total area. The proportionality factor of 0.20 for urban and 0.50 for peri-urban and rural were used to estimate the biomass turnover (Guo *et al.*, 2006). The total biomass added and available for recycling was expressed in t ha^{-1} on oven dry weight basis.

3.6 Nutrient turnover

The biomass collected from each land use system was oven dried, powdered and stored in air tight containers for further analysis. The major nutrients added through FYM/composts were quantified based on their dosage and nutrient composition of 0.50:0.25:0.50 of N:P:K content. The methodology used to estimate the nutrient content in biomass and their turnover is detailed in the following section.

3.6.1 Nitrogen content

The processed samples were digested by adding 10 ml conc. H_2SO_4 along with digestion mixture in a solid block digester. Total nitrogen content in digested samples

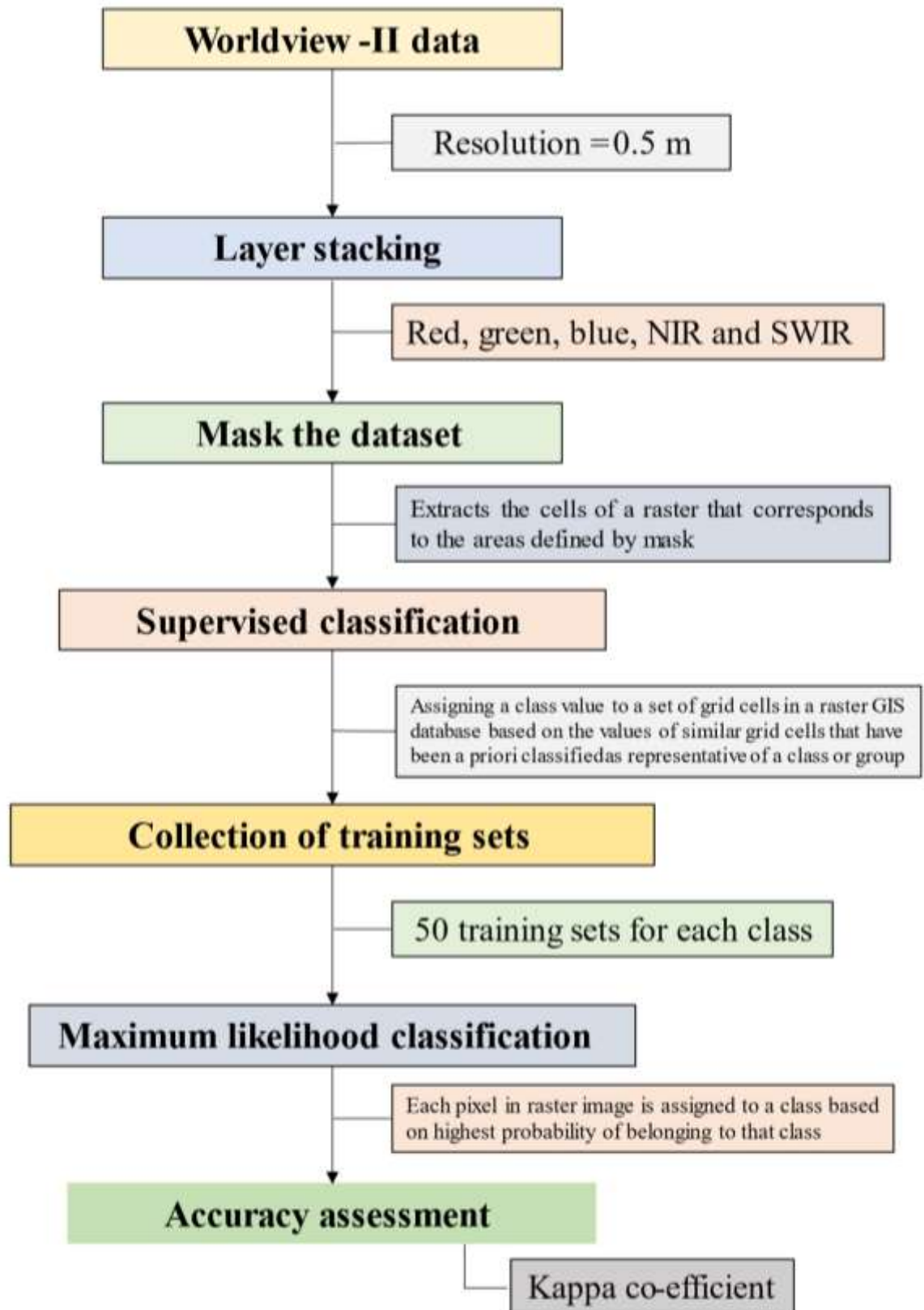


Fig. 6. Workflow of image acquisition and pre-processing

was distilled and determined by Kjeldhal's method using Kjeldhal distillation unit (Kelplus Classic DX, Pelican) as described by Piper (1966).

3.6.3 Phosphorus content

Biomass samples were subjected for diacid digestion on a hot plate. The powdered sample of 0.2 gram was pre-digested with 8 ml concentrated HNO₃ and 2 ml H₂O₂, then digested on hot plate or sand bath. The volume of the digest was made up to 100 ml with distilled water.

A known volume of the diacid digested sample (5 ml) added with vanadomolybdate reagent. The intensity of yellow colour developed by vanadomolybdate-phosphoric complex was read at 420 nm using Systronics spectrophotometer (Baruah and Barthakur, 1997).

3.6.4 Potassium content

The diacid digested sample was fed to Systronics flame photometer. The intensity of red colour developed in the flame was measured using K-filter and the concentration of K was determined using the standard curve (Piper, 1966).

3.6.5 Assessment of nutrient turnover

The amount of nutrients (N, P and K) added/recycled per hectare (kg ha⁻¹) through biomass were quantified separately for all three zones by considering the biomass added (kg ha⁻¹) and nutrient content (%) using the below given formula.

Nutrient recycled (kg ha⁻¹) = (Biomass added) X (Nutrient Content in Biomass)

3.7 Soil characterization

Soil ecological functions and processes are vital for the various ecosystem services with respect to climate regulation and other provisioning services. These soil ecological functions and processes can be evaluated by assessing the soil physical, chemical and biological properties. The selected soil parameters were determined by adopting standard procedures.

3.7.1 Soil sampling and preparation

Soil sampling sites were randomly selected for different land uses all along the rural-urban gradient (Fig. 7). Surface soil (0-15 cm) samples were collected from forest/forest plantations, mango, arecanut, coconut, paddy, rainfed crop, waste lands, compacted area and sealed area. With respect to sealed areas, soil sampling was opportunistic. Road and construction sites were selected across the rural-urban gradient, where road or pavement surface were temporarily removed and soil pits were being

exposed. Sampling was done from such selected sites across the rural-urban gradient. Five soil samples were collected from each land use in each zone and in total around 200 samples were collected across the study area. Soil samples were dried under shade and gently grounded and sieved (2 mm). The sieved samples were stored in polypropylene bottles and analyzed for various physical, chemical and biological properties of soil by adopting standard procedures.

3.7.2 Bulk density

Bulk density was determined by core sampler method and expressed in Mg m^{-3} as given by Piper (1966). Bulk density was calculated as dry weight of soil per unit volume of the core collected by the core sampler in the field.

3.7.3 Particle density

Particle density was determined by Pycnometer and expressed in Mg m^{-3} as given by Piper (1966). The weight of pycnometer added with soil was measured (m_p+m_s) followed by adding water and measuring the weight (m_{H_2O}). The volume of solid V_s was the difference between the volume of water that fills empty pycnometer (V) and water added along with the presence of soil (V_{H_2O}). The density of soil was then calculated using the formula,

$$d_s = \frac{m_s}{V_s}$$

3.7.4 Porosity

Porosity of soil was computed by substituting the values of particle density (PD) and bulk density (BD) densities in the following equation (Black, 1965).

$$\text{Porosity (\%)} = \left(1 - \frac{BD}{PD}\right) \times 100$$

3.7.5 Soil pH and Electrical conductivity (1:2.5)

The soil pH was determined for the soil-water suspension in the ratio of 1:2.5 by potentiometric method. The soil pH was measured using digital pH meter having combined electrode and electrical conductivity was determined by Conductivity Bridge. The EC values were expressed in dS m^{-1} at 25°C as described by Jackson, 1973.

3.7.6 Soil organic carbon

The soil organic carbon was determined by wet oxidation method (Walkley and Black, 1934). A known weight of 0.2 mm sieved soil sample was treated with known excess volume of standard potassium dichromate solution in a 500 ml conical flasks and 20 ml of concentrated sulphuric acid was added. The soil suspension was swirled immediately and kept undisturbed for 30 minutes. The unused potassium dichromate

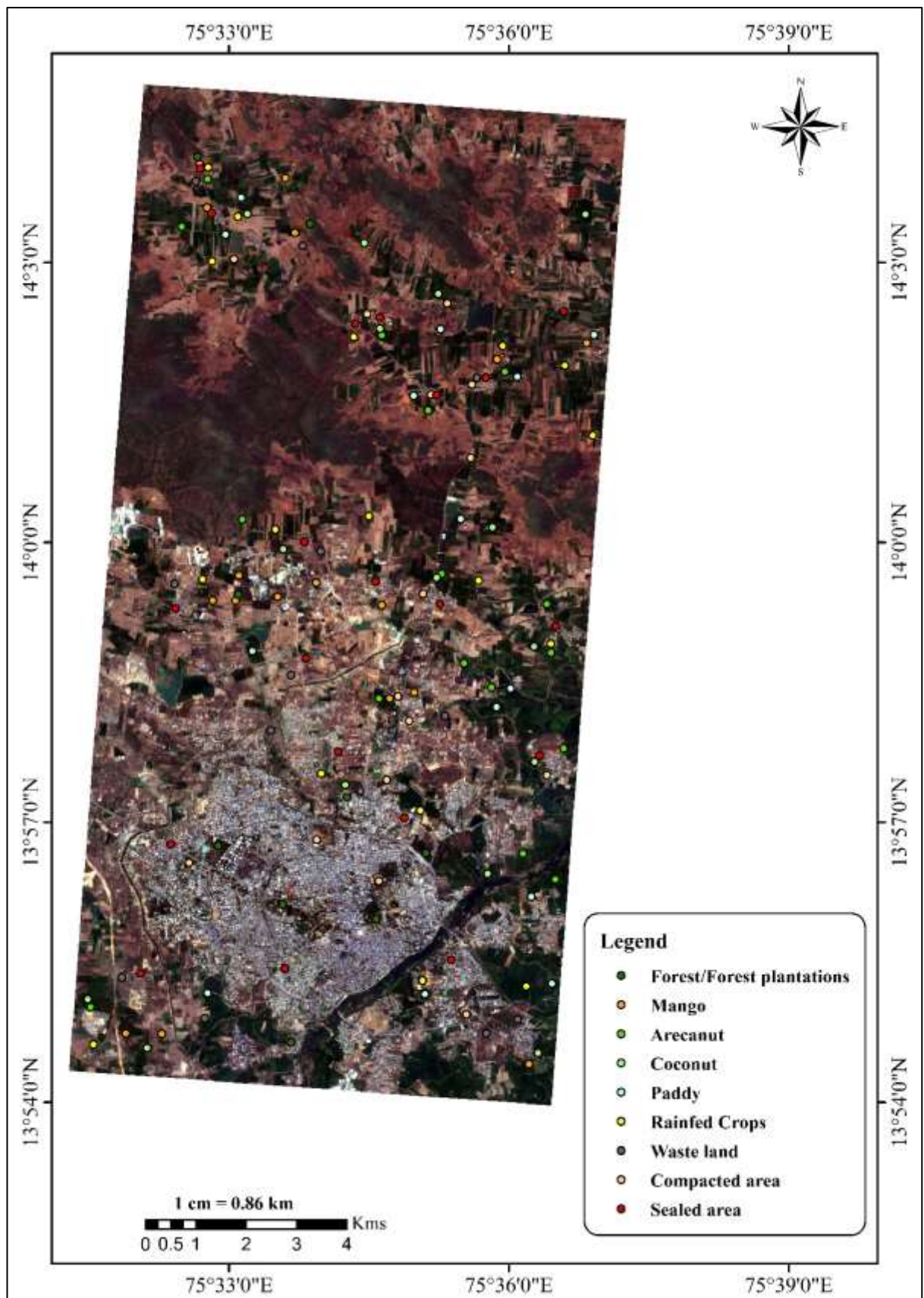


Fig. 7. Map showing the sampling sites of different land uses in the study area

was determined by back titration with standard ferrous ammonium sulphate solution in the presence of ferroin indicator. The strength of ferrous ammonium sulphate solution was determined separately during each batch titration. Soil organic carbon contents were expressed in g kg^{-1} .

3.7.7 Available nitrogen

Available N content in soil was determined by alkaline permanganate method (Subbiah and Asija, 1956). The ammonia evolved during mild oxidation was trapped in boric acid and it was determined by titrating with standard acid. The available-N content was calculated and expressed in kg ha^{-1} .

3.7.8 Available phosphorus

The collected soil samples were acidic in nature and hence for acidic soil, Bray's No. I reagent was used for determination of available phosphorous (Jackson, 1973). The phosphorous in the extract was quantified by using phosphomolybdate complex reduction method. The intensity of blue colour was read at 660 nm. The available-P contents were expressed in kg ha^{-1} .

3.7.9 Available potassium

Available potassium content in soil was extracted with neutral normal ammonium acetate (pH 7.0) at soil: extractant ratio of 1:5. The potassium content in the extract was determined by flame photometer (Page *et al.*, 1982). The available K_2O content was expressed in kg ha^{-1} .

3.7.10 Exchangeable calcium and magnesium

The extract obtained for available K was used for the determination of exchangeable calcium and magnesium by Versenate titration method. The exchangeable calcium and magnesium together were determined by titrating against EDTA (Ethylene diamine tetra acetic acid) at pH 8.5 in presence of EBT (Eriochrome Black T) indicator. The calcium alone was determined by adding 2.5 per cent NaOH to get a pH of 12 in the presence of murexide indicator (Page *et al.*, 1982). Both exchangeable calcium and magnesium values were expressed in $\text{cmol (p}^+) \text{ kg}^{-1}$.

3.7.11 Available sulphur

The available sulphur was extracted with 0.15 per cent CaCl_2 solution. The sulphate in the extract was determined by developing turbidity with addition of BaCl_2 in presence of stabilizing agent, gum acacia (Black, 1965). The turbidity was measured using spectrophotometer at 420 nm. The available-S content was expressed in mg kg^{-1} .

3.7.12 DTPA extractable micronutrients

The metal ions namely iron, zinc, manganese and copper were extracted using DTPA buffer solution (pH 7.3) in 1:2 soil: extractant ratio. The concentration in the extract was determined by atomic absorption spectrometry using appropriate hollow cathode lamps (Lindsay and Norvell, 1978) and their concentrations were expressed in mg kg⁻¹.

3.7.13 Soil Carbon fractions

Analyzing soil carbon fractions is essential for understanding the role of soil in carbon cycling and its potential for mitigating climate change. Soil carbon majorly exists in three fractions, such as total organic carbon (TOC), total inorganic carbon (TIC) and total carbon (TC).

3.7.13.1 Total inorganic carbon (TIC)

The rapid titration method of Piper (1944), also called the acid neutralization method, was used to estimate soil inorganic carbon. The sample was treated with dilute acid, and the unreacted acid was titrated with alkali. Ten grams of 2 mm sieved soil was taken in the Erlenmeyer flask including blank and 25 ml of 0.5 N HCl was added. Later, it was transferred to water bath and heated for 5 min, cooled, and filtered it with two washings. 5 drops of phenolphthalein indicator were added and titrated it with 0.25 N NaOH.

$$TIC \text{ or } CaCO_3 \text{ \% equivalent C} = \frac{(a - b) \times 0.050 \times 100}{S}$$

Where,

a = ml of NaOH used for blank,

b = ml of NaOH used for sample and

S = air dry sample weight in grams (Piper, 1944).

3.7.13.2 Total organic carbon (TOC)

0.25 g of processed soil sample was weighed out into a 25 ml volumetric flask, 20 ml of 0.4 N chromic acid solution was added to the soil samples and similar quantity was taken for the blank. The mixture was heated in H₃PO₄ bath and heated on hot plate at such a rate that a temperature of 155 °C is reached in 20 to 25 minutes. The temperature was held at 155 °C to 160 °C for an additional 5 minutes. The chromic acid solution, cooled to room temperature, was diluted with distilled water to 200 ml, and then 1g NaF and 2-4 drops diphenyl ammine indicator was added. The solution was back titrated with the 0.2 N ferrous ammonium sulphate until the solutions colour turns from violet to light green (Jackson, 1973).

$$\text{TOC (\%)} = \frac{10.67 \times (B - S)}{B}$$

Where,

B= Blank Titration (ml), S= Sample titration (ml)

3.7.13.3 Total carbon (TC)

Total carbon (TC) was obtained by adding both TOC and TIC and is given as below.

$$\text{TC} = \text{TOC} + \text{TIC}$$

3.7.14 Soil Nitrogen fractions

The fractionation of soil nitrogen was carried out by an adopted version of the procedure given by Page *et al.* (1982). About ten g of soil in a polyethylene bottle was mixed with 50 mL of 2 M KCl solution and shaken for one hour and then filtered.

3.7.14.1 Ammonical nitrogen ($\text{NH}_4^+\text{-N}$)

An aliquot of 20 ml of the above filtrate was steam distilled with freshly ignited MgO in Bremner's distillation apparatus and distillate collected in 4 per cent boric acid containing mixed indicator and was titrated with standard H_2SO_4 .

3.7.14.2 Nitrate nitrogen ($\text{NO}_3^-\text{-N}$)

After distillation of the ammonical nitrogen, immediately the sample was added with a pinch of Devarda's alloy and the distillation was continued. The amount of nitrate nitrogen was determined as described for exchangeable $\text{NH}_4^+\text{-N}$.

$$\text{NH}_4^+ - \text{N or NO}_3^- - \text{N (mg kg}^{-1}\text{)} = \frac{(V_s - V_b) \times N \times 0.014 \times 10^6}{w}$$

Where,

V_s = Volume of standard acid used for sample titration (ml)

V_b = Volume of standard acid used for blank titration (ml)

N = Normality of standard acid

w = Weight of soil sample taken

3.7.14.3 Total nitrogen (TN)

The total nitrogen content in soil was estimated by the micro-Kjeldahl method as per the procedure suggested by Page *et al.* (1982). Total nitrogen content in soil sample was determined by taking 0.25 to 0.5 gram of uniformly processed soil sample

in a digestion tube along with a 0.5 g of digestion mixture (K_2SO_4 : $CuSO_4 \cdot 5H_2O$ and selenium powder in the ratio of 100:20:1) and 10 mL of concentrated sulphuric acid was added and was kept for pre-digestion. After pre-digestion, the soil sample was digested at 360 °C in a digestion block till the material turns colourless. After cooling, the digested material was distilled by an automatic Kel-Plus distillation assembly. After completion of the distillation, the liberated ammonia will be trapped in boric acid mixed indicator solution and the distillate was titrated against standard acid. The amount of total nitrogen was determined and expressed in $mg\ kg^{-1}$.

$$Total\ nitrogen\ (mg\ kg^{-1}) = \frac{(V_s - V_b) \times N \times 0.014 \times 10^6}{w}$$

Where,

V_s = Volume of standard acid used for sample titration (ml)

V_b = Volume of standard acid used for blank titration (ml)

N = Normality of standard acid

w = Weight of soil sample taken

3.7.14.4 Organic nitrogen (ON)

The organic nitrogen was determined by subtracting the soil inorganic nitrogen from the total nitrogen content of soil as below:

$$Organic\ nitrogen\ (ON) = Total\ nitrogen\ (TN) - Inorganic\ nitrogen\ (IN)$$

3.7.15 Soil Enzymatic activities

3.7.15.1 Dehydrogenase activity ($\mu g\ TPF\ g^{-1}\ day^{-1}$)

The dehydrogenase activity was determined as described by Casida *et al.* (1964). Three grams of air-dried soil was thoroughly mixed with 0.2 g of $CaCO_3$. To each sample tube, 1 ml of 3 per cent aqueous solution of 2, 3, 5-triphenyl tetrazolium chloride (TTC) and 2.5 ml of distilled water were added. The contents of each tube were mixed using vortex mixer and the tubes were incubated at 37 °C. After 24 hours, ten ml of methanol was added to the tube and shaken for one minute. Then, the suspension was filtered through a glass funnel plugged with absorbent cotton into a 100 ml volumetric flask. The soil in the tube was washed repeatedly with methanol till the filtrate coming from the tip of the funnel was colorless. Then, the volume was made to 100 ml with methanol. The intensity of reddish colour was read using spectrophotometer at a wavelength of 485 nm with methanol as the blank. The amount of TPF (2, 3, 5-triphenyl formazan) produced was calculated using a standard graph prepared by using different concentrations of TPF standards. For this, ten ml of the TPF

standard solution containing 1000 $\mu\text{g ml}^{-1}$ was diluted to 100 ml with methanol (100 μg of TPF ml^{-1}). Aliquots of 5, 10, 15 and 20 ml were pipetted into 100 ml volumetric flasks (500, 1000, 1500 and 2000 μg of TPF 100 ml^{-1}). The volume was made to 100 ml with methanol and mixed thoroughly. Then, the intensity of red colour formed by TPF was measured as described earlier.

Calculation:

$$\text{Dehydrogenase activity } (\mu\text{g TPF/g/d}) = \frac{(S - C)}{W} \times 100 \text{ (dilution factor)}$$

Where,

S- Concentration of TPF in samples as calculated from the calibration curve

C- Concentration of TPF in control as calculated from the calibration curve

W- Dry weight of soil (g) obtained from oven drying of the fresh soil

3.7.15.2 Acid phosphatase activity ($\mu\text{g p-nitrophenol g}^{-1} \text{ h}^{-1}$)

One gram of the fresh soil (particle size < 2mm) was placed in 50 mL Erlenmeyer flask. To that 0.2 mL of toluene and 4 mL of modified universal buffer (pH 6.5) was added, followed by 1mL of 0.05 M p-nitro phenyl phosphate (pH 6.5) and swirled the flask for few seconds and kept in an incubator at 37 °C. After 1 hour of incubation, 1 mL of 0.5 M calcium chloride and 4 mL of 0.5 M sodium hydroxide was added and again swirled the flask for few seconds. The soil suspension was filtered through Whatman No. 2 filter paper. The intensity of the yellow colour was measured immediately in an UV-Vis spectrophotometer (CARY 50 model of VARIAN, Australia) at 420 nm. The p-nitro phenol content of the filtrate was computed from standard curve plotted from the results obtained with standards containing 0, 10, 20, 30, 40, 50 g of the p-nitro phenol. Control was performed by following the same procedure described for the assay of phosphatase activity without PNP solution as substrate just before filtration of soil suspension. The enzyme activity was expressed in $\mu\text{g p-nitro phenol per g of soil per hour on dry weight basis}$ (Tabatabai and Bremner, 1972)

$$\text{Acid phosphate activity } (\mu\text{g PNP g}^{-1} \text{ soil hr}^{-1}) = \frac{\text{OD}}{(0.0492 - 0.022)}$$

3.7.15.3 Urease activity ($\mu\text{g NH}_4\text{-N g}^{-1} \text{ h}^{-1}$)

Five gram of fresh soil was taken in a 50 mL volumetric flask. To this 0.2 mL of toluene and 9 mL of tris hydroxymethyl aminomethane buffer (THAM-0.05M, pH 9.0) was added and swirled the flask for few seconds, followed by the addition of 1 mL of 0.2 M urea solution and incubated for 2 hours at 37 °C. Then 35 mL of KCl-Ag₂SO₄

solution was added and mixed by shaking immediately and allowed to cool to room temperature. Then the volume was made up to 50 mL by addition of KCl-Ag₂SO₄, stoppered and mixed thoroughly. From this solution 20 mL aliquot was pipetted out into 100 mL distillation flask and steam distilled with 0.2 g of MgO for 6 minutes. Then 25 mL of the distillate was collected in 50 mL Erlenmeyer flask containing 5 mL of 2 per cent boric acid with double indicator solution. Ammonium N concentration of distillate was estimated by titrating with standard acid (0.01 N H₂SO₄) with the color change from green to pink as the end point. A control was also maintained and the same procedure was followed with the addition of 1 mL of 0.5 M urea solution after the addition of 35 mL of KCl-Ag₂SO₄ solution (Tabatabai and Bremner, 1972).

$$\text{Urease activity } (\mu\text{g NH}_4^+ - \text{N g}^{-1} \text{ soil h}^{-1}) = \frac{\text{Quantity of ammonium produced}}{\text{Weight of soil} \times \text{Time of incubation}}$$

3.7.16 Microbial biomass carbon (mg kg⁻¹)

The microbial biomass carbon was analyzed by following chloroform fumigation method. Ten grams of soil sample was fumigated for 24 hours in vacuum desiccator using ethanol-free chloroform. After fumigation, chloroform fumes were removed by evacuation. Non-fumigated and fumigated soil samples were extracted using 50 ml of 0.5 M K₂SO₄ and extracts were used for determining carbon (Vance *et al.*, 1987).

$$\text{Microbial biomass carbon in soil (MBC)} = \frac{EC_F - EC_{NF}}{K_{EC}}$$

Where,

EC_F = Total weight of extractable C in the fumigated soil samples

EC_{NF} = Total weight of extractable C in non-fumigated soil samples

K_{EC} = 0.25 ± 0.05 and it represents the efficiency of extraction of MBC

3.7.17 Microbial biomass nitrogen (mg kg⁻¹)

The microbial biomass nitrogen was estimated by following chloroform fumigation method. Five grams of soil sample was fumigated for 24 hours in vacuum desiccator using ethanol-free chloroform. After fumigation, chloroform fumes were removed by evacuation. Non-fumigated soil and fumigated soils were extracted using 50 ml of 0.5 M K₂SO₄ and extracts were used for determining microbial biomass nitrogen (Shen *et al.*, 1984).

3.7.18 Soil microbial populations (Bacteria, fungi and actinomycetes)

Dilution and plate count technique was used for enumerating the microbial population in the soil. Ten gram soil was suspended in 100 ml water to obtain 10¹

dilutions. One ml of this suspension was added to 9 ml water to get a dilution of 10^2 . Similarly, the dilution was continued until 10^6 dilutions were obtained. From 10^6 dilution 1 ml was added to sterile petri plate for enumeration of bacteria and 1ml from 10^4 and 10^3 dilutions for fungi and actinomycetes, respectively.

Then, 15 ml of the appropriate medium (Table 1) was added to each plate and rotated in the clockwise and anticlockwise direction. After solidification, the plates were incubated in an inverted position at room temperature. After the incubation period, the colonies were counted assuming that each viable cell will give rise to a single colony. Finally, the number of colonies (CFU) in 1g of soil was calculated by using the following formula as described by Skinner *et al.* (1952).

$$\text{No. of colonies per gram of soil (CFU)} = \frac{\text{No. of microbial count} \times \text{Dilution factor}}{\text{ml taken for dilution} \times \text{Weight of soil}}$$

Table 1. Specific media used for enumeration of soil microorganisms

Microbial group	Media used
Bacteria	Nutrient agar (NA)
Fungi	Martin's rose Bengal agar (MRBA)
Actinomycetes	Kusters Agar (KA)

3.7.19 Soil meso-fauna

Soil meso-faunal population was estimated for the freshly collected samples. The meso-fauna was extracted from the soil samples through the Berlese-Tullgren funnel method (Murphy, 1962). Soil samples were placed inside the funnel fitted with sieve of size 2×2 mm. The electric bulbs (25 w) were fixed at the top in the baffle board served as the source of light and heat. The apparatus was run for 48 hours. The invertebrates passing through sieve of the sample holder were collected in vials containing 70 per cent ethyl alcohol fixed to the lower end of the funnel. These vials were periodically checked to keep the alcohol at required levels. A stereo binocular microscope (35x magnifications) was used for enumerating the insect population present in the respective soil samples.

3.8 Identification of key indices and development of management strategies

The key indicators responsible to maintain critical ecosystem functions and processes were identified by soil quality index (SQI) approach. The steps involved in the SQI approach is detailed in the following section.

3.8.1 Soil Quality Index (SQI)

Soil quality assessment entails three primary steps: selecting the minimum data set (MDS) through principal component analysis (PCA) and determining the significance difference in correlation ($p < 0.05$), scoring soil indicators and amalgamating scores to formulate the SQI (Fig. 8). PCA, employing the varimax rotation technique, was also conducted to explore the relationships among these indicators. Principal components (PCs) explaining a minimum of 5 per cent of the variance and possessing eigenvalues > 1 was considered for indicator selection. Within each PC, indicators with weighted loading values within 10 per cent of the highest loading were selected for the MDS, irrespective of their sign. Multivariate correlation was used to detect and eliminate redundant data when multiple factors were retained within a single PC. In instances of high correlation ($r > 0.60$) among variables, only the variable with the highest correlation was retained for the MDS and considered a “key indicator” used for computing the SQI (Uthappa *et al.*, 2024).

A linear scoring method was used to convert the data of each identified critical MDS indicator into scores. For indicators where higher values indicated better function, each observation was divided by the highest observed value. Conversely, for indicators where lower values were preferable, the lowest observed value was divided by each observation (Tejashvini *et al.*, 2023). This process was performed using the following formula. Linear normalization (S_L) was carried out using the maximum (X_{max}) and minimum (X_{min}) values for each soil indicator (X), as shown in Eqs. 1 and 2.

$$S_L = \frac{X}{X_{max}} \text{ ----- “more is the better” approach (1)}$$

$$S_L = \frac{X_{min}}{X} \text{ ----- “Less is the better” approach (2)}$$

Based on the PCA results, the MDS indicators for each observation were weighted following conversion into linear scores. Each PC in the data set represented a certain percentage of variance, and the weighted factor for each MDS indicator was determined by dividing the percentage variance by the cumulative variance for all PCs with eigenvalues > 1 . Equation 3 was used to calculate the SQI by the weighted scores of the MDS indicators for each observation.

$$SQI = \sum_{i=1}^n W_i S_i \text{ ----- (3)}$$

The subscripted variable’s score is denoted as (S_i), with its weighting factor from PCA represented as (W_i). The SQI values were standardized to a range of 0 to 1 by dividing all the SQI values by the maximum SQI value. Soil quality indicators can be a valuable asset for ecosystem monitoring and assessment in restoration programs,

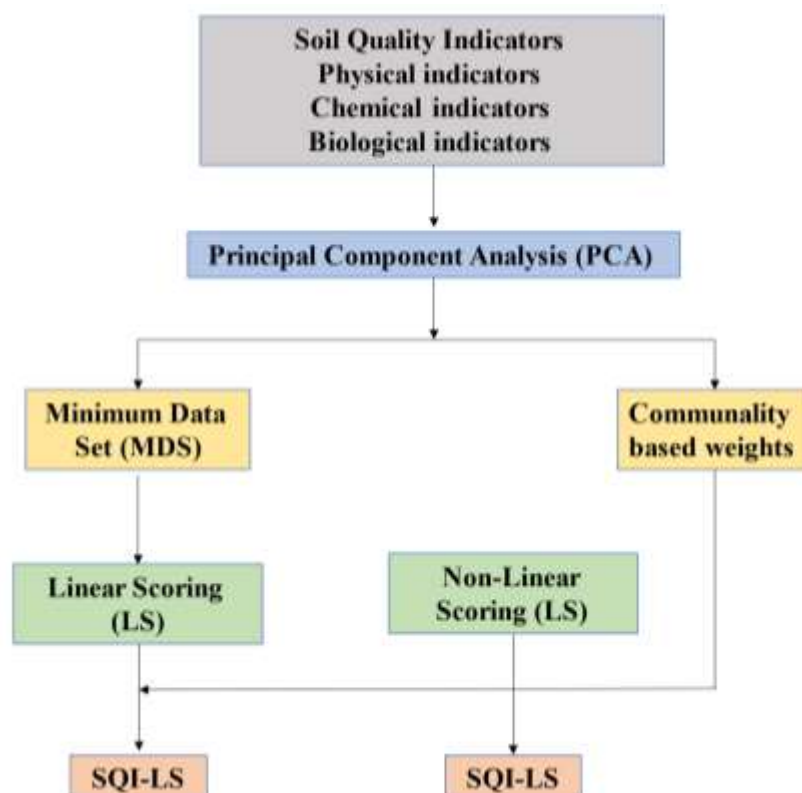


Fig. 8. Flowchart of the methodology adopted for enumerating SQI

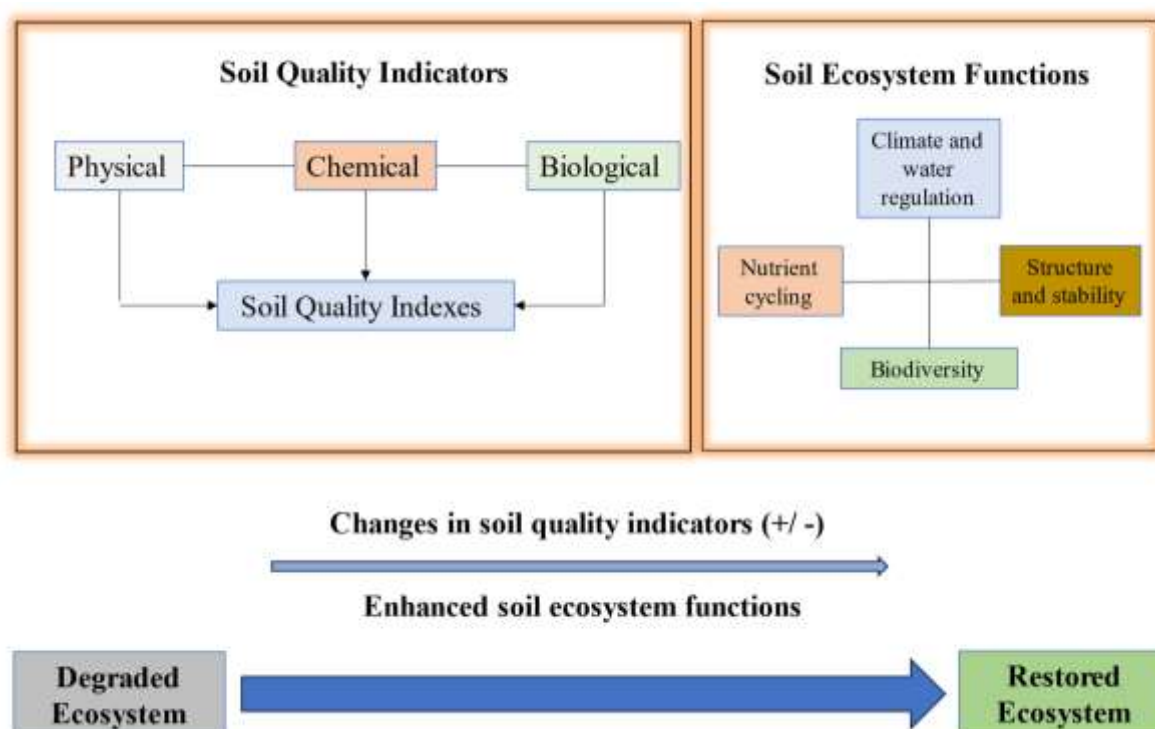


Fig. 9. Role of Soil quality indicators in restoration of soil ecosystem services

predominantly with respect to understanding the role of soil properties and plant–soil relationships that maintains the critical ecological functions (Fig. 9). By regular and careful monitoring and assessment of key soil quality indicators will aid in restoration of ecological services. Data obtained from SQI analysis can be used in allocating the optimum resources to maintain the critical ecological functions and processes in urban areas.

3.9 Statistical Analysis

Urban, peri-urban and rural data were analysed statistically for different soil attributes by using two-way analysis of variance (ANOVA) with unequal number of observations and ANOVA was performed in OriginPro 2024. Means for different land uses were compared by using Fisher's least significant difference (LSD) test. Differences in land-use mean at $p < 0.05$ were considered to be statistically significant at 5 per cent level of significance (Chew, 1977). Strength of different soil quality properties was determined by the Pearson's correlation coefficient. Selection of minimum data set (MDS) through PCA for SQI was performed by using SPSS software. Data visualization was done with the help of OriginPro 2024 software.

EXPERIMENTAL RESULTS AND DISCUSSION

IV EXPERIMENTAL RESULTS AND DISCUSSION

The results obtained for the study entitled “Assessment of Soil Ecological Functions and Processes along the Rural-Urban Gradient in Shivamogga District,” are discussed through the data tables and relevant figures in this chapter with the following headings.

4.1 Magnitude of different land uses and extent of soil sealing along rural-urban gradient

The land use land cover (LULC) map of the study area (Fig.10.) was derived using the FCC, NDVI, NDWI and NDBI in ArcMap 10.8. This resultant image provided the spatial representation of different land cover types across the rural-urban gradient, highlighting variations in vegetation cover and impervious surfaces with a kappa coefficient of 0.94.

The urban zone data highlighted the dominance of sealed areas and compacted areas, indicative of significant urbanization (Table 2). Constituting the largest land use, the sealed areas ranged from 225.54 ha (U4) to 240.50 ha (U3). This reflects intensive infrastructure development and urban sprawl. Rainfed crops account for approximately 24.71 to 30.30 per cent of the total land use across different regions, demonstrating their persistence despite urban pressures. Coconut and arecanut plantations were notable, with areas between 10.14 ha (U3) and 19.15 ha (U1) for coconut and 6.76 ha (U2) to 11.38 ha (U3) for arecanut. A significant percentage of land remains under waterbodies (7.16 ha to 12.78 ha), contributing to ecological stability.

The peri-urban zone exhibits a transitional land-use pattern, balancing between urban influences and rural characteristics (Table 3). Rainfed crops was most extensive land use, ranging from 70.04 ha (P5) to 91.01 ha (P2). This dominance signifies the zone's agricultural reliance. Arecanut (17.38 ha to 52.53 ha) and coconut plantations (24.30 ha to 43.50 ha) are significant, reflecting economic diversification. Sealed areas were substantial, yet lesser than urban zones, ranging from 92.15 ha (P3) to 130.23 ha (P1). Higher proportions of waterbodies were present compared to the urban zone, with a maximum of 27.41 ha (P3), emphasizing their prominence in the peri-urban landscape.

The rural zone was dominated by agricultural and horticultural-based land uses, with minimal urban influences (Table 4). Rainfed Crops was the largest category, ranging from 116.78 ha (R5) to 130.03 ha (R2), emphasizing agriculture's centrality. Significantly high proportion of paddy land use system was present compared to urban and peri-urban zones, with a maximum of 69.53 ha (R1). Coconut plantations dominate (40.73 ha to 69.79 ha), followed by arecanut (32.24 ha to 48.10 ha). Sealed areas were

Table 2. Magnitude of different land uses and extent of soil sealing (in ha) in five circular areas of urban zone (r = 1 sq. km)

Land uses	U1	U2	U3	U4	U5
Forest/forest plantations	0.95	1.67	1.15	1.45	2.54
Waterbodies	11.35	8.76	7.16	9.76	12.78
Mango	1.34	3.23	1.10	3.11	2.11
Arecanut	8.87	6.76	11.38	9.71	10.63
Coconut	19.15	16.77	10.14	14.31	15.64
Paddy	4.94	4.70	4.20	6.86	3.43
Rainfed Crop	26.00	25.70	27.04	30.30	24.71
Waste lands	8.89	10.01	6.67	9.44	9.44
Compacted area	5.00	3.04	4.79	3.33	5.00
Sealed Area	227.11	233.45	240.50	225.54	227.34

Note: U1 to U5 are the circular plots selected randomly in urban zone to assess extent of soil sealing and magnitude of different land uses

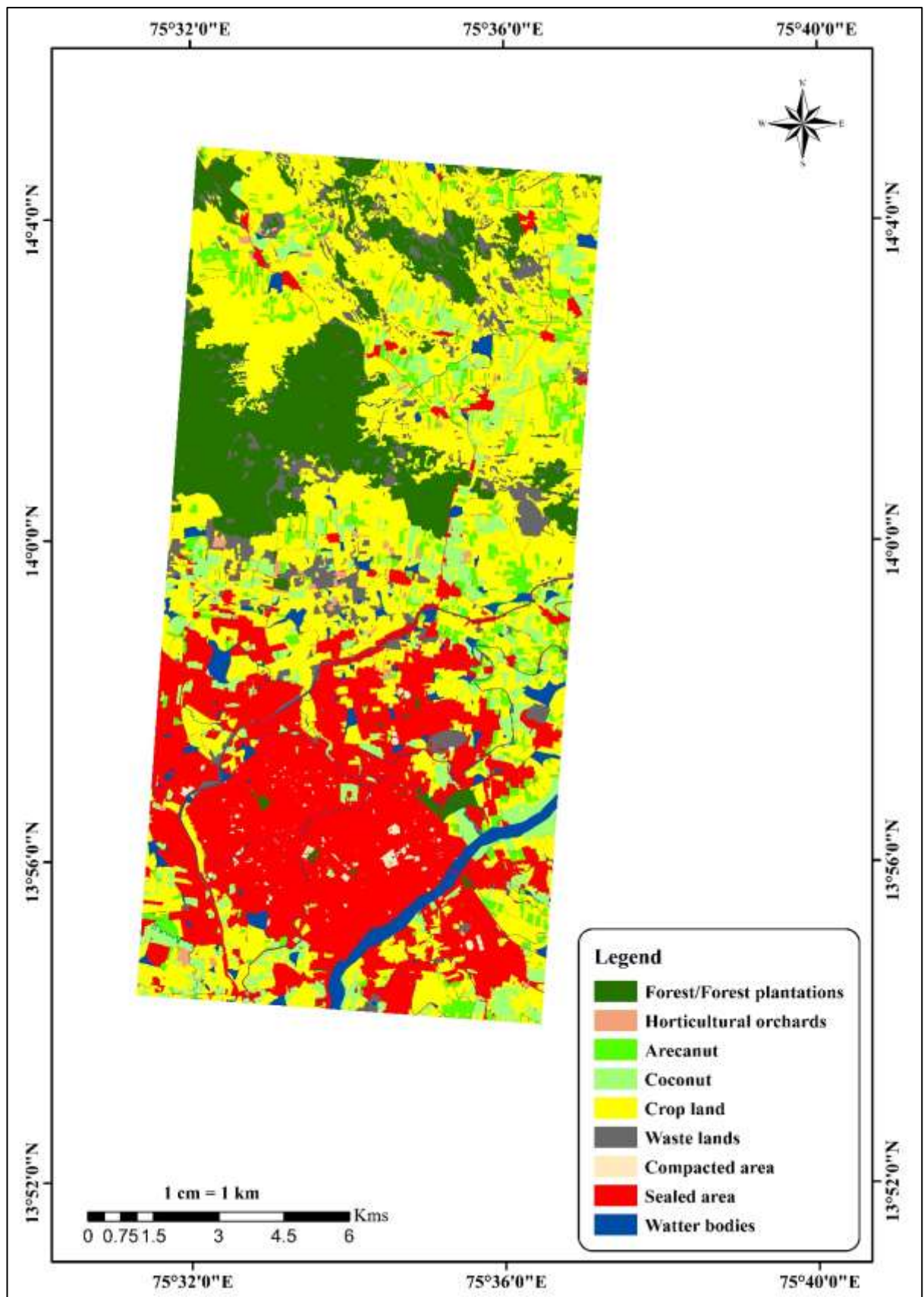


Fig. 10. Land use land cover map of the study area

Table 3. Magnitude of different land uses and extent of soil sealing (in ha) in five circular areas of peri-urban zone (r = 1 sq. km)

Land uses	P1	P2	P3	P4	P5
Forest/forest plantations	3.37	2.11	1.67	1.11	2.44
Waterbodies	21.87	18.64	27.41	22.14	11.74
Mango	8.19	7.93	4.56	11.45	10.33
Arecanut	17.38	34.98	29.40	52.53	24.32
Coconut	25.37	34.44	43.50	24.30	38.74
Paddy	20.46	20.92	18.49	13.49	23.76
Rainfed Crop	78.83	91.01	88.66	80.75	70.04
Waste lands	28.65	28.85	23.05	24.94	34.09
Compacted area	1.56	1.01	0.98	0.67	1.02
Sealed Area	130.23	96.10	92.15	98.76	125.67

Note: P1 to P5 are the circular plots selected randomly in peri-urban zone to assess extent of soil sealing and magnitude of different land uses

Table 4. Magnitude of different land uses and extent of soil sealing (in ha) in five circular areas of rural zone (r = 1 sq. km)

Land uses	R1	R2	R3	R4	R5
Forest/forest plantations	4.01	2.23	4.11	3.90	3.34
Waterbodies	2.23	1.78	1.87	1.37	1.89
Mango	11.41	10.80	12.87	8.88	7.72
Arecanut	33.89	44.70	32.24	45.98	48.10
Coconut	40.73	56.78	69.79	60.97	46.24
Paddy	69.53	39.25	37.05	41.06	58.78
Rainfed Crop	128.78	130.03	125.87	117.83	116.78
Waste lands	4.69	5.66	5.26	6.92	7.02
Compacted area	0.09	0.23	0.14	0.12	0.10
Sealed Area	18.64	22.46	24.66	26.55	24.16

Note: R1 to R5 are the circular plots selected randomly in rural zone to assess extent of soil sealing and magnitude of different land uses

found to be minimal, ranging from 18.64 ha (R1) to 26.55 ha (R4), reflecting low infrastructure development. Waterbodies were limited in rural, ranging from 1.37 ha (R4) to 2.23 ha (R1).

Fig. 11 to 14 emphasized the spatial distribution and proportional significance of various land-use types across zones, providing a clear representation of urbanization's impact and the persistence of agriculture in rural zones.

The gradient analysis (Table 5) provided a comparative summary of land-use patterns across urban, peri-urban and rural zones. Urban zone exhibited the highest extent of sealed areas, amounting to 230.79 ha. Forest/forest plantations and water bodies in urban areas covered only about 1.55 ha and 9.96 ha, respectively. Agricultural land use was constrained, with areas for paddy and rainfed crops being relatively small, 4.83 ha and 26.75 ha, respectively. Compacted areas (4.23 ha) and mango (2.18 ha) were also limited.

The peri-urban zone presented a transitional landscape between urban and rural settings. The extent of sealed areas was reduced to 108.58 ha. Agricultural uses dominated in peri-urban areas, with rainfed crops (81.86 ha) and paddy cultivation (19.42 ha) occupying substantial portions. Coconut (33.27 ha) and arecanut (31.72 ha) plantations were prominent. Waterbodies covered 20.36 ha, substantially higher than in urban areas. Waste lands and compacted areas were minor components at 4.32 ha and 1.05 ha, respectively.

Rural zone was characterized by expansive agricultural and natural landscapes, with minimal soil sealing (23.29 ha). Rainfed crops (123.86 ha) and paddy fields (49.13 ha) were the dominant land uses in the rural set-up. Coconut (54.90 ha) and arecanut (40.98 ha) were significant plantation crops. Forest/forest plantations were comparatively larger (3.52 ha). However, waterbodies in rural zones were minimal (1.83 ha). Waste lands (0.54 ha) and compacted areas (0.14 ha) were found to be minor components.

The proportional distribution of sealed, arable and open lands (Fig. 14.) indicated that, urban areas were largely dominated by sealed lands, reflecting infrastructure and population density (Tardieu *et al.*, 2021). While rural areas are predominantly arable, highlighting their dependence on agriculture (Giller *et al.*, 2021). Peri-urban areas exhibited mixed land-use characteristics, acting as a buffer zone between urbanized and rural landscapes (Halder *et al.*, 2024). These findings emphasize the significant impact of urbanization on land use patterns.

Table 5. Magnitude of different land uses and extent of soil sealing in rural-urban gradient of Shivamogga

Land uses	Urban	Peri-urban	Rural
	Area in ha		
Forest/forest plantations	1.55	2.14	3.52
Waterbodies	9.96	20.36	1.83
Mango	2.18	8.49	10.34
Arecanut	9.47	31.72	40.98
Coconut	15.20	33.27	54.90
Paddy	4.83	19.42	49.13
Rainfed Crop	26.75	81.86	123.86
Waste lands	8.89	6.86	5.91
Compacted area	4.23	1.05	0.14
Sealed Area	230.79	108.58	23.29
S.Em±	0.34	1.07	0.86
CD @ 0.05	0.96	3.04	2.43

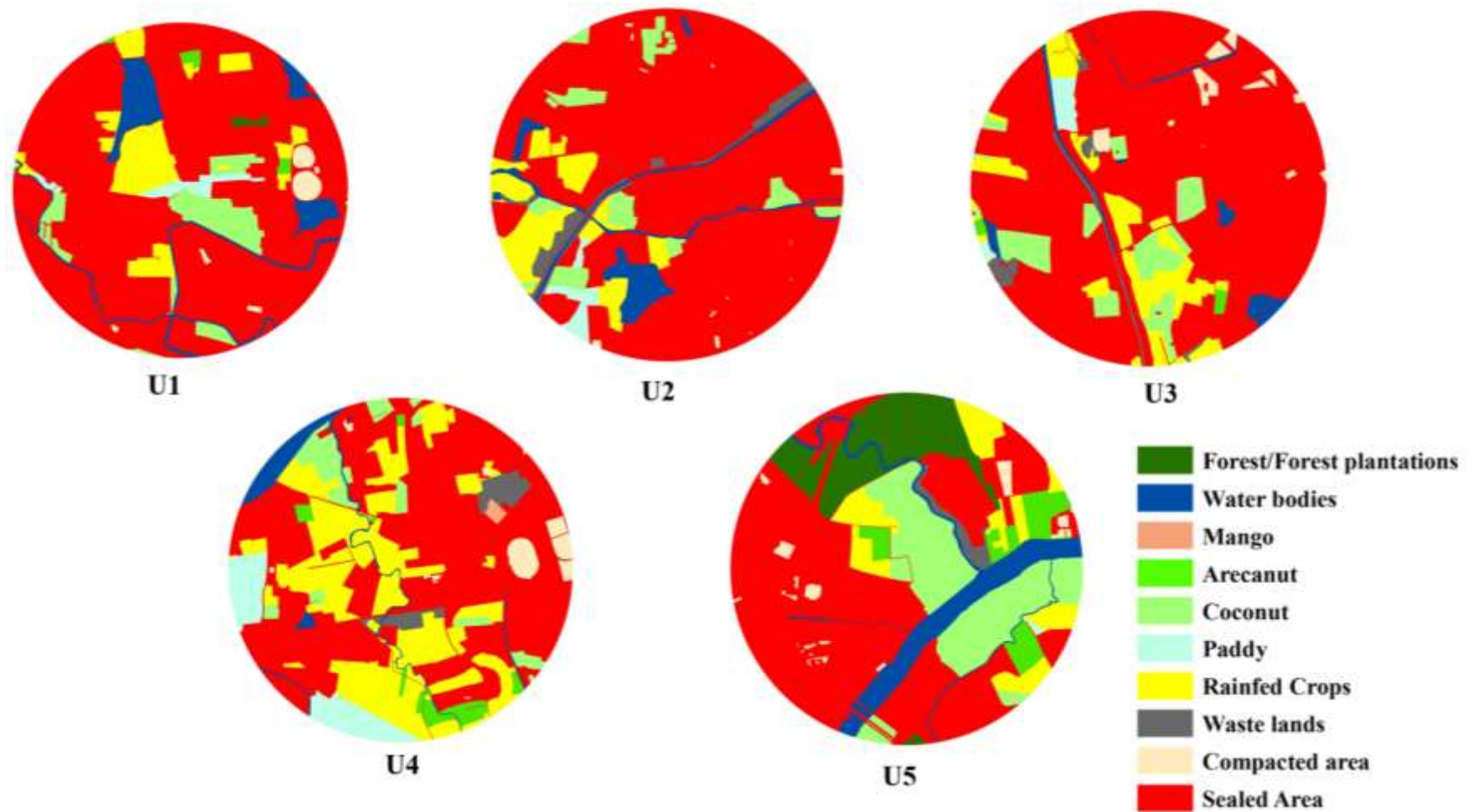


Fig. 11. Pictorial depiction of magnitude of different land uses and extent of soil sealing in five circular areas of urban zone
(Note: U1 to U5 represents the five circular areas of urban zone, $r = 1$ sq. km)

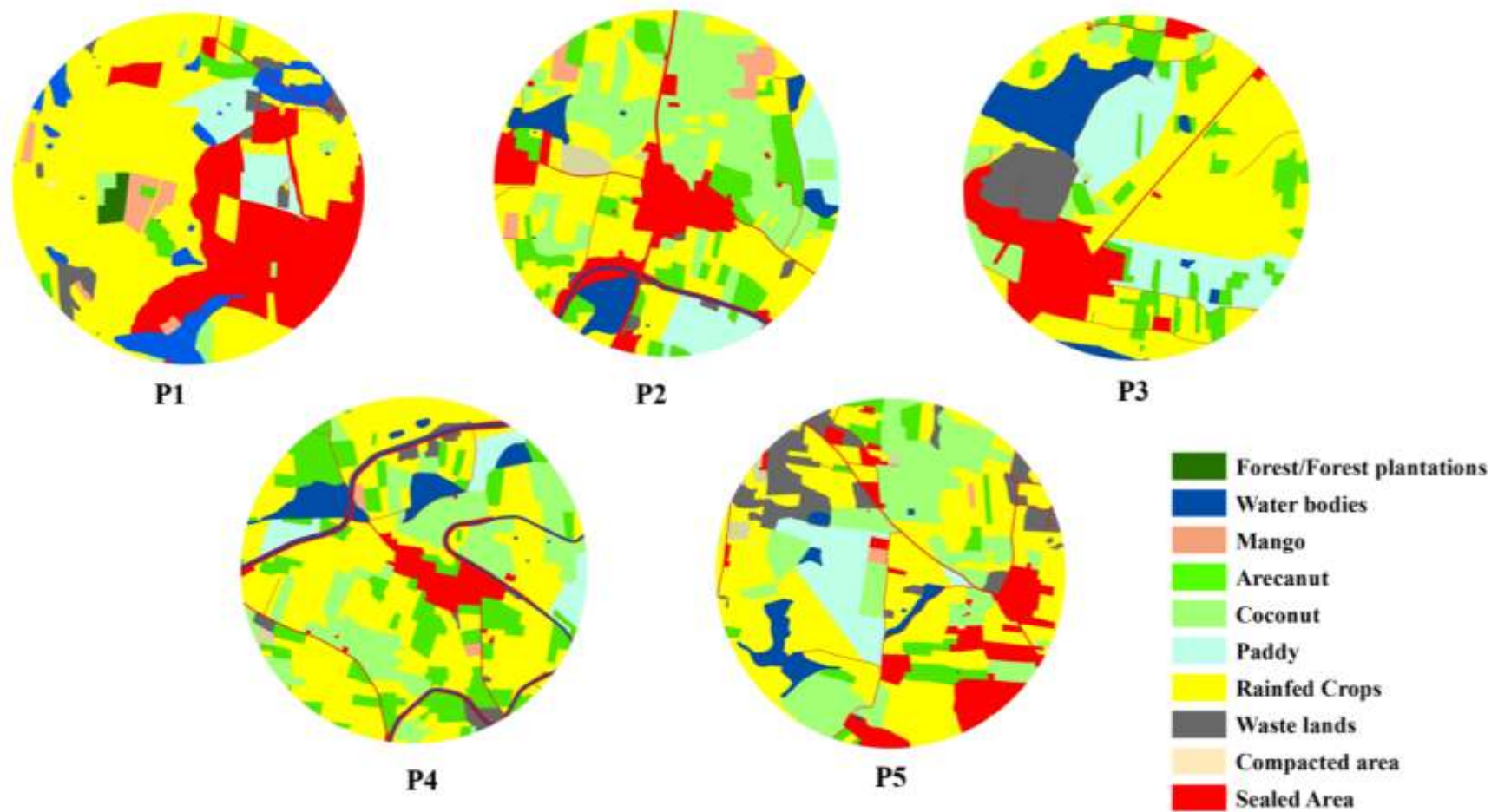


Fig.12. Pictorial depiction of magnitude of different land uses and extent of soil sealing in five circular areas of peri-urban zone
 (Note: P1 to P5 represents the five circular areas of peri-urban zone, $r = 1$ sq. km)

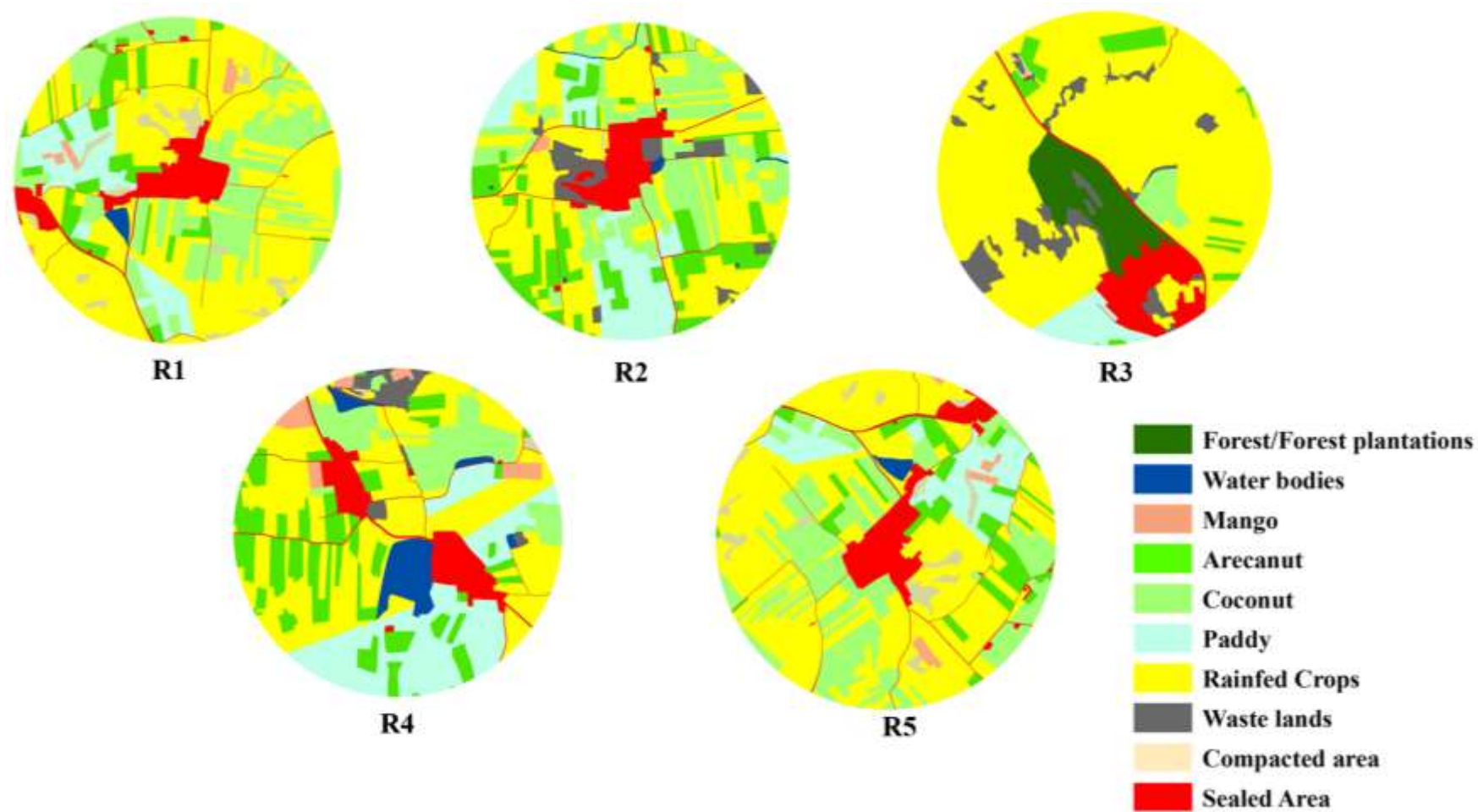


Fig. 13. Pictorial depiction of magnitude of different land uses and extent of soil sealing in five circular areas of rural zone
 (Note: R1 to R5 represents the five circular areas of rural zone, $r = 1$ sq. km)

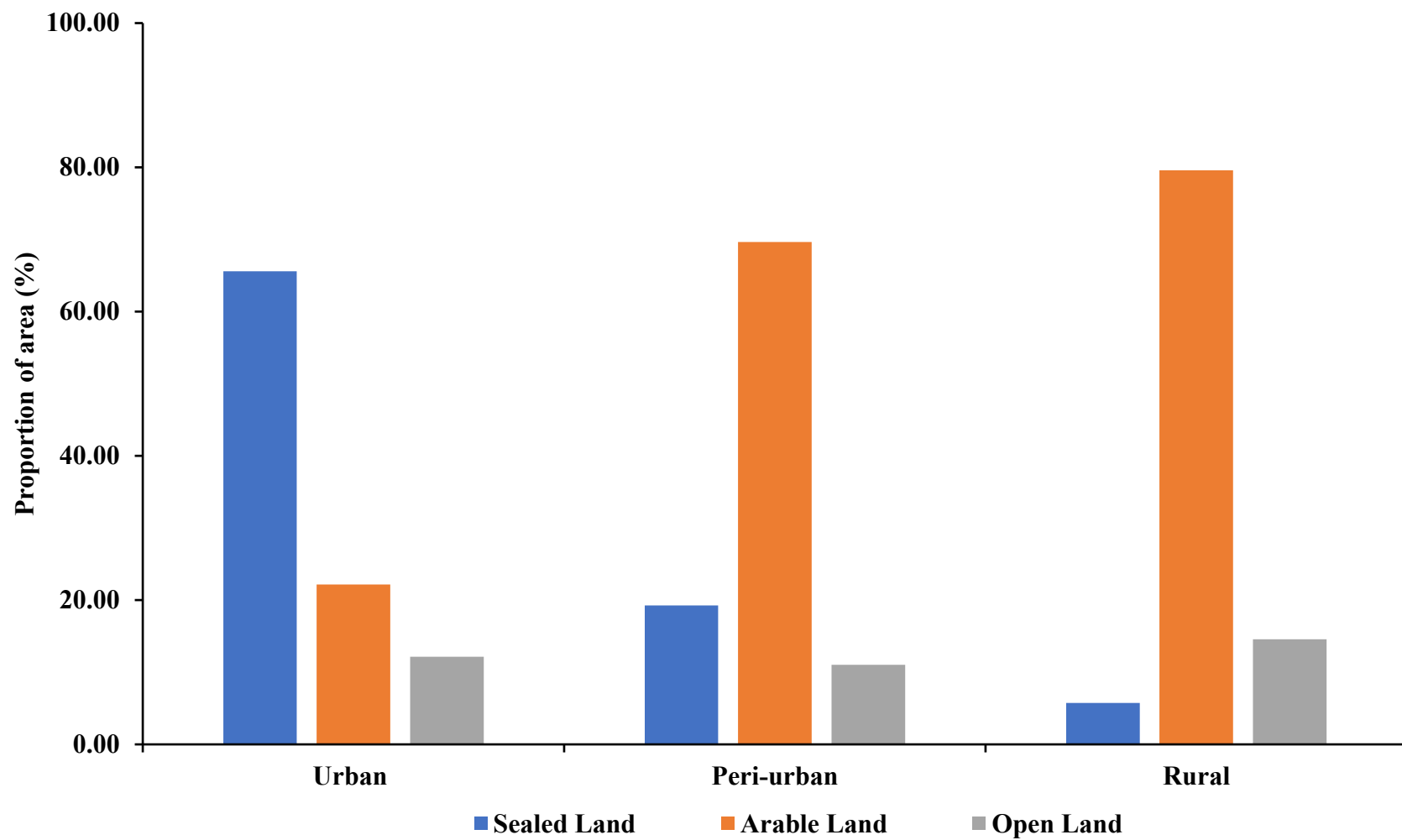


Fig. 14. Proportion of sealed land, arable land and open land across the rural-urban gradient

4.2 Quantification of litter biomass and nutrient turnovers in different land uses along rural-urban gradient

4.2.1 Net organic matter recycled

The analysis of land use systems across the rural-urban gradient highlighted the diverse nature of agricultural and natural ecosystems (Table 6). Natural systems, such as forests and waste lands, were characterized by the absence of irrigation and external organic inputs, whereas man-made systems, including mango, arecanut, coconut, paddy and rainfed crops, relied on irrigation and farmyard manure (FYM) for nutrient supplementation. Compacted and sealed areas, primarily indicative of urban infrastructure, lacked both irrigation and organic inputs, signifying their non-productive status in terms of biomass recycling and nutrient dynamics.

The quantity of organic matter recycled through biomass and FYM varied significantly across zones and land use systems (Table 7). Forest plantations exhibited the highest biomass recycling across all zones, with values of 9.01, 9.19 and 9.38 Mg ha⁻¹ yr⁻¹ in urban, peri-urban and rural areas, respectively. Among man-made systems, arecanut contributed the highest net organic matter recycling (10.50 to 10.65 Mg ha⁻¹ yr⁻¹), followed by mango (8.87 to 9.01 Mg ha⁻¹ yr⁻¹). Rainfed crops demonstrated moderate recycling capacities (6.09 to 6.16 Mg ha⁻¹ yr⁻¹), reflecting their dependence on FYM inputs. Compacted and sealed areas had the lowest recycling rates (1.11–1.21 Mg ha⁻¹ yr⁻¹ and 0.32 to 0.67 Mg ha⁻¹ yr⁻¹, respectively), highlighting their limited ecological contribution.

Arecanut land use systems exhibited the highest organic matter recycling rates across all zones. This can be attributed to the perennial nature of arecanut plantations, which contribute to higher biomass returns through leaf litter, fronds and organic amendments (Sadasivuni *et al.*, 2015). Forest/forest plantation systems demonstrated consistent organic matter recycling across zones, highlighting their ecological stability and contribution to nutrient cycling (Nagaraja *et al.*, 2018; Macdonald *et al.*, 2019). Rainfed crops and paddy fields also showed higher recycling rates, where periodic organic inputs from crop residues and crop stubbles enhanced biomass turnovers (Pooniya *et al.*, 2018). Similarly, mango and coconut plantations recorded substantial levels of OM recycling, benefiting from managed vegetation and external organic inputs (Nagaraja *et al.*, 2016; Thomas and Krishnakumar, 2024).

Sealed areas showed the lowest organic matter recycling, emphasizing the detrimental effects of soil sealing due to urbanization (Hendrix *et al.*, 2018). Waste lands and compacted soils reflected limited organic matter inputs, primarily due to overgrazing and compaction, which hinder microbial activity and organic matter decomposition (Cusack *et al.*, 2016). The rural zone displayed the highest organic

Table 6. Description of system type and management practices followed in different land uses

Land uses	Abbreviation used	System type	Management	
			Irrigation	External organic inputs
Forest/forest plantations	FP	Natural	Nil	Nil
Mango	M	Man-made	Irrigated	Farm yard manure
Arecanut	A	Man-made	Irrigated	Farm yard manure
Coconut	C	Man-made	Irrigated	Farm yard manure
Paddy	P	Man-made	Irrigated	Farm yard manure
Rainfed crop	R	Man-made	Nil	Farm yard manure
Waste lands	WL	Natural	Nil	Nil
Compacted area	CA	Man-made	Nil	Nil
Sealed area	SA	Man-made	Nil	Nil

Table 7. Quantity of organic matter recycled and added through FYM (Mg ha⁻¹ yr⁻¹) in different land use systems of rural-urban gradient

Zones	Urban			Peri-urban			Rural		
	Litter biomass recycled	FYM added	Net organic matter recycled	Litter biomass recycled	FYM added	Net organic matter recycled	Litter biomass recycled	FYM added	Net organic matter recycled
Forest/forest plantations	9.01	-	9.01	9.19	-	9.19	9.38	-	9.38
Mango	1.34	7.53	8.87	1.37	7.58	8.95	1.40	7.61	9.01
Arecanut	2.30	8.20	10.50	2.35	8.21	10.56	2.40	8.25	10.65
Coconut	1.21	6.23	7.44	1.23	6.23	7.46	2.26	6.25	8.51
Paddy	2.72	4.44	7.16	2.77	4.47	7.24	2.83	4.51	7.34
Rainfed crop	1.79	4.30	6.09	1.82	4.30	6.12	1.86	4.30	6.16
Waste lands	1.26	-	1.26	1.33	-	1.33	1.42	-	1.42
Compacted area	1.11	-	1.11	1.19	-	1.19	1.21	-	1.21
Sealed area	0.32	-	0.32	0.45	-	0.45	0.67	-	0.67

matter recycling, followed by peri-urban and urban zones. The relatively higher recycling rates in rural areas align with less intensive land use, traditional farming practices and higher organic inputs (Tadesse *et al.*, 2021). The proportion of sealed area is less in rural zone, which also might have attributed in higher biomass turnovers in the rural zone (Salata *et al.*, 2024).

4.2.2 Nutrient turnover

The data pertaining to nutrient turnovers *viz.*, nitrogen, phosphorus and potassium of different land use systems across the zones are presented in Table 9 to 10.

4.2.2.1 Nitrogen turnover

Nitrogen (N) recycling showed distinct trends across land use systems and zones (Table 8). Forest plantations were the largest contributors, recycling up to 90.99 kg N ha⁻¹ yr⁻¹ in rural areas. Arecanut systems demonstrated significant nitrogen recycling, with net contributions increasing from 87.00 kg N ha⁻¹ yr⁻¹ in urban zones to 89.25 kg N ha⁻¹ yr⁻¹ in rural zones. Rainfed crops and paddy systems exhibited consistent yet lower nitrogen recycling (32.60 to 35.57 kg N ha⁻¹ yr⁻¹). Urban zones showed reduced recycling rates across most land uses compared to peri-urban and rural zones, correlating with increased soil sealing and reduced biomass input.

4.2.2.2 Phosphorus turnover

Phosphorus (P) recycling mirrored the trends observed for nitrogen (Table 9). Forest plantations and arecanut systems exhibited the highest net P recycling, with arecanut contributing up to 29.03 kg P ha⁻¹ yr⁻¹ in rural areas. Coconut systems displayed notable increases in rural zones, reaching 23.54 kg P ha⁻¹ yr⁻¹, compared to 17.27 kg P ha⁻¹ yr⁻¹ in urban zones. Rainfed crops and paddy systems showed moderate recycling capacities, with net P contributions ranging from 14.87 to 21.18 kg P ha⁻¹ yr⁻¹ across zones. Urban areas consistently exhibited lower P recycling due to limited agricultural activity and higher compacted and sealed land proportions.

4.2.2.3 Potassium turnover

Potassium (K) recycling also demonstrated the similar trends as that of nitrogen and phosphorus (Table 10). Arecanut systems were the most significant recyclers, with net K recycling reaching 68.13 kg ha⁻¹ yr⁻¹ in rural areas. Paddy systems also showed high K recycling rates, with contributions increasing from 60.85 kg ha⁻¹ yr⁻¹ in urban zones to 61.89 kg ha⁻¹ year⁻¹ in rural zones. Coconut systems demonstrated substantial variation, with rural zones achieving the highest net K recycling (59.95 kg ha⁻¹ yr⁻¹). The results highlight the pivotal role of perennial crop systems in maintaining soil fertility, particularly in rural and peri-urban zones.

Table 8. Quantity of N recycled and added through FYM (kg ha⁻¹ yr⁻¹) in different land use systems of rural-urban gradient

Zones	Urban			Peri-urban			Rural		
	N recycled through biomass	N added through FYM	Net N recycled	N recycled through biomass	N added through FYM	Net N recycled	N recycled through biomass	N added through FYM	Net N recycled
Forest/forest plantations	87.40	-	87.40	89.14	-	89.14	90.99	-	90.99
Mango	26.80	37.65	64.45	27.40	37.90	65.30	28.00	38.05	66.05
Areca nut	46.00	41.00	87.00	47.00	41.05	88.05	48.00	41.25	89.25
Coconut	14.64	31.15	45.79	14.88	31.15	46.03	27.35	31.25	58.60
Paddy	12.51	22.20	34.71	12.74	22.35	35.09	13.02	22.55	35.57
Rainfed crop	11.10	21.50	32.60	11.28	21.50	32.78	11.53	21.50	33.03
Waste lands	15.75	-	15.75	16.63	-	16.63	17.75	-	17.75
Compacted area	5.55	-	5.55	5.95	-	5.95	6.05	-	6.05
Sealed area	1.95	-	1.95	2.75	-	2.75	4.09	-	4.09

Table 9. Quantity of P recycled and added through FYM (kg ha⁻¹ yr⁻¹) in different land use systems of rural-urban gradient

Zones	Urban			Peri-urban			Rural		
	P recycled through biomass	P added through FYM	Net P recycled	P recycled through biomass	P added through FYM	Net P recycled	P recycled through biomass	P added through FYM	Net P recycled
Forest/forest plantations	31.54	-	31.54	32.17	-	32.17	32.83	-	32.83
Mango	2.68	18.83	21.51	4.80	18.95	23.75	4.90	19.03	23.93
Arecanut	4.83	20.50	25.33	8.23	20.53	28.75	8.40	20.63	29.03
Coconut	1.69	15.58	17.27	4.31	15.58	19.88	7.91	15.63	23.54
Paddy	4.08	11.10	15.18	9.70	11.18	20.87	9.91	11.28	21.18
Rainfed crop	4.12	10.75	14.87	6.37	10.75	17.12	6.51	10.75	17.26
Waste lands	3.15	-	3.15	4.66	-	4.66	4.97	-	4.97
Compacted areas	2.11	-	2.11	4.17	-	4.17	4.24	-	4.24
Scaled area	0.67	-	0.67	1.58	-	1.58	2.35	-	2.35

Table 10. Quantity of K recycled and added through FYM (kg ha⁻¹ yr⁻¹) in different land use systems of rural-urban gradient

Zones	Urban			Peri-urban			Rural		
	K recycled through biomass	K added through FYM	Net K recycled	K recycled through biomass	K added through FYM	Net K recycled	K recycled through biomass	K added through FYM	Net K recycled
Land use system									
Forest/forest plantations	48.65	-	49.63	49.63	-	49.63	50.65	-	50.65
Mango	20.10	37.90	58.45	20.55	37.90	58.45	21.00	38.05	59.05
Arecanut	25.76	41.05	67.37	26.32	41.05	67.37	26.88	41.25	68.13
Coconut	15.37	31.15	46.77	15.62	31.15	46.77	28.70	31.25	59.95
Paddy	37.81	22.35	60.85	38.50	22.35	60.85	39.34	22.55	61.89
Rainfed crop	8.41	21.50	30.05	8.55	21.50	30.05	8.74	21.5	30.24
Waste land	14.36	-	15.16	15.16	-	15.16	16.19	-	16.19
Compacted land	9.88	-	10.59	10.59	-	10.59	10.77	-	10.77
Scaled area	2.02	-	2.84	2.84	-	2.84	4.22	-	4.22

Arecanut plantations exhibited the highest nutrient recycling. The higher nutrient turnovers in arecanut could be attributed to higher biomass production and organic matter inputs (Sadasivuni *et al.*, 2015). Forest/forest plantations demonstrated high nutrient recycling rates. This consistent recycling across urban, peri-urban and rural zones highlights the resilience of forest ecosystems in nutrient turnover (Macdonald *et al.*, 2019). Similarly, mango showed substantial amounts of nutrient recycling, indicating efficient management practices that sustain soil fertility (Thomas and Krishnakumar, 2024). Coconut plantations also displayed substantial amount of nutrient recycling, which can be attributed to external nutrient additions and organic matter retention (Subramanian *et al.*, 2024). Paddy and rainfed cropping systems recycled considerable amount nutrients, underscoring the role of managed cropping systems in enhancing soil nutrient cycling (Pooniya *et al.*, 2018).

Sealed areas showed lower nutrient recycling. This underlines the profound impact of impervious surfaces on nutrient cycling (Hendrix *et al.*, 2018). Compacted areas and sealed areas had the least nutrient recycling, reflecting the reduced biological activity and organic matter inputs in these highly disturbed environments (Cusack *et al.*, 2016). Nutrient recycling was highest in rural zone, followed by peri-urban and urban zone. These findings reflect the influence of anthropogenic pressures, with rural zone benefiting from larger areas under natural vegetation or low-intensity agriculture, which facilitate organic matter accumulation and nutrient cycling (Tadesse *et al.*, 2021).

4.3 Impact of soil sealing on soil ecological functions along rural-urban gradient

There are three main ecosystem services that are linked in to soil *i.e.*, global climate regulation via carbon storage, biomass provisioning and regulation of health and environmental risks linked to anthropic stresses in urban context. Soils provide multiple functions that support the delivery of key ecosystem services. However, to achieve the above-mentioned ecosystem services, four key soil functions can be taken in to consideration (Sere *et al.*, 2024). The four key soil functions that are linked to main ecosystem services are nutrient cycle, organic matter cycle, water cycle and biodiversity support (Sere *et al.*, 2024). The soil physical, chemical and biological properties govern the soil functions, which in turn regulates the critical soil ecosystems services. This section presents the impacts of land sealing on soil ecological functions along the rural-urban gradient, highlighting the changes in soil physical, chemical and biological properties.

4.3.1 Bulk density

The data pertaining to soil bulk density of different land uses across three different zones is presented in Table 11. The bulk density values ranged from 1.23 Mg m⁻³ in forest plantations of rural zone to 1.72 Mg m⁻³ in sealed areas of urban zone.

Table 11. Bulk density of soils (Mg m⁻³) among different land uses across the rural-urban gradient

Land uses (L)	Zones (Z)			Land uses (L) Mean
	Urban	Peri-urban	Rural	
Forest/forest plantations	1.29	1.25	1.23	1.25
Mango	1.31	1.32	1.30	1.31
Arecanut	1.27	1.30	1.32	1.30
Coconut	1.34	1.30	1.28	1.31
Paddy	1.41	1.40	1.38	1.40
Rainfed crop	1.36	1.33	1.33	1.34
Waste lands	1.48	1.45	1.43	1.45
Compacted area	1.65	1.61	1.60	1.62
Sealed area	1.72	1.70	1.68	1.70
Zones (Z) Mean	1.42	1.41	1.39	
Factors	S.Em±		CD @ 0.05	
Zones (Z)	0.01		0.02	
Land uses (L)	0.01		0.03	
Interaction (Z x L)	0.02		NS	

Among different land uses, the soils of forest/forest plantations recorded significantly lower values of bulk density (1.25 Mg m^{-3}). Among the agricultural and horticultural-based land use systems, the soils of mango orchards (1.31 Mg m^{-3}) arecanut (1.30 Mg m^{-3}) and coconut plantations (1.31 Mg m^{-3}) recorded lower bulk density values. However, paddy soils recorded slightly higher bulk density (1.40 Mg m^{-3}). In contrast, the soils of sealed area (1.70 Mg m^{-3}) and compacted area (1.62 Mg m^{-3}) recorded significantly higher values of bulk density. Comparison among the zones indicated that the soils of urban zones exhibited significantly higher bulk density (1.42 Mg m^{-3}). Whereas, the soils of rural (1.39 Mg m^{-3}) and peri-urban zones (1.41 Mg m^{-3}) recorded significantly lower bulk density values and were found to be statistically on par with each other.

The lower bulk density observed in forest and forest plantation soils indicate higher porosity in these soils (Fig. 15). Better soil structure due to higher organic matter content may be because of higher porosity and lower soil bulk density (Robinson *et al.*, 2022). These findings are in line with previous studies that have shown forest soils to exhibit superior structural stability and reduced bulk density compared to other land uses (Shete *et al.*, 2016; Reichert *et al.*, 2021). Slightly higher bulk density in paddy soils could be attributed to the loss of soil structure by puddling in paddy soil (Alam *et al.*, 2020). Significantly higher bulk density recorded in sealed areas in urbanized environments can be attributed to the compaction induced by surface sealing and construction activities. High bulk density in sealed soils is a well-documented phenomenon. The compactness is known to result in decreased soil aeration, reduced water infiltration and increased surface runoff, which may exacerbate urban hydrological challenges (Nagaraja *et al.*, 2016; Fokaides *et al.*, 2016). This trend aligns with urbanization studies that report increased soil bulk density in built-up areas due to reduced vegetation cover, lower organic matter and higher mechanical stress on soils (Demina *et al.*, 2018). Among the zones, urban zone exhibited higher bulk density followed by rural and peri-urban zones. The higher bulk density in urban zones is consistent with the intense anthropogenic activities and infrastructure development that led to soil compaction (Howard, 2021).

4.3.2 Particle density

Table 12 depicts the data related to soil particle density (PD) of different land use systems across the zones. The PD values ranged from 2.60 Mg m^{-3} in rainfed crop in urban zone to 2.69 Mg m^{-3} in waste lands under rural zone. However, the data on soil PD revealed no significant differences in land uses across the rural-urban gradient.

The data on soil particle density demonstrated minimal variation across the rural-urban gradient and among different land uses (Fig. 16). The results indicated that particle density, which primarily reflects the mineral composition of the soil, remains

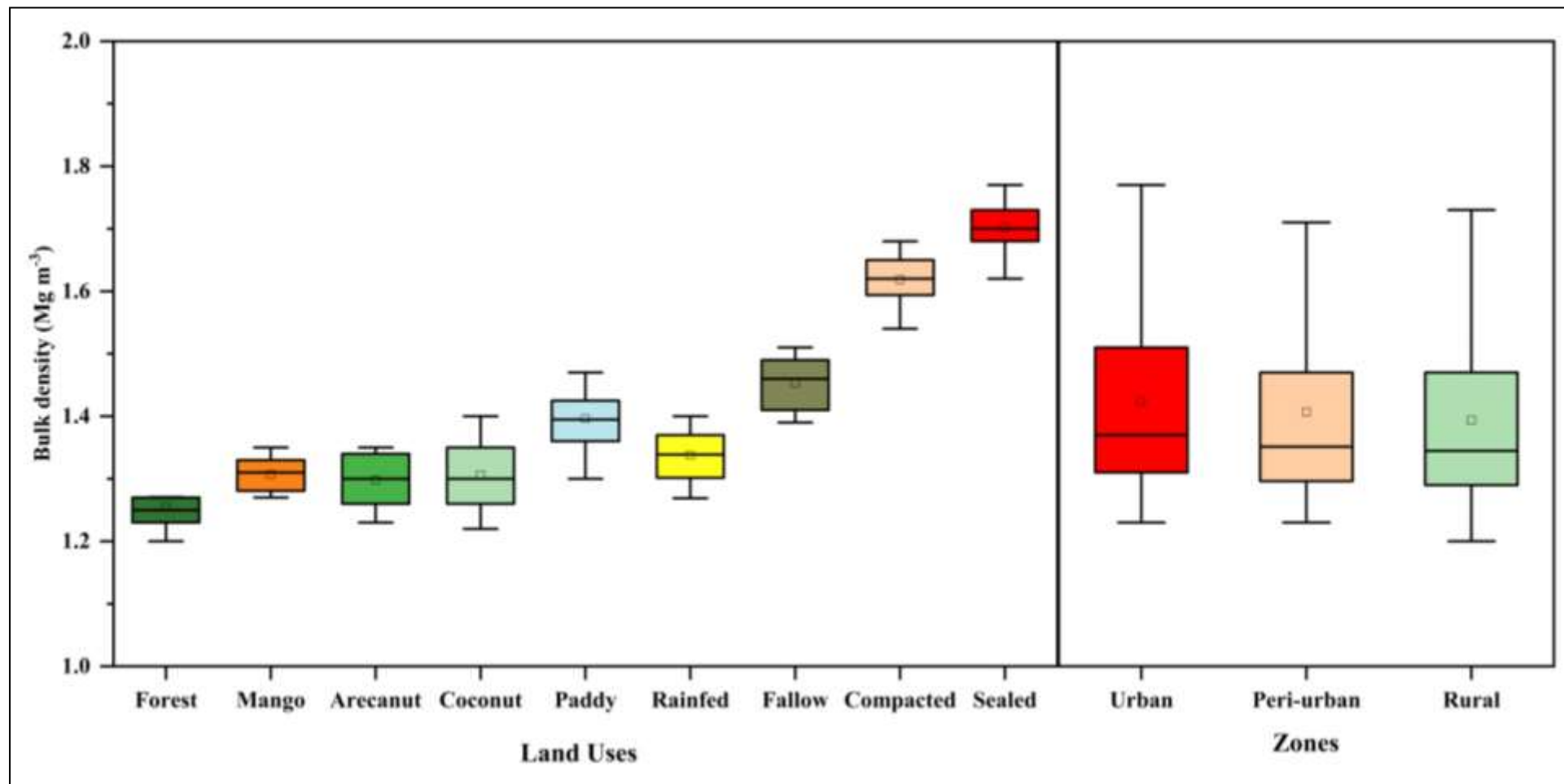


Fig. 15. Bulk density of soils (Mg m⁻³) among different land uses across the rural-urban gradient

Table 12. Particle density of soils (Mg m⁻³) among different land uses across the rural-urban gradient

Land uses (L)	Zones (Z)			Land uses (L) Mean
	Urban	Peri-urban	Rural	
Forest/forest plantations	2.61	2.62	2.62	2.62
Mango	2.62	2.61	2.63	2.62
Arecanut	2.63	2.62	2.65	2.63
Coconut	2.65	2.66	2.64	2.65
Paddy	2.61	2.61	2.62	2.61
Rainfed crop	2.60	2.61	2.63	2.61
Waste lands	2.65	2.65	2.69	2.66
Compacted area	2.63	2.65	2.62	2.63
Sealed area	2.63	2.68	2.65	2.65
Zones (Z) Mean	2.63	2.63	2.64	
Factors	S.Em±		CD @ 0.05	
Zones (Z)	0.02		NS	
Land uses (L)	0.03		NS	
Interaction (Z x L)	0.05		NS	

consistent across diverse land use systems, suggesting that anthropogenic activities and land-use changes in the studied gradient have limited impact on particle density (He *et al.*, 2023).

4.3.3 Porosity

Table 13 presents data on soil porosity across various land-use systems within three different zones. The porosity levels ranged from a low of 34.40 per cent in sealed areas of urban zone to a high of 53.11 per cent in forest/forest plantations of rural zone.

Among the different land-use types, forest/forest plantations exhibited the highest soil porosity (51.99 %), closely followed by coconut plantations (50.66 %). Within agricultural and horticultural land-use systems, mango (50.05 %), arecanut (50.59 %) and rainfed crop lands (48.77 %) displayed relatively high porosity. In contrast, paddy soils had a slightly lower porosity of 46.56 per cent. On the other end of the spectrum, sealed areas (35.75 %) and compacted zones (38.50 %) registered the lowest porosity levels. A comparative analysis of land-use systems across zones revealed that rural soils had the highest porosity (47.14 %), followed by peri-urban zone (46.47 %), while urban soils recorded the lowest porosity (45.70 %).

Among land use systems, higher porosity was observed in the soils of forest/forest plantations (Fig. 17). Forest ecosystems typically exhibit good soil structure and porosity which may be due to minimal disturbance and high organic matter content (Osman, 2013). Soils of arecanut plantations also demonstrated similar higher porosity values. The arecanut-growing systems often incorporate high amount of organic matter in the form of fronds and FYM. The agroforestry practices and maintenance of soil cover through mulching, which often contributes to higher organic carbon content and better soil structure (Acharya *et al.*, 2015).

Soils of sealed areas exhibited least porosity. Urbanization often leads to the sealing of soils through the construction of impervious surfaces such as roads, pavements and buildings. This process compresses the soil and reduces air-filled and water-filled pores, leading to compacted and structurally degraded soils (Nagaraja *et al.*, 2016; Wang *et al.*, 2018). Similarly, compacted soils also showed lower porosity values reflecting their destroyed structure. Among three zones, rural zone in general exhibited higher porosity and it followed the order rural > peri-urban > urban. Higher porosity in rural zones may be attributed to extensive vegetation cover, lesser anthropogenic disturbance and which collectively enhance soil structure and porosity (Herrmann *et al.*, 2020).

4.3.4 pH

The soil pH ranged from 5.36 in paddy fields of rural zones to 7.12 in sealed areas under urban zones (Table 14). The soils in the study area were found to be in slightly acidic to neutral range.

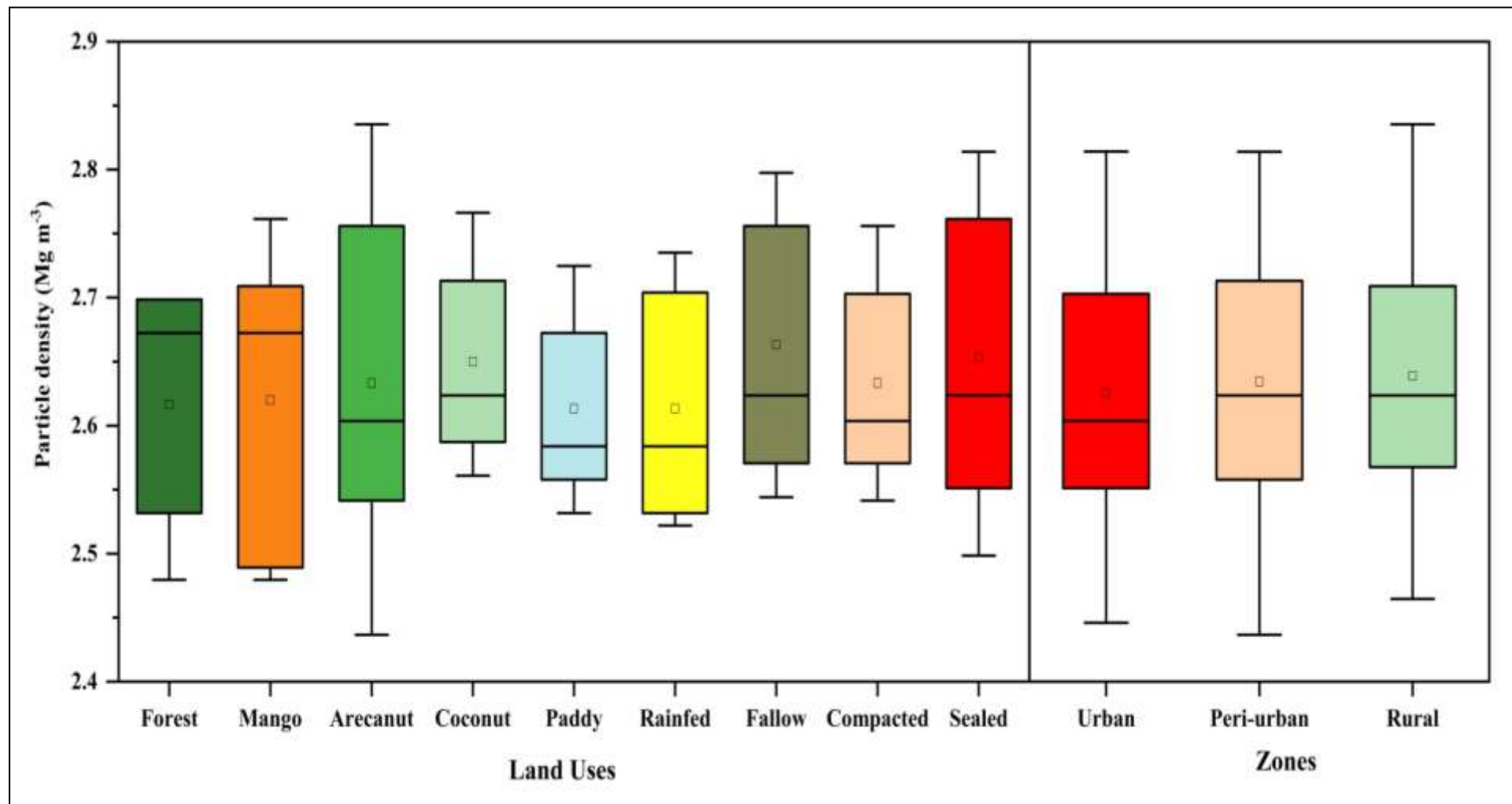


Fig. 16. Particle density of soils (Mg m⁻³) among different land uses across the rural-urban gradient

Table 13. Porosity of soils (%) among different land uses across the rural-urban gradient

Land uses (L)	Zones (Z)			Land uses (L) Mean
	Urban	Peri-urban	Rural	
Forest/forest plantations	50.49	52.38	53.11	51.99
Mango	49.98	49.53	50.65	50.05
Arecanut	51.49	50.27	49.99	50.59
Coconut	49.44	50.92	51.61	50.66
Paddy	46.06	46.31	47.32	46.56
Rainfed crop	47.83	49.01	49.47	48.77
Waste lands	44.29	45.28	46.60	45.39
Compacted area	37.34	39.08	39.07	38.50
Sealed area	34.40	36.37	36.47	35.75
Zones (Z) Mean	45.70	46.57	47.14	
Factors	S.Em±		CD @ 0.05	
Zones (Z)	0.37		1.05	
Land uses (L)	0.65		1.81	
Interaction (Z x L)	1.12		NS	

Table 14. Soil pH among different land uses across the rural-urban gradient

Land uses (L)	Zones (Z)			Land uses (L) Mean
	Urban	Peri-urban	Rural	
Forest/forest plantations	6.52	5.80	5.66	5.99
Mango	6.18	6.06	5.88	6.03
Arecanut	6.05	6.08	5.92	6.01
Coconut	6.15	5.82	5.92	5.95
Paddy	5.48	5.41	5.36	5.41
Rainfed crop	5.68	5.78	5.46	5.64
Waste lands	6.15	6.08	5.93	6.05
Compacted area	6.89	6.42	6.27	6.51
Sealed area	7.12	7.02	6.78	6.96
Zones (Z) Mean	6.23	6.05	5.91	
Factors	S.Em±		CD @ 0.05	
Zones (Z)	0.07		0.19	
Land uses (L)	0.07		0.21	
Interaction (Z x L)	0.11		NS	

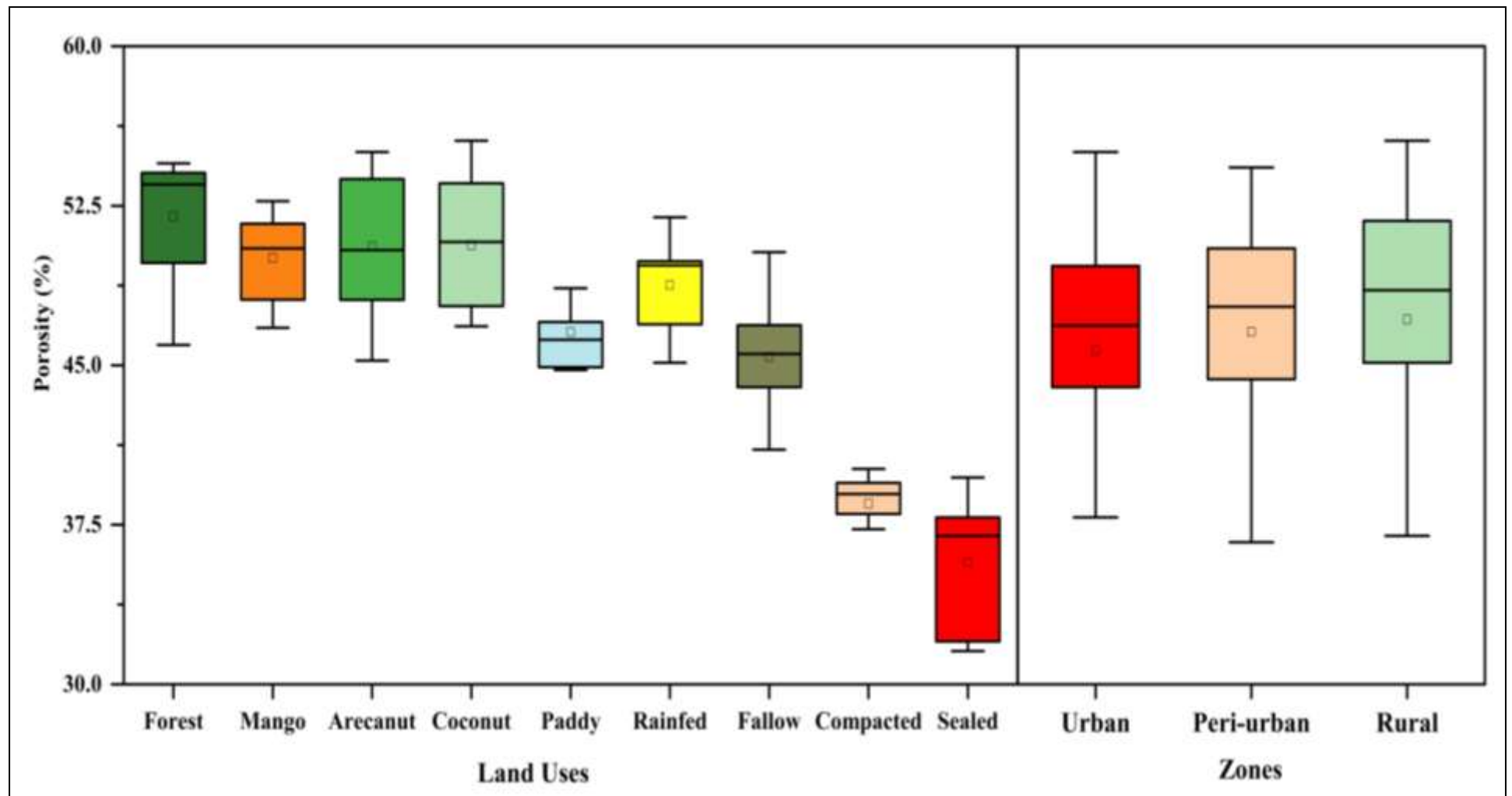


Fig. 17. Porosity of soils (%) among different land uses across the rural-urban gradient

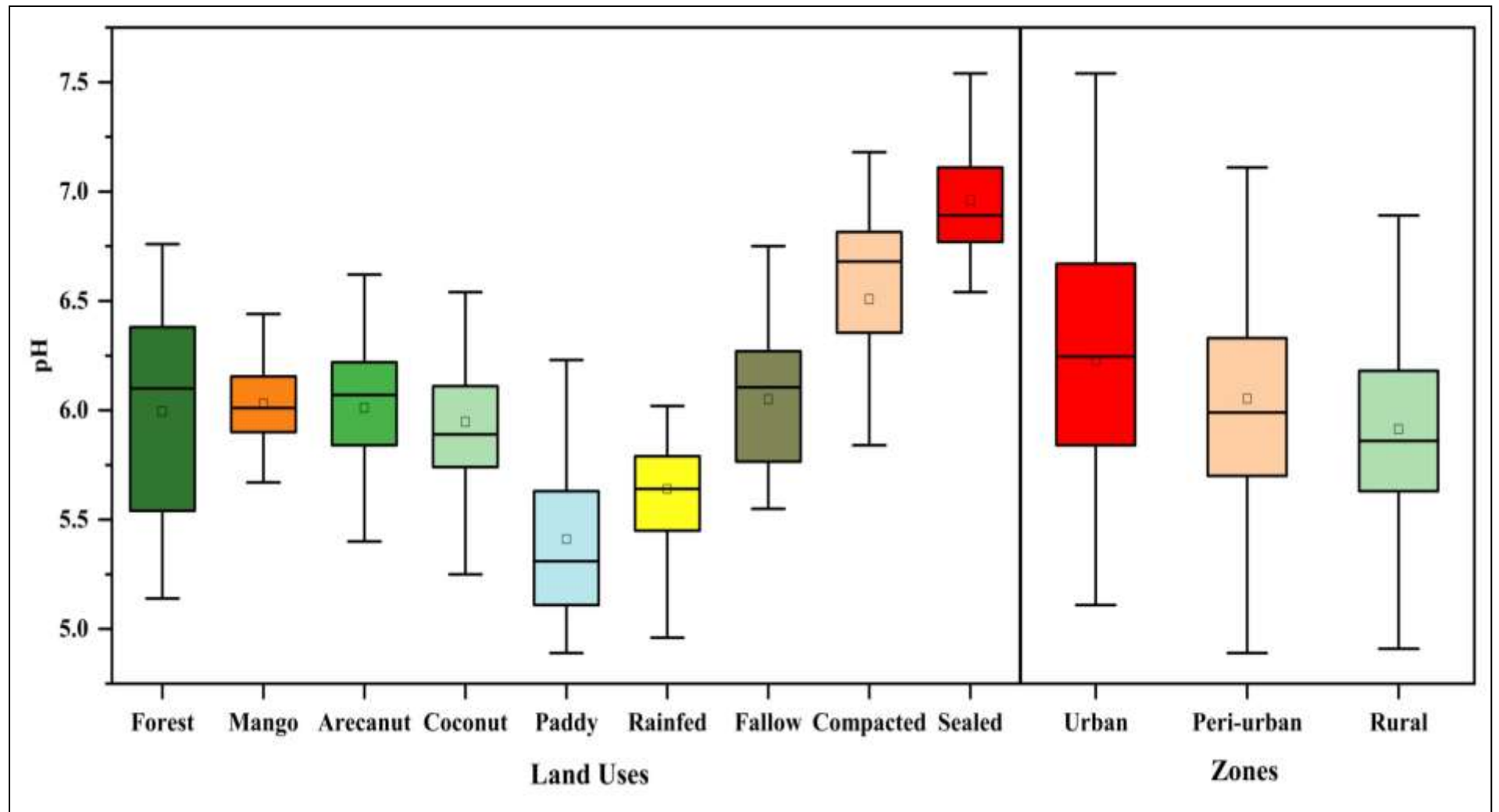


Fig. 18. Soil pH among different land uses across the rural-urban gradient

Among different land use systems, the paddy soils were found relatively more acidic (5.41) followed by rainfed agricultural systems (5.64). The soils of sealed areas were found to be neutral with pH ranging from 6.78 to 7.12 followed by compacted areas with pH ranging from 6.27 to 6.89. Comparison among land use systems across three zones indicated that the soils of urban areas recorded slightly higher pH compared to soils of other two zones. However, significantly higher pH was recorded in forest plantations, mango, coconut, waste lands, compacted areas and sealed areas of urban zones. However, such differences were not recorded in arecanut and paddy soils.

Neutral to slightly acidic pH of these soils (Fig. 18) may be attributed to acidic parent material comprising of granite and granite gneiss (da Silva *et al.*, 2022). Higher rainfall in the region which might have leached out bases and resulting in development of acidic soils (Dong *et al.*, 2022). The variations in soil pH among different land use systems may be due to the differences in management practices and biomass turnovers. Lower pH in paddy land use system may be due to submergence effect by impounding water (Wang *et al.*, 2013). Similar observations were also reported in paddy systems by Niranjana *et al.* (2018) and Shruthi *et al.* (2019). Among agricultural systems, higher pH was recorded in forest plantations, mango, arecanut, coconut and waste lands which may be due to lesser leaching losses (Choudhury *et al.*, 2021) and addition of cement dust (Agbede *et al.*, 2024). Addition of bases in the form of cement and lime in sealed areas might have increased the soil pH (Demina *et al.*, 2018). The variation in soil pH across the zones may be attributed to difference in degree of anthropogenic and urbanization activities across the zones (Mónok *et al.*, 2021). This aligns with findings from previous studies that have reported higher pH in urban zones compared to peri-urban and rural zones (Kuntoji *et al.*, 2021; O'Riordan *et al.*, 2021).

4.3.5 Electrical conductivity

The data regarding soil electrical conductivity (EC) for various land-use systems along a rural-urban gradient is presented in Table 15. The soil electrical conductivity ranged from 0.53 dS m⁻¹ in rainfed cropping system of peri-urban zones to 1.25 dS m⁻¹ in sealed areas under urban zones. The EC of soils in the study area was found to be non-saline.

Among different land use systems, the soils of sealed areas recorded higher EC values ranging from 1.03 to 1.25 dS m⁻¹, followed by compacted areas with EC values ranging from 0.78 to 0.86 dS m⁻¹ and the values were found to be significantly different. However, the rainfed crop (0.55 dS m⁻¹) and paddy land use systems (0.56 dS m⁻¹) recorded lower soil EC values and no significant differences among them. Significantly higher EC values in urban zones was recorded in forest/forest plantations, coconut, waste lands, compacted area and sealed areas. However, the EC values ranged from 0.53 to 0.73 dS m⁻¹ in mango, arecanut, paddy and rainfed crops. Comparison across

Table 15. Soil EC (dS m⁻¹) among different land uses across the rural-urban gradient

Land uses (L)	Zones (Z)			Land uses (L) Mean
	Urban	Peri-urban	Rural	
Forest/forest plantations	0.68	0.54	0.55	0.59
Mango	0.62	0.56	0.56	0.58
Arecanut	0.72	0.73	0.68	0.71
Coconut	0.62	0.58	0.55	0.58
Paddy	0.58	0.56	0.56	0.56
Rainfed crop	0.58	0.53	0.55	0.55
Waste lands	0.79	0.67	0.65	0.69
Compacted area	0.86	0.78	0.82	0.82
Sealed area	1.25	1.09	1.03	1.11
Zones (Z) Mean	0.74	0.68	0.66	
Factors	S.Em±		CD @ 0.05	
Zones (Z)	0.02		0.06	
Land uses (L)	0.01		0.03	
Interaction (Z x L)	0.01		0.03	

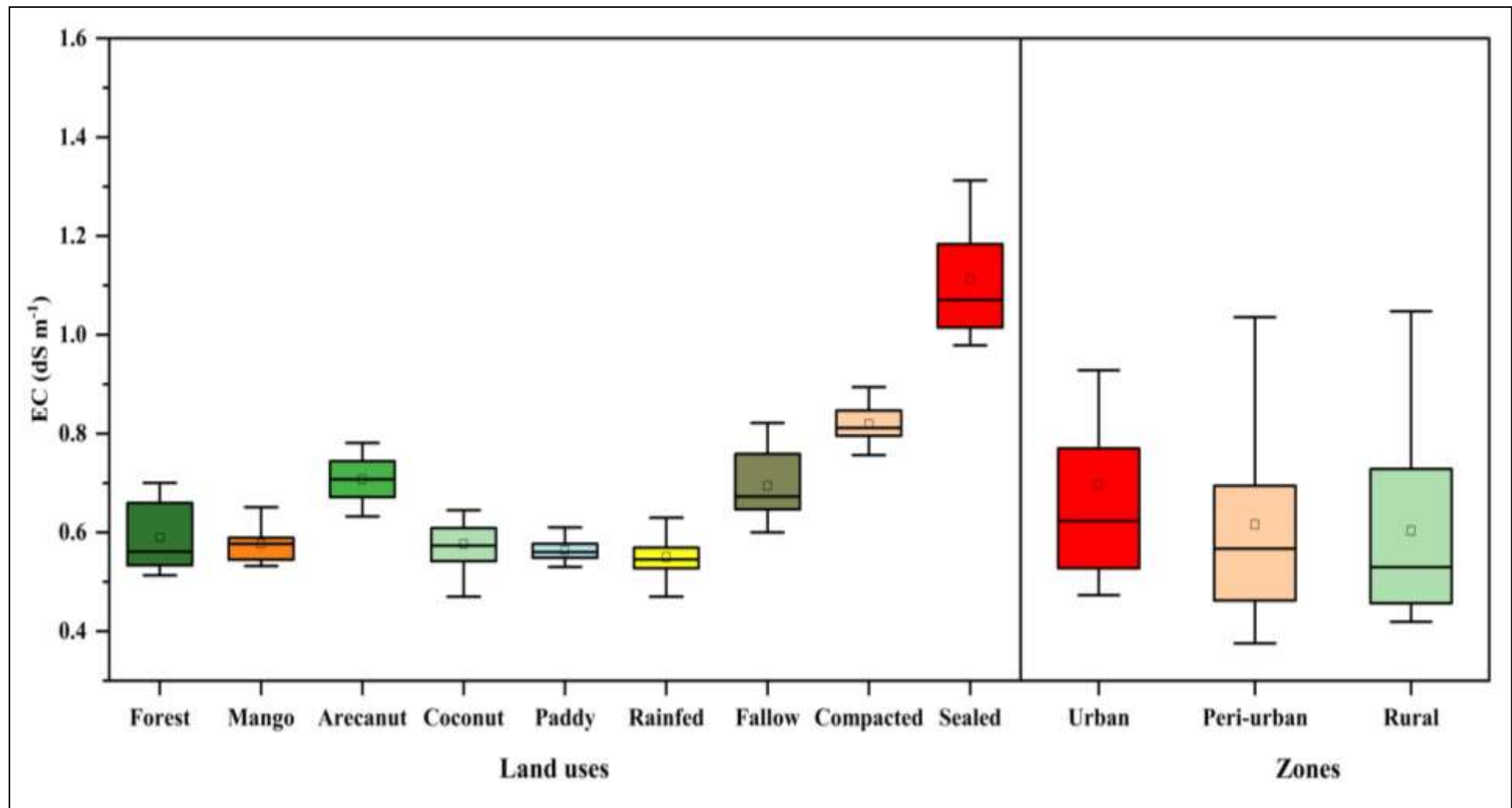


Fig. 19. Soil EC (dS m⁻¹) among different land uses across the rural-urban gradient

three zones showed that the soils of urban zone exhibited slightly higher EC compared to the soils of the rural zone and was found to be on par with peri-urban zone.

Lower conductivity values in the study area may be attributed to high rainfall (Corwin, 2021). The variations in EC among different land use systems (Fig. 19) could be attributed to the management practices *viz.*, use of ground water and application of fertilizers (Gowthamchand *et al.*, 2023). Anthropogenic addition of sewage and industrial effluents also might have contributed for higher EC (Peralta and Costa, 2013). The higher EC values in the urban zones could be attributed to excessive use of construction materials, industrial effluents and sewage (Kuntoji *et al.*, 2021). Among the land uses, significantly higher EC was recorded in sealed areas across all the three zones. This could be due attributed to the use of cement and asphalt for construction (O'Riordan *et al.*, 2021). This aligns with the findings of previous studies who have reported higher EC values in urban areas in comparison to rural areas (Kuntoji *et al.*, 2021; Tejashvini *et al.*, 2023).

4.3.6 Soil organic carbon

The soil organic carbon (SOC) content in the study area varied, with the lowest value recorded in sealed urban areas (2.58 g kg⁻¹) and the highest in forest plantations within the urban zone (6.68 g kg⁻¹) (Table 16). Overall, the soils exhibited SOC levels ranging from low to medium.

Among different land use systems, forest and forest plantations had the highest SOC (6.83 g kg⁻¹), followed by arecanut-based systems (6.41 g kg⁻¹). Within agricultural and horticultural-based land uses, mango orchards (5.99 g kg⁻¹), coconut plantations (5.17 g kg⁻¹), paddy fields (5.63 g kg⁻¹) and rainfed croplands (5.05 g kg⁻¹) maintained notable SOC levels. Conversely, sealed surfaces (2.72 g kg⁻¹) and compacted areas (3.20 g kg⁻¹) registered the lowest values. A comparison across zones showed that rural soils (5.17 g kg⁻¹) had slightly higher SOC than peri-urban (5.00 g kg⁻¹) and urban soils (4.56 g kg⁻¹).

The soil organic C content varied significantly across the land uses and zones (Fig. 20). These results may be due to the effects of biomass turnovers, land management practices and degree of urbanization (Xu *et al.*, 2024). Forest plantations had higher SOC levels which could be attributed to continuous addition of leaf litter, minimal soil disturbance and active microbial community (Bruun *et al.*, 2015). Arecanut plantations also recorded higher SOC levels, which may due to addition of higher amount of organic manures and incorporation of areca frond residues and litterfall (Vasundhara *et al.*, 2021). The variation in loss of soil through erosion may also be the reason for the varied levels of soil organic C. The sealed areas exhibited the lower soil organic C levels. The land sealing process is known to block organic matter

Table 16. Organic-C (g kg^{-1}) in soils of different land uses across the rural-urban gradient

Land uses (L)	Zones (Z)			Land uses (L) Mean
	Urban	Peri-urban	Rural	
Forest/forest plantations	6.73	6.87	6.87	6.83
Mango	5.80	6.09	6.04	5.99
Arecanut	6.68	6.18	6.36	6.41
Coconut	5.01	5.13	5.33	5.17
Paddy	5.49	5.53	5.83	5.63
Rainfed crop	4.80	5.07	5.23	5.05
Waste lands	4.03	4.34	4.41	4.28
Compacted area	2.87	3.29	3.40	3.20
Sealed area	2.58	2.71	2.83	2.72
Zones (Z) Mean	4.56	5.00	5.17	
Factors	S.Em$\pm$		CD @ 0.05	
Zones (Z)	0.21		0.59	
Land uses (L)	0.22		0.61	
Interaction (Z x L)	0.38		NS	

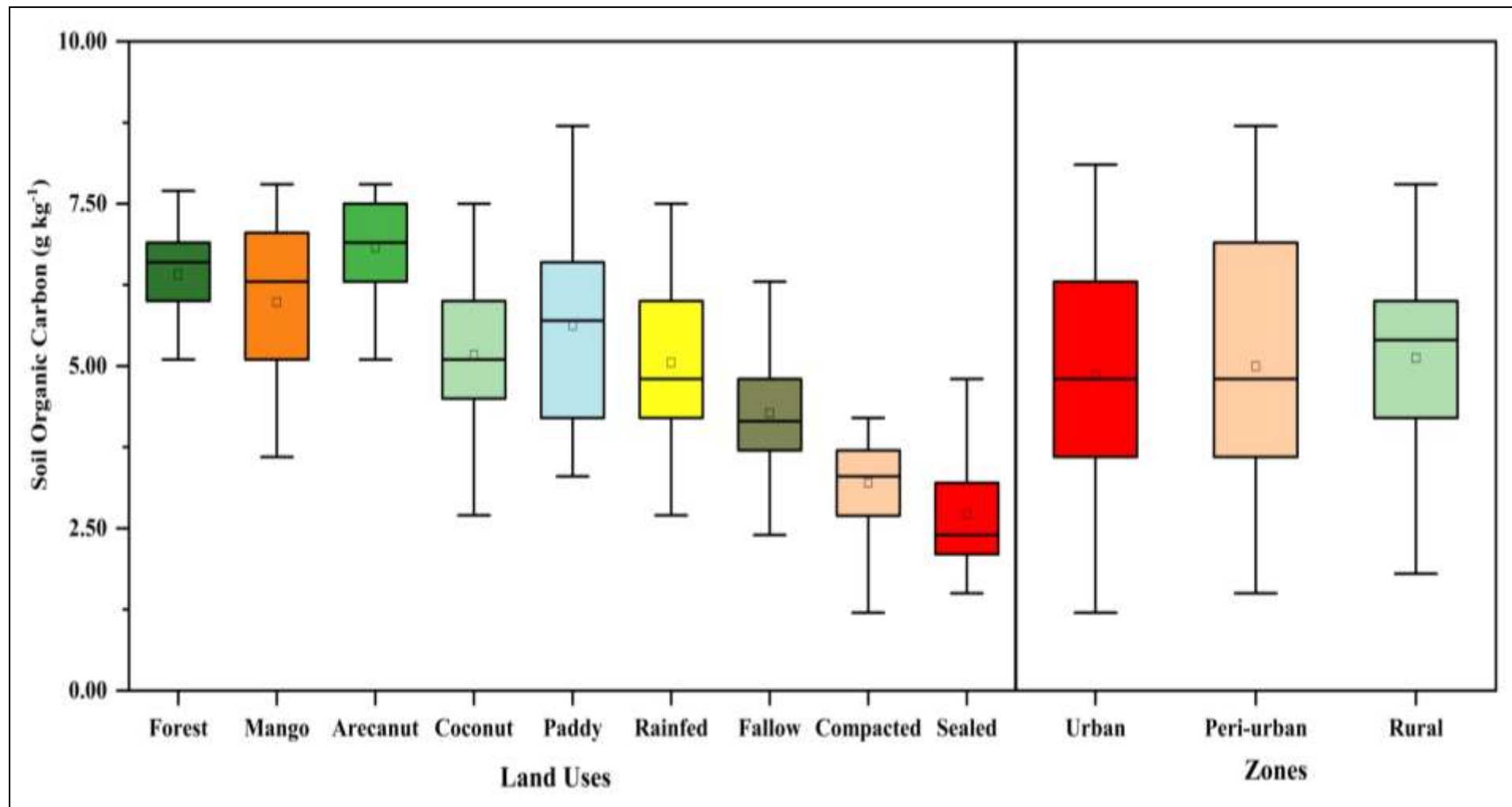


Fig. 20. Organic-C (g kg^{-1}) in soils of different land uses across the rural-urban gradient

Input (O'Riordan *et al.*, 2021). Similarly, compacted areas also recorded lower SOC levels. Compaction due to heavy machinery and foot traffic limits root growth and reduced organic matter incorporation also might have influenced SOC (Busse *et al.*, 2021).

In general, higher SOC was observed in rural zone compared to urban and peri-urban zone. This can be attributed to difference in biomass turnovers across three zones (Kuntoji *et al.*, 2021). Contrastingly, the urban zone is influenced by anthropogenic activities such as construction of buildings, roads etc are known to alter biomass turnovers and affects SOC levels (O'Riordan *et al.*, 2021). Peri-urban areas often represent transitional zones where mixed land uses and intermediate levels of soil disturbance result in moderate SOC levels (Seifollahi-Aghmiuni *et al.*, 2022).

4.3.7 Available nitrogen

The data on soil available nitrogen (N) for various land-use systems along the rural-urban gradient is given in Table 17. The soil available N ranged from 75.20 kg ha⁻¹ in sealed areas of urban zone to 266.56 kg ha⁻¹ in forest/forest plantations under urban zone. The available N status in the study area found to be lower in soil available N.

Among different land use systems, the soils of forest/forest plantations (258.72 kg ha⁻¹) and paddy soils (252.22 kg ha⁻¹) recorded higher available N. Among agricultural-based land use systems, mango (237.16 kg ha⁻¹), arecanut (242.14 kg ha⁻¹), coconut (230.72 kg ha⁻¹) and rainfed crop (229.38 kg ha⁻¹) exhibited notable amount of soil available N. In contrast, the soils of sealed areas (82.48 kg ha⁻¹) and compacted areas (86.44 kg ha⁻¹) recorded lower soil available N. Comparison of three zones indicated that the available N was 187.80 kg ha⁻¹, 191.39 kg ha⁻¹ and 194.68 kg ha⁻¹ for urban, peri-urban and rural zone, respectively. However, there was no significant differences in the soil available N.

The results revealed significant variations in soil available N across different land use systems and zones (Fig. 21), emphasizing the influence of land use practices on soil nutrient dynamics (Xu *et al.*, 2024). This aligns with findings from previous studies that have reported declining soil fertility in intensively managed landscapes (Kuntoji *et al.*, 2021; Tejashvini *et al.*, 2023). Soils under forest/forest plantations recorded significantly higher available N levels by continuous organic matter inputs through litterfall and root exudates (Xu *et al.*, 2024). Among, agricultural systems, paddy soils exhibited higher soil available N. This could be due to the external addition of N fertilizers and manures (Hu *et al.*, 2021).

Sealed areas exhibited the lowest soil available N levels. The reduced availability may be attributed to surface sealing, which restricts organic matter inputs,

Table 17. Available-N (kg ha⁻¹) in soils of different land uses across the rural-urban gradient

Land uses (L)	Zones (Z)			Land uses (L) Mean
	Urban	Peri-urban	Rural	
Forest/forest plantations	266.56	258.72	250.88	258.72
Mango	243.60	230.40	238.40	237.16
Arecanut	245.60	238.93	242.67	242.14
Coconut	222.40	231.47	236.44	230.72
Paddy	242.40	254.49	257.60	252.22
Rainfed crop	220.80	228.98	236.44	229.38
Waste lands	94.63	103.20	103.89	100.87
Compacted area	80.93	88.00	89.61	86.44
Sealed area	75.20	84.36	87.38	82.88
Zones (Z) Mean	187.80	191.39	194.68	
Factors	S.Em±		CD @ 0.05	
Zones (Z)	9.23		NS	
Land uses (L)	4.44		12.38	
Interaction (Z x L)	7.33		NS	

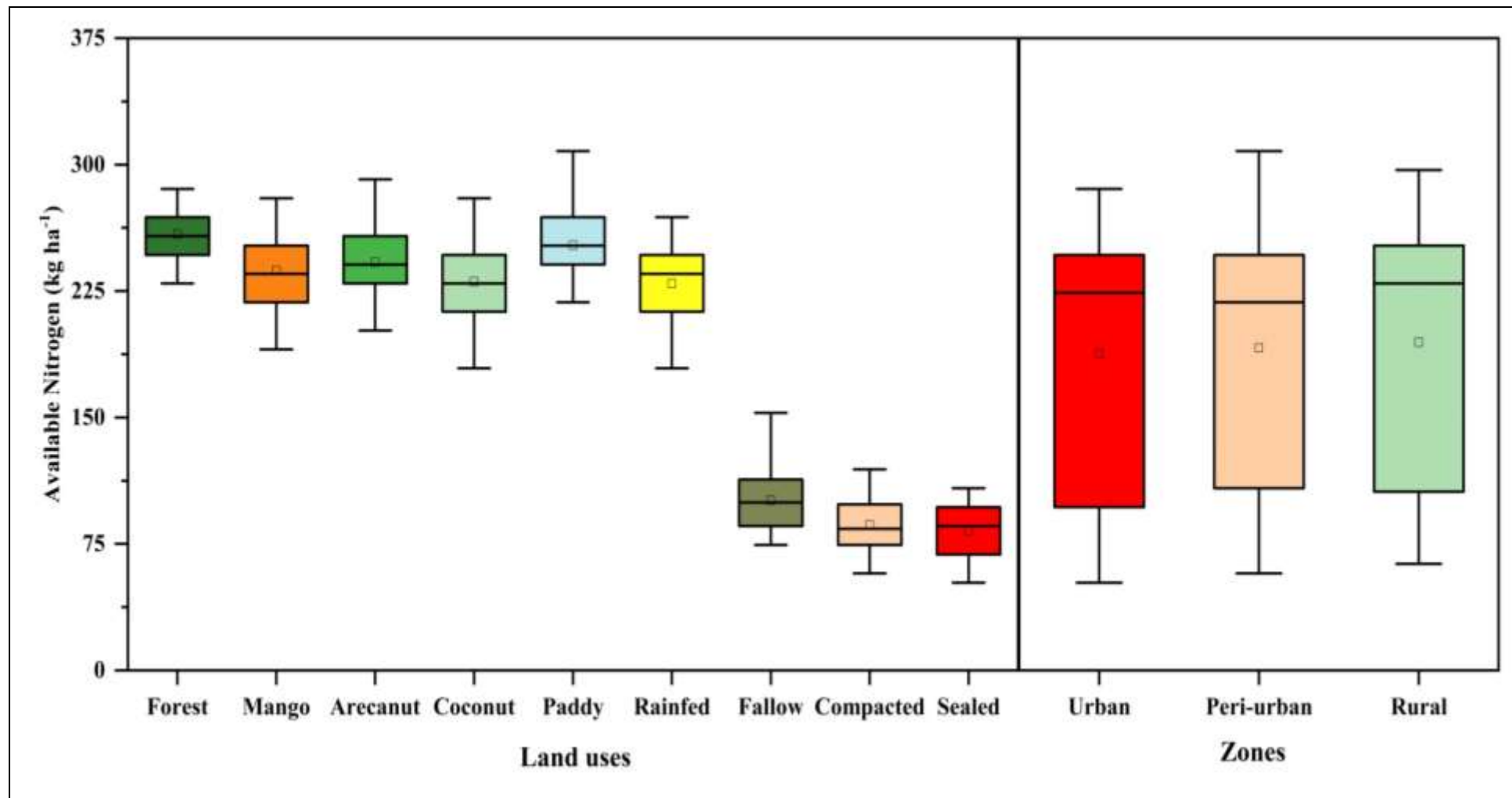


Fig. 21. Available-N (kg ha⁻¹) in soils of different land uses across the rural-urban gradient

microbial activity and mineralization (Kuntoji *et al.*, 2021). This may be also attributed to the fact that exposed sub-surface soil layer as surface soil layer is removed for construction activities (Pouyat *et al.*, 2007). Similarly, compacted areas showed lower soil available N levels, which is likely due to reduced biomass recycling and less mineralization as affected by restricted aeration. (Busse *et al.*, 2021). Rural zones exhibited the highest soil available N, followed by peri-urban zones and urban zones. These trends reflect the cumulative impact of land use intensity, organic matter inputs and soil management practices across the gradient (Li *et al.*, 2013).

4.3.8 Available Phosphorus

Soil available phosphorus (P) varied significantly across the land use systems and zones (Table 18). The soil available P ranged from 11.30 kg ha⁻¹ in sealed area of rural zone to 64.75 kg ha⁻¹ in forest/forest plantations of urban zone. The soils in the study area were found to be in medium to higher range of available P.

Among various land use systems, arecanut plantations had the highest levels of available phosphorus (63.74 kg ha⁻¹), with paddy soils (62.75 kg ha⁻¹) and forest/forest plantations (54.65 kg ha⁻¹) following closely. Within agricultural and horticultural and use categories, mango orchards (51.26 kg ha⁻¹), coconut plantations (49.98 kg ha⁻¹) and rainfed cropping areas (55.59 kg ha⁻¹) contained notable amounts of available phosphorus. In contrast, compacted (14.07 kg ha⁻¹) and sealed areas (12.64 kg ha⁻¹) exhibited significantly lower levels. A zonal comparison revealed that urban zone soils had the highest available phosphorus (46.06 kg ha⁻¹), while the levels in urban (43.85 kg ha⁻¹) and rural zones (44.67 kg ha⁻¹) were statistically similar.

Among land use systems, the forest/forest plantations exhibited significantly higher levels of available P (Fig. 22). This was in consistent with findings that forest soils often have elevated phosphorus levels due to substantial organic matter inputs from litterfall and reduced anthropogenic disturbances, promoting enhanced phosphorus mineralization (Xu *et al.*, 2024). Agricultural systems such as arecanut and paddy soils also recorded higher phosphorus availability. This could be attributed to regular phosphorus inputs through fertilizers, organic amendments and crop residues as observed in intensively managed agricultural lands (Hu *et al.*, 2021). Compacted areas recorded the lower availability of phosphorus. This could be attributed to reduced organic matter content and limited biological activity due to soil compaction, which restricts soil aeration and hence the phosphorus mineralization and microbial-mediated phosphorus cycling (Busse *et al.*, 2021). Similarly, sealed areas also exhibited reduced P availability resulting from impermeable surfaces that reduces water infiltration, organic matter decomposition and altered phosphorus dynamics (O'Riordan *et al.*, 2021).

Table 18. Available-P₂O₅ (kg ha⁻¹) in soils of different land uses across the rural-urban gradient

Land uses (L)	Zones (Z)			Land uses (L) Mean
	Urban	Peri-urban	Rural	
Forest/forest plantations	64.75	47.94	51.24	54.65
Mango	58.35	48.80	47.64	51.26
Arecanut	63.90	62.00	65.34	63.74
Coconut	47.87	50.14	51.45	49.98
Paddy	61.77	62.59	63.66	62.75
Rainfed crop	53.34	56.11	56.83	55.59
Waste lands	37.86	34.03	35.37	35.65
Compacted area	14.53	13.86	13.88	14.07
Sealed area	13.62	13.22	11.30	12.64
Zones (Z) Mean	46.06	43.85	44.67	
Factors	S.Em±		CD @ 0.05	
Zones (Z)	0.35		1.04	
Land uses (L)	1.14		3.19	
Interaction (Z x L)	1.65		4.61	

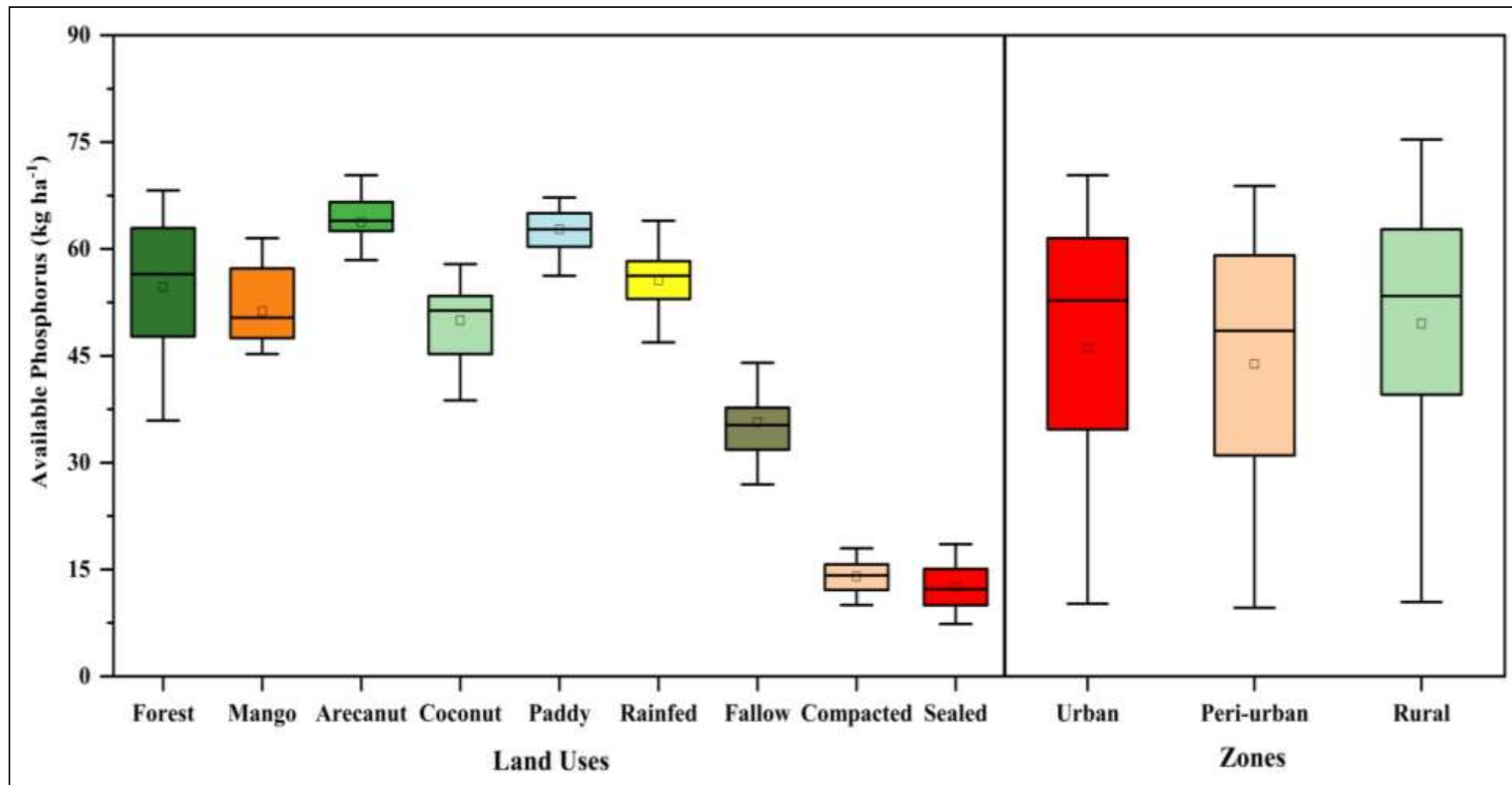


Fig. 22. Available-P₂O₅ (kg ha⁻¹) in soils of different land uses across the rural-urban gradient

Comparison across the three zones exhibited that soils in urban zones recorded the higher available P which was statistically on par with rural zones. The elevated phosphorus levels in urban zones could be due to intensified anthropogenic activities, fertilizer application and addition of organic waste (Li *et al.*, 2013). The soils of peri-urban zone exhibited significantly lower available P. Peri-urban zones characterized by both urban and rural influences, which may affect the P dynamics in peri-urban soils (Chaurasia *et al.*, 2024).

4.3.9 Available potassium

The available K varied from 45.74 kg ha⁻¹ in sealed areas of peri-urban zone to 249.50 kg ha⁻¹ in arecanut land use of urban zone (Table 19). The available K content of the soils in the study area was found to be in low to medium range.

Among different land uses systems, the arecanut growing soils recorded significantly higher amounts of available K (244.01 kg ha⁻¹) followed by paddy soils (224.65 kg ha⁻¹). Among the agricultural-based land use systems, mango (217.84 kg ha⁻¹), coconut (217.13 kg ha⁻¹) and rainfed crop (204.93 kg ha⁻¹) also recorded notable amount of soil available K. Substantially higher available K was also recorded in forest/forest plantations (176.58 kg ha⁻¹). The soils of sealed areas (46.32 kg ha⁻¹) and compacted areas (55.24 kg ha⁻¹) recorded lower available K. Comparison of land uses across the zones revealed no significant differences among the zones. However, the soils of compacted area exhibited significant variations in terms of available K across the zones.

Higher available K in arecanut and paddy land use systems could be due to application of K fertilizers (Kuntoji *et al.*, 2021). *In-situ* recycling of areca fronds and paddy stubble as practised by the farmers also might have contributed for higher available K (Paramesh *et al.*, 2019). Higher available K in soils of arecanut land use system may be due to the higher K application and recycling of K (Fig. 23). Similar observations have been reported by Sadasivuni *et al.* (2015) and Sujatha and Bhat (2016). The soils of sealed and compacted areas were low in available K as the nutrient rich surface soils have been removed leading to the exposure of less weathered sub-surface soils. Low available K in sub-surface layers have been reported by Kuntoji *et al.* (2021) and Tejashvini *et al.* (2023).

4.3.10 Exchangeable calcium

Table 20 depicts the data related to soil exchangeable calcium (Ca) of different land uses across the rural-urban gradient. The soil exchangeable Ca ranged from 3.69 cmol(p⁺) kg⁻¹ in waste lands of rural zone to 5.94 cmol(p⁺) kg⁻¹ in arecanut land use under rural zone.

Table 19. Available-K₂O (kg ha⁻¹) in soils of different land uses across the rural-urban gradient

Land uses (L)	Zones (Z)			Land uses (L) Mean
	Urban	Peri-urban	Rural	
Forest/forest plantations	180.31	176.69	172.73	176.58
Mango	214.56	217.71	220.78	217.84
Arecanut	249.50	243.07	240.68	244.01
Coconut	218.16	220.13	213.32	217.13
Paddy	225.50	222.88	225.76	224.65
Rainfed crop	201.15	207.35	205.45	204.93
Waste lands	72.57	74.08	71.72	72.80
Compacted area	53.66	54.95	56.89	55.24
Sealed area	45.74	46.81	46.29	46.32
Zones (Z) Mean	164.25	165.81	164.69	
Factors	S.Em±		CD @ 0.05	
Zones (Z)	9.64		NS	
Land uses (L)	2.81		7.85	
Interaction (Z x L)	4.92		NS	

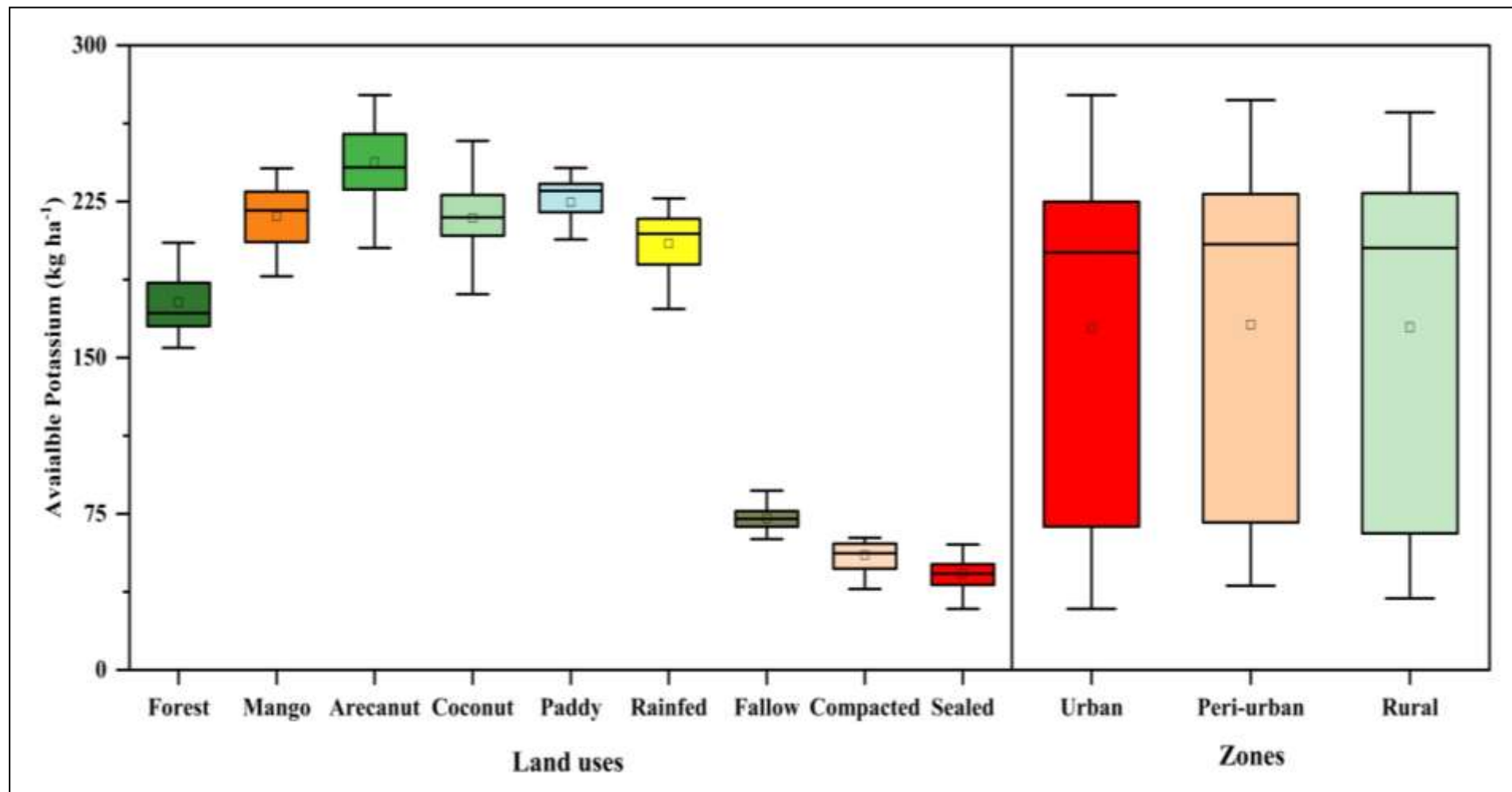


Fig. 23. Available-K₂O (kg ha⁻¹) in soils of different land uses across the rural-urban gradient

Table 20. Exchangeable-Ca [cmol(p⁺) kg⁻¹] in soils of different land uses across the rural-urban gradient

Land uses (L)	Zones (Z)			Land uses (L) Mean
	Urban	Peri-urban	Rural	
Forest/forest plantations	5.82	4.90	4.94	5.22
Mango	4.58	4.46	4.56	4.53
Arecanut	5.84	5.71	5.94	5.83
Coconut	4.80	4.39	4.69	4.61
Paddy	4.19	3.94	3.86	3.98
Rainfed crop	3.77	3.84	3.69	3.77
Waste lands	3.82	3.74	3.69	3.75
Compacted area	4.82	4.87	4.76	4.82
Sealed area	5.67	5.41	5.09	5.37
Zones (Z) Mean	4.90	4.59	4.58	
Factors	S.Em±		CD @ 0.05	
Zones (Z)	0.11		0.30	
Land uses (L)	0.11		0.31	
Interaction (Z x L)	0.18		NS	

Among different land use systems, the arecanut growing soils exhibited significantly higher soil exchangeable Ca [$5.83 \text{ cmol(p}^+) \text{ kg}^{-1}$] followed by the soils of sealed areas [$5.37 \text{ cmol(p}^+) \text{ kg}^{-1}$]. Among the agricultural-based land use systems, mango [$4.53 \text{ cmol(p}^+) \text{ kg}^{-1}$], coconut [$4.61 \text{ cmol(p}^+) \text{ kg}^{-1}$] and paddy [$3.98 \text{ cmol(p}^+) \text{ kg}^{-1}$] exhibited considerable amount of soil exchangeable Ca. However, soils of waste lands and rainfed crops recorded significantly lower amount of soil exchangeable Ca with values ranging from 3.69 to $3.82 \text{ cmol(p}^+) \text{ kg}^{-1}$ and 3.69 to $3.84 \text{ cmol(p}^+) \text{ kg}^{-1}$, respectively. Comparison among land use systems across the three zones indicated the significant variations in soil exchangeable Ca contents. Among the zones, soils of urban zone recorded notably higher amount of exchangeable Ca [$4.90 \text{ cmol(p}^+) \text{ kg}^{-1}$]. While, soils of peri-urban [$4.59 \text{ cmol(p}^+) \text{ kg}^{-1}$] and rural zones [$4.58 \text{ cmol(p}^+) \text{ kg}^{-1}$] were lesser and found to be on par with each other.

Among different land uses, arecanut growing soils exhibited highest exchangeable Ca levels (Fig. 24). This can be attributed to intensive management practices, including regular application of lime and fertilizers which contribute to Ca availability (Xu *et al.*, 2024). Soils in sealed areas also recorded relatively higher exchangeable Ca values. High Ca content in these soils may be attributed to the addition of Ca containing construction materials (Liu *et al.*, 2021). Lower Ca in waste lands and rainfed land use systems could be due to its continuous removal through biomass. Loss of surface soils by erosion in these sparsely vegetative lands might have caused Ca depletion (Smith *et al.*, 2016).

4.3.11 Exchangeable magnesium

The data in Table 21 illustrates the variation in soil exchangeable magnesium (Mg) across different land use systems in three zones. The soil exchangeable Mg varied from $1.30 \text{ cmol(p}^+) \text{ kg}^{-1}$ in waste lands of peri-urban zone to $2.44 \text{ cmol(p}^+) \text{ kg}^{-1}$ in forest/forest plantations of urban zone.

Among different land use systems, considerably higher amount of soil exchangeable Mg was found in arecanut growing soils [$2.17 \text{ cmol(p}^+) \text{ kg}^{-1}$] followed by forest/forest plantations [$1.79 \text{ cmol(p}^+) \text{ kg}^{-1}$]. Among the agricultural-based land use systems, mango [$1.56 \text{ cmol(p}^+) \text{ kg}^{-1}$], paddy [$1.48 \text{ cmol(p}^+) \text{ kg}^{-1}$] and rainfed crop [$1.47 \text{ cmol(p}^+) \text{ kg}^{-1}$] also recorded notable amounts of soil exchangeable Mg. The soils of waste lands and coconut land use system exhibited substantially lower levels of soil exchangeable Mg with the values ranging from 1.30 to $1.38 \text{ cmol(p}^+) \text{ kg}^{-1}$ and 1.30 to $1.61 \text{ cmol(p}^+) \text{ kg}^{-1}$, respectively. A comparison across the zones indicated that exchangeable Mg levels did not differ significantly among them.

Among various land use types, soils under arecanut cultivation exhibited the highest levels of exchangeable Mg (Fig. 25). This can be linked to intensive management practices, including the frequent application of dolomitic lime stone and secondary nutrient fertilizers, which enhance Mg availability (Xu *et al.*, 2024).

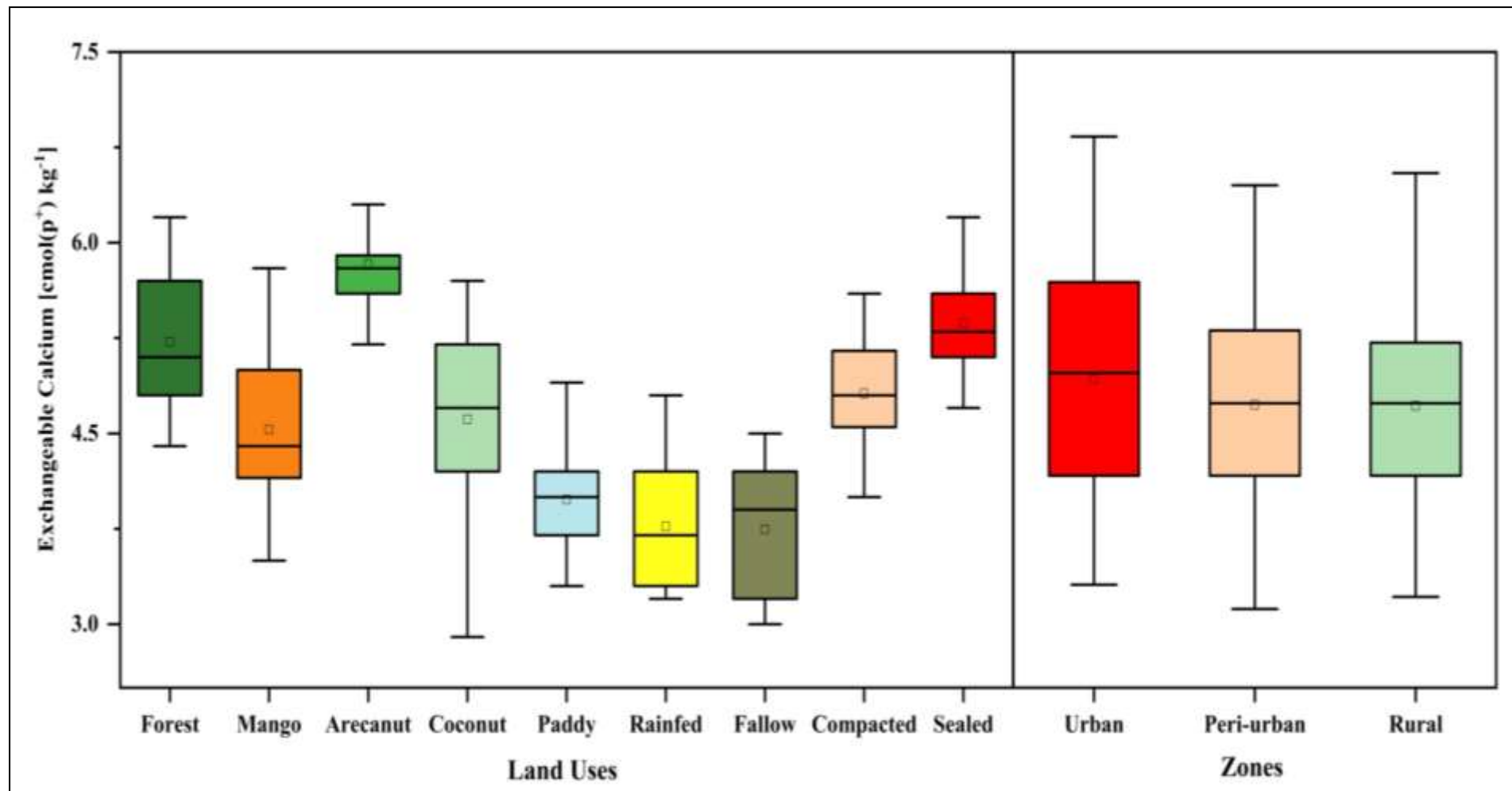


Fig. 24. Exchangeable-Ca [cmol(p⁺) kg⁻¹] in soils of different land uses across the rural-urban gradient

Table 21. Exchangeable-Mg [cmol(p⁺) kg⁻¹] in soils of different land uses across the rural-urban gradient

Land uses (L)	Zones (Z)			Land uses (L) Mean
	Urban	Peri-urban	Rural	
Forest/forest plantations	2.44	1.48	1.44	1.79
Mango	1.60	1.51	1.56	1.56
Arecanut	2.24	2.16	2.12	2.17
Coconut	1.61	1.49	1.30	1.46
Paddy	1.53	1.42	1.51	1.48
Rainfed crop	1.50	1.43	1.49	1.47
Waste lands	1.38	1.30	1.36	1.35
Compacted area	1.60	1.54	1.66	1.60
Sealed area	1.53	1.57	1.76	1.62
Zones (Z) Mean	1.70	1.56	1.59	
Factors	S.Em±		CD @ 0.05	
Zones (Z)	0.06		NS	
Land uses (L)	0.09		0.25	
Interaction (Z x L)	0.14		NS	

Similarly, soils in sealed areas showed relatively elevated Mg concentrations, likely due to the incorporation of lime-rich construction materials (Liu *et al.*, 2021). In contrast, waste lands and coconut plantations had lower exchangeable Mg levels, possibly due to continuous biomass removal and soil erosion in these sparsely vegetated areas, leading to significant Mg depletion (Smith *et al.*, 2016).

4.3.12 Available sulphur

The available sulphur (S) content in the study area varied from 6.82 mg kg⁻¹ in sealed areas of urban zone to a peak of 25.92 mg kg⁻¹ in mango orchards within rural zones (Table 22). Overall, the soil S availability ranged from low to high across different land use systems.

Among these systems, paddy soils exhibited the highest levels of available S (23.44 mg kg⁻¹), followed by arecanut plantations (22.67 mg kg⁻¹). Within agricultural land use types, mango (22.66 mg kg⁻¹), coconut (18.96 mg kg⁻¹) and rainfed crops (19.94 mg kg⁻¹) also maintained substantial S content. However, the soils of sealed and compacted areas had significantly lower S availability, with values ranging from 6.82 to 10.67 mg kg⁻¹ and 11.47 to 15.08 mg kg⁻¹, respectively. A comparison across different zones revealed that rural soils had the highest sulphur levels, averaging 19.82 mg kg⁻¹. Meanwhile, peri-urban (18.43 mg kg⁻¹) and urban soils (17.42 mg kg⁻¹) displayed similar S concentrations, with no significant statistical difference between them.

The results revealed a clear variation in soil available S across different land use systems and zones (Fig. 26), emphasizing the influence of land use practices on soil nutrient dynamics (Muluaem *et al.*, 2021). This aligns with findings from previous studies that have reported declining soil fertility in intensively managed landscapes (Kuntoji *et al.*, 2021; Tejashvini *et al.*, 2023). Among land use systems, soils under arecanut plantations recorded significantly higher available S levels by continuous organic matter inputs through litterfall and root exudates (Xu *et al.*, 2024). Paddy soils also exhibited higher soil available S which could be due to the external addition of S fertilizers and manures (Hu *et al.*, 2021). Sealed areas exhibited the lowest soil available S. This reduction can be attributed to surface sealing as it restricts organic matter inputs and microbial activity (Kuntoji *et al.*, 2021). This may be also attributed to that fact that the surface soil layer is removed during construction leaving behind sub-surface soil layer exposed (Pouyat *et al.*, 2007). Similarly, compacted areas showed lower soil available S levels which is likely due to restricted root penetration and reduced soil aeration (Busse *et al.*, 2021). Rural zones exhibited higher soil available S followed by peri-urban zones and urban zones. These trends reflect the cumulative impact of land use intensity, organic matter inputs and soil management practices across the gradient (Li *et al.*, 2013).

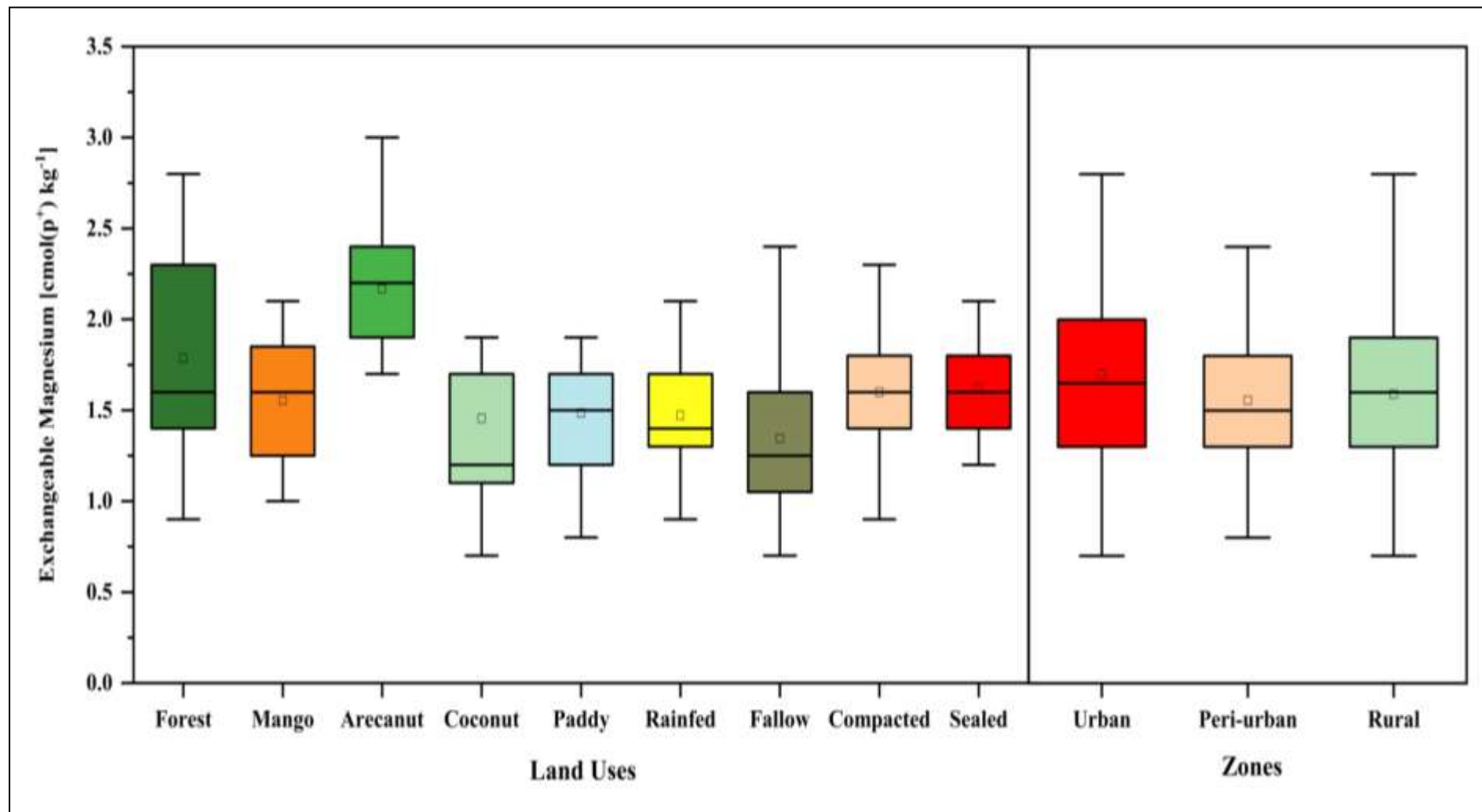


Fig. 25. Exchangeable-Mg [cmol(p⁺) kg⁻¹] in soils of different land uses across the rural-urban gradient

Table 22. Available-S (mg kg⁻¹) in soils of different land uses across the rural-urban gradient

Land uses (L)	Zones (Z)			Land uses (L) Mean
	Urban	Peri-urban	Rural	
Forest/forest plantations	23.47	22.72	21.41	22.53
Mango	20.19	21.50	25.92	22.66
Arecanut	21.28	22.01	24.41	22.67
Coconut	16.73	18.57	21.07	18.96
Paddy	22.67	23.78	23.71	23.44
Rainfed crop	18.38	21.32	19.79	19.94
Waste lands	13.72	14.88	16.50	15.17
Compacted area	11.47	13.29	15.08	13.47
Sealed area	6.82	8.47	10.76	8.83
Zones (Z) Mean	17.42	18.43	19.82	
Factors	S.Em±		CD @ 0.05	
Zones (Z)	0.67		1.86	
Land uses (L)	0.51		1.42	
Interaction (Z x L)	0.67		1.88	

4.3.13 DTPA extractable micronutrients

The data pertaining to DTPA extractable micronutrients of different land use systems across the zones are presented in Table 23 to 26. The results obtained with respect to DTPA extractable micronutrients and discussed collectively in this section.

The data on DTPA extractable iron (DTPA-Fe) of different land use systems across the zones are presented in Table 23. The DTPA-Fe ranged from 5.16 mg kg⁻¹ in sealed areas of peri-urban zone to 11.13 mg kg⁻¹ in forest/forest plantations under rural zone. The soils of the study area exhibited higher levels of DTPA-Fe. Among different land use systems, significantly higher amounts of DTPA-Fe were recorded in soils of forest/forest plantations (10.81 mg kg⁻¹) followed by the soils of mango (10.58 mg kg⁻¹). Among agricultural-based land use systems, arecanut (8.99 mg kg⁻¹), coconut (8.96 mg kg⁻¹), paddy soils (10.19 mg kg⁻¹) and rainfed crop (9.85 mg kg⁻¹) also exhibited significantly higher amount of DTPA-Fe. Contrastingly, the sealed area and compacted areas recorded significantly lower DTPA-Fe with values ranging from 5.16 to 5.62 mg kg⁻¹ and 8.80 to 10.13 mg kg⁻¹, respectively. Comparison among different land use systems across the three zones showed that the soils of rural zones had significantly higher DTPA-Fe (9.49 mg kg⁻¹) followed by peri-urban (9.22 mg kg⁻¹) and urban zone (9.02 mg kg⁻¹). However, the soils of urban and peri-urban zones were found to be on par with each other (Fig. 27).

Table 24 depicts the data related to DTPA extractable zinc (DTPA-Zn) of different land use systems across the three zones. The DTPA-Zn ranged from 0.33 mg kg⁻¹ in sealed areas of urban zone to 0.95 mg kg⁻¹ in forest plantations under rural zone. Among different land use systems, the soils of forest/forest plantations recorded significantly higher DTPA-Zn (0.93 mg kg⁻¹) followed by mango (0.80 mg kg⁻¹). Among agricultural-based land use systems, paddy (0.76 mg kg⁻¹) and rainfed crop (0.73 mg kg⁻¹) also exhibited substantially higher DTPA-Zn. The soils of the sealed areas (0.40 mg kg⁻¹) coconut (0.51 mg kg⁻¹) and arecanut land use systems (0.55 mg kg⁻¹) recorded significantly lower DTPA-Zn (Fig. 28). Comparison among the different land use systems exhibited that the higher DTPA-Zn was recorded in the order of rural (0.75 mg kg⁻¹), peri-urban (0.62 mg kg⁻¹) and urban zone (0.58 mg kg⁻¹).

The DTPA extractable Manganese (DTPA-Mn) ranged from 7.68 mg kg⁻¹ in sealed areas of rural zone to 23.76 mg kg⁻¹ in forest plantations under urban zone (Table 25). The DTPA-Mn of the soils in the study area was found to be higher than the critical range (Fig. 29). Among different land use systems, paddy soils recorded significantly higher amounts of DTPA-Mn (23.42 mg kg⁻¹) followed by forest plantations (23.33 mg kg⁻¹). Among the agricultural-based land use systems, mango (23.07 mg kg⁻¹), arecanut (23.28 mg kg⁻¹), coconut (22.92 mg kg⁻¹) and rainfed crop

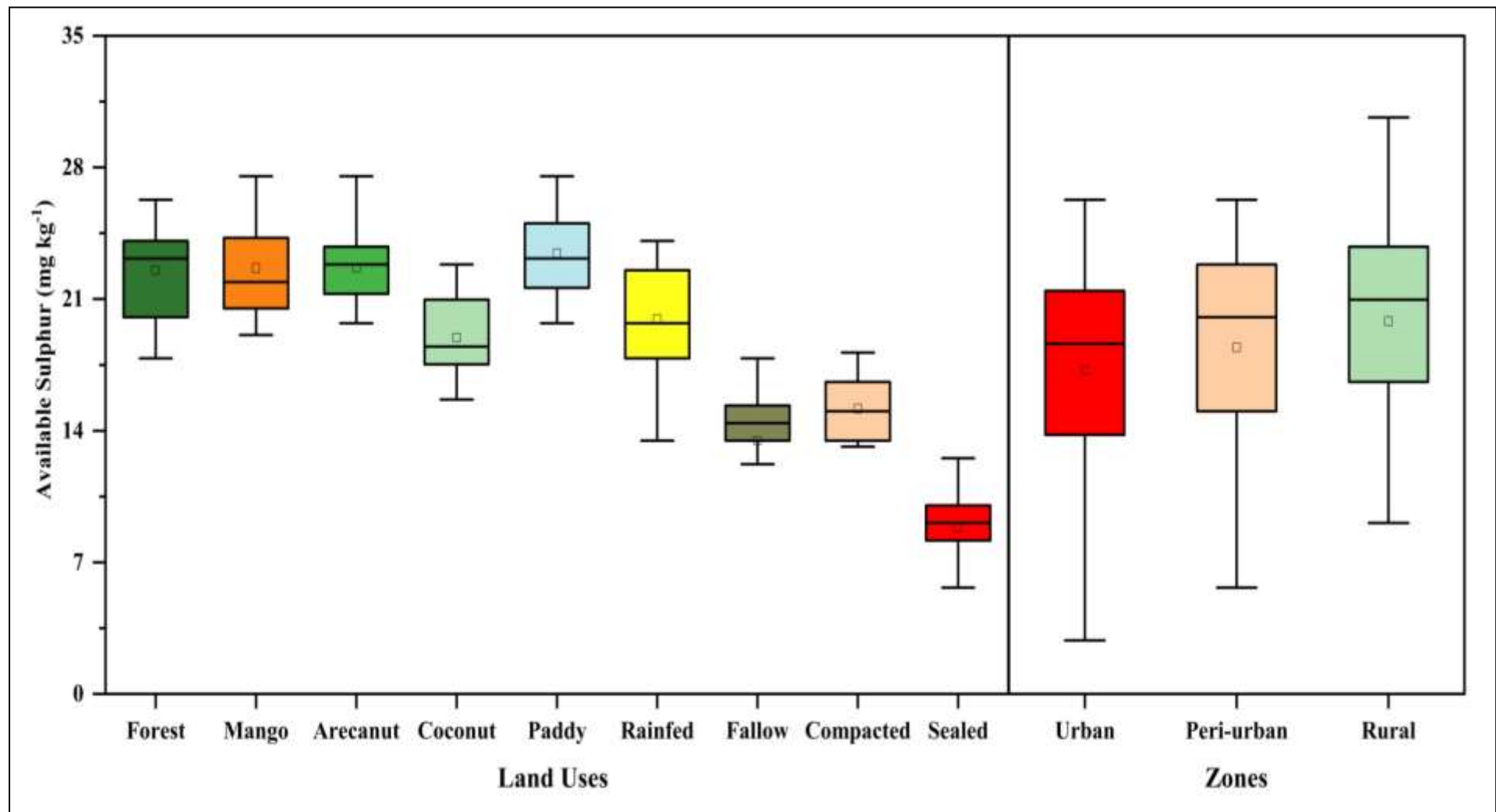


Fig. 26. Available-S (mg kg⁻¹) in soils of different land uses across the rural-urban gradient

Table 23. DTPA-Fe (mg kg⁻¹) in soils of different land uses across the rural-urban gradient

Land uses (L)	Zones (Z)			Land uses (L) Mean
	Urban	Peri-urban	Rural	
Forest/forest plantations	10.47	10.83	11.13	10.81
Mango	10.11	10.57	10.98	10.58
Arecanut	8.69	9.16	9.04	8.99
Coconut	8.72	9.11	8.99	8.96
Paddy	9.52	10.17	10.73	10.19
Rainfed crop	9.28	9.92	10.23	9.85
Waste lands	10.93	10.05	9.89	10.26
Compacted area	8.80	9.22	10.13	9.41
Sealed area	5.46	5.16	5.62	5.41
Zones (Z) Mean	9.02	9.22	9.49	
Factors	S.Em±		CD @ 0.05	
Zones (Z)	0.15		0.42	
Land uses (L)	0.14		0.40	
Interaction (Z x L)	0.22		0.60	

Table 24. DTPA-Zn (mg kg⁻¹) in soils of different land uses across the rural-urban gradient

Land uses (L)	Zones (Z)			Land uses (L) Mean
	Urban	Peri-urban	Rural	
Forest/forest plantations	0.91	0.94	0.95	0.93
Mango	0.73	0.75	0.92	0.80
Arecanut	0.48	0.51	0.70	0.55
Coconut	0.45	0.48	0.63	0.51
Paddy	0.69	0.74	0.88	0.76
Rainfed crop	0.66	0.69	0.85	0.73
Waste lands	0.63	0.68	0.72	0.67
Compacted area	0.56	0.59	0.59	0.58
Sealed area	0.33	0.37	0.54	0.40
Zones (Z) Mean	0.58	0.62	0.75	
Factors	S.Em±		CD @ 0.05	
Zones (Z)	0.02		0.07	
Land uses (L)	0.02		0.06	
Interaction (Z x L)	0.05		0.13	

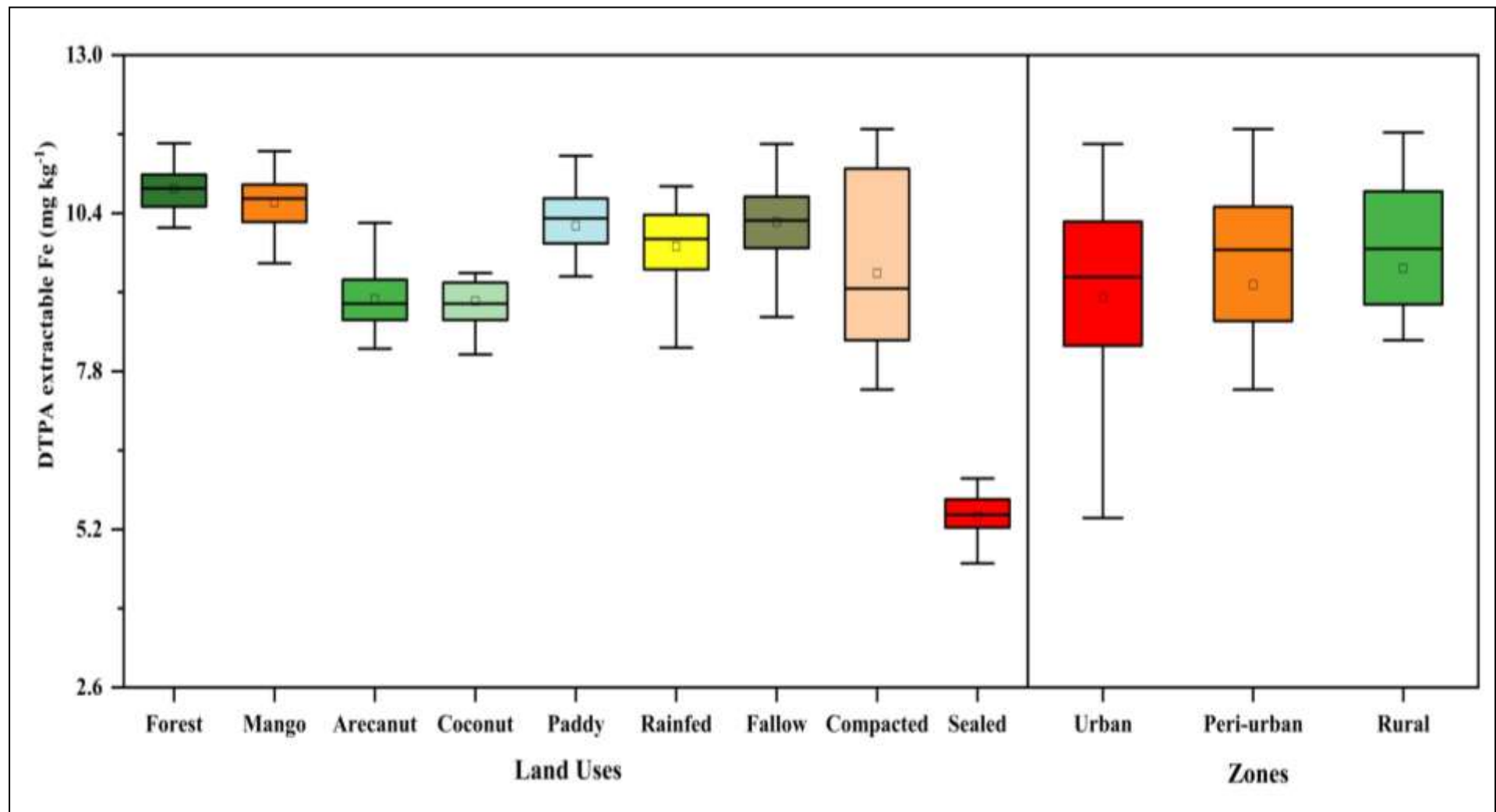


Fig. 27. DTPA-Fe (mg kg⁻¹) in soils of different land uses across the rural-urban gradient

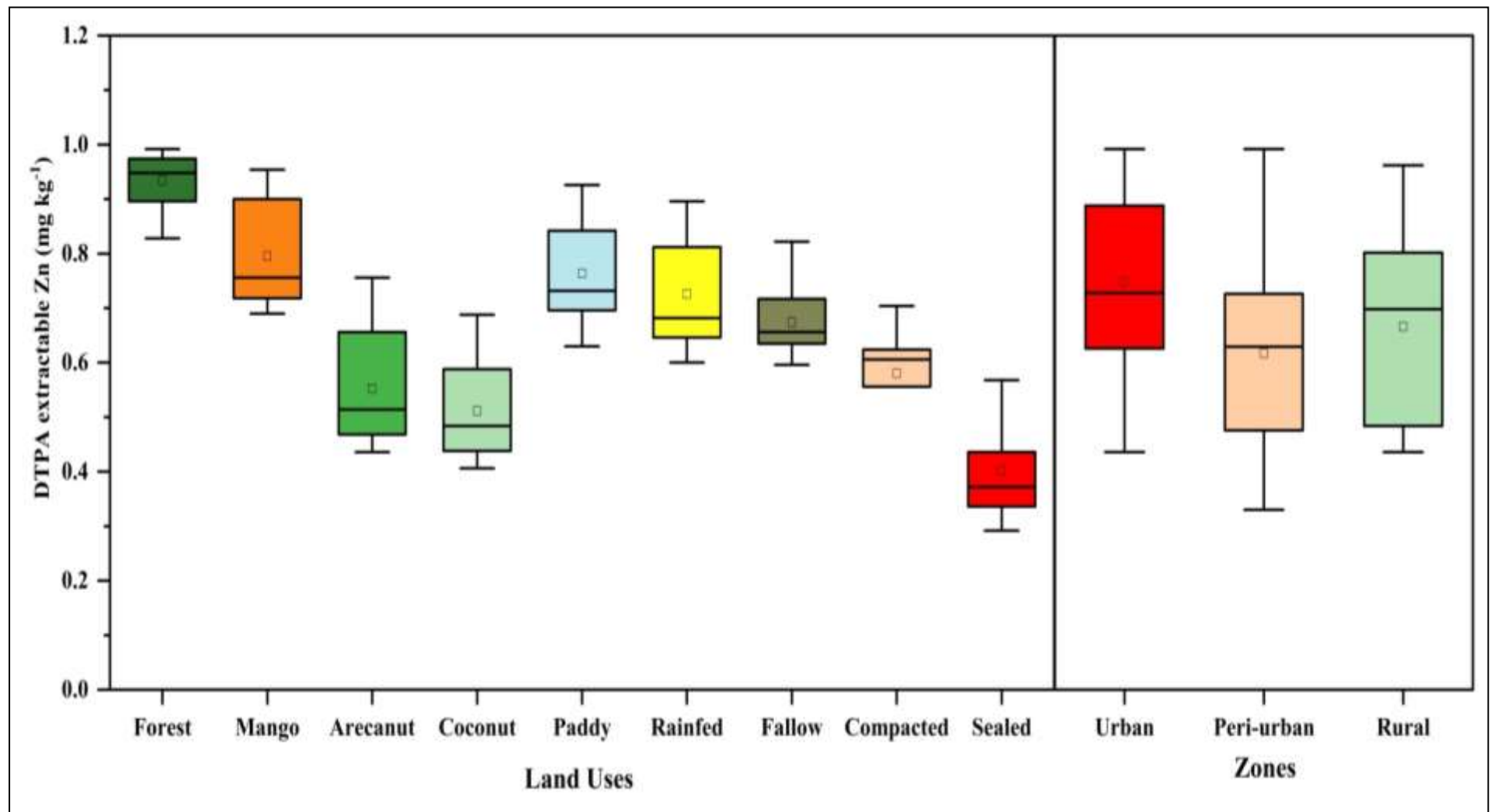


Fig. 28. DTPA-Zn (mg kg⁻¹) in soils of different land uses across the rural-urban gradient

Table 25. DTPA-Mn (mg kg⁻¹) in soils of different land uses across the rural-urban gradient

Land uses (L)	Zones (Z)			Land uses (L) Mean
	Urban	Peri-urban	Rural	
Forest/forest plantations	23.76	23.26	22.96	23.33
Mango	23.21	23.07	22.96	23.07
Arecanut	23.50	23.26	23.12	23.28
Coconut	23.20	22.96	22.66	22.92
Paddy	23.23	23.57	23.42	23.42
Rainfed crop	22.93	23.27	22.97	23.07
Waste lands	16.23	16.50	16.40	16.39
Compacted area	12.92	13.45	13.35	13.25
Sealed area	7.90	7.82	7.68	7.79
Zones (Z) Mean	19.63	19.65	19.47	
Factors	S.Em±		CD @ 0.05	
Zones (Z)	0.70		NS	
Land uses (L)	0.16		0.44	
Interaction (Z x L)	0.27		NS	

(23.07 mg kg⁻¹) also recorded higher amounts of DTPA-Mn. The sealed areas (7.79 g kg⁻¹) and compacted areas (13.25 mg kg⁻¹) were found to have significantly lower amounts of DTPA-Mn. Comparison of land uses across the three zones revealed that DTPA-Mn did not vary significantly across the zones.

The DTPA extractable Cu (DTPA-Cu) varied from 0.17 mg kg⁻¹ in sealed area of rural zone to 2.85 mg kg⁻¹ in paddy soils under rural zone (Table 26). The DTPA-Cu was ranged from low to high across the study area. Among different land uses, the paddy soils recorded with significantly higher amounts of DTPA-Cu (2.83 mg kg⁻¹) followed by the soils of rainfed crop (2.80 mg kg⁻¹). Among the agricultural-based land use systems, mango (2.06 mg kg⁻¹), arecanut (2.32 mg kg⁻¹) and coconut (2.28 mg kg⁻¹) also exhibited significantly higher amount of DTPA-Cu. The soils of forest/forest plantations also recorded notably higher amount of DTPA-Cu. The soils of sealed areas (0.20 mg kg⁻¹) and compacted areas (1.36 mg kg⁻¹) recorded significantly lower amounts of DTPA-Cu. Comparison among different land use systems across the zones indicated that the higher amount of DTPA-Cu was recorded in the order of rural (2.06 mg kg⁻¹), peri-urban (1.97 mg kg⁻¹) and urban zones (1.96 mg kg⁻¹). However, among the different land uses, mango, paddy, rainfed crop and compacted soils did not exhibit significant variations in DTPA-Cu across the zones (Fig. 30).

The variability in micronutrient availability *i.e.*, DTPA extractable Fe, Zn, Mn and Cu among different land use systems could be due to variation in management practices (Choudhury *et al.*, 2021). Complexation of micronutrients by organic acids produced during decomposition of organic matter in arecanut and forest/forest plantations might have resulted in their higher availability (Dhaliwal *et al.*, 2019). Lower soil pH may be the cause for their higher availability in mango and paddy land use systems (Wang *et al.*, 2022). Continuous inundation of paddy soils may also be contributed for higher micronutrient availability (Fageria *et al.*, 2011; Sahrawat, 2012). Lower micronutrient availability in sealed and compacted areas could be due to the lower organic matter content and reduced microbial activity (Charzyński *et al.*, 2017). The higher soil pH in these soils might have also influenced micronutrient availability in the above land use systems (Wang *et al.*, 2022). Rural zones recorded substantially higher concentrations of DTPA extractable micronutrients followed by peri-urban zones and urban zones. These trends reflect the cumulative impact of land use intensity, organic matter inputs and soil management practices across the gradient (Li *et al.*, 2013).

4.3.14 Soil carbon fractions

The data regarding to soil carbon fractions *viz.*, total organic carbon, total inorganic carbon and total carbon of different land use systems across the zones are presented in Table 27 to 29.

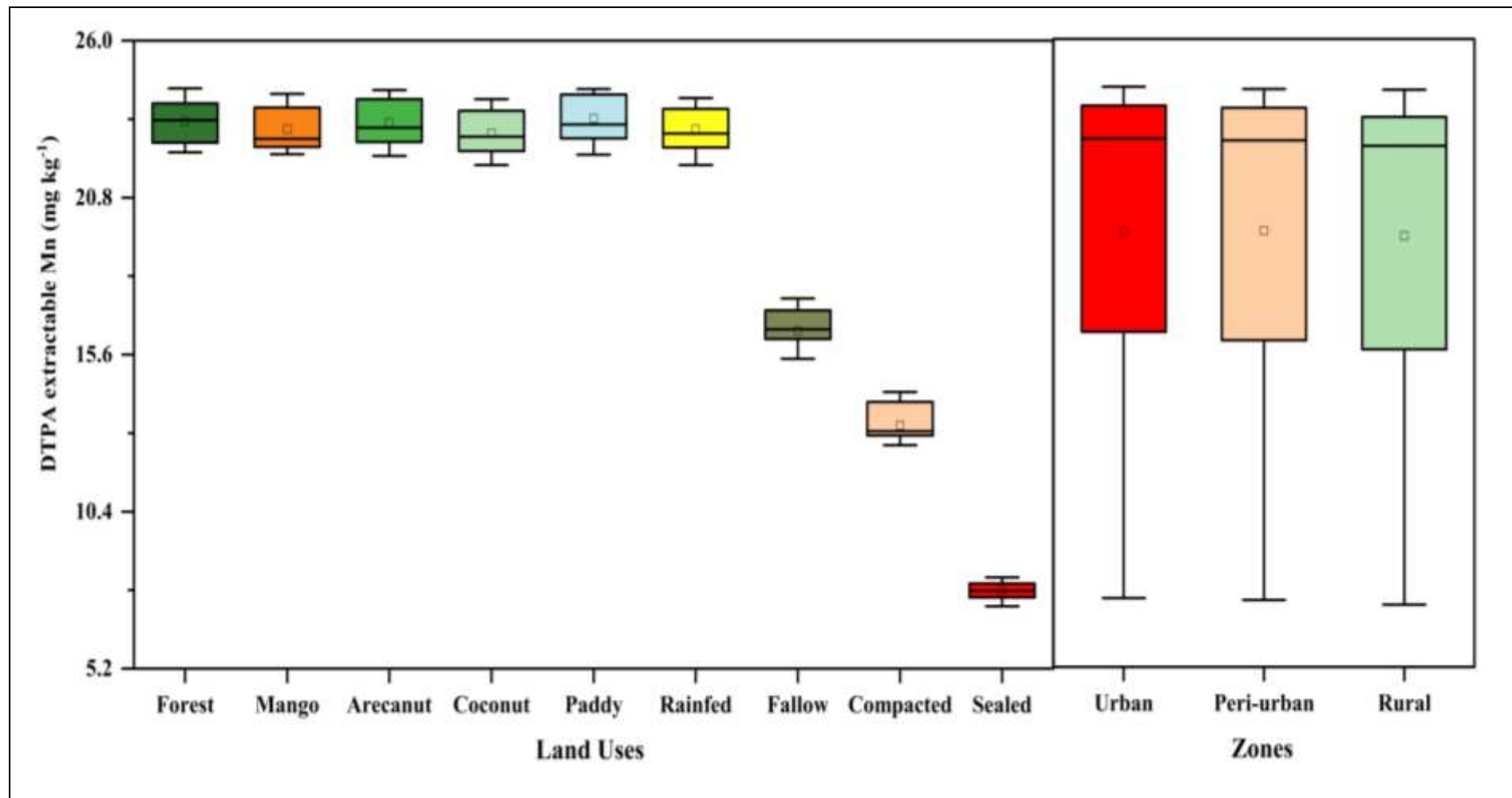


Fig. 29. DTPA-Mn (mg kg⁻¹) in soils of different land uses across the rural-urban gradient

Table 26. DTPA-Cu (mg kg⁻¹) in soils of different land uses across the rural-urban gradient

Land uses (L)	Zones (Z)			Land uses (L) Mean
	Urban	Peri-urban	Rural	
Forest/forest plantations	2.71	2.66	2.63	2.66
Mango	2.03	2.05	2.08	2.06
Arecanut	2.10	2.26	2.55	2.32
Coconut	2.05	2.23	2.53	2.28
Paddy	2.84	2.81	2.85	2.83
Rainfed crop	2.81	2.78	2.82	2.80
Waste lands	1.73	1.61	1.53	1.62
Compacted area	1.33	1.36	1.38	1.36
Sealed area	0.24	0.20	0.17	0.20
Zones (Z) Mean	1.97	1.96	2.06	
Factors	S.Em±		CD @ 0.05	
Zones (Z)	0.02		0.06	
Land uses (L)	0.03		0.09	
Interaction (Z x L)	0.05		0.13	

4.3.14.1 Total organic carbon

The data pertaining to total organic-C of different land use systems across the zones is presented in Table 27. The total organic-C values ranged from 2.76 g kg⁻¹ in sealed areas of urban zone to 7.35 g kg⁻¹ in forest/forest plantations of rural zone.

Among different land use systems, forest/forest plantations recorded significantly higher amounts of total organic-C (7.31 g kg⁻¹) followed by arecanut plantations (6.86 g kg⁻¹). Among the agricultural-based land use systems, mango (6.40 g kg⁻¹), coconut (5.53 g kg⁻¹), paddy (6.02 g kg⁻¹) and rainfed crop (5.41 g kg⁻¹) exhibited substantial amount of total organic-C. The soils of sealed areas (2.87 g kg⁻¹) and compacted areas (3.42 g kg⁻¹) recorded significantly lower values of total organic-C. Comparison among the different land use systems across the three zones revealed that there was no significant difference among the zones in terms of total organic-C.

The total organic C content varied significantly across land uses and zones (Fig. 31), indicating the effects of biomass turnovers, land management practices and degree of urbanization (Xu *et al.*, 2024). Forest plantations showed highest total organic-C levels which could be attributed to continuous addition of leaf litter, minimal soil disturbance and active microbial community (Bruun *et al.*, 2015). Arecanut plantations also recorded higher total organic-C levels, which may be due to addition of higher amount of organic manures and incorporation of crop residues and litterfall while reducing soil erosion (Nagaraja *et al.*, 2016; Vasundhara *et al.*, 2021). The sealed areas exhibited the lowest total organic-C levels. Sealing prevents organic matter input, reduces soil aeration and disrupts microbial activity, ultimately leading to reduced total organic-C accumulation (O'Riordan *et al.*, 2021). Similarly, compacted areas also recorded lower total organic-C levels. Compaction due to heavy machinery and foot traffic limits root growth and organic matter incorporation (Busse *et al.*, 2021). In general, higher total organic-C was observed in rural zone. This can be attributed to higher biomass turnovers in the rural zone compared to urban and peri-urban zone (Kuntoji *et al.*, 2021).

Contrastingly, the urban zone is influenced by anthropogenic activities such as construction, paving and industrialization, which affects total organic-C levels (O'Riordan *et al.*, 2021). Peri-urban areas often represent transitional zones where mixed land uses and intermediate levels of soil disturbance result in moderate total organic-C levels (Seifollahi-Aghmiuni *et al.*, 2022).

4.3.14.2 Total inorganic carbon

Table 28 presents data on total inorganic carbon (C) levels across various land use systems in three zones. The values ranged from a low of 0.17 g kg⁻¹ in rural wastelands to a peak of 0.34 g kg⁻¹ in urban forest and forest plantations.

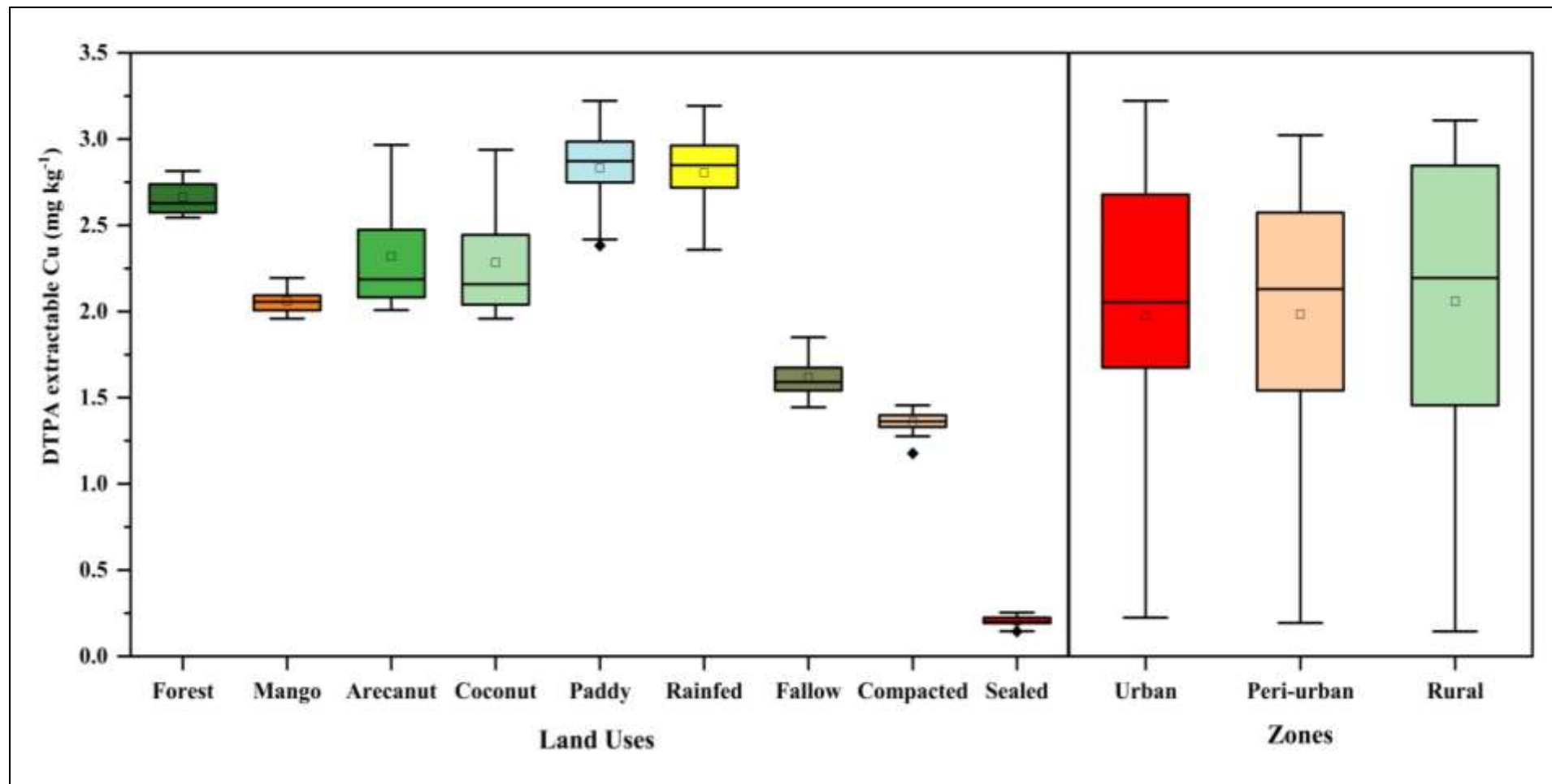


Fig. 30. DTPA-Cu (mg kg⁻¹) in soils of different land uses across the rural-urban gradient

Table 27. Total organic-C (g kg⁻¹) in soils of different land uses across the rural-urban gradient

Land uses (L)	Zones (Z)			Land uses (L) Mean
	Urban	Peri-urban	Rural	
Forest/forest plantations	7.20	7.35	7.35	7.31
Mango	6.21	6.51	6.47	6.40
Arecanut	7.15	6.61	6.81	6.86
Coconut	5.37	5.49	5.71	5.53
Paddy	5.87	5.92	6.24	6.02
Rainfed crop	5.14	5.42	5.60	5.41
Waste lands	4.32	4.65	4.72	4.57
Compacted area	3.07	3.52	3.64	3.42
Sealed area	2.76	2.90	2.92	2.87
Zones (Z) Mean	5.20	5.35	5.47	
Factors	S.Em±		CD @ 0.05	
Zones (Z)	0.22		NS	
Land uses (L)	0.23		0.65	
Interaction (Z x L)	0.40		NS	

Table 28. Total inorganic-C (g kg⁻¹) in soils of different land uses across the rural-urban gradient

Land uses (L)	Zones (Z)			Land uses (L) Mean
	Urban	Peri-urban	Rural	
Forest/forest plantations	0.34	0.33	0.25	0.31
Mango	0.31	0.32	0.26	0.30
Arecanut	0.36	0.35	0.32	0.34
Coconut	0.27	0.26	0.23	0.25
Paddy	0.29	0.28	0.25	0.27
Rainfed crop	0.26	0.26	0.22	0.25
Waste lands	0.21	0.19	0.17	0.19
Compacted area	0.25	0.23	0.21	0.23
Sealed area	0.31	0.29	0.27	0.29
Zones (Z) Mean	0.29	0.28	0.24	
Factors	S.Em±		CD @ 0.05	
Zones (Z)	0.01		0.02	
Land uses (L)	0.01		0.03	
Interaction (Z x L)	0.02		NS	

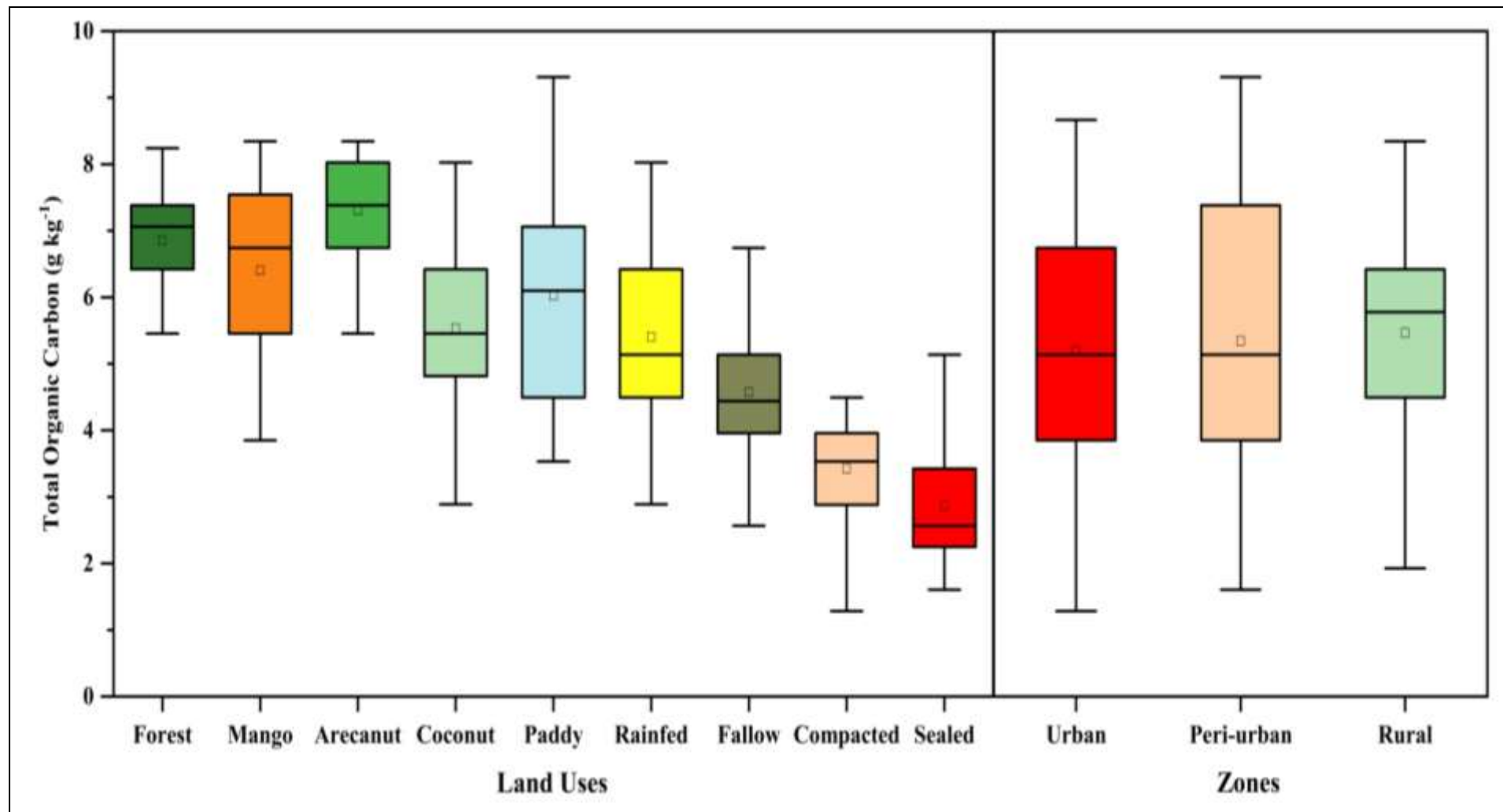


Fig. 31. Total organic-C (g kg^{-1}) in soils of different land uses across the rural-urban gradient

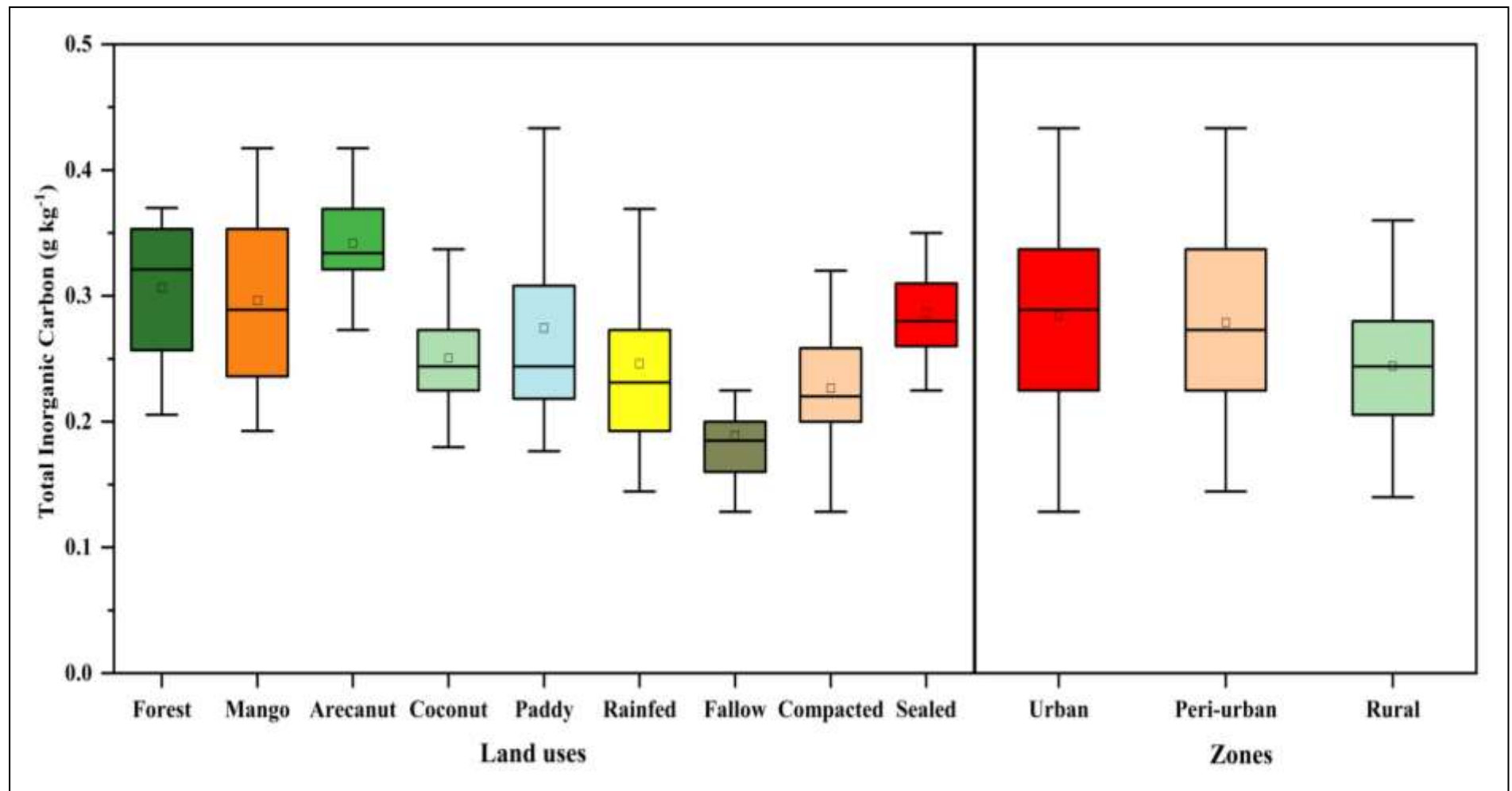


Fig. 32. Total inorganic-C (g kg⁻¹) in soils of different land uses across the rural-urban gradient

Among the different land use types, arecanut plantations exhibited the highest total inorganic-C content (0.34 g kg^{-1}), closely followed by forest and forest plantations (0.31 g kg^{-1}). Among agricultural and horticultural-based systems, mango orchards recorded 0.30 g kg^{-1} , while coconut and rainfed crops each held 0.25 g kg^{-1} of total inorganic-C. Paddy fields, with 0.27 g kg^{-1} , also showed notable levels. Additionally, sealed areas (0.29 g kg^{-1}) and compacted areas (0.23 g kg^{-1}) contained considerable amounts of total inorganic-C. Comparing zones, rural soils exhibited the lowest total inorganic-C content (0.24 g kg^{-1}), whereas urban soils recorded the highest (0.29 g kg^{-1}). Meanwhile, peri-urban and urban zones displayed similar levels, with no significant statistical difference between them.

Among different land uses, arecanut plantations exhibited the highest total inorganic-C levels (Fig. 32). This can be attributed to intensive management practices (Xu *et al.*, 2024). Forest plantations in urban zone recorded higher total inorganic-C values, which may be due to the deposition and distribution of anthropogenic materials in the urban zones (Chudecka *et al.*, 2024). Soils in sealed areas also recorded relatively higher exchangeable total inorganic-C values. Higher total inorganic-C content in these soils may be attributed to the addition of CaCO_3 containing construction materials (Liu *et al.*, 2013). The urban zones exhibited slightly higher total inorganic-C values and was on par with peri-urban zone. However, rural zone recorded significantly lower total inorganic-C values. This could be due to addition of anthropogenic substances (concrete and asphalt) and higher intensity of urbanization prevalent in urban zones (Wong *et al.*, 2022).

4.3.14.3 Total carbon

The data pertaining to total-C of different land use systems across the zones is presented in Table 29. The total-C values ranged from 3.06 g kg^{-1} in sealed areas of rural zone to 7.56 g kg^{-1} in forest/forest plantations under urban zone.

Among different land use systems, the soils of forest/forest plantations exhibited significantly higher values of total-C (7.65 g kg^{-1}) followed by the arecanut growing soils (7.16 g kg^{-1}). Among the agricultural-based land use systems, mango (6.70 g kg^{-1}), coconut (5.78 g kg^{-1}), paddy (6.30 g kg^{-1}) and rainfed crop (5.65 g kg^{-1}) exhibited substantial amounts of total-C. The soils of the sealed areas (3.15 g kg^{-1}) and compacted areas (3.65 g kg^{-1}) recorded significantly lower values of total-C. Comparison among the different land use system revealed that there was no significant difference among the zones in terms of total-C.

The higher total-C content in forest and arecanut plantations in rural zone could be attributed to the higher SOC content in respective land use systems in rural zones (Fig. 33). This aligns with the findings of previous studies who have reported higher total-C values in rural zones in comparison to urban zones (Nagaraja *et al.*, 2018;

Table 29. Total-C (g kg⁻¹) in soils of different land uses across the rural-urban gradient

Land uses (L)	Zones (Z)			Land uses (L) Mean
	Urban	Peri-urban	Rural	
Forest/forest plantations	7.56	7.70	7.66	7.65
Mango	6.52	6.83	6.72	6.70
Arecanut	7.49	6.94	7.05	7.16
Coconut	5.63	5.75	5.93	5.78
Paddy	6.16	6.20	6.49	6.30
Rainfed crop	5.39	5.68	5.82	5.65
Waste lands	4.52	4.84	4.90	4.76
Compacted area	3.32	3.74	3.84	3.65
Sealed area	3.06	3.19	3.19	3.15
Zones (Z) Mean	5.49	5.62	5.71	
Factors	S.Em±		CD @ 0.05	
Zones (Z)	0.23		NS	
Land uses (L)	0.24		0.67	
Interaction (Z x L)	0.41		NS	

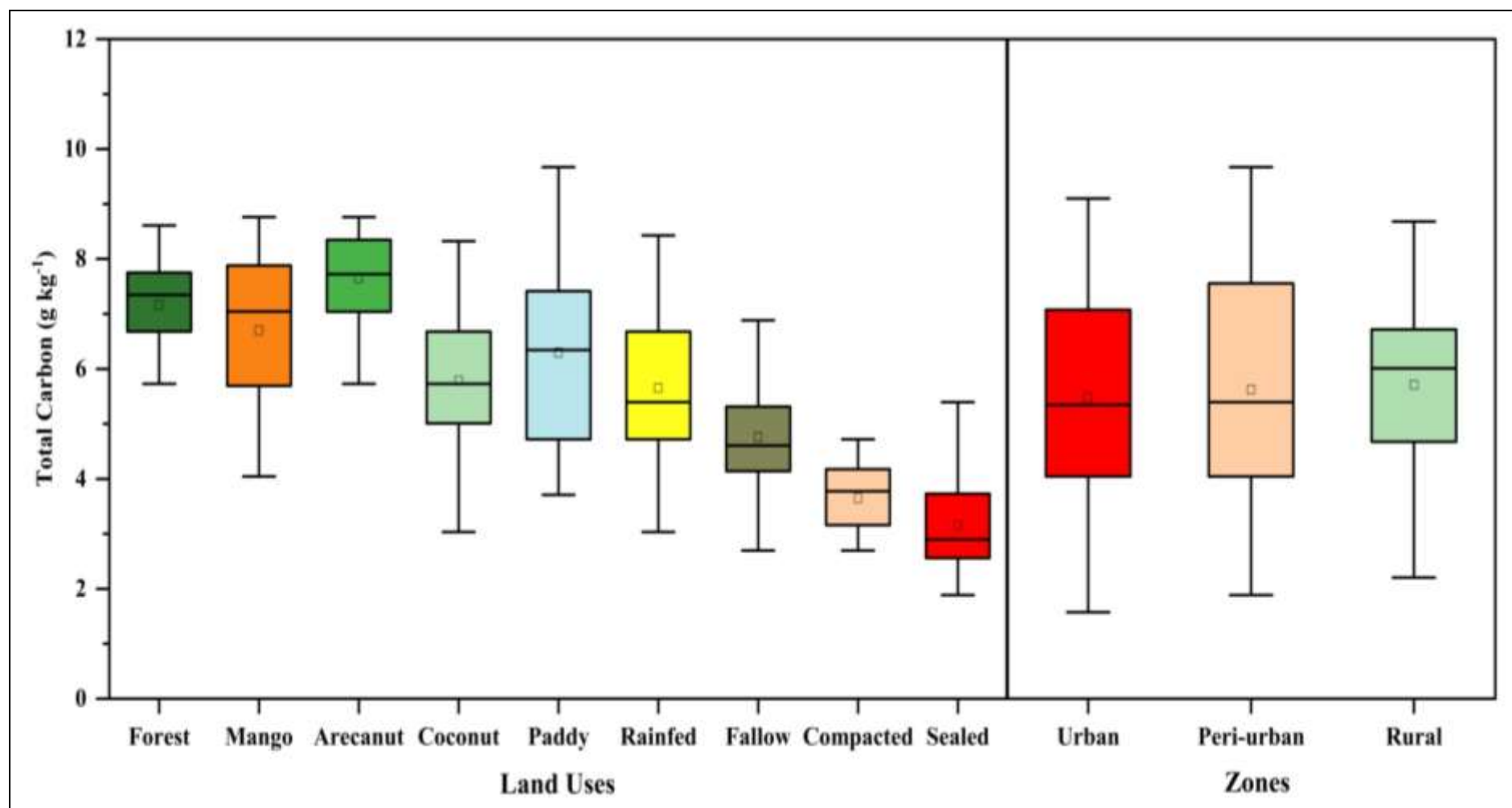


Fig. 33. Total-C (g kg⁻¹) in soils of different land uses across the rural-urban gradient

Lu *et al.*, 2020; Asabere *et al.*, 2018). Sealed areas exhibited lower total-C values, which is likely due to the negative effect of soil sealing, which limits the organic inputs, retards microbiological processes resulting in reduced organic matter decomposition (O'Riordan *et al.*, 2021).

4.3.15 Soil nitrogen fractions

The data on various soil nitrogen fractions, including ammoniacal nitrogen, nitrate nitrogen, organic nitrogen and total nitrogen, across different land use systems and zones are summarized in Tables 30 to 33.

4.3.15.1 Ammonical nitrogen

Table 30 depicts the data related to soil ammonical nitrogen ($\text{NH}_4^+\text{-N}$) of different land use systems across the urban, peri-urban and rural zones. The soil $\text{NH}_4^+\text{-N}$ values ranged from 10.28 mg kg⁻¹ in sealed areas of urban zone to 39.98 mg kg⁻¹ in forest plantations of urban zone.

Among different land use systems, the soils of forest plantations recorded significantly higher values of $\text{NH}_4^+\text{-N}$ (38.81 mg kg⁻¹) followed by the paddy soils (37.83 mg kg⁻¹). Among the agricultural-based land use systems, mango (35.57 mg kg⁻¹), arecanut (35.55 mg kg⁻¹), coconut (34.61 mg kg⁻¹) and rainfed crop (34.41 mg kg⁻¹) also exhibited significant amounts of $\text{NH}_4^+\text{-N}$. The soils of sealed areas (11.67 mg kg⁻¹) and compacted areas (12.34 mg kg⁻¹) recorded significantly lower values of $\text{NH}_4^+\text{-N}$. Comparison among the different land use systems across the three zones indicated that there was no significant difference among the zones.

Ammonium nitrogen ($\text{NH}_4^+\text{-N}$) levels varied notably across different land uses and zones (Fig. 34), reflecting the influence of biomass turnover, land management strategies and urbanization intensity (Xu *et al.*, 2024). Among the land types, forest plantations exhibited the highest $\text{NH}_4^+\text{-N}$ concentrations, likely due to consistent leaf litter deposition, minimal soil disturbance and a thriving microbial community (Bruun *et al.*, 2015). Similarly, arecanut plantations demonstrated elevated $\text{NH}_4^+\text{-N}$ levels, which can be attributed to the regular application of organic manures, integration of crop residues and natural litterfall, all of which enhance soil fertility while mitigating erosion (Vasundhara *et al.*, 2021).

In contrast, sealed surfaces recorded the lowest $\text{NH}_4^+\text{-N}$ levels, as the absence of organic matter input, reduced soil aeration and disrupted microbial activity hinder nitrogen accumulation (O'Riordan *et al.*, 2021). Compacted soils also exhibited diminished $\text{NH}_4^+\text{-N}$ concentrations, likely due to restricted root penetration and organic matter incorporation caused by heavy machinery and frequent foot traffic (Busse *et al.*, 2021). Higher concentrations of $\text{NH}_4^+\text{-N}$ are commonly found in rural zone, primarily due to increased biomass turnover compared to urban and peri-urban regions

Table 30. Ammonical-N (mg kg⁻¹) in soils of different land uses across the rural-urban gradient

Land uses (L)	Zones (Z)			Land uses (L) Mean
	Urban	Peri-urban	Rural	
Forest/forest plantations	39.98	38.81	37.63	38.81
Mango	36.54	34.56	35.76	35.57
Arecanut	36.84	35.84	34.25	35.55
Coconut	33.36	34.72	35.47	34.61
Paddy	36.36	38.17	38.64	37.83
Rainfed crop	33.12	34.35	35.47	34.41
Waste lands	12.68	14.48	15.01	14.13
Compacted area	11.14	12.20	13.52	12.34
Sealed area	10.28	11.65	12.77	11.67
Zones (Z) Mean	27.79	28.38	28.84	
Factors	S.Em±		CD @ 0.05	
Zones (Z)	1.43		NS	
Land uses (L)	0.69		1.94	
Interaction (Z x L)	1.14		NS	

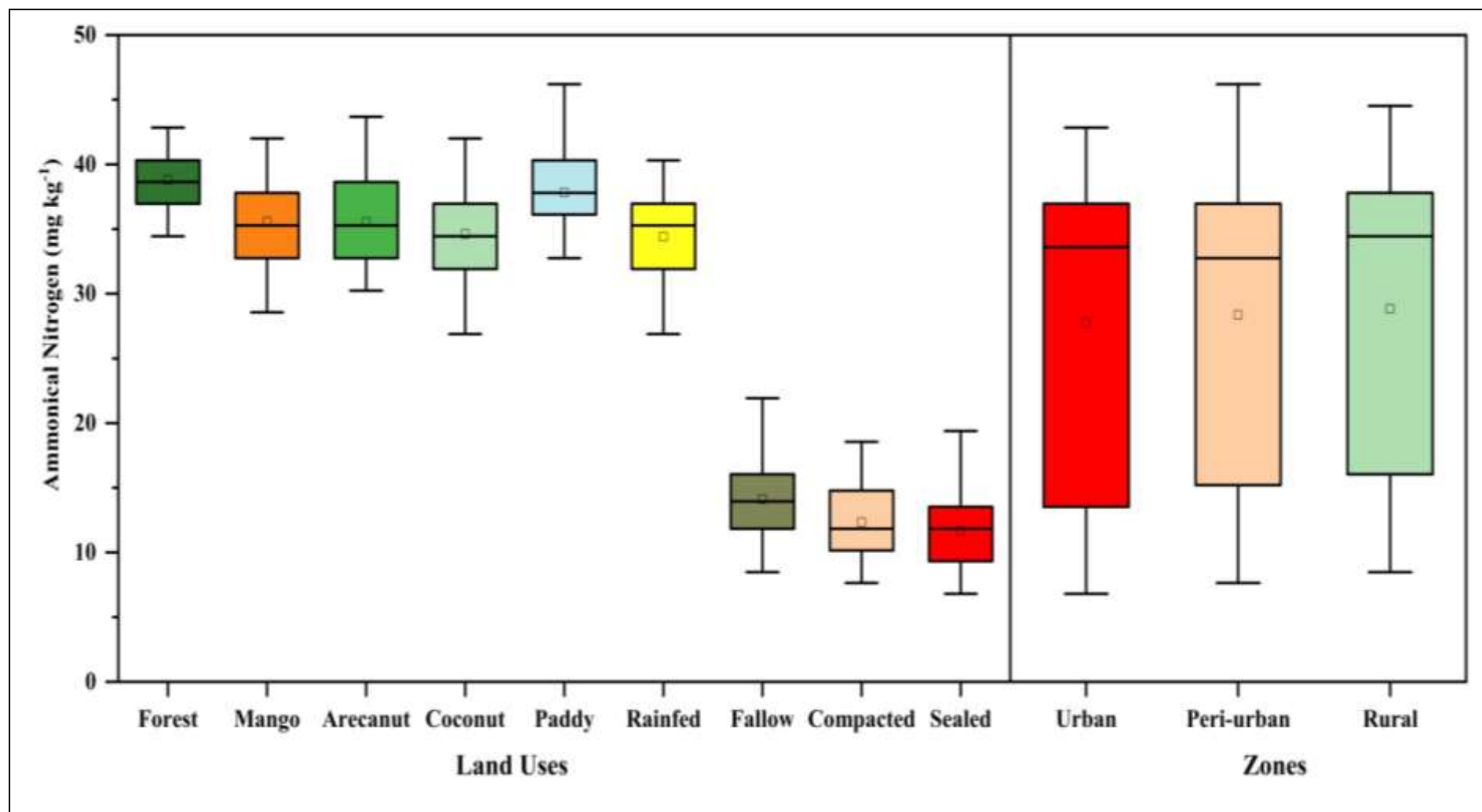


Fig. 34. Ammonical-N (mg kg⁻¹) in soils of different land uses across the rural-urban gradient

(Kuntoji *et al.*, 2021). In contrast, urban environments experience significant human-induced changes, including construction, paving and industrial activities, which influence $\text{NH}_4^+\text{-N}$ dynamics (O'Riordan *et al.*, 2021). Peri-urban zones, acting as transitional spaces, exhibit moderate $\text{NH}_4^+\text{-N}$ levels due to their mixed land use patterns and intermediate levels of soil disturbance (Seifollahi-Aghmiuni *et al.*, 2022).

4.3.15.2 Nitrate nitrogen

The data pertaining to soil nitrate nitrogen ($\text{NO}_3^-\text{-N}$) of different land use systems across three zones is presented in Table 31. The soil $\text{NO}_3^-\text{-N}$ values ranged from 6.64 mg kg^{-1} in sealed areas of urban zone to 19.99 mg kg^{-1} in forest plantations of urban zone.

Among different land use systems, the soils of forest plantations recorded significantly higher values of $\text{NO}_3^-\text{-N}$ (19.40 mg kg^{-1}) followed by the paddy soils (18.92 mg kg^{-1}). Among the agricultural-based land use systems, mango (17.79 mg kg^{-1}), arecanut (17.77 mg kg^{-1}), coconut (17.30 mg kg^{-1}) and rainfed crop (17.20 mg kg^{-1}) exhibited substantial amounts of $\text{NO}_3^-\text{-N}$. The soils of sealed areas (7.34 mg kg^{-1}) and compacted areas (7.67 mg kg^{-1}) recorded significantly lower values of $\text{NO}_3^-\text{-N}$. Comparison among the different land use systems across the three zones indicated that there was no significant difference among the zones.

Forest/forest plantations had higher levels of $\text{NO}_3^-\text{-N}$ which is likely due to enhanced biomass turnovers and microbial decomposition which sustain nitrification process (Yuan *et al.*, 2024). Higher $\text{NO}_3^-\text{-N}$ levels in paddy soils were reported (Fig. 35). Paddy soils are often associated with anaerobic conditions due to waterlogging, periods of aerobic conditions during land preparation and post-harvest management could stimulate nitrification (Soliman *et al.*, 2024). Additionally, the application of nitrogen-based fertilizers in paddy cultivation could have contributed to the observed nitrate levels (Hu *et al.*, 2021). The lower $\text{NO}_3^-\text{-N}$ contents in sealed and compacted areas can be attributed to reduced infiltration, root penetration and microbial activity. These limitations hinder nitrification process (O'Riordan *et al.*, 2021).

4.3.15.3 Organic nitrogen

Table 32 presents data on soil organic nitrogen (organic-N) levels across various land use systems in three zones. The values ranged from a minimum of 91.08 mg kg^{-1} in sealed areas of urban zone to a peak of 371.22 mg kg^{-1} in forest plantations under urban zone.

Among the land use types, forest plantations exhibited the highest organic-N content, averaging 363.65 mg kg^{-1} , followed by soils in arecanut plantations (361.08 mg kg^{-1}). Within agricultural and horticultural-based land uses systems, mango orchards (359.65 mg kg^{-1}), coconut soils (321.61 mg kg^{-1}), paddy fields

Table 31. Nitrate-N (mg kg⁻¹) in soils of different land uses across the rural-urban gradient

Land uses (L)	Zones (Z)			Land uses (L) Mean
	Urban	Peri-urban	Rural	
Forest/forest plantations	19.99	19.40	18.82	19.40
Mango	18.27	17.28	17.88	17.79
Arecanut	18.42	17.92	17.13	17.77
Coconut	16.68	17.36	17.73	17.30
Paddy	18.18	19.09	19.32	18.92
Rainfed crop	16.56	17.17	17.73	17.20
Waste lands	7.84	8.74	9.22	8.64
Compacted area	7.07	7.60	8.26	7.67
Sealed area	6.64	7.33	7.89	7.34
Zones (Z) Mean	14.39	14.68	14.93	
Factors	S.Em±		CD @ 0.05	
Zones (Z)	0.63		NS	
Land uses (L)	0.35		0.98	
Interaction (Z x L)	0.57		NS	

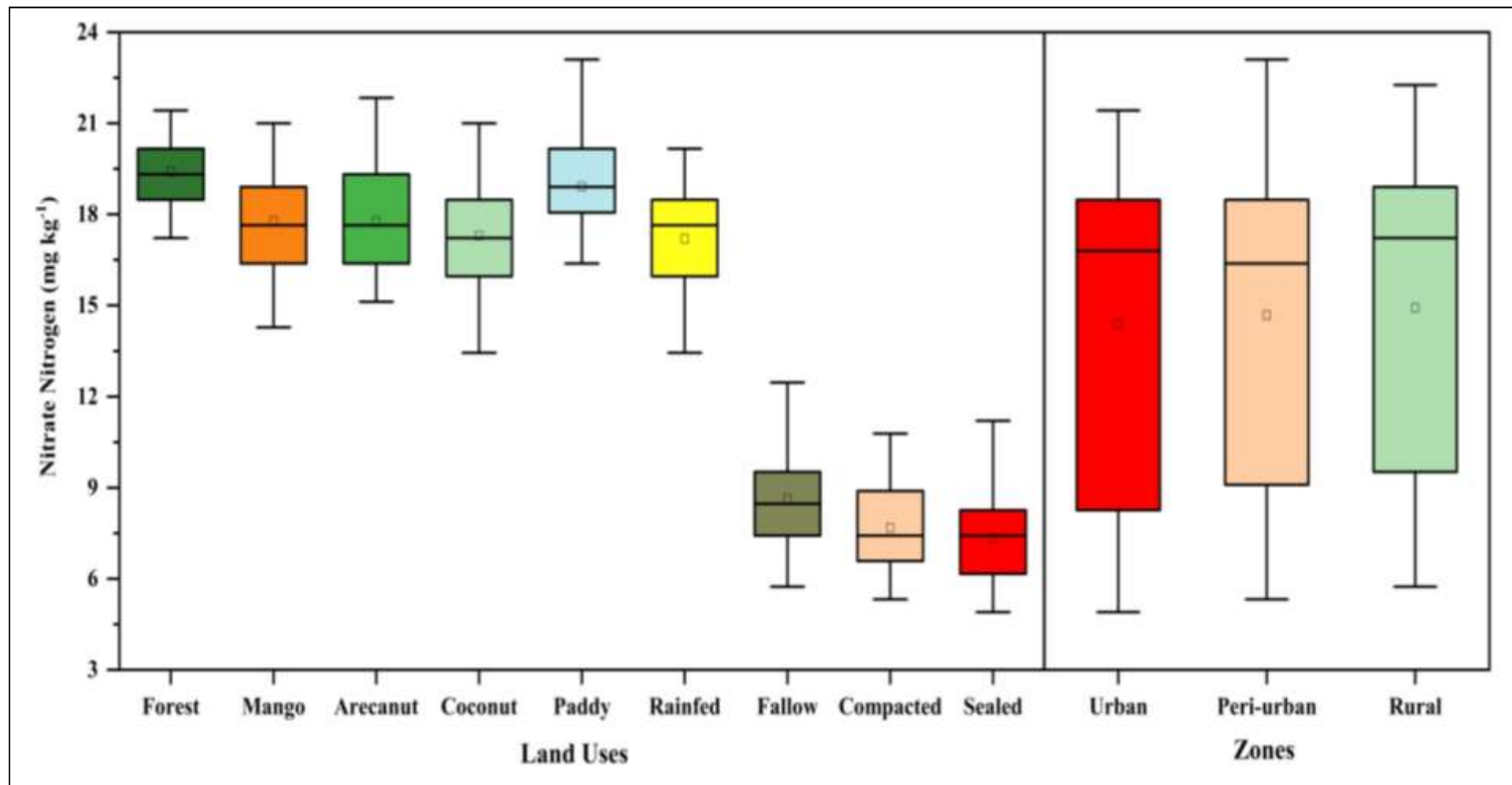


Fig. 35. Nitrate-N (mg kg⁻¹) in soils of different land uses across the rural-urban gradient

Table 32. Organic-N (mg kg⁻¹) in soils of different land uses across the rural-urban gradient

Land uses (L)	Zones (Z)			Land uses (L) Mean
	Urban	Peri-urban	Rural	
Forest/forest plantations	371.22	364.59	355.15	363.65
Mango	362.86	356.16	360.36	359.64
Arecanut	368.74	360.02	356.18	361.08
Coconut	319.96	324.36	320.13	321.61
Paddy	327.46	331.63	326.26	328.53
Rainfed crop	312.32	314.04	320.13	315.75
Waste lands	167.81	170.78	177.77	172.34
Compacted area	116.46	120.20	124.22	120.48
Sealed area	91.08	103.24	112.45	103.15
Zones (Z) Mean	270.27	271.16	272.23	
Factors	S.Em±		CD @ 0.05	
Zones (Z)	12.98		NS	
Land uses (L)	5.64		15.75	
Interaction (Z x L)	10.02		NS	

(328.53 mg kg⁻¹) and rainfed croplands (315.75 mg kg⁻¹) also showed considerably high organic-N levels. Conversely, soils in sealed (103.15 mg kg⁻¹) and compacted (120.48 mg kg⁻¹) areas recorded the lowest values. A comparative analysis across the three zones revealed no significant variation in organic-N levels among different land use systems.

Organic nitrogen content varied notably across different land uses and zones, reflecting the influence of biomass turnover, land management strategies and urbanization levels (Xu *et al.*, 2024). Forest plantations exhibited higher organic-N levels, likely due to consistent leaf litter deposition, minimal soil disturbance and a thriving microbial community (Bruun *et al.*, 2015). Similarly, arecanut plantations showed elevated organic-N content, which can be attributed to the extensive use of organic manure, incorporation of crop residues and natural litterfall, all of which enhance soil fertility while mitigating erosion (Vasundhara *et al.*, 2021).

Conversely, sealed surfaces had the lowest organic-N levels (Fig. 36), as soil sealing restricts organic matter input, reduces aeration and disrupts microbial processes, ultimately limiting N accumulation (O'Riordan *et al.*, 2021). Compacted soils also displayed reduced organic nitrogen content, likely due to the impact of heavy machinery and foot traffic, which hinder root penetration and organic matter incorporation (Busse *et al.*, 2021). Overall, organic-N levels were highest in rural zone, where greater biomass turnover contributes to soil enrichment compared to urban and peri-urban zones (Kuntoji *et al.*, 2021). In contrast, urban zones experience significant anthropogenic influences, including construction, paving and industrial activities, which negatively affect soil nitrogen content (O'Riordan *et al.*, 2021). Peri-urban areas, serving as transitional zones, exhibit moderate organic nitrogen levels due to mixed land uses and intermediate levels of soil disturbance (Seifollahi-Aghmiuni *et al.*, 2022).

4.3.15.4 Total nitrogen

The data on total nitrogen (TN) levels across various land use systems and zones is displayed in the Table 33. The TN content in soils varied from a minimum of 108.00 mg kg⁻¹ in sealed areas of urban zone to a maximum of 431.20 mg kg⁻¹ in forest plantations under urban zone.

Among the different land use categories, soils of forest/forest plantation recorded the highest TN concentration (421.87 mg kg⁻¹), closely followed by arecanut-growing soils (414.40 mg kg⁻¹). Within agricultural and horticultural-based land use systems, mango orchards (413.00 mg kg⁻¹), coconut plantations (373.52 mg kg⁻¹), paddy fields (385.28 mg kg⁻¹) and rainfed croplands (367.36 mg kg⁻¹) also showed notable TN levels. In contrast, sealed (122.16 mg kg⁻¹) and compacted (140.50 mg kg⁻¹) areas had the lowest TN content. A comparison across the three zones revealed no statistically significant differences in TN levels among them.

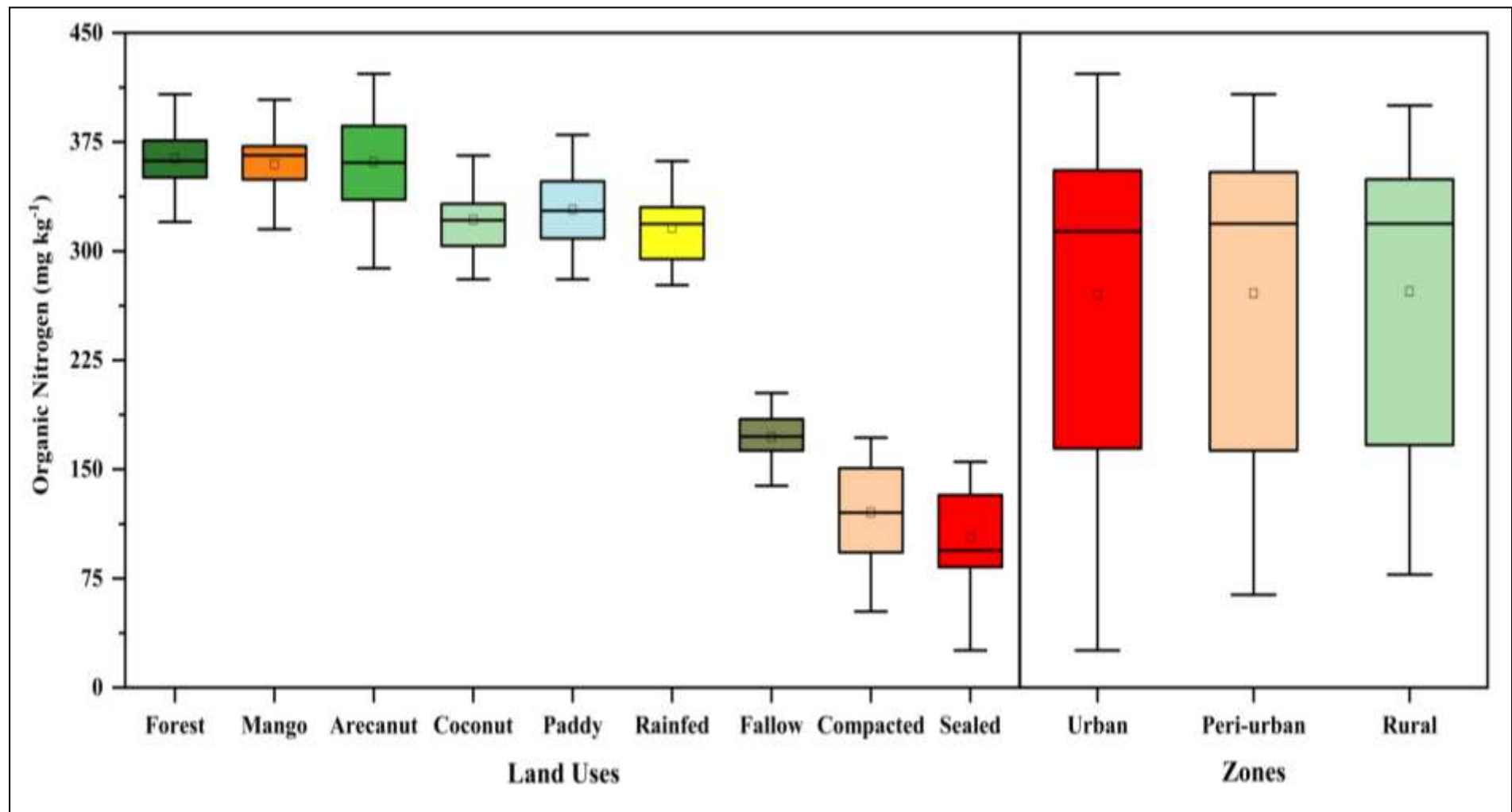


Fig. 36. Organic-N (mg kg⁻¹) in soils of different land uses across the rural-urban gradient

Table 33. Total-N (mg kg⁻¹) in soils of different land uses across the rural-urban gradient

Land uses (L)	Zones (Z)			Land uses (L) Mean
	Urban	Peri-urban	Rural	
Forest/forest plantations	431.20	422.80	411.60	421.87
Mango	417.67	408.00	414.00	413.00
Arecanut	424.00	413.78	407.56	414.40
Coconut	370.00	376.44	373.33	373.52
Paddy	382.00	388.89	384.22	385.28
Rainfed crop	362.00	365.56	373.33	367.36
Waste lands	188.33	194.00	202.00	195.10
Compacted area	134.67	140.00	146.00	140.50
Sealed area	108.00	122.22	133.11	122.16
Zones (Z) Mean	312.45	314.23	316.00	
Factors	S.Em±		CD @ 0.05	
Zones (Z)	14.85		NS	
Land uses (L)	5.53		15.44	
Interaction (Z x L)	9.68		NS	

The higher TN content in forest and arecanut plantations in rural zone (Fig. 37) could be attributed to the higher $\text{NH}_4^+\text{-N}$ and organic-N content in respective land use systems in rural zones. This aligns with the findings of previous studies who have reported higher TN values in rural zones in comparison to urban zones (Lu *et al.*, 2020; Asabere *et al.*, 2018). Sealed areas exhibited the lowest TN values which could be due to the reduced organic matter inputs and microbial activity (O'Riordan *et al.*, 2021). Compacted soils also recorded lower TN values, likely due to similar constraints in soil structure, porosity and aeration (Kahsay *et al.*, 2023). The comparison of TN across the zones revealed no significant difference.

4.3.16 Soil microbial population

The data pertaining to soil microbial population *viz.*, bacteria, fungi and actinomycetes of different land use systems across the zones are presented in Table 34 to 36. The results obtained with respect to soil microbial populations are discussed collectively in this section.

4.3.16.1 Soil bacterial population

The data in Table 34 illustrates the soil bacterial population across various land use systems in three distinct zones. Bacterial counts varied widely, ranging from 3.23×10^6 CFU g^{-1} in sealed areas of the peri-urban zone to 12.31×10^6 CFU g^{-1} in forest plantations within the urban zone.

Among different land use types, paddy soils exhibited the highest bacterial population at 11.10×10^6 CFU g^{-1} , closely followed by forest and forest plantations (11.08×10^6 CFU g^{-1}), indicating comparable microbial activity. Within agricultural land use, mango orchards (10.59×10^6 CFU g^{-1}), arecanut (10.43×10^6 CFU g^{-1}), coconut (10.28×10^6 CFU g^{-1}) and rainfed croplands (9.93×10^6 CFU g^{-1}) also supported the bacterial communities (Fig. 38). Conversely, soils of sealed areas (3.37×10^6 CFU g^{-1}) and compacted soils (5.23×10^6 CFU g^{-1}) harbored significantly lower microbial populations. When comparing zones, soils of rural zone exhibited the highest bacterial populations (9.46×10^6 CFU g^{-1}), surpassing peri-urban (9.17×10^6 CFU g^{-1}) and urban soils (7.51×10^6 CFU g^{-1}), highlighting the influence of land use and environmental conditions on microbial abundance.

4.3.16.2 Soil fungal population

Table 35 depicts the data related to soil fungal population of different land use systems across three zones. The soil fungal population ranged from 2.08×10^4 CFU g^{-1} in sealed areas of urban zone to 12.35×10^4 CFU g^{-1} in paddy soils of rural zone.

Among different land use systems, the soils of forest/forest plantations exhibited significantly higher fungal population (11.25×10^4 CFU g^{-1}). All the agricultural-based land use systems recorded significantly higher soil fungal population with values ranging from 9.97×10^4 CFU g^{-1} to 10.45×10^4 CFU g^{-1} . Among the

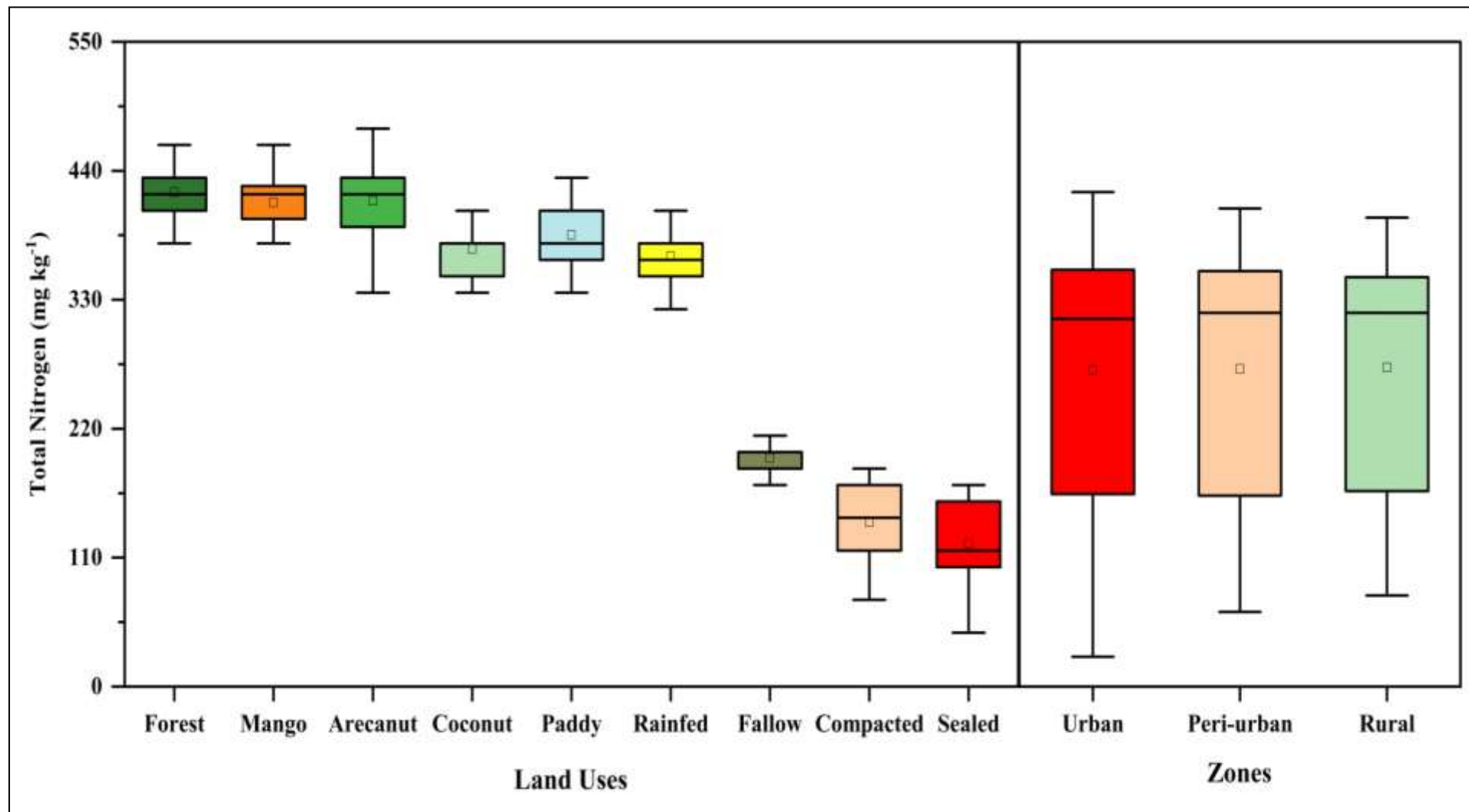


Fig. 37. Total-N (mg kg⁻¹) in soils of different land uses across the rural-urban gradient

Table 34. Bacterial population (10^6 CFU g⁻¹) in soils of different land uses across the rural-urban gradient

Land uses (L)	Zones (Z)			Land uses (L) Mean
	Urban	Peri-urban	Rural	
Forest/forest plantations	12.31	11.68	9.26	11.08
Mango	9.19	11.22	11.35	10.59
Arecanut	8.45	11.02	11.81	10.43
Coconut	8.30	10.87	11.66	10.28
Paddy	9.46	11.68	12.15	11.10
Rainfed crop	8.28	10.51	11.00	9.93
Waste lands	4.59	6.70	7.95	6.41
Compacted area	3.37	5.60	6.73	5.23
Sealed area	3.62	3.23	3.26	3.37
Zones (Z) Mean	7.51	9.17	9.46	
Factors	S.Em±		CD @ 0.05	
Zones (Z)	0.07		0.19	
Land uses (L)	0.12		0.34	
Interaction (Z x L)	0.21		0.58	

Table 35. Fungal population (10^4 CFU g⁻¹) in soils of different land uses across the rural-urban gradient

Land uses (L)	Zones (Z)			Land uses (L) Mean
	Urban	Peri-urban	Rural	
Forest/forest plantations	11.81	11.76	10.17	11.25
Mango	7.75	8.54	11.66	9.32
Arecanut	7.99	10.12	12.24	10.12
Coconut	7.84	9.97	12.12	9.98
Paddy	8.10	10.89	12.35	10.45
Rainfed crop	6.95	10.74	12.22	9.97
Waste lands	4.61	6.07	6.02	5.57
Compacted area	3.88	3.35	4.13	3.79
Sealed area	2.08	2.62	2.18	2.29
Zones (Z) Mean	6.78	8.23	9.23	
Factors	S.Em±		CD @ 0.05	
Zones (Z)	0.07		0.18	
Land uses (L)	0.11		0.32	
Interaction (Z x L)	0.20		0.55	

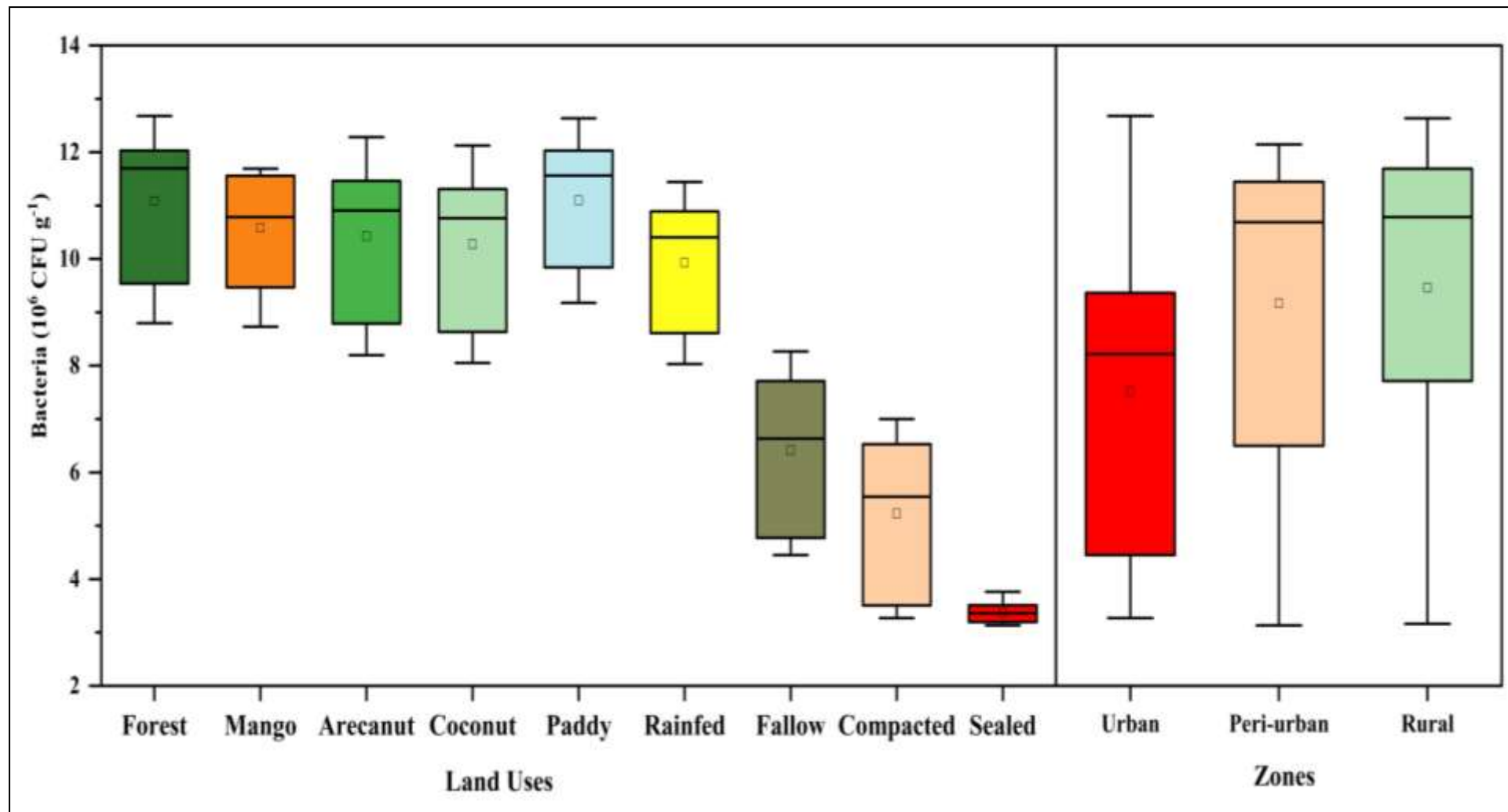


Fig. 38. Bacterial population (10^6 CFU g^{-1}) in soils of different land uses across the rural-urban gradient

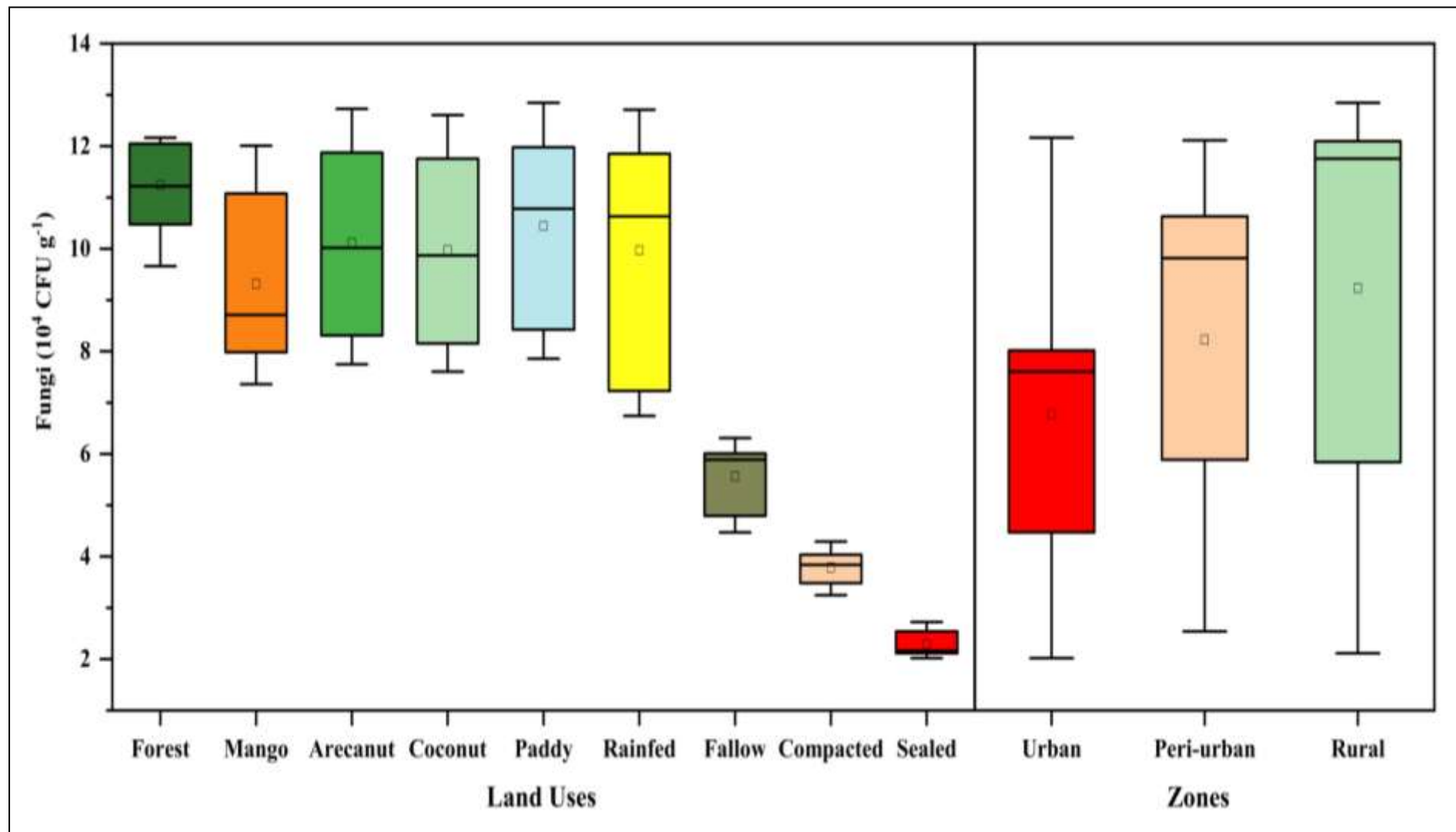


Fig. 39. Fungal population (10^4 CFU g^{-1}) in soils of different land uses across the rural-urban gradient

agricultural-based land use systems, paddy (10.45×10^4 CFU g⁻¹) and arecanut plantations (10.12×10^4 CFU g⁻¹) exhibited significantly higher soil fungal populations (Fig. 39). Similarly, significant soil fungal populations were also observed in mango (9.32×10^4 CFU g⁻¹), coconut (9.98×10^4 CFU g⁻¹) and rainfed crop (9.97×10^4 CFU g⁻¹). The soils of sealed area (2.29×10^4 CFU g⁻¹) and compacted area (3.79×10^4 CFU g⁻¹) recorded significantly lower soil fungal population. Comparison across the zones indicated that the soils of rural zone exhibited significantly higher fungal population (9.23×10^4 CFU g⁻¹) over peri-urban (8.23×10^4 CFU g⁻¹) and urban zone (6.78×10^4 CFU g⁻¹).

4.3.16.3 Soil actinomycetes population

The data on soil actinomycetes population of different land use systems across three zones is presented in the Table 36. The soil actinomycetes population ranged from 1.30×10^3 CFU g⁻¹ in sealed areas of rural zone to 8.66×10^3 CFU g⁻¹ in paddy soils of rural zone.

Among different land use systems, the soils of forest plantations exhibited significantly higher actinomycetes population (8.11×10^3 CFU g⁻¹). All the agricultural-based land use systems recorded significantly higher soil actinomycetes population with values ranging from 7.32×10^3 CFU g⁻¹ to 7.64×10^3 CFU g⁻¹. Among the agricultural-based land use systems, paddy (7.64×10^3 CFU g⁻¹) and arecanut plantation (7.46×10^3 CFU g⁻¹) recorded significantly higher soil actinomycetes (Fig. 40). Similarly, substantial values of soil actinomycetes populations were also observed in mango (7.32×10^3 CFU g⁻¹), coconut (7.39×10^3 CFU g⁻¹) and rainfed crop (7.35×10^3 CFU g⁻¹). However, soil actinomycetes population did not vary significantly in these land uses. In contrast, the soils of sealed areas (1.58×10^3 CFU g⁻¹) and compacted areas (2.92×10^3 CFU g⁻¹) recorded significantly lower actinomycetes population. Comparison across zones indicated that the soils of rural zones exhibited significantly higher actinomycetes population (6.29×10^3 CFU g⁻¹) over peri-urban (5.85×10^3 CFU g⁻¹) and urban zones (5.54×10^3 CFU g⁻¹).

Higher soil microbial populations in forest/forest plantations could be attributed to higher biomass turnovers and enhanced organic matter decomposition which favours the microbial proliferation (Bhattacharyya *et al.*, 2022). Paddy and arecanut plantations also recorded significantly higher soil microbial populations which is likely due to the addition of organic residues and amendments which might have created favourable conditions for the microbial growth (Bamdad *et al.*, 2022). Lower soil microbial populations in sealed areas could be due to the negative impacting of soil sealing on soil biological properties (Pereira *et al.*, 2021). Similarly, lower microbial population in compacted areas may be attributed to the reduced aeration, water movement and nutrient availability in compacted soils (Hu *et al.*, 2021). Higher soil microbial

Table 36. Actinomycetes population (10^3 CFU g⁻¹) in soils of different land uses across the rural-urban gradient

Land uses (L)	Zones (Z)			Land uses (L) Mean
	Urban	Peri-urban	Rural	
Forest/forest plantations	8.49	7.99	7.85	8.11
Mango	7.29	7.01	7.65	7.32
Arecanut	6.98	7.36	8.03	7.46
Coconut	6.87	7.25	8.05	7.39
Paddy	6.93	7.32	8.66	7.64
Rainfed crop	6.32	7.20	8.53	7.35
Waste lands	2.59	3.64	3.64	3.29
Compacted area	2.60	3.25	2.92	2.92
Sealed area	1.83	1.60	1.30	1.58
Zones (Z) Mean	5.54	5.85	6.29	
Factors	S.Em±		CD @ 0.05	
Zones (Z)	0.05		0.13	
Land uses (L)	0.08		0.23	
Interaction (Z x L)	0.14		0.40	

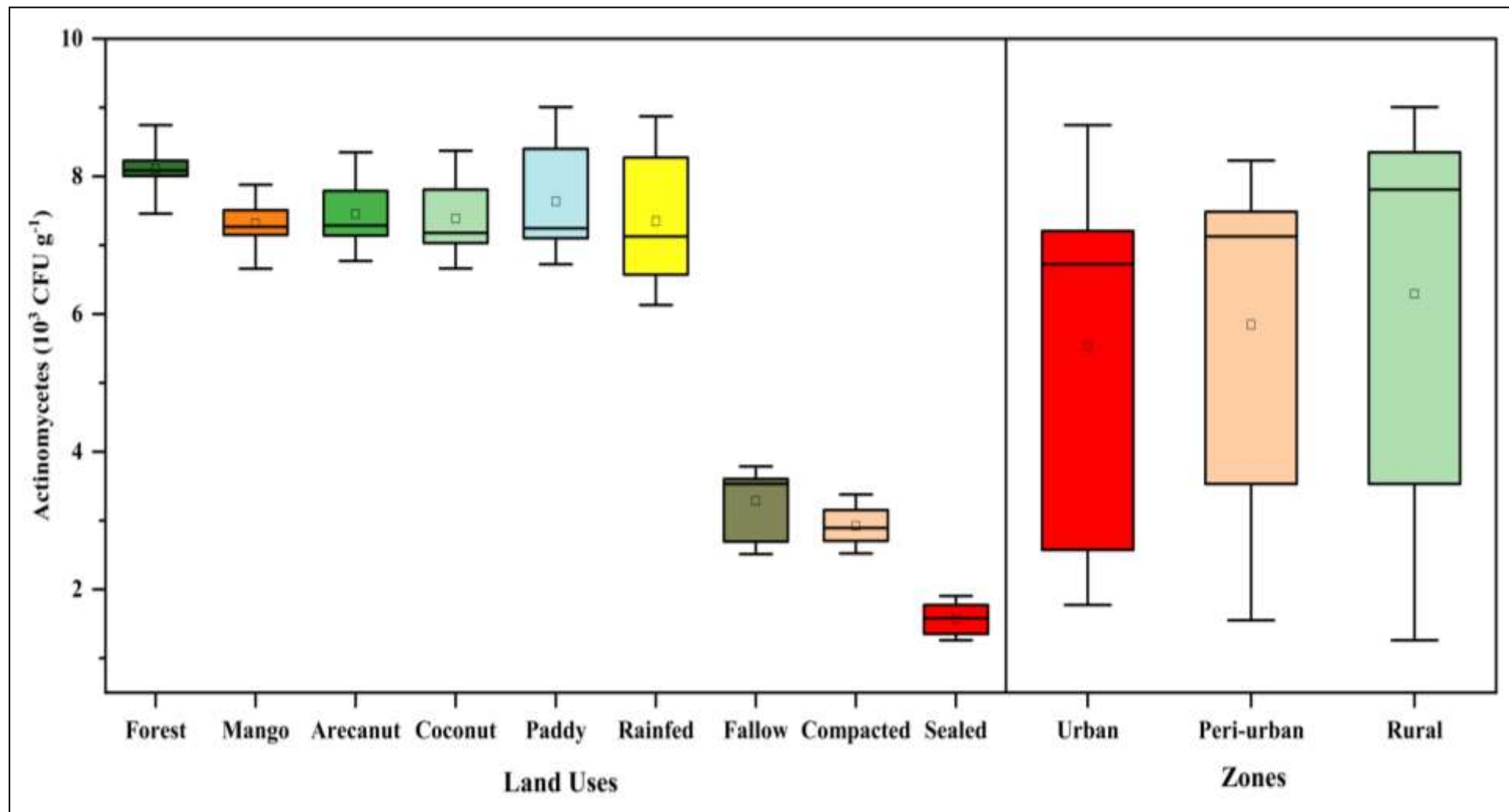


Fig. 40. Actinomycetes population (10^3 CFU g^{-1}) in soils of different land uses across the rural-urban gradient

populations in rural zone over urban and peri-urban zones may be due to the lesser anthropogenic interventions and higher organic matter content in the soils (Mónok *et al.*, 2021). This aligns with the findings of previous studies who have reported higher soil microbial population in rural areas in comparison to urban areas (Rusterholz and Baur, 2023; Pereira *et al.*, 2021).

4.3.17 Soil enzyme activities

The data on soil enzymatic activities, including urease, dehydrogenase and acid phosphatase, across various land use systems in three different zones is presented in Tables 37 to 39. This section provides a comprehensive discussion of the observed results, analyzing the variations in enzymatic activity collectively.

4.3.17.1 Urease activity

Table 37 depicts the data related soil urease activity of different land use systems across the zones. The urease activity of the soils in the study area ranged from 2.33 $\mu\text{g NH}_4^+\text{-N g}^{-1} \text{ h}^{-1}$ in sealed areas of rural zone to 9.51 $\mu\text{g NH}_4^+\text{-N g}^{-1} \text{ h}^{-1}$ in forest plantations of urban zone.

Among different land use systems, the soils of forest plantations exhibited significantly higher urease activity (9.22 $\mu\text{g NH}_4^+\text{-N g}^{-1} \text{ h}^{-1}$) followed by mango (8.68 $\mu\text{g NH}_4^+\text{-N g}^{-1} \text{ h}^{-1}$). The urease activity in waste lands (3.59 $\mu\text{g NH}_4^+\text{-N g}^{-1} \text{ h}^{-1}$) and compacted areas (3.38 $\mu\text{g NH}_4^+\text{-N g}^{-1} \text{ h}^{-1}$) were found to be significantly higher than that of sealed areas across all zones (2.54 $\mu\text{g NH}_4^+\text{-N g}^{-1} \text{ h}^{-1}$). However, the enzyme activity was substantially lower when compared with all other land use systems. Comparison of different land use systems across the three zones indicated that rural zone (6.68 $\mu\text{g NH}_4^+\text{-N g}^{-1} \text{ h}^{-1}$) significantly higher urease activity compared to urban zone (6.39 $\mu\text{g NH}_4^+\text{-N g}^{-1} \text{ h}^{-1}$). Whereas, peri-urban and urban zone was found to be significantly on par in terms of urease activity (Fig. 41).

4.3.17.2 Dehydrogenase activity

Table 38 displays the dehydrogenase activity of different land use systems across three zones. The dehydrogenase activity of the soils in the study area ranged from 1.25 $\mu\text{g TPF g}^{-1} 24 \text{ h}^{-1}$ in sealed areas of urban zone to 10.30 $\mu\text{g TPF g}^{-1} 24 \text{ h}^{-1}$ in forest plantations under urban zone.

Among different land use systems, the soils of forest plantations exhibited significantly higher dehydrogenase activity (8.10 $\mu\text{g TPF g}^{-1} 24 \text{ h}^{-1}$) followed by arecanut plantations (7.04 $\mu\text{g TPF g}^{-1} 24 \text{ h}^{-1}$). It was observed that activity of dehydrogenase in all agriculture-based land use systems was substantially higher compared to compacted (1.88 $\mu\text{g TPF g}^{-1} 24 \text{ h}^{-1}$) and sealed areas (1.30 $\mu\text{g TPF g}^{-1} 24 \text{ h}^{-1}$). However, its activity varied significantly across agricultural land use

Table 37. Urease activity ($\mu\text{g NH}_4^+\text{-N g}^{-1} \text{ h}^{-1}$) in soils of different land uses across the rural-urban gradient

Land uses (L)	Zones (Z)			Land uses (L) Mean
	Urban	Peri-urban	Rural	
Forest/forest plantations	9.51	9.22	8.93	9.22
Mango	8.60	8.75	8.70	8.68
Arecanut	7.67	8.30	8.49	8.15
Coconut	7.46	8.00	8.15	7.87
Paddy	7.92	8.08	8.26	8.09
Rainfed crop	6.67	7.39	7.97	7.34
Waste lands	3.60	3.39	3.79	3.59
Compacted area	3.26	3.42	3.46	3.38
Sealed area	2.82	2.46	2.33	2.54
Zones (Z) Mean	6.39	6.56	6.68	
Factors	S.Em$\pm$		CD @ 0.05	
Zones (Z)	0.04		0.12	
Land uses (L)	0.07		0.20	
Interaction (Z x L)	0.12		0.35	

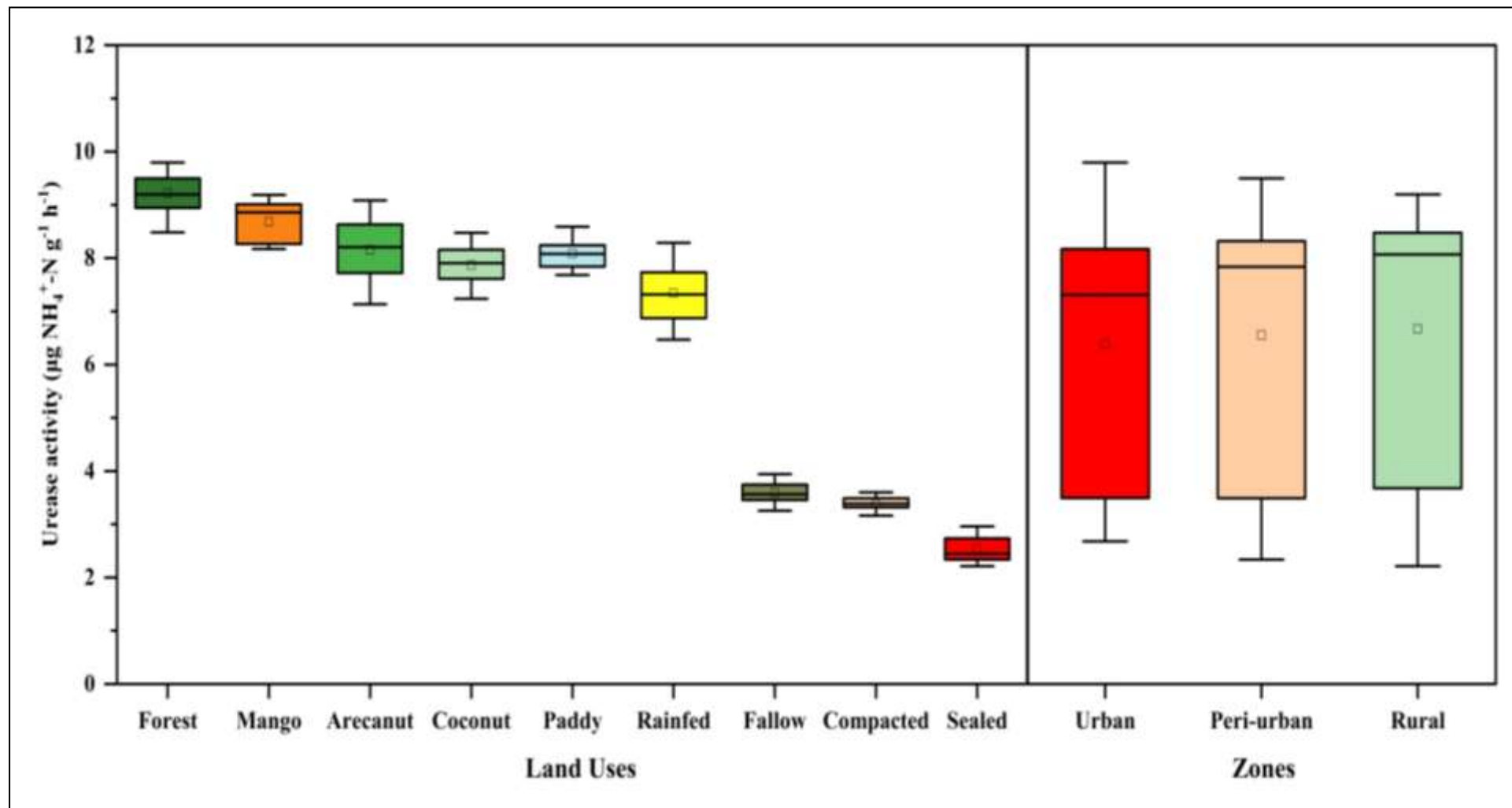


Fig. 41. Urease activity ($\mu\text{g NH}_4^+\text{-N g}^{-1} \text{h}^{-1}$) in soils of different land uses across the rural-urban gradient

Table 38. Dehydrogenase activity ($\mu\text{g TPF g}^{-1} 24 \text{ h}^{-1}$) in soils of different land uses across the rural-urban gradient

Land uses (L)	Zones (Z)			Land uses (L) Mean
	Urban	Peri-urban	Rural	
Forest/forest plantations	10.30	7.36	6.63	8.10
Mango	6.61	6.95	7.42	6.99
Arecanut	6.52	6.73	7.86	7.04
Coconut	5.79	6.36	7.05	6.40
Paddy	6.14	6.94	7.14	6.74
Rainfed crop	4.30	5.45	6.77	5.51
Waste lands	2.87	3.56	4.64	3.69
Compacted area	1.71	1.92	2.02	1.88
Sealed area	1.25	1.29	1.37	1.30
Zones (Z) Mean	5.05	5.17	5.66	
Factors	S.Em$\pm$		CD @ 0.05	
Zones (Z)	0.03		0.10	
Land uses (L)	0.06		0.17	
Interaction (Z x L)	0.10		0.29	

systems. Comparison of different land use systems across the three zones indicated that soils of rural zone exhibited higher dehydrogenase activity ($5.66 \mu\text{g TPF g}^{-1} 24 \text{ h}^{-1}$) compared to peri-urban ($5.17 \mu\text{g TPF g}^{-1} 24 \text{ h}^{-1}$) and urban ($5.05 \mu\text{g TPF g}^{-1} 24 \text{ h}^{-1}$) zones (Fig. 42).

4.3.17.3 Acid phosphatase activity

The data on acid phosphatase activity of different land use systems across the three is illustrated in Table 39. The acid phosphatase activity of the soils in the study area ranged from $9.74 \mu\text{g PNP g}^{-1} \text{ h}^{-1}$ in sealed areas of urban zone to $39.38 \mu\text{g PNP g}^{-1} \text{ h}^{-1}$ in forest plantations of urban zone.

Among different land use systems, the soils of forest plantations exhibited considerably higher acid phosphatase activity ($37.04 \mu\text{g PNP g}^{-1} \text{ h}^{-1}$) followed by the soils of mango ($34.48 \mu\text{g PNP g}^{-1} \text{ h}^{-1}$). Among the agricultural and horticultural-based land use systems, arecanut ($29.98 \mu\text{g PNP g}^{-1} \text{ h}^{-1}$), coconut ($28.85 \mu\text{g PNP g}^{-1} \text{ h}^{-1}$), paddy ($32.72 \mu\text{g PNP g}^{-1} \text{ h}^{-1}$) and rainfed crop ($21.62 \mu\text{g PNP g}^{-1} \text{ h}^{-1}$) also exhibited significant amounts of dehydrogenase activity. The soils of sealed areas ($10.11 \mu\text{g PNP g}^{-1} \text{ h}^{-1}$) and compacted soils ($11.82 \mu\text{g PNP g}^{-1} \text{ h}^{-1}$) recorded notably lower acid phosphatase activity. Comparison of different land use systems across the three zones depicted that urban zone exhibited lower acid phosphatase activity ($22.90 \mu\text{g PNP g}^{-1} \text{ h}^{-1}$). Whereas, rural zone displayed considerably higher acid phosphatase activity ($25.51 \mu\text{g PNP g}^{-1} \text{ h}^{-1}$) and peri-urban zone ($24.88 \mu\text{g PNP g}^{-1} \text{ h}^{-1}$) exhibited intermediate values of acid phosphatase activity (Fig. 43).

Higher enzyme activities were found in all agriculture-based land use systems along with forest/forest plantations. The enzyme activities varied significantly across the above land use systems, except arecanut and paddy system. Forest/forest plantations exhibited significantly higher soil enzymatic activities. This could be attributed to the presence of vegetative cover and organic inputs, which might have enhanced the microbial and enzymatic activity (Louwagie *et al.*, 2017). Arecanut plantations also recorded higher dehydrogenase activity, which could be due to the regular application of organic amendment and biomass turnovers, which might have improved the microbial proliferation and associated enzymatic activities (Sharan Bhoopal Reddy *et al.*, 2020; Mencil *et al.*, 2022). The lowest dehydrogenase activity was recorded in sealed area. These findings align with previous research indicating that the sealed area limits the biological activity due to physical barriers retarding natural soil gas exchange processes, moisture availability and microbial habitat (Gorbov *et al.*, 2021). Compacted areas were also exhibited lower dehydrogenase activity which could be attributed to the increased soil compaction which in turn decrease the soil microbial activity (Kuntoji *et al.*, 2021; Tejashvini *et al.*, 2023). Comparison of enzyme activities across the zones however did not vary significantly.

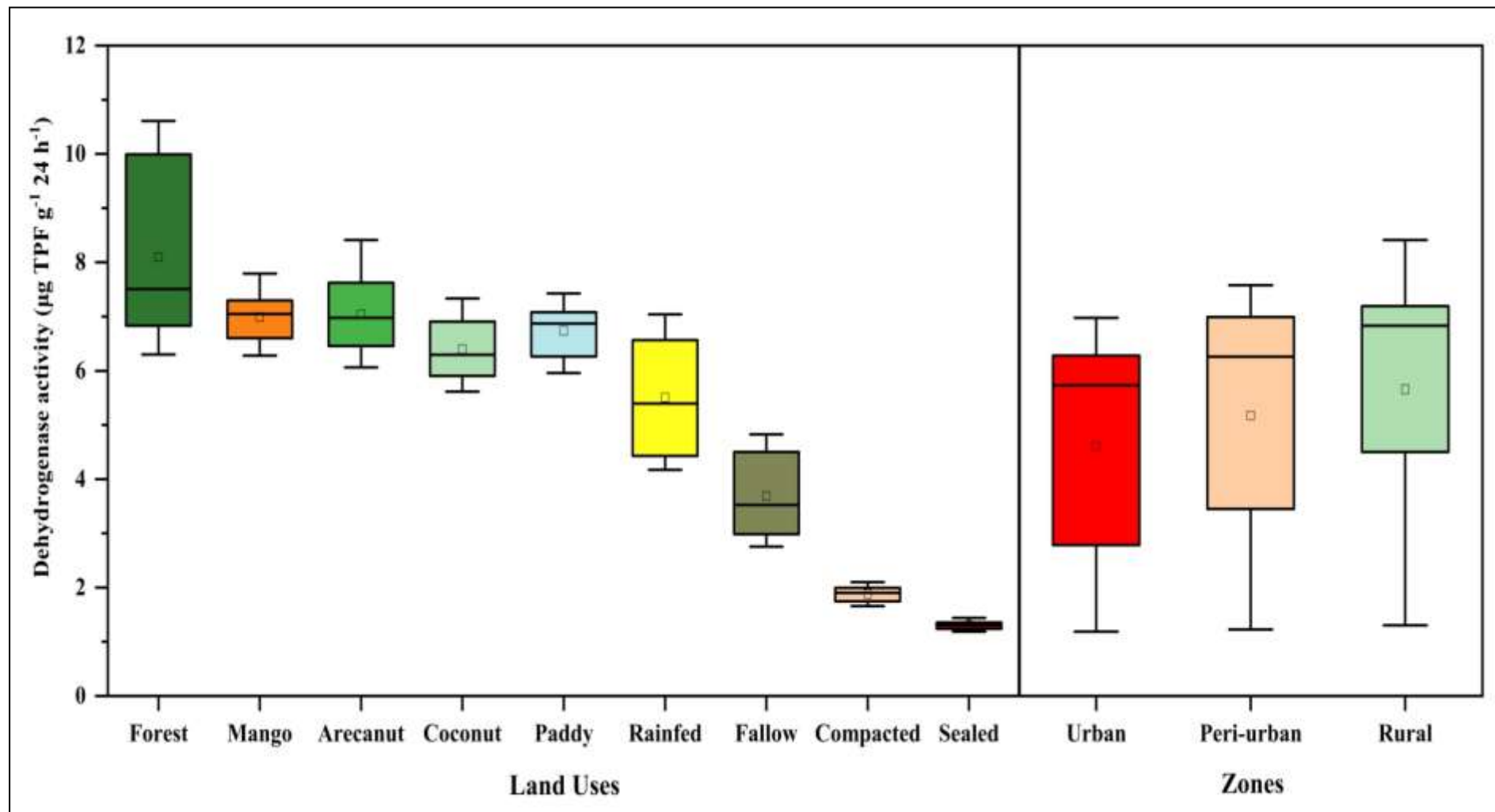


Fig. 42. Dehydrogenase activity ($\mu\text{g TPF g}^{-1} 24 \text{ h}^{-1}$) in soils of different land uses across the rural-urban gradient

Table 39. Acid phosphatase activity ($\mu\text{g PNP g}^{-1} \text{h}^{-1}$) in soils of different land uses across the rural-urban gradient

Land uses (L)	Zones (Z)			Land uses (L) Mean
	Urban	Peri-urban	Rural	
Forest/forest plantations	39.38	37.88	33.86	37.04
Mango	31.72	35.11	36.61	34.48
Arecanut	25.87	31.29	32.79	29.98
Coconut	24.67	30.19	31.69	28.85
Paddy	30.66	33.00	34.50	32.72
Rainfed crop	19.56	21.90	23.40	21.62
Waste lands	13.00	12.60	14.10	13.23
Compacted area	11.48	11.93	12.06	11.82
Sealed area	9.74	10.04	10.54	10.11
Zones (Z) Mean	22.90	24.88	25.51	
Factors	S.Em$\pm$		CD @ 0.05	
Zones (Z)	0.17		0.47	
Land uses (L)	0.29		0.82	
Interaction (Z x L)	0.51		1.42	

4.3.18 Soil microbial biomass carbon and nitrogen

The data pertaining to soil microbial biomass carbon (SMB-C) and soil microbial biomass nitrogen (SMB-N) are presented in Table 40 to 41. The SMB-C values varied from 60.0 mg kg⁻¹ in sealed areas sealed areas of peri-urban zone to 184.0 mg kg⁻¹ in forest/forest plantations of urban zone.

Similarly, the SMB-N values ranged from 8.17 mg kg⁻¹ in sealed areas of peri-urban zone to 37.10 mg kg⁻¹ in forest plantations of urban zone. Among different land use systems, the soils of forest/forest plantations recorded significantly higher SMB-C (174.7 mg kg⁻¹) and SMB-N values (35.31 mg kg⁻¹). Among the agricultural-based land use systems, mango exhibited significantly higher SMB-C (156.7 mg kg⁻¹) and SMB-N values (29.32 mg kg⁻¹). In contrast, the soils of sealed area exhibited significantly lower SMB-C (62.3 mg kg⁻¹) and SMB-N (8.81 mg kg⁻¹). The compacted areas also exhibited significantly lower values of SMB-C (86.0 mg kg⁻¹) and SMB-N (12.13 mg kg⁻¹). Comparison among the land use systems across the zones indicated that the SMB-C and SMB-N varied significantly in the order of rural (SMB-C: 128.8 mg kg⁻¹; SMB-N: 23.58 mg kg⁻¹), peri-urban (SMB-C: 122.4 mg kg⁻¹; SMB-N: 21.64 mg kg⁻¹) and urban zone (SMB-C: 111.6 mg kg⁻¹; SMB-N: 19.85 mg kg⁻¹).

Forest/plantations recorded significantly higher soil microbial biomass (Fig 44 & Fig 45). This could be attributed to the higher biomass turnovers and enhanced organic matter decomposition which might have enhanced the microbial biomass (Bhattacharyya *et al.*, 2022). Arecanut, mango and paddy land use systems also exhibited relatively higher soil microbial biomass which is likely due to organic amendments, diverse root systems and frequent management practices that enrich soil microbial growth (Bamdad *et al.*, 2022). The lower soil microbial biomass values in sealed areas may be attributed to the limited organic inputs and disturbed natural soil gaseous exchange which disrupts the microbial habitats (Rai *et al.*, 2018). Similarly, lower soil microbial biomass carbon in compacted areas could be attributed to reduced aeration and water movement which retards the microbial growth (Pradeep *et al.*, 2018).

Lower soil microbial biomass in rural zone could be attributed to lesser soil contamination and reduced anthropogenic pressures (Demina *et al.*, 2018). Whereas, the urban areas are known to be subjected for various anthropogenic disturbances. This might have resulted in lower soil microbial biomass (Medina *et al.*, 2015).

4.3.19 Soil meso-faunal population

Table 42 depicts the data on soil meso-faunal population of different land use systems across three zones. The soil meso-faunal population data was ranged from 0.33 individuals per sample in sealed areas of urban zone to 9.67 individuals per sample in forest plantations of rural zone.

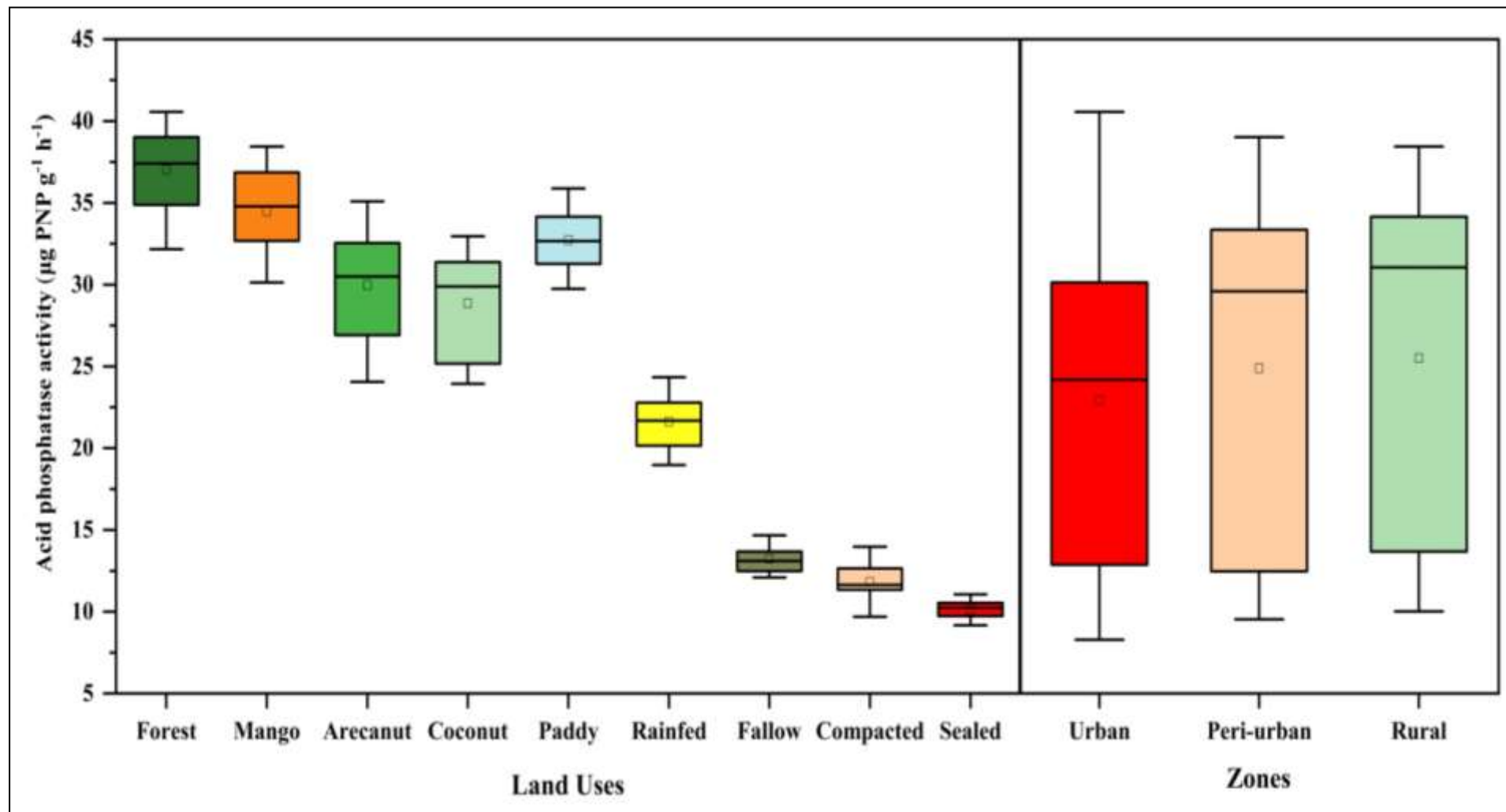


Fig. 43. Acid phosphatase activity ($\mu\text{g PNP g}^{-1} \text{h}^{-1}$) in soils of different land uses across the rural-urban gradient

Table 40. Microbial biomass-C (mg kg⁻¹) in soils of different land uses across the rural-urban gradient

Land uses (L)	Zones (Z)			Land uses (L) Mean
	Urban	Peri-urban	Rural	
Forest/forest plantations	184.0	178.0	162.0	174.7
Mango	138.0	154.0	178.0	156.7
Arecanut	126.0	140.0	152.0	139.3
Coconut	108.0	126.0	134.0	122.7
Paddy	118.0	138.0	142.0	132.7
Rainfed crop	96.0	113.0	126.0	111.7
Waste lands	90.0	106.0	111.0	102.3
Compacted area	78.0	87.0	93.0	86.0
Sealed area	66.0	60.0	61.0	62.3
Zones (Z) Mean	111.6	122.4	128.8	
Factors	S.Em±		CD @ 0.05	
Zones (Z)	0.75		2.11	
Land uses (L)	1.30		3.65	
Interaction (Z x L)	2.25		6.32	

Table 41. Microbial biomass-N (mg kg⁻¹) in soils of different land uses across the rural-urban gradient

Land uses (L)	Zones (Z)			Land uses (L) Mean
	Urban	Peri-urban	Rural	
Forest/forest plantations	37.10	34.77	34.07	35.31
Mango	24.27	28.23	31.73	28.08
Arecanut	24.27	30.10	33.60	29.32
Coconut	20.53	23.57	25.43	23.18
Paddy	22.63	26.83	29.40	26.29
Rainfed crop	17.03	19.13	23.10	19.76
Waste lands	11.70	12.03	13.30	12.34
Compacted area	11.07	11.97	13.37	12.13
Sealed area	10.03	8.17	8.23	8.81
Zones (Z) Mean	19.85	21.64	23.58	
Factors	S.Em±		CD @ 0.05	
Zones (Z)	0.26		0.74	
Land uses (L)	0.46		1.28	
Interaction (Z x L)	0.79		2.22	

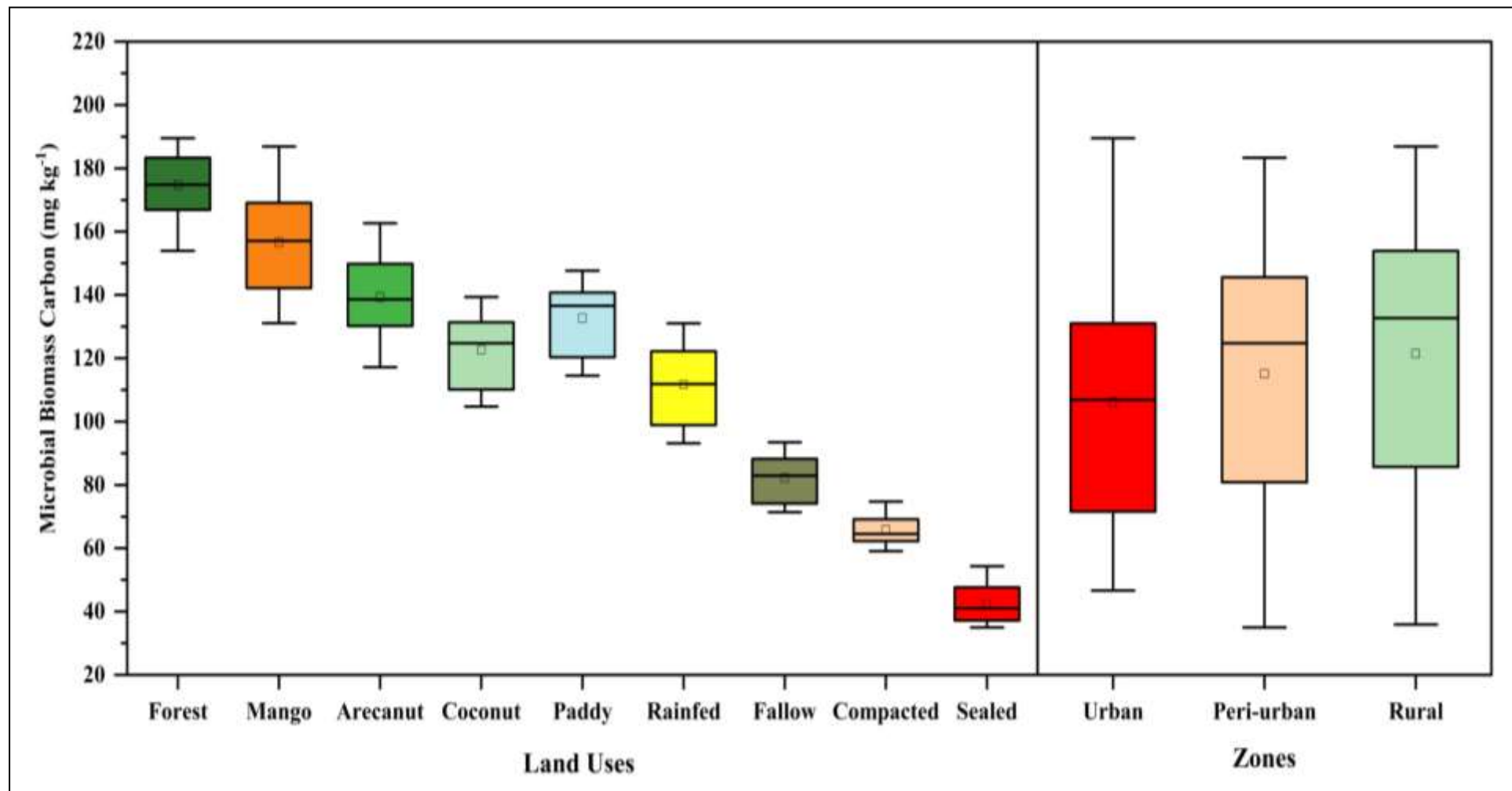


Fig. 44. Microbial biomass-C (mg kg⁻¹) in soils of different land uses across the rural-urban gradient

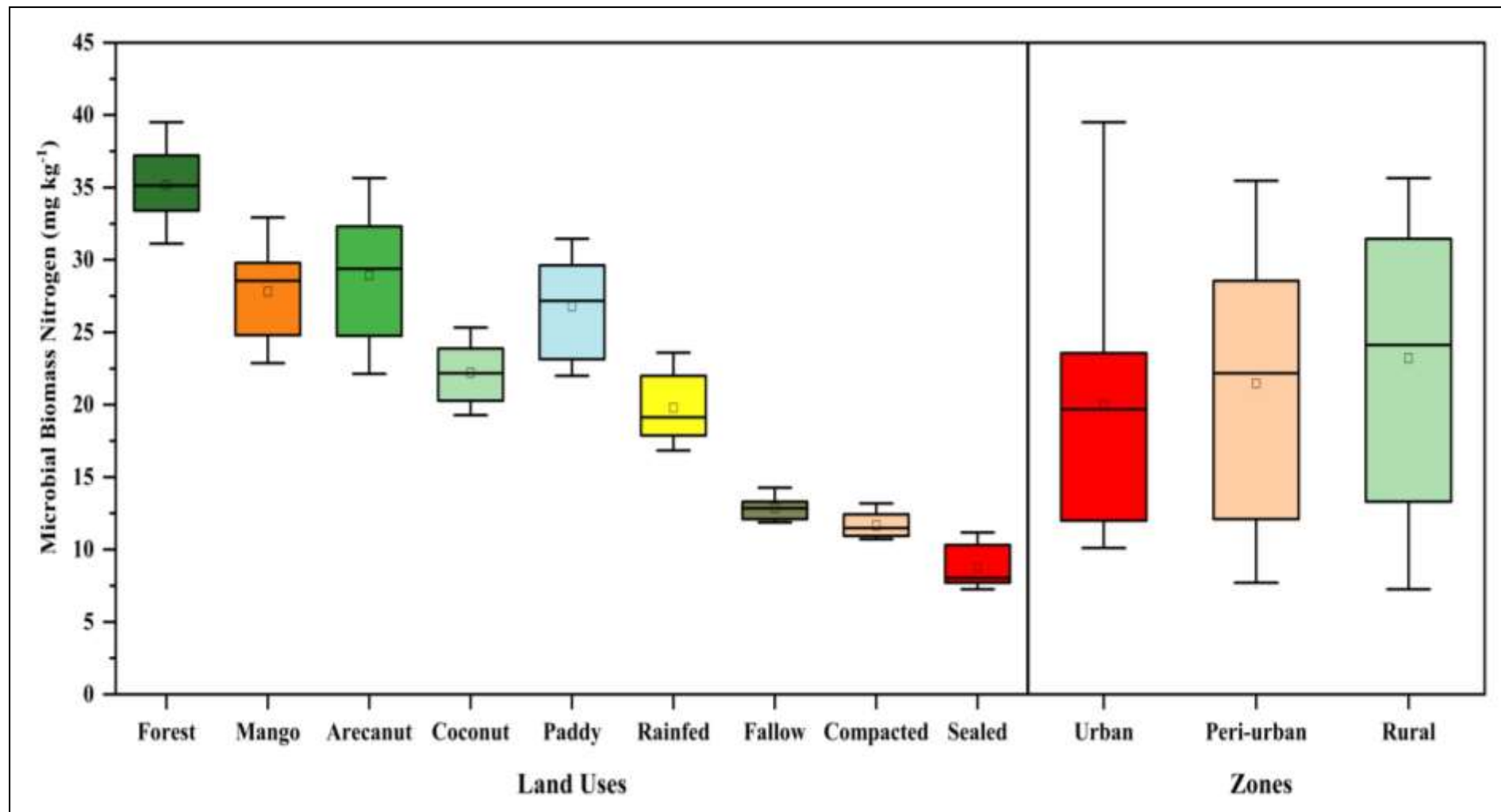


Fig 45. Microbial biomass-N (mg kg⁻¹) in soils of different land uses across the rural-urban gradient

Table 42. Meso-faunal population (number of individuals per sample) in soils of different land uses across the rural-urban gradient

Land uses (L)	Zones (Z)			Land uses (L) Mean
	Urban	Peri-urban	Rural	
Forest/forest plantations	7.67	8.33	9.67	8.56
Mango	7.00	6.67	8.33	7.33
Arecanut	6.33	7.33	8.67	7.44
Coconut	6.33	7.67	8.00	7.33
Paddy	6.04	7.33	7.67	7.01
Rainfed crop	6.00	6.67	7.33	6.67
Waste lands	1.57	1.67	2.33	1.86
Compacted area	0.68	0.83	1.00	0.84
Sealed area	0.33	0.33	0.67	0.44
Zones (Z) Mean	4.66	5.20	5.96	
Factors	S.Em±		CD @ 0.05	
Zones (Z)	0.12		0.33	
Land uses (L)	0.20		0.56	
Interaction (Z x L)	0.35		NS	

Among different land use systems, the soils of forest/forest plantations recorded significantly higher soil meso-faunal population (8.56 individuals per sample). All the agricultural-based land use systems exhibited significantly higher soil meso-faunal population. Among the agricultural-based land use systems, mango (7.33 individuals per sample) arecanut (7.44 individuals per sample) and coconut land use systems (7.33 individuals per sample) recorded significantly higher soil meso-faunal population. Paddy (7.01 individuals per sample) and rainfed cropping systems (6.67 individuals per sample) exhibited substantial amounts of soil meso-fauna. In contrast, the soils of sealed area (0.44 individuals per sample) and compacted area (0.84 individuals per sample) exhibited significantly lower soil meso-faunal population. Comparison among different land use systems across the zones revealed that the soils of rural zones exhibited significant variations in soil meso-faunal population (5.96 individuals per sample) over peri-urban (5.20 individuals per sample) and urban zones (4.66 individuals per sample).

Forest plantations recorded the highest meso-faunal densities, reflecting the critical role of vegetation cover and organic matter input in fostering a diverse soil biota (Fig. 46). These results align with findings from (Louwagie *et al.*, 2017; Braschler *et al.*, 2020), who emphasized the positive correlation between forest ecosystems and soil biodiversity. Mango plantations also recorded higher meso-faunal populations, though slightly lower than the forest plantations. This is likely due to organic amendments and minimal soil disturbance practices typically associated with horticultural land use systems (Huang *et al.*, 2022). Substantial amounts of soil meso-fauna in arecanut, coconut, paddy and rainfed cropping systems could be attributed to organic amendments, diverse rooting systems and frequent management practices that enrich soil meso-faunal population (Juceviča *et al.*, 2023). Sealed areas exhibited the lowest soil meso-faunal density. This can be attributed to the reduced organic matter, disrupted soil structure and diminished water infiltration capacity in sealed areas. Such conditions are inhospitable to meso-fauna, which depend on organic substrates and aerated soil microhabitats for survival (Braschler *et al.*, 2020). Rural zones exhibited the highest meso-faunal population, followed by peri-urban and urban zones. Soils of rural zone often benefit from lower pollution levels, lesser extent of soil sealing and greater organic matter inputs, thereby supporting a richer meso-faunal community (Yu *et al.*, 2021; Yu *et al.*, 2022).

4.4 Identification of key indices and develop management strategies to maintain soil ecological functions

4.4.1 Selection of soil quality indicators to evaluate soil quality index of different land use systems by PCA

The variables with significant differences among different land use systems ($p < 0.05$) were subjected to principal component analysis (PCA) in SPSS software (Version 12.0, SPSS Inc., Chicago IL). The results of PCA and per cent contribution of

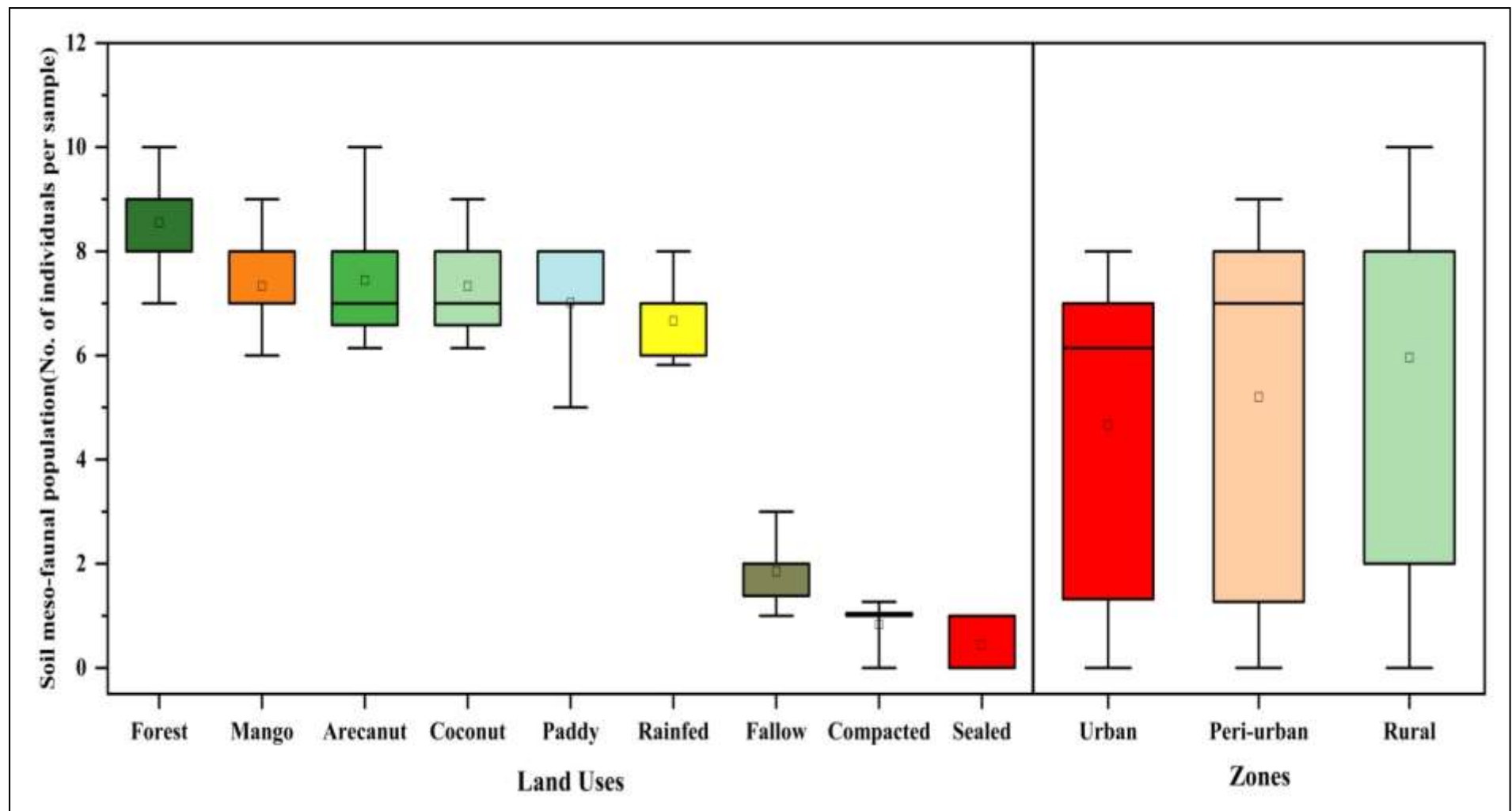


Fig. 46. Meso-faunal population (number of individuals per sample) in soils of different land uses across the rural-urban gradient

each soil attributes towards of different land use systems were indicated by communalities are presented in Table 43. The results of PC provided the three components with eigenvalues >1 and explaining at least 5 per cent of the variance in dataset and which together explains around 91.16 per cent variation in the entire data set.

The eigenvalue decreased under PC-1 to PC-3. Similarly, variance explained decreased under PC-1 to PC-3. The variation of 75.30, 10.09 and 5.77 per cent was explained by first PC, second PC and third PC, respectively in the data set. The Eigenvalues indicated by PC-1: 15.813, PC-2: 2.118 and PC-3: 1.002 (Fig. 47). Based on rotated factor loadings of soil attributes, selected indicators were: under PC-1, bulk density, porosity, SOC, available phosphorus, available S, urease, dehydrogenase, acid phosphatase, bacteria, fungi, actinomycetes, MBN, MBC and soil meso-fauna were the fourteen highest weighted variables with the rotated factor loadings of -0.839, 0.822, 0.933, 0.838, 0.876, 0.921, 0.922, 0.869, 0.916, 0.921, 0.925, 0.848, 0.908 and 0.933, respectively. Since these attributes (except bulk density) were significantly correlated to SOC (Fig. 48), to avoid any redundancy in indicator selection, only SOC and bulk density were selected on the basis of their highest factor loading.

Under PC-2, soil pH (-0.746), EC (-0.674) and exchangeable Ca (-0.942) were considered as highly weighted variables. There was strong correlation between soil pH and EC ($r = 0.88$) therefore the variable with high rotated factor loading was selected *i.e.*, soil pH (-0.746). However, exchangeable Ca exhibited weak correlation with soil pH and EC (Fig. 48). Hence, soil pH and exchangeable Ca were selected under PC-2. Whereas, zinc (0.874) was the only highly weighted variables in PC-3. Thus, key soil quality indicators selected from the PC were bulk density, SOC, soil pH, calcium and Zn which have been used as key indicators of soil quality by various other researchers (Mukherjee and Lal, 2014; Zhang *et al.*, 2014; Debi *et al.*, 2019, Raiesi and Tavakoli, 2022).

SOC and bulk density play a crucial role in determining soil quality. SOC is a key indicator of soil fertility as it influences nutrient availability, water retention and microbial activity, thereby supporting plant growth and maintaining ecosystem health (Topa *et al.*, 2021). Bulk density, on the other hand, reflects soil compaction and porosity, impacting root penetration, water infiltration and air exchange (Dörner, *et al.*, 2022). Soil pH affects nutrient solubility and microbial activity, determining the overall chemical balance of the soil (Naz *et al.*, 2022). Calcium contributes to soil structure by improving aggregation and reducing soil compaction, enhancing water and nutrient movement (Sharma and Kumar, 2023). Zinc is a critical micronutrient necessary for plant enzyme activation, growth and development. Its availability directly influences crop productivity and overall soil health (Nandal and Solanki, 2021).

Table 43. Principal Component Analysis (PCA) and communalities of soil quality parameters to evaluate soil quality index of different land uses of rural-urban gradients of Shivamogga

Rotated component matrix				
Component				
Soil Attributes	PC1	PC2	PC3	Communalities
Bulk density	<u>-0.839</u>	-0.232	-0.262	0.826
Porosity	0.822	0.231	0.218	0.777
pH	-0.576	<u>-0.746</u>	-0.111	0.901
EC	-0.652	-0.674	-0.265	0.950
SOC	<u>0.933</u>	0.031	0.463	0.909
P	0.838	0.015	0.277	0.779
Ca	0.222	<u>-0.942</u>	-0.043	0.938
S	0.876	0.257	0.255	0.898
Fe	0.417	0.595	0.580	0.864
Zn	0.230	0.285	<u>0.874</u>	0.898
Cu	0.734	0.510	0.323	0.903
TIC	0.671	-0.086	0.632	0.857
Urease	0.921	0.148	0.283	0.950
Dehydrogenase	0.922	0.116	0.305	0.956
Acid phosphatase	0.869	0.052	0.319	0.859
Bacteria	0.916	0.294	0.148	0.947
Fungi	0.921	0.298	0.096	0.947
Actinomycetes	0.925	0.258	0.199	0.961
MBN	0.848	0.101	0.418	0.904
MBC	0.908	-0.010	0.348	0.945
Soil Mesofauna	0.933	0.255	0.175	0.963
Maximum value	0.933	0.595	0.874	
10 per cent of the Max. value	0.0932	0.0595	0.0874	
Eigen Values	15.813	2.118	1.002	
Variance explained (%)	75.299	10.086	5.771	
Cumulative variance explained (%)	75.299	85.384	91.156	
Extraction method: Principal component analysis: Rotation method: Varimax with Kaiser normalization				

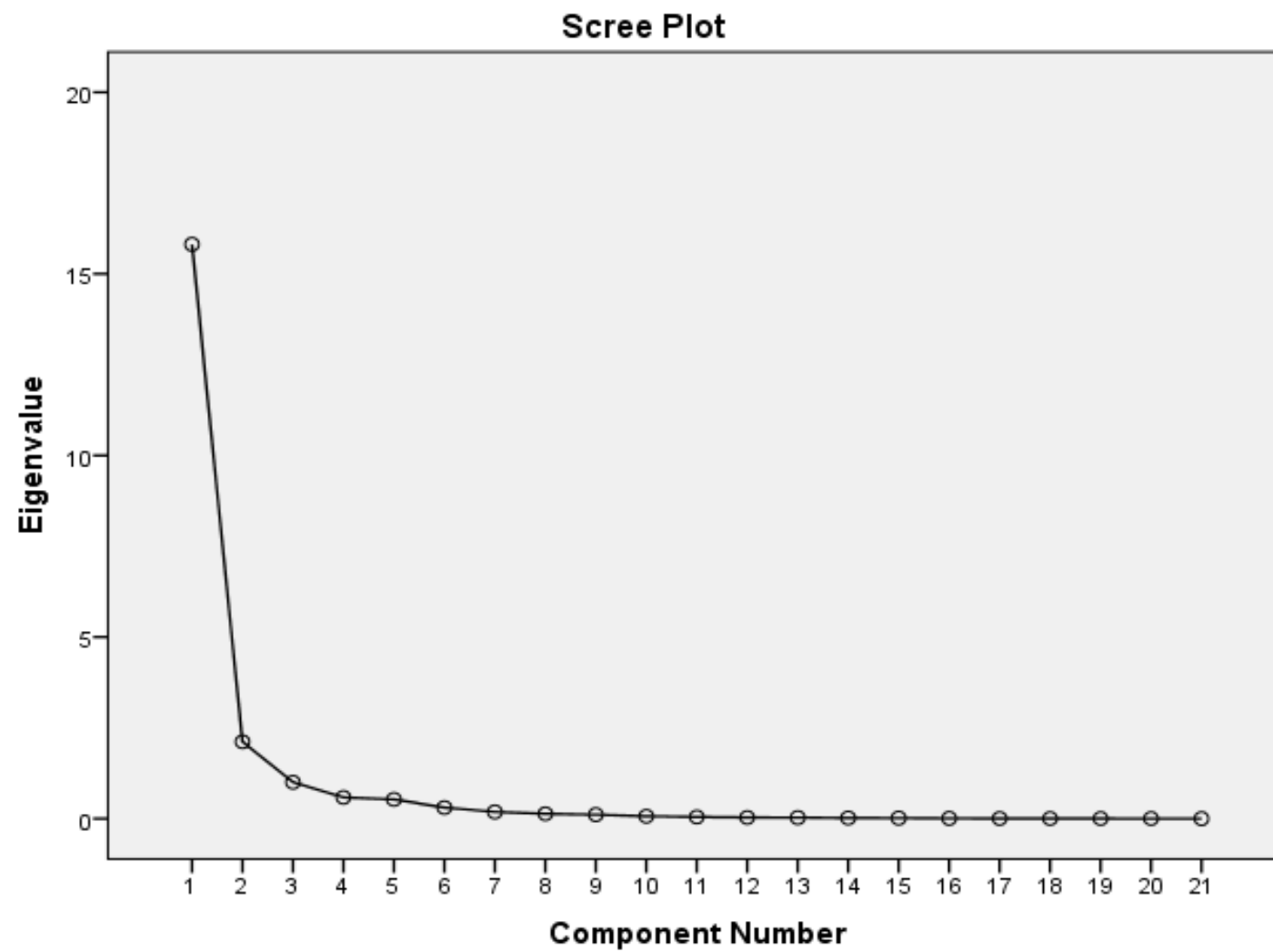


Fig. 47. Relationship between Eigenvalue and principal component analysis of different land uses of urban, peri-urban and rural zones

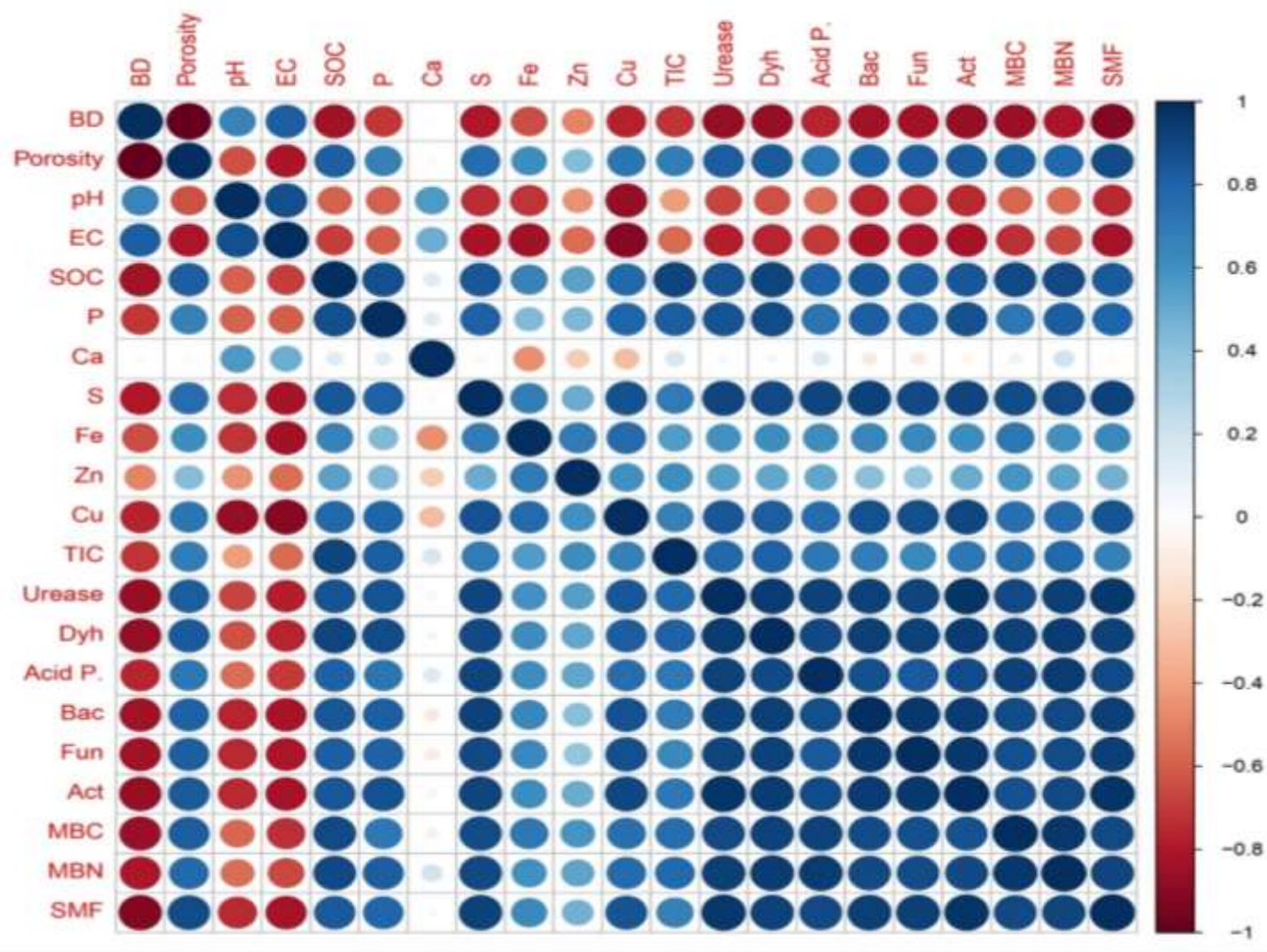


Fig. 48. Correlation of significant soil indicators in different land uses of urban, peri-urban and rural zones

4.4.2 Scoring of indicators

Because of different indicator units, linear scoring functions (Andrews *et al.*, 2002) were used to score soil indicators. “More is better” approach was followed for SOC, Ca and Zn, while “optimum is better” approach is followed for soil pH (6.5-7.5) and for bulk density “less is better approach” was employed. The score was then multiplied by the weighting factor derived from the PCA to obtain the soil quality index value for different land use systems of urban, peri-urban and rural zones. The weight of each PC on the basis of per cent variance to total variance ranged from 0.053 to 0.835. The weighted factor for each PC was: 0.835 (PC-1), 0.112 (PC-2) and 0.053 (PC-3).

4.4.3 Soil quality index of different land uses

The data pertaining to soil quality index (SQI) of different land uses across the zones is presented in Table 44 to 46. The SQI values for different land uses across urban, peri-urban and rural zones of Shivamogga reflected notable variations in soil quality influenced by land management practices and environmental factors. The SQI classification ranged from “very low” to “very high,” highlighting distinct patterns in soil quality.

In the urban zone, the SQI ranged from a minimum of 0.20 (very low) for sealed areas (SA) to a maximum of 0.59 (high) for forest/forest plantations (FP). Land uses such as FP, mango (M), arecanut (A) and paddy soils (P) consistently exhibited high soil quality, emphasizing the positive impact of vegetation cover and organic inputs (Sathish *et al.*, 2024). Conversely, categories like SA and compacted areas (CA) demonstrated “very low” SQI, attributed to soil sealing, compaction and reduced biological activity (Hyun *et al.*, 2022). Waste lands (WL) and rainfed crop (RC) were classified as “low” and “moderate,” respectively, reflecting limited vegetation and anthropogenic disturbances in WL and imbalanced cultural management in RC.

The peri-urban zone displayed a slightly improved SQI compared to the urban zone, with values ranging from 0.23 (very low) for SA to 0.59 (high) for FP and P. High soil quality was consistently observed in FP, M, A and P, indicating the importance of organic matter and minimal soil disturbance (Kuntoji and Subbarayappa, 2022). Moderate SQI in WL and “very low” values in CA and SA were indicative of degraded soils due to intermediate land-use practices and developmental activities (Hyun *et al.*, 2022).

The rural zone demonstrated the highest average SQI, ranging from 0.30 (very low) in SA to 0.64 (very high) in FP. The “very high” classification for FP, M and A is likely due to organic inputs and limited urban encroachment, which enhance soil properties (Séré *et al.*, 2024). Even the lower classifications, such as WL

Table 44. Soil quality index and relevant descriptive statistics for different land uses and their classification for urban zone

Land uses	SQI (mean)	Std. deviation	Std. Error	Min.	Max.	Soil quality classification
Forest/forest plantations	0.59	0.21	0.10	0.37	0.88	High
Mango	0.56	0.09	0.04	0.48	0.71	High
Areca nut	0.57	0.08	0.03	0.50	0.67	High
Coconut	0.55	0.17	0.08	0.43	0.85	High
Paddy	0.57	0.07	0.03	0.48	0.67	High
Rainfed crop	0.54	0.08	0.03	0.47	0.67	Moderate
Waste lands	0.42	0.12	0.05	0.25	0.58	Low
Compacted area	0.30	0.13	0.06	0.10	0.40	Very low
Sealed area	0.20	0.05	0.02	0.15	0.28	Very low

Table 45. Soil quality index and relevant descriptive statistics for different land uses and their classification for peri-urban zone

Land uses	SQI (mean)	Std. deviation	Std. Error	Min.	Max.	Soil quality classification
Forest/forest plantations	0.59	0.09	0.04	0.45	0.70	High
Mango	0.57	0.07	0.03	0.46	0.64	High
Areca nut	0.57	0.11	0.05	0.40	0.68	High
Coconut	0.56	0.19	0.09	0.22	0.71	High
Paddy	0.59	0.09	0.04	0.49	0.73	High
Rainfed crop	0.57	0.15	0.07	0.44	0.80	High
Waste lands	0.45	0.09	0.04	0.32	0.55	Moderate
Compacted area	0.35	0.08	0.03	0.25	0.43	Very low
Sealed area	0.23	0.07	0.03	0.14	0.30	Very low

Table 46. Soil quality index and relevant descriptive statistics for different land uses and their classification for rural zone

Land uses	SQI (mean)	Std. deviation	Std. Error	Min.	Max.	Soil quality classification
Forest/forest plantations	0.64	0.45	0.85	0.15	0.07	Very high
Mango	0.63	0.49	0.79	0.11	0.05	Very high
Areca nut	0.62	0.44	0.86	0.15	0.07	Very high
Coconut	0.59	0.49	0.76	0.11	0.05	High
Paddy	0.61	0.51	0.73	0.08	0.04	Very high
Rainfed crop	0.60	0.55	0.75	0.08	0.04	High
Waste lands	0.54	0.33	0.68	0.13	0.06	Moderate
Compacted area	0.38	0.27	0.60	0.13	0.06	Very low
Sealed area	0.30	0.20	0.39	0.09	0.04	Very low

(moderate) and CA (very low), performed relatively better than their urban counterparts, underscoring the lesser impact of urbanization in rural areas (Hyun *et al.*, 2022).

4.4.4 Soil quality index of different zones (urban, peri-urban and rural)

The SQI varied significantly across the urban, peri-urban and rural zones, reflecting the influence of land-use patterns and management practices on soil health. Table 47 revealed that the rural zone exhibited the highest mean SQI (0.55), followed by the peri-urban (0.50) and urban zones (0.48).

The rural zone's classification as "High" in soil quality indicated a more favorable environment for soil biological activity, nutrient availability and structure, supported by practices like organic amendments and crop rotation (Hyun *et al.*, 2022). In contrast, the urban zone, with a "Moderate" classification, reflected the adverse effects of urbanization, including soil compaction, reduced organic matter and potential contamination (Kuntoji and Subbarayappa, 2022). The peri-urban zone, also classified as "Moderate," likely represents a transition influenced by both urban and rural factors, such as mixed land use and intermediate levels of soil management (Hyun *et al.*, 2022).

Standard deviation values were relatively consistent across zones (0.16–0.17), suggesting moderate variability in soil quality within each zone. However, the broader range in the urban zone (minimum 0.10 to maximum 0.88) indicates a more heterogeneous soil environment, potentially driven by diverse land uses such as industrial areas, residential developments and urban green spaces (Hyun *et al.*, 2022). Similarly, the peri-urban zone (range: 0.14–0.80) reflects mixed influences, whereas the rural zone's narrower range (0.20–0.86) signified more uniform soil quality conditions (Sathish *et al.*, 2024).

These findings highlight the critical need for targeted soil management strategies. In urban and peri-urban areas, practices such as reducing soil sealing, promoting urban agriculture and enhancing organic matter inputs are essential to improve soil quality. Meanwhile, sustaining traditional, low-impact farming methods in rural zones can help maintain their high soil quality.

To mitigate the impacts of land sealing on soil ecological functions, several strategies can be implemented, particularly in urban planning and development (Vieillard *et al.*, 2024). Incorporating green infrastructure, such as green roofs, permeable pavements and urban green spaces, can help reduce the extent of land sealing and promote soil health in urban areas. These practices allow for water infiltration, support vegetation and enhance biodiversity. In areas where land has already been sealed, soil restoration efforts, such as decompaction, re-vegetation and the addition of organic matter, can help to restore some of the lost soil functions (Vieillard *et al.*, 2024).

Table 47. Soil quality index and relevant descriptive statistics for urban, peri-urban and rural zones

Zones	SQI (mean)	Std. deviation	Std. Error	Min.	Max.	Soil quality classification
Urban	0.48	0.17	0.03	0.10	0.88	Moderate
Peri-urban	0.50	0.16	0.02	0.14	0.80	Moderate
Rural	0.55	0.16	0.02	0.20	0.86	High

Conclusion

The assessment of soil ecological functions and processes along the rural-urban gradient in Shivamogga District revealed significant variations in soil attributes influenced by land use and anthropogenic activities.

The analysis of land use and soil sealing along the rural-urban gradient in Shivamogga city highlighted the substantial transformations associated with urbanization. Urban zones exhibited the highest extent of sealed areas, driven by infrastructure and population density, while rural areas retained significant vegetative and agricultural land cover. The peri-urban zone, acting as a transitional landscape, presented a mosaic of land uses reflecting both urban influences and rural practices. These findings emphasize the spatial heterogeneity in land use patterns and the critical impact of urban expansion on soil sealing and ecological functions. The distinct shifts in land allocation and the extent of impervious surfaces across the gradient underline the necessity of strategic land management to balance urban growth with the preservation of ecological integrity.

The results of biomass and nutrient turnovers underscore the significant variability in biomass and nutrient turnovers across land use systems and zones along the rural-urban gradient. Arecanut plantations consistently emerged as the most efficient land use for biomass and nutrient recycling, attributed to their perennial nature and high organic inputs. Forest plantations also showed robust ecological stability and nutrient turnover, emphasizing their critical role in sustaining soil health. Agricultural systems such as coconut, paddy and mango plantations demonstrated substantial biomass and nutrient recycling due to active management practices and periodic organic matter inputs. Conversely, sealed and compacted areas exhibited the lowest rates of organic matter and nutrient turnover, highlighting the adverse effects of urbanization and soil sealing on soil ecological functions. Rural zones exhibited the highest levels of biomass and nutrient recycling, followed closely by peri-urban zones, while urban areas lagged due to higher impervious surfaces and anthropogenic pressures. These findings reinforce the importance of sustainable land management practices to enhance soil productivity and ecological stability, particularly in peri-urban and urban landscapes undergoing rapid transformation.

Across three zones, soil bulk density, porosity, pH and nutrient content demonstrated significant variations influenced by land-use practices and anthropogenic pressures. Urban zones exhibited higher bulk density and pH levels, attributed to compaction and construction activities, while rural zones retained higher porosity and organic matter due to minimal disturbance and traditional farming practices. Soil sealing drastically reduced nutrient cycling, organic carbon levels, microbial populations and meso-faunal densities, highlighting its detrimental effects on soil

health. In contrast, forest and agricultural systems such as arecanut and paddy consistently supported higher biological and biochemical activity, reflecting their critical role in maintaining soil ecological balance.

The results of the Principal Component Analysis (PCA) revealed a set of critical soil quality indicators that comprehensively reflect variations in soil functions across different land use systems along the rural-urban gradient. These indicators, which capture over 91% of the variance in the dataset, emphasize the multi-dimensional impact of anthropogenic activities on soil quality. The identification of these indices offers a robust framework for assessing and monitoring soil ecological health under varying land use practices. The comparative analysis across zones revealed that rural areas generally sustained better soil quality indicators, while urban areas experienced severe ecological degradation. These findings emphasize the need for targeted soil management practices to mitigate the adverse effects of urbanization. Strategies such as promoting green infrastructure, minimizing soil sealing and enhancing organic matter inputs can help to preserve soil functions, particularly in urban and peri-urban zones.

Furthermore, these insights provide the foundation for developing tailored management strategies. Such strategies can mitigate the adverse effects of urbanization on soil functions, promote sustainable agricultural practices and enhance ecological resilience. By integrating the identified indicators into soil quality assessment protocols, stakeholders can better prioritize interventions and optimize land management to sustain soil ecological functions in rural, peri-urban and urban zones.

Future line of work

- Develop predictive models to assess long-term impacts of urbanization on soil ecological functions across diverse rural-urban gradients.
- Investigate seasonal dynamics of soil properties and nutrient cycles to enhance temporal understanding of ecological processes.
- Explore sustainable urban planning techniques that mitigate soil sealing and promote ecological restoration in urban zones.
- Study pollutant dynamics in urban soils to understand their impact on microbial activity and soil ecological functions.

SUMMARY

V SUMMARY

The study titled “Assessment of Soil Ecological Functions and Processes along the Rural-Urban Gradient in Shivamogga District” was undertaken to evaluate effect of urbanization intensity on variations in land-use patterns, extent of soil sealing, litter biomass turnovers, nutrient turnovers, soil physical, chemical and biological parameters and their influence on soil ecological functions across urban, peri-urban and rural zones. The results duly obtained in this present study are summarized in this chapter.

I. To assess the magnitude of different land use systems and extent of land sealing along rural-urban gradient using satellite data

The LULC map of the study area describing the extent of soil sealing and magnitude of different land uses with a kappa coefficient of 0.94 was derived using the combination of FCC, NDVI, NDWI and NDBI.

Urban zone exhibited the higher extent of sealed areas (230.79) compared to open lands and agricultural systems. Forest/forest plantations (1.5 ha) and water bodies (9.96 ha) occupied limited area. Agricultural and horticultural land use was constrained with area for arecanut (9.47 ha), mango (2.18 ha), paddy (4.83 ha) and rainfed crops (26.75 ha) being relatively small. Peri-urban zone presented a transitional landscape between urban and rural zone. The extent of soil sealing was reduced to 108.58 ha. Agricultural and horticultural-based land uses were dominated in peri-urban zones with arecanut (31.72 ha), coconut (33.27 ha), rainfed crops (81.86 ha) and paddy (19.42 ha) occupying substantial portions of the peri-urban zone. The rural zone was dominated by agricultural and horticultural-based land uses. Rainfed crops occupied the largest portion of the rural zone (123.86 ha). Significantly high proportion of paddy (49.13 ha), coconut (54.90 ha) and arecanut (40.98 ha) land use systems were present compared to urban and peri-urban zones. Sealed areas were found to be minimal (23.29 ha).

II. To quantify biomass and nutrient turnovers in different land use systems along rural-urban gradient

The quantity of organic matter recycled through biomass and FYM varied significantly across zones and land use systems. Forest plantations exhibited highest biomass recycling with values ranging from 9.01 to 9.38 Mg ha⁻¹ yr⁻¹. Among man-made systems, arecanut contributed the highest net organic matter recycling (10.65 Mg ha⁻¹ yr⁻¹) followed by mango (9.01 Mg ha⁻¹ yr⁻¹). Urban zones showed reduced recycling rates across most land uses compared to peri-urban and rural zones.

With respect to N-turnover, forest/forest plantations exhibited higher turnover (90.99 kg ha⁻¹ yr⁻¹) followed by arecanut plantations (89.25 kg ha⁻¹ yr⁻¹). Lower N turnovers were recorded in sealed and compacted areas. A similar trend was recorded in P turnover with forest/forest plantations exhibited higher P turnovers

(32.83 kg ha⁻¹ yr⁻¹) followed by arecanut plantations (29.03 kg ha⁻¹ yr⁻¹). Coconut plantations also displayed notable amount of P turnovers (23.54 kg ha⁻¹ yr⁻¹). Sealed and compacted areas recorded lower P turnovers. Arecanut plantations recorded high amounts of K turnover (68.13 kg ha⁻¹ yr⁻¹) followed by paddy systems (61.89 kg ha⁻¹ yr⁻¹). Other agricultural-based land uses (mango, coconut and rainfed crop) also recorded substantial amounts of K turnovers. However, they were significantly lesser. Among the zones, rural zone exhibited higher nutrient turnovers followed by peri-urban and urban zone.

III. To assess the impacts of land sealing on soil ecological functions along rural-urban gradient

Among different land uses, the soils of forest/forest plantations recorded significantly lesser values of bulk density (1.25 Mg m⁻³). In contrast, the soils of sealed area (1.70 Mg m⁻³) and compacted area (1.62 Mg m⁻³) recorded significantly higher bulk densities. Among three zones, the soils of urban zone exhibited significantly higher bulk density (1.42 Mg m⁻³) and was on par with peri-urban zone (1.41 Mg m⁻³). Whereas, the soils of rural (1.39 Mg m⁻³) recorded lower bulk density. Particle density did not vary significantly across different land uses and zones. Porosity varied significantly across the zones and land uses. Forest soils recorded higher porosity (51.99 %) followed by coconut plantations (50.66 %). Lower porosity was observed in sealed (35.75 %) and compacted areas (38.50 %). Among zones, porosity was found to be in the order rural zone > peri-urban zone > urban zone.

The soils in the study area were found to be slightly acidic to neutral range. Among different land use systems, the paddy soils were found relatively more acidic (5.41) followed by rainfed agricultural systems (5.64). The soils of sealed areas exhibited soil pH. Among three zones, higher soil pH was found in the order urban zone > peri-urban zone > rural zone. Sealed areas recorded the higher conductivity (1.11 dS m⁻¹) followed by compacted areas (0.82 dS m⁻¹). Lower EC values were noted in rainfed (0.55 dS m⁻¹) and paddy systems (0.56 dS m⁻¹). Urban zone had slightly higher EC than the soils of rural zone and the EC did not vary significantly between peri-urban and rural zone. The SOC varied significantly across different land uses. Among the land use systems, the soils of forest/forest plantations recorded higher SOC (6.83 g kg⁻¹) and lower SOC was recorded in sealed areas (2.72 g kg⁻¹). Among three zones, rural zone exhibited higher SOC (5.17 g kg⁻¹) and urban zone recorded lower SOC (4.56 g kg⁻¹).

Forest/forest plantations exhibited significantly higher available N (258.72 kg ha⁻¹). Contrastingly, sealed areas registered lower available N (82.88 kg ha⁻¹). Soil available N did not vary significantly across three zones. Soils of arecanut plantations (63.74 kg ha⁻¹) recorded significantly higher amount of available

P (57.98 kg ha^{-1}) followed by paddy soils (62.75 kg ha^{-1}) and lower available P was found in sealed areas (12.64 kg ha^{-1}). Urban zone recorded higher available P (46.06 kg ha^{-1}) than rural (44.67 kg ha^{-1}) and peri-urban zones (43.85 kg ha^{-1}). Similar to available N and P, the arecanut growing soils also recorded significantly higher amounts of available K ($244.01 \text{ kg ha}^{-1}$). Lower available K was recorded in sealed areas (46.32 kg ha^{-1}). Soil available K did not vary significantly across zones.

Arecanut plantations exhibited significantly higher soil exchangeable Ca [$5.83 \text{ cmol(p}^+) \text{ kg}^{-1}$] followed by the soils of sealed areas [$5.37 \text{ cmol(p}^+) \text{ kg}^{-1}$]. Among the three zones, soils of urban zones recorded significantly higher amount of exchangeable Ca [$4.90 \text{ cmol(p}^+) \text{ kg}^{-1}$]. Significantly higher amount of exchangeable Mg was found in arecanut plantations [$2.17 \text{ cmol(p}^+) \text{ kg}^{-1}$] and rainfed crop recorded lower soil exchangeable Mg [$1.47 \text{ cmol(p}^+) \text{ kg}^{-1}$]. Soil exchangeable Mg did not vary significantly across the zones. The paddy soils recorded significantly higher amounts of available S (23.44 mg kg^{-1}) compared to other land use systems. Contrastingly, lower available S was noticed in sealed areas (8.83 mg kg^{-1}). Rural zone exhibited higher available S (19.82 mg kg^{-1}) compared to peri-urban (18.43 mg kg^{-1}) and urban zone (17.42 mg kg^{-1}).

Among different land use systems, significantly higher amount of DTPA extractable micronutrients was recorded in soils of forest/forest plantations, followed by the soils of mango plantations. Whereas, the sealed and compacted areas recorded significantly lower DTPA extractable micronutrients. Across zone, soils of rural zone exhibited higher micronutrient status compared to urban and peri-urban zones. Among different land use systems, forest/forest plantations recorded significantly higher amounts of total organic-C (7.31 g kg^{-1}), total-C (7.65 g kg^{-1}). However, lower total organic-C and total-C were recorded in sealed areas 2.87 g kg^{-1} and 3.15 g kg^{-1} , respectively. Whereas, arecanut land use systems exhibited significantly higher values of total inorganic-C (0.34 g kg^{-1}) and even sealed areas recorded considerable amount of total inorganic-C (0.29 g kg^{-1}). No significant variations were recorded with respect to total organic-C and total-C across the zones. However, urban zones recorded higher total-inorganic-C (0.29 g kg^{-1}) over peri-urban (0.28 g kg^{-1}) and rural zone (0.24 g kg^{-1}). Among the different land use systems, forest/forest plantations recorded significantly higher amounts of ammonical-N (38.81 mg kg^{-1}), nitrate-N (19.40 mg kg^{-1}), organic-N ($363.65 \text{ mg kg}^{-1}$), total-N ($421.87 \text{ mg kg}^{-1}$) and lowest ammonical-N (11.67 mg kg^{-1}), nitrate-N (7.34 mg kg^{-1}), organic-N ($103.15 \text{ mg kg}^{-1}$), total-N ($122.16 \text{ mg kg}^{-1}$) in sealed areas.

Paddy soils exhibited significantly higher bacterial population ($11.10 \times 10^6 \text{ CFU g}^{-1}$), forest/forest plantation soils exhibited significantly higher fungal ($11.25 \times 10^4 \text{ CFU g}^{-1}$) and actinomycetes population ($8.11 \times 10^3 \text{ CFU g}^{-1}$). Sealed areas

showed lower bacterial (3.37×10^6 CFU g⁻¹), fungal (2.29×10^4 CFU g⁻¹) and actinomycetes population (1.58×10^3 CFU g⁻¹). Among the zones, the soils of rural zones exhibited significantly higher microbial populations over peri-urban and urban zone. Forest plantations had significantly higher urease activity ($9.22 \mu\text{g NH}_4^+\text{-N g}^{-1} \text{ h}^{-1}$), dehydrogenase activity ($8.10 \mu\text{g TPF g}^{-1} 24 \text{ h}^{-1}$) and acid phosphatase activity

($37.04 \mu\text{g PNP g}^{-1} \text{ h}^{-1}$). Lower urease ($2.54 \mu\text{g NH}_4^+\text{-N g}^{-1} \text{ h}^{-1}$), dehydrogenase ($1.30 \mu\text{g TPF g}^{-1} 24 \text{ h}^{-1}$) and acid phosphatase activity ($10.11 \mu\text{g PNP g}^{-1} \text{ h}^{-1}$) was recorded in sealed areas. Urban, peri-urban and rural zones did not significantly influence the soil enzymatic activities.

Soil microbial biomass carbon (SMB-C) and nitrogen (SMB-N) revealed significant variations across land use systems and zones. Forest plantations recorded the highest SMB-C (174.7 mg kg^{-1}) and SMB-N (35.31 mg kg^{-1}), while sealed areas exhibited the lower microbial biomass values (SMB-C: 62.3 mg kg^{-1} ; SMB-N: 8.81 mg kg^{-1}). Across zones, rural soils had the highest SMB-C (128.8 mg kg^{-1}) and SMB-N (23.58 mg kg^{-1}), followed by peri-urban (SMB-C: 122.4 mg kg^{-1} ; SMB-N: 21.64 mg kg^{-1}) and urban zones (SMB-C: 111.6 mg kg^{-1} ; SMB-N: 19.85 mg kg^{-1}). Soil meso-faunal population across land use systems revealed the highest population in forest plantations (8.56 individuals per sample). Sealed areas (0.44) and compacted areas (0.84) had significantly lower populations. Rural zone exhibited higher soil meso-faunal populations (5.96 individuals per sample) compared to peri-urban (5.20 individuals per sample) and urban zone (4.66 individuals per sample).

IV. To identify key indices and develop management strategies to maintain soil ecological functions

The Soil Quality Index (SQI) of different land use systems across urban, peri-urban and rural zones of Shivamogga varied significantly due to land management and environmental factors. SQI ranged from "very low" to "very high," reflecting distinct patterns.

In urban areas, sealed areas (SA) and compacted areas (CA) had "very low" SQI (0.20), while forest/plantations (FP) and crops like mango (M) and paddy (P) exhibited high SQI (0.59) due to vegetation and organic inputs. Peri-urban zones showed improved SQI, with FP and P maintaining high values (0.59). However, SA and CA remained degraded with "very low" SQI. Rural zones had the highest average SQI, ranging from 0.30 (SA) to 0.64 (FP). Forests, mango and arecanut (A) achieved "very high" SQI due to minimal disturbance and organic inputs. Waste lands (WL) and rainfed crops (RC) performed better in rural areas, highlighting lesser urbanization impacts. The SQI of urban, peri-urban and rural zones of Shivamogga varied significantly due to the variation in intensity of urbanization across the zones. The rural

zone exhibited the highest mean SQI (0.55), followed by the peri-urban (0.50) and urban zones (0.48).

The study demonstrated that urbanization leads to increased soil sealing, reducing organic matter turnover, nutrient recycling, microbial activity and soil ecological functions. Rural zones maintained higher ecological functions due to lower anthropogenic disturbances, whereas peri-urban areas act as transitional zones with mixed characteristics. The findings highlight the importance of sustainable land management practices to mitigate soil degradation due to urban expansion. Furthermore, these insights provide the foundation for developing tailored management strategies. Such strategies can mitigate the adverse effects of urbanization on soil functions, promote sustainable agricultural practices and enhance ecological resilience. By integrating the identified indicators into soil quality assessment protocols, stakeholders can better prioritize interventions and optimize land management to sustain soil ecological functions in rural, peri-urban and urban zones.

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