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ORIGINAL PAPER

Microplastic Accumulation and Soil Quality Changes Under Varying Plastic Mulching Durations in the Mediterranean Region

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Abstract

Research on microplastics (MPs) in soils has been widespread, but their impact on soil quality and the effects of mulching duration are still unclear. Therefore, the aim of this study is to determine how MP accumulation in mulched agricultural soils affects soil quality across three different mulching durations. Soil samples were collected from two depths (0–10 and 10–20 cm) at 15 agricultural sites in Adana, Türkiye, representing mulching durations of 5, 10, and 30 years. Microplastics were extracted using density separation and identified with ATR-FTIR. Soil physical, chemical, and biological properties were analyzed, and Soil Quality Index (SQI) was assessed with the Soil Management Assessment Framework (SMAF), which considers soil taxonomy, climate class, slope, sampling time, texture, mineralogy, organic matter, and analytical techniques to estimate the functional potential of each soil, considering the relevant soil indicators under MP accumulation. In this study, the evaluated soil indicators included physical properties such as bulk density (BD), aggregate stability (AS), available water content (AWC), and water-filled pore space (WFPS); chemical properties such as soil pH, electrical conductivity (EC), available phosphorus (P), and exchangeable potassium (K); and biological properties such as soil organic carbon (SOC), microbial biomass carbon (MBC), and potentially mineralizable nitrogen (PMN). Our data showed clear differences in both MPs and SQI across the three mulching periods, allowing the observed patterns to be better understood. The results showed that prolonged mulch application substantially increased MP accumulation and negatively affected key physical and biological soil indicators. Long-term mulching consistently reduced SQI values, indicating a cumulative decline in soil health and ecosystem functioning. At 0–10 cm depth, MP abundances were 69.3 ± 18.2, 64.0 ± 14.39, and 48.0 ± 9.53 particles kg^{-1} for short-, medium-, and long-term durations, respectively, while at 10–20 cm depth, the values were 37.33 ± 6.18, 48.0 ± 9.75, and 78.66 ± 27.76 particles kg^{-1}. These depth-specific patterns were accompanied by increases in bulk density (10%), substantial reductions in aggregate stability (48%), nitrogen mineralization (24%), and microbial biomass carbon (14%) with increasing plastic use (from S to L) at 0–10 cm depth. Overall, these analytical outcomes corresponded with a 17% reduction in SQI under long-term mulch application. These findings provide a solid foundation for predicting and monitoring MP contamination in agricultural soils with different mulch durations. The main limitation of this study is that, as a field-based investigation conducted under real agricultural conditions, it lacks full control over environmental and management variability, which should be considered when interpreting the results.

Keywords Plastic debris · Soil health indicators · SMAF approach · Pollution

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1 Introduction

Plastics, made from synthetic polymers, are widely used across many industries. As industrial activity expands in developing regions, plastic waste pollution has become a significant issue, with over 6,300 million tons generated by 2015 (Naz et al. 2024). Increased production and use have resulted in detection of higher plastic residues in the environment (MacLeod et al. 2021). If current trends continue, landfills and natural ecosystems together are projected to contain nearly 12 billion tons of plastic waste within the next three decades (Geyer et al. 2017). In landfills, this accumulation is an expected outcome of waste deposition; however, in natural soils, such accumulation is undesirable, as plastics can gradually fragment into microplastics (MPs) (less than 5 mm) through mechanical abrasion, ultraviolet radiation, and microbial activity (Lv et al. 2024). Unlike landfill areas, where plastics remain relatively contained, MPs disperse widely in natural ecosystems from agricultural soils to aquatic systems and even remote polar regions (Akdemir et al. 2025; Isobe et al. 2017). Consequently, the threats of MP pollution to ecosystem stability have become a major global concern, particularly in terrestrial environments, where soils act as the primary sink and accumulate plastic residues in even greater quantities than the oceans each year (Cao et al. 2023; Huffer et al. 2019). However, reported concentrations of MPs in mulched soils vary widely across regions. Globally, studies have demonstrated that agricultural practices, particularly the use of plastic mulches, are contributing to elevated MP concentrations in soils. An extensive review highlighted that agricultural and horticultural sites frequently experience contamination due to factors such as sewage sludge application and remnants from plastic mulching practices (Büks and Kaupenjohann 2020). For instance, a study indicated that residues from plastic mulches significantly contribute to the abundance of MPs in agricultural lands, showing a higher concentration of MPs (1412 ± 529 particles kg^{-1}) in mulched fields compared to control sites (72 ± 41 particles kg^{-1}) (Kumari and Chakraborty 2024). Focusing on the Mediterranean region, there is increasing concern regarding the specific impacts of MP pollution due to its semi-closed nature, which exacerbates the accumulation of pollutants. Research indicates that agricultural activities, combined with the unique climatic conditions of the Mediterranean, contribute significantly to MP burdens within this basin (De Ruijter et al. 2019). Coastal ecosystems in the Mediterranean are at risk, as studies reveal that these areas can display marked seasonal variations in MP concentrations, influenced by hydrological factors (Gündoğdu et al. 2025). Events such as heavy rainfall and subsequent agricultural runoff can lead to fluctuating concentrations, where limited water exchange can result in higher MP accumulation during dry seasons, mirroring trends observed in other regions (Gündoğdu et al. 2025). In Türkiye, Akça et al. (2022) reported 75.5–377.3 particles kg^{-1} in Konya City, whereas in Adana it was ~17 particles kg^{-1} (Gundogdu et al. 2022).

Adana is a major agricultural production center in the Mediterranean Region of Türkiye. The extensive use of agricultural mulch in the agricultural production locations (Akca et al. 2024; Gundogdu et al. 2022) creates locations with high concentration of MP contamination. The widespread use of plastic mulch in intensive agricultural systems has led to increased MP residues in soils. The high adoption of plastic mulching is due to enhanced crop productivity as it is cost-effective, lightweight, easy to install and manage, and durable (Vox et al. 2016). Serrano-Ruiz et al. (2021) indicated that plastic mulching also increases soil temperature, modifies soil properties, and accelerates harvest and crop development. The low recovery rate of applied PE mulch has resulted in significant residual film pollution in Adana's soils. Prolonged mulching in agricultural fields increases MP load, which negatively affects soil quality (Yang et al. 2021). For this reason, soil-biodegradable mulch films have increasingly been developed, studied, and applied as sustainable alternatives to conventional plastics (Hajilou et al. 2024; Payanthoth et al. 2024). However, the risks of micro-bioplastic residues from soil biodegradable plastics and their effects on soil quality and biodiversity require further research (Sadeleer and Woodhouse, 2024; Gündoğdu et al. 2026). The accumulation of MPs in soil ecosystems has far-reaching ecological implications, extending beyond soil degradation to the broader food web. MPs from various sources disrupt water-stable aggregates, reduce soil porosity, and alter microbial community structure (Qi et al. 2020a). These disruptions hinder essential soil processes including root growth, water retention, and nutrient cycling and ultimately threaten agroecosystem sustainability (Qi et al. 2018). Soil organisms play a key role in this process: micro- and mesofauna, particularly earthworms, act as bioindicators of soil health and pollutant accumulation. Recent findings show that MPs and biodegradable mulch residues can affect earthworm activity, soil ecotoxicity, and microbial community composition (Francioni et al. 2025), highlighting a direct pathway for MP transfer within the soil food web. Moreover, MPs can move upward through trophic levels, reaching crops and, ultimately humans (Gómez-Pliego et al. 2025; Horton et al. 2017). These observations underscore the importance of evaluating MP-driven changes using systematic approaches. Assessing soil quality in agricultural soils contaminated with MPs has received special interest in environmental management and soil science. The Soil Management Assessment Framework (SMAF) provides an effective method for evaluating these effects by integrating key soil quality indicators across physical, chemical, and biological properties using non-linear scoring (Karlen et al. 2013). SMAF considers factors such as soil taxonomy, climate class, slope, sampling time, soil texture, mineralogy, organic matter, and analytical techniques to estimate the functional potential of each soil (Nunes et al. 2020). Previous studies have successfully applied SMAF to evaluate management decisions and the effects on land use change to maintains sustainability (Celik et al. 2021; da Luz et al. 2019; Pacci et al. 2024). SMAF can also systematically measure soil function deterioration and identify thresholds where soil quality is severely degraded in mulched fields with known MP contamination. Soil quality assessment is essential for understanding how agricultural practices influence soil functions and sustainability. Widely used frameworks such as the SMAF, and the Cornell Soil Health Assessment (CASH) combine physical, chemical, and biological indicators to evaluate soil health comprehensively. Introducing these internationally recognized methods helps contextualize how mulching duration affects soil degradation or improvement and aligns local findings with global standards (Alaoui et al. 2020; Idowu et al. 2009; Vincent-Caboud et al. 2019).

Despite growing interest in MP contamination in agricultural soils, the combined effects of mulching duration on MP accumulation and on the integrated physical, chemical, and biological indicators of soil quality remain insufficiently understood. Additionally, the relationship between MPs in various soil layers and the duration of mulching practices has not been adequately investigated (Khan et al. 2023; Kim et al. 2021; Tian et al. 2025). Accordingly, this study addresses the following research question: How does the duration of plastic mulch application influence MP accumulation across soil depths, and how do these changes affect soil physical, chemical, and biological quality indicators? Based on this question, we hypothesized that increasing MP accumulation resulting from longer mulch application durations would lead to measurable declines in soil physical, chemical, and biological indicators, ultimately reducing overall soil quality.

The present study examines how mulching duration affects soil quality focusing on the accumulation of MPs from residual mulching materials in fruit and vegetable cultivation areas in the Mediterranean Region, Adana, Türkiye. The objectives are to assess: (i) MP distribution across mulching durations and soil depths; (ii) how key soil properties respond to increasing MP loads; and (iii) the relationship among MP accumulation, mulching duration, and soil quality indices.

2 Materials and Methods

2.1 Study Area and Sample Collection

The study area is in Adana, located in the Mediterranean Region of Türkiye (Fig. 1), and the geographic coordinates and agricultural area sizes of the sampling sites are

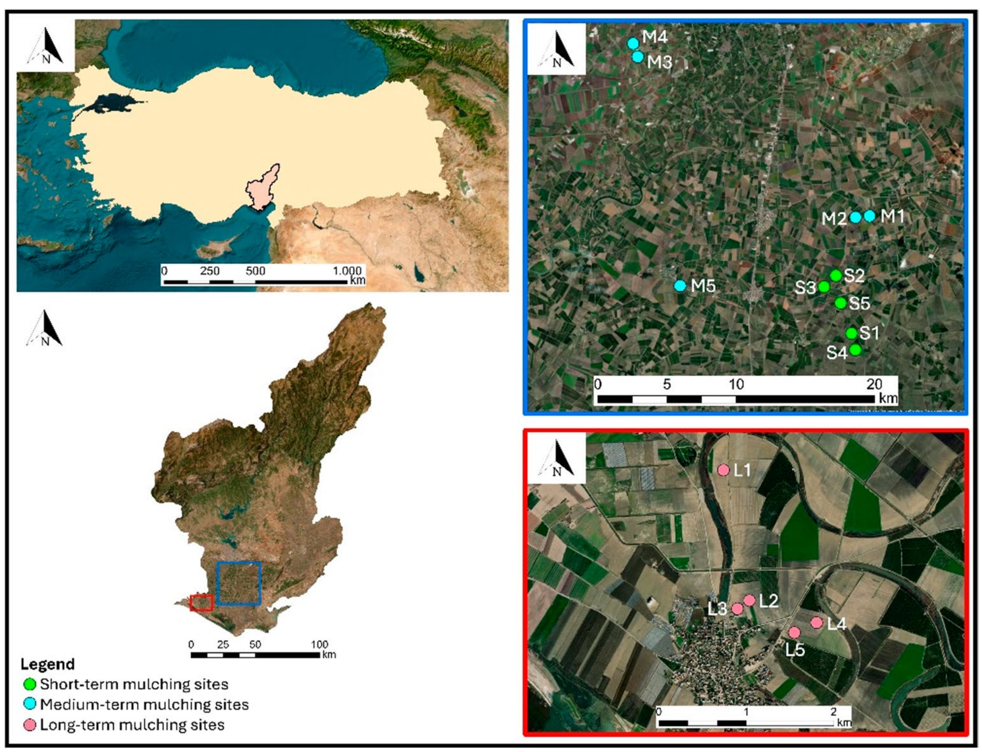

Fig. 1 Location of the sampling sites in Adana, Türkiye. Green: short duration mulching sites (S), blue: medium duration mulching sites (M), pink: long duration mulching sites (L). Coordinate system: WGS 84 / UTM Zone 36N

provided in Table S1. The region has a mean annual temperature of 19.3 °C and an average annual precipitation of 667.5 mm (General Directorate of State Meteorology Affairs, 2025). The climate is classified as Csa, indicating a hot-summer Mediterranean climate with dry winters (Kottek et al. 2006). Adana has an intensive agricultural structure with large, irrigated plains and a high proportion of cropland compared to urban land, which makes it a representative area for examining mulch-related MP accumulation. Agricultural lands were selected based on consistent plastic mulch application for approximately 5, 10, and 30 years, corresponding respectively to short-duration (S), medium-duration (M), and long-duration (L) mulching. For each duration category, five independent sites were sampled, resulting in a total of 15 sampling locations (Fig. S1a-c). Short-duration mulching sites (S) were located in the Yüreğir district, medium-duration mulching sites (M) were distributed across Yüreğir (three sites) and Seyhan (two sites), and long-duration mulching sites (L) were located in the Karataş district, where typical Mediterranean vegetable-based cropping systems dominate. Population density varied markedly among the sampling districts, with the highest concentration observed in the central district of Seyhan, followed by Karataş, which is located closer to the city center (Table S1). In contrast, Yüreğir represents a transitional zone between urban and rural areas. Within Yüreğir, sampling sites located closer to the urban core (Yunusoğlu and Hürriyet) were characterized by relatively higher population density, whereas sites situated in predominantly rural areas (Şeyhmurat and Şahinağa) exhibited lower population density.

Field interviews and site observations confirmed that polyethylene (PE) mulch films are the primary soil-covering materials used across all sampling locations, whereas polypropylene (PP) has never been employed as a mulching film in the region. PP-based items occur only in minor auxiliary uses such as bale strings, irrigation accessories, and greenhouse fixtures. In S and M sites, PE films are applied at the beginning of each growing season and partially removed after harvest, though small residues remain due to handling and weathering. In L sites, PE has been continuously used for approximately 30 years, and aged mulch is no longer removed because fragmentation makes complete retrieval impractical. A summary of mulch type, mulching duration, and years of mulch use characteristics is provided in Table S1. In the Adana region, customary agricultural practices differ between areas with short-to-medium and long histories of mulch use. At the S and M sites, plastic mulch is typically removed after each growing season, supported by regular crop rotations involving watermelon, tomato, peanut, soybean, and cotton. In contrast, the L sites, located in the Karataş district, are characterized by continuous melon-watermelon cultivation without crop rotation. In these long-duration fields, plastic mulch is not removed at the end of the season; instead, aged and fragmented mulch remains in the soil, as its disintegration makes complete removal difficult. Conventional tillage with a mouldboard plow (25–30 cm depth) is carried out at all sampled sites, but at the L sites this tillage is performed without prior removal of residual mulch pieces, resulting in greater incorporation of fragmented plastics into the soil profile.

During July and August 2024, soil samples were collected at two depths (0–10 cm and 10–20 cm), including undisturbed samples for soil moisture, porosity, and bulk density, and disturbed cores for MP distribution, soil structure, and routine soil analyses. Undisturbed samples were obtained using a cylindrical core (5.0 cm height × 5.1 cm internal diameter), and disturbed samples were collected with a stainless-steel spade. For each site and each depth (0–10 cm and 10–20 cm), three subsamples were collected and composited to obtain a representative sample. After homogenization, a subsample of 5 kg of fresh soil was weighed into glass containers (Gundogdu et al. 2022) and stored at 4 °C until analysis.

The soils across the study area were fine-textured, predominantly classified as clay, silty clay, and clay loam, with clay contents ranging from 25% to 74% (Table S2). According to Soil Taxonomy (2022), the soils corresponded to Typic Xerofluvent, Chromic Haploxerert, and Typic Calcixerept. Typic Xerofluvent, Chromic Haploxerert, and Typic Calcixerept soils are located on flat to nearly flat slopes and show an AC horizon sequence. Typic Xerofluvent soils are young, moderately textured, and highly calcareous; Chromic Haploxerert soils are clay-textured and highly calcareous; and Typic Calcixerept soils are classified as older soils with moderate texture and highly calcareous. These characteristics were generally consistent across the short (S), medium (M), and long (L) mulching durations.

2.2 Soil Analyses

Standard pre-analysis preparation was carried out on all soil samples (air-drying, 2-mm sieving, and homogenization), while fresh soils were used for biological analyses. In this study, the measured soil indicators included bulk density (BD), aggregate stability (AS), available water content (AWC), water-filled pore space (WFPS), pH, electrical conductivity (EC), available phosphorus, exchangeable potassium, soil organic carbon (SOC), microbial biomass carbon (MBC), and potentially mineralizable nitrogen (PMN). Detailed descriptions of all analytical methods and their references are provided in Table S3.

2.3 Microplastic Extraction

To extract MPs, soil subsamples were dried for 24 h at 60 °C, weighed, and passed through a 5-mm sieve. From these, 150 g of sieved soil was weighed into glass beakers, and MPs were extracted with saturated NaCl solution ($\rho = 1.2$ g cm^{-3}) (Akca et al. 2024; Gundogdu et al. 2025; Vural et al. 2025). Saturated NaCl was chosen because of its inertness, non-toxicity, cost-effectiveness, availability, and environmental compatibility, making it suitable for extracting low-density MPs. The solution was added to the weighed soil samples (3–5 cm above), stirred with a glass rod, and 12 h for density separation. Floating particles were filtered (33 µm); the process was repeated in triplicate. Organic matter was digested with 100 mL of 30% H_2O_2, heated at 70 °C for 72 h, and filtered through Whatman GF/C filters (1822-047) under vacuum (Value-CL VE 135).

2.3.1 Microplastic Particle Identification

Filter-retained MPs were examined using a stereomicroscope (Leica S8 AP0, transmitted light, 1.0–8.0× zoom). Suspected MPs were isolated, their pictures taken for size, shape, and color. The large component of the filtered residues was scanned from 400 to 4000 cm^{-1} at a resolution of 2 cm^{-1} with Attenuated Total Reflectance-Fourier Transform Infrared Spectroscopy (ATR-FTIR, Shimadzu IRTracerTM-100, Japan) (Akca et al. 2024). Spectral interpretation used Lab Solution IR Software with automatic baseline correction. The obtained images and scans are presented in Fig. S2a–d.

2.3.2 Quality Control and Recovery Test

All laboratory apparatus was cleaned with deionized water before each analytical step. A blank sample confirmed the absence of laboratory-derived contamination. Plastic-based materials were avoided, and the NaCl solution was prefiltered to remove impurities. The beakers were stoppered during density separation and digestion to minimize contamination. Recovery tests using clean soil spiked with high density polyethylene (HDPE), low density polyethylene (LDPE), polypropylene (PP), polystyrene (PS), and polyethylene terephthalate (PET) all non-biodegradable polymers showed recovery efficiencies above 86%.

Table 1 Classification of soil indicators used in SQI calculation

Indicator	Category	Interpretation
AS, AWC, K, SOC, MBC, PMN	Physical/Biological	More is better
BD	Physical	Less is better
WFPS, EC, pH, P	Physical/Chemical	Mid-point is optimum

AS Aggregate stability, *AWC* Available water content, *K* Exchangeable potassium, *SOC* Soil organic carbon, *MBC* Microbial biomass carbon, *PMN* Potentially mineralizable nitrogen, *BD* Bulk density, *WFPS* Water filled pore space, *EC* Electrical conductivity, *pH* Soil reaction, *P* Available phosphorus

2.4 Evaluation of Soil Quality

The Soil Quality Index (SQI) is a composite metric used to quantify soil functioning, and its calculation relies on converting measured soil indicators into unitless scores. To achieve this, the present study employed the Soil Management Assessment Framework (SMAF), which provides a standardized scoring system for evaluating soil physical, chemical, and biological properties (Andrews et al. 2004). SMAF evaluates each soil indicator using non-linear scoring functions that consider soil taxonomy, climate, texture, slope, sampling time, and analytical techniques, thereby enabling context-specific assessments relevant to conditions such as MP accumulation.

In this study, the assessment process involved four steps: (i) establishing a dataset of relevant indicators, (ii) normalizing the data and generating unitless scores using SMAF, (iii) weighting the indicators with the Fuzzy Analytic Hierarchy Process (FAHP), and (iv) combining the outcome into a soil quality index (Gozukara et al. 2022). Following this procedure, soil quality was evaluated not only through the overall Soil Quality Index (SQI) but also through three component indices representing distinct functional domains: the physical quality index (PQI), the chemical quality index (CQI), and the biological quality index (BQI). The final SQI was derived from these component indices.

The SMAF model was applied using major soil indicators widely recognized as sensitive to management impacts (Bagavthsingh and Duraisamy 2024; da Silva Souza et al. 2025; Reis and Dindaroğlu 2024). The selected indicators were classified into three categories: physical (water-filled pore space-WFPS, aggregate stability-AS, available water content-AWC, bulk density-BD), chemical (available phosphorus-P, exchangeable potassium-K, soil pH, electrical conductivity-EC), and biological (organic carbon-OC, microbial biomass carbon-MBC, potentially mineralizable nitrogen-PMN).

All indicator values were normalized into unitless scores ranging from 0 to 1 to ensure comparability (Koca et al. 2019). The soil indicators used in the SQI calculation were classified according to SMAF scoring rules, and their categories are summarized in Table 1. After normalization and scoring, the Fuzzy Analytic Hierarchy Process (FAHP) was applied to weight the indicators, PQI, CQI, and BQI according to Chang (1996). The final FAHP formulation was presented in Eq. (1).

$$\mu M\left(x\right)=\left\{\begin{array}{c}\frac{x}{m\text{-}l}\text{-}\frac{l}{m\text{-}l},x\in \left[l,m\right],\\ \frac{x}{m\text{-}u}\text{-}\frac{u}{m\text{-}u},x\in \left[m,u\right],\\ 0,otherwise\end{array}\right\} \quad (1)$$

Where M is the triangular fuzzy number, x is the data set, l is the lower values of M, u is the upper values of M, and m is the modal value. When the m and M derivatives are equal their outcome is termed crisp number else fuzzy number. The overall SQI was calculated based on the outcome of the FAHP results (Eq. 2) and the weighted linear combination (WLC) method was used to integrate the weighted indicators into a single soil quality index according to Özkan et al. (2020),

$$SQI=\sum\nolimits_{i=1}^{n}W_i\times S_i \quad (2)$$

SQI represents the soil quality index, W represents the weight of an indicator/quality component according to FAHP, and S represents the indicator/quality component score. The classification ranges for very low (<40%), low (40–55%), medium (55–70%), high (70–85%), and very high (>85%) levels of the physical, chemical, biological, and overall soil quality indices were defined according to Gugino et al. (2009).

2.5 Statistical Analyses

The Kolmogorov-Smirnov test was employed to determine whether the number of plastics followed a normal distribution, and Levene's homogeneity of variance test was conducted on the data. Variations among mulching durations were analyzed using a one-way ANOVA, and significant differences were determined by Tukey's HSD test at $p<0.05$.

Depth-related variations in soil parameters were analyzed using paired-sample t-tests. Statistical significance was evaluated at the 95% and 99% confidence levels. Principal component analysis (PCA) was employed to investigate the relationships between soil quality indices and soil MP abundance. All statistical analyses (normality tests, homogeneity tests, ANOVA, Tukey's HSD, and paired-sample t-tests), multivariate analysis (PCA) and figures were performed using OriginPro 2024.

3 Results

3.1 Soil Indicators

The impacts of MPs on soil physical properties were primarily observed at the soil surface (Table 2). Available water capacity (AWC) differed significantly among mulching durations at the 0–10 cm depth ($p<0.05$), with the lowest values under M and the highest values under L. Depth-related differences were significant for M and L ($p<0.05$). Bulk density (BD) increased with mulching duration, with higher values under M and L at the soil surface and under L at the subsurface (Table 2). BD SMAF scores decreased with increasing MP abundance. Significant differences between S and L were observed for both measured BD and SMAF scores ($p<0.05$). Aggregate stability (AS) decreased significantly with mulching duration ($p<0.05$), with the lowest values consistently under L at both depths (Table 2). Significant depth-related reductions were also detected across durations. Water-filled pore space (WFPS) differed significantly among mulching durations ($p<0.05$; Table 2). WFPS increased under M relative to S but decreased under

Table 2 Soil physical indicators: measured means, standard errors, and SMAF scores across mulching durations and soil depths. Lowercase letters indicate significant differences among treatments ($p<0.05$)

Durations	Average Measured Values				SMAF Scores			
	AWC	BD	AS	WFPS	AWC	BD	AS	WFPS
	0–10 cm				**0–10 cm**			
S	10.7±0.42 a	1.24±0.02 b	49.9±4.49 a	25.9±2.80 b	0.48±0.02 a	0.58±0.03 a	0.93±0.04 a	0.67±0.04 ab
M	7.7±1.17 b	1.32±0.03 a	46.4±4.45 a	37.3±4.31 a	0.27±0.06 b	0.53±0.04 a	0.90±0.04 a	0.79±0.05 a
L	12.7±0.37 a	1.37±0.02 a	26.0±4.44 b	21.1±2.31 b	0.51±0.02 a	0.38±0.02 b	0.53±0.07 b	0.61±0.03 b
	10–20 cm				**10–20 cm**			
S	12.5±1.33 a	1.33±0.02 b	20.3±3.55 a	34.4±2.34 b	0.51±0.06 a	0.44±0.03 a	0.45±0.10 ab	0.79±0.03 ab
M	9.9±0.52 a	1.36±0.03 b	29.0±3.95 a	44.7±3.59 a	0.38±0.02 a	0.41±0.04 ab	0.66±0.08 a	0.85±0.02 a
L	12.0±0.66 a	1.45±0.02 a	18.0±2.75 a	31.8±1.00 b	0.44±0.03 a	0.32±0.02 b	0.37±0.06 b	0.77±0.01 b
Sdepth	ns	**	**	ns	ns	**	**	ns
Mdepth	*	ns	*	ns	*	**	*	*
Ldepth	ns	**	**	ns	*	*	**	**

S short-duration mulching, *M* medium-duration mulching, *L* long-duration mulching, ± represents standard errors, * indicates significant differences between different depths within the same duration time at $p<0.05$ level according to paired sample t-test; ** indicates significant differences between different depths within the same duration time at $p<0.01$ level according to paired sample t-test; ns, non-significant; AWC (%), available water content; BD (g cm^{-3}), bulk density; AS (%), aggregate stability; WFPS (%), water filled pore space

Table 3 Soil chemical indicators: measured means, standard errors, and SMAF scores across mulching durations and soil depths. Lowercase letters indicate significant differences among treatments ($p<0.05$)

Durations	Average Measured Values				SMAF Scores			
	pH	EC	Available *P*	Exchangeable K	pH	EC	Available *P*	Exchangeable K
	0–10 cm				**0–10 cm**			
S	8.23 ± 0.04 a	0.21 ± 0.02 b	5.05 ± 0.22 a	266 ± 8.7 b	0.73 ± 0.01 ab	0.66 ± 0.06 b	0.74 ± 0.04 b	1.00 ± 0.00 a
M	8.17 ± 0.05 a	0.27 ± 0.03 ab	5.71 ± 0.88 a	269 ± 21.5 b	0.65 ± 0.05 b	0.79 ± 0.07 ab	0.71 ± 0.05 b	0.96 ± 0.02 a
L	8.09 ± 0.04 a	0.30 ± 0.02 a	6.26 ± 0.35 a	324 ± 13.9 a	0.77 ± 0.01 a	0.87 ± 0.05 a	0.88 ± 0.01 a	1.00 ± 0.00 a
	10–20 cm				**10–20 cm**			
S	8.23 ± 0.05 a	0.24 ± 0.03 a	4.62 ± 0.24 b	249 ± 12.5 a	0.73 ± 0.01 a	0.74 ± 0.06 a	0.67 ± 0.05 b	0.99 ± 0.00 a
M	8.20 ± 0.06 a	0.21 ± 0.01 a	5.46 ± 0.82 ab	260 ± 21.3 a	0.64 ± 0.06 a	0.72 ± 0.05 a	0.68 ± 0.06 b	0.98 ± 0.01 a
L	8.21 ± 0.04 a	0.30 ± 0.03 a	6.75 ± 0.36 a	299 ± 21.9 a	0.74 ± 0.01 a	0.81 ± 0.06 a	0.90 ± 0.01 a	0.98 ± 0.01 a
Sdepth	ns	*	**	**	ns	**	*	ns
Mdepth	ns	*	ns	ns	ns	*	ns	ns
Ldepth	**	ns	ns	ns	**	ns	ns	ns

S short-duration mulching, *M* medium-duration mulching, *L* long-duration mulching, ± represents standard errors, * indicates significant differences between different depths within the same duration time at $p<0.05$ level according to paired sample t-test; ** indicates significant differences between different depths within the same duration time at $p<0.01$ level according to paired sample t-test; pH, soil reaction; EC (dS m^{-1}), electrical conductivity; Available P (mg P kg^{-1}), available phosphorus; Exchangeable K (mg K kg^{-1}), exchangeable potassium

Table 4 Soil biological indicators: measured means, standard errors, and SMAF scores across mulching durations and soil depths. Lowercase letters indicate significant differences among treatments ($p<0.05$)

Durations	Average Measured Values			SMAF Scores		
	SOC	MBC	PMN	SOC	MBC	PMN
	0–10 cm			**0–10 cm**		
S	1.06 ± 0.05 a	834 ± 38.4 ab	35.4 ± 4.2 a	0.52 ± 0.09 a	0.97 ± 0.01 a	0.86 ± 0.06 a
M	1.11 ± 0.06 a	931 ± 70.2 a	19.6 ± 3.5 b	0.49 ± 0.08 a	1.00 ± 0.00 a	0.45 ± 0.11 b
L	1.24 ± 0.06 a	717 ± 38.3 b	26.9 ± 5.4 ab	0.27 ± 0.04 a	0.99 ± 0.00 a	0.48 ± 0.12 b
	10–20 cm			**10–20 cm**		
S	0.97 ± 0.04 b	667 ± 32.3 ab	15.7 ± 1.2 a	0.47 ± 0.09 a	0.99 ± 0.00 a	0.46 ± 0.09 a
M	1.13 ± 0.06 ab	819 ± 54.6 a	18.7 ± 2.3 a	0.48 ± 0.09 a	1.00 ± 0.00 a	0.57 ± 0.10 a
L	1.21 ± 0.06 a	540 ± 38.9 b	29.8 ± 6.6 a	0.26 ± 0.04 a	0.96 ± 0.01 b	0.50 ± 0.12 a
Sdepth	ns	**	ns	**	ns	**
Mdepth	ns	*	ns	ns	ns	ns
Ldepth	*	**	ns	ns	*	ns

S short-duration mulching, *M* medium-duration mulching, *L* long-duration mulching, ± represents standard errors, * indicates significant differences between different depths within the same duration time at $p<0.05$ level according to paired sample t-test; ** indicates significant differences between different depths within the same duration time at $p<0.01$ level according to paired sample t-test; ns, non-significant; SOC (%), soil organic carbon; MBC (mg C kg^{-1}), microbial biomass carbon; PMN (mg NH_4^+-N kg^{-1} $28d^{-1}$), potential mineralizable nitrogen

L at both depths. SMAF scores increased significantly for M and L ($p<0.05$).

Soil pH showed only minor variation among mulching durations (Table 3). All values remained within a narrow alkaline range (8.09–8.23) and given the absence of MP-free control soils, these small differences likely reflect natural spatial variability. A slight but significant increase was observed only under L ($p<0.01$). Electrical conductivity (EC) increased significantly with mulching duration at the soil surface (Table 3), whereas subsurface values remained within a narrower range. Significant depth differences were detected for S and M ($p<0.05$). Available P at the subsurface increased significantly with mulching duration ($p<0.05$; Table 3). SMAF scores were highest under L. Because soil P can vary considerably over short distances and no control soil was available, these differences may partly reflect spatial heterogeneity. A moderate decrease in available P was observed under S ($p<0.05$). Exchangeable K increased at the soil surface with increasing mulching duration, while subsurface values remained stable (Table 3).

Soil biological properties also varied with mulching duration (Table 4). Soil organic carbon (SOC) increased significantly at the subsurface under L ($p<0.05$), with measured values increasing in the order S < M < L. SOC SMAF scores decreased slightly from S to L at both depths. Microbial biomass carbon (MBC) differed significantly among durations ($p<0.05$; Table 4), with values ordered as S > M > L at both depths. Surface SMAF scores remained close to 1.00. Potentially mineralizable N (PMN) decreased under M and showed higher values under L at the surface ($p<0.05$), while subsurface PMN increased progressively from S to L (Table 4).

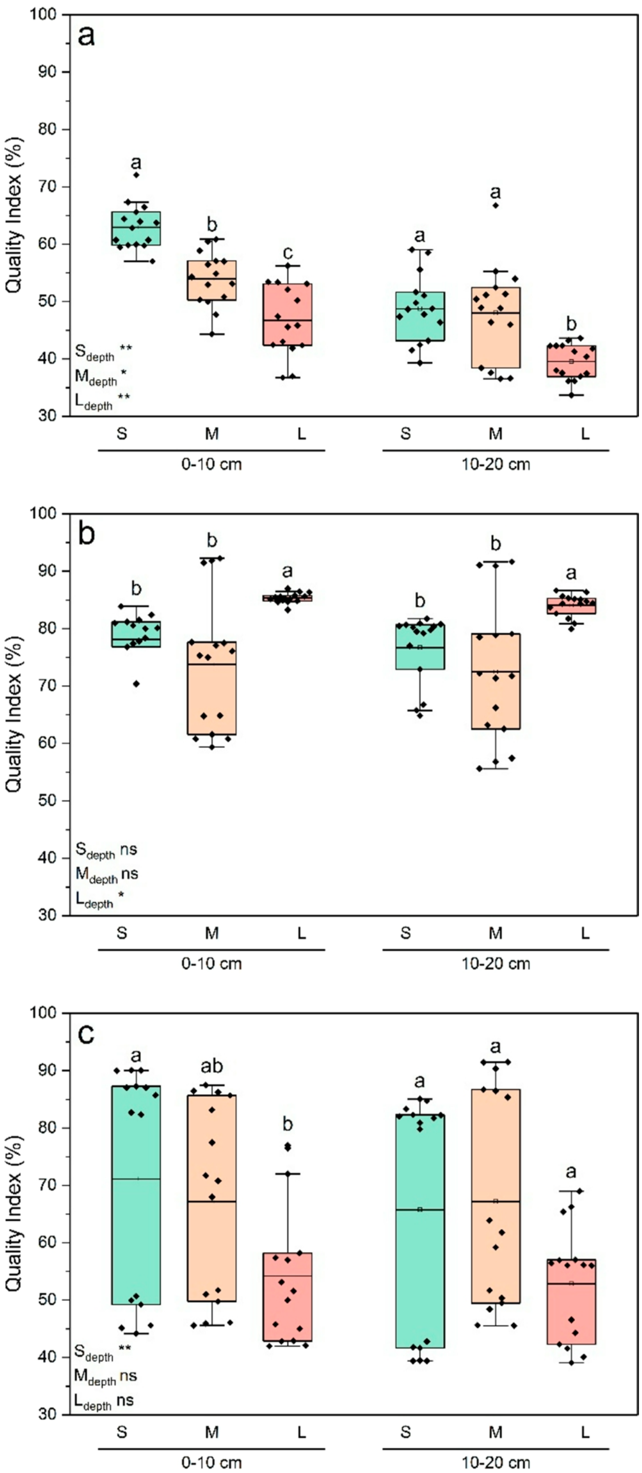

Fig. 2 Box plot showing soil physical, chemical, and biological quality**.** Different letters represent significant differences among different treatments (p<0.05). S, short-duration; M, medium-duration; L, long-duration; Bars represents standard errors; * indicates significant differences between different depths within the same duration time at p<0.05 level according to paired sample t-test; ** indicates significant differences between different depths within the same duration time at p<0.01 level according to paired sample t-test; ns, non-significant; **a**) PQI, physical quality index; **b**) CQI, chemical quality index; **c**) BQI, biological quality index

3.2 Indicator Weights and Soil Quality Index

Physical quality contributed the most to overall soil quality (0.449), followed by biological (0.351) and chemical quality (0.200) (Table S4, Table S5). Within physical quality, BD and AWC had 0.338 and 0.337, relative to 0.053 obtained in WFPS. For chemical quality, soil pH > EC having 0.446 and 0.056, respectively. Among the biological indicators, SOC had the highest weight at 0.567, and PMN the lowest at 0.077.

The PQI decreased significantly ($p < 0.05$) with increasing plastic mulch duration (L < M < S; Fig. 2a). At the soil surface, the PQI was classified as medium under S, whereas it was low under both M and L, corresponding to 14% and 26% declines compared to S. In the subsurface, the PQI was classified as low under both S and M, but decreased to very low under L, indicating a 19% decline compared to S. PQI degradation at the surface was continuous with longer mulch duration, whereas subsurface effects appeared only after extended exposure. Furthermore, the CQI differed significantly ($p < 0.05$) (Fig. 2b). At the surface, the CQI was classified as high under S and M but very high under L. In the subsurface, all treatments remained within the high class, with L showing the greatest improvement. Overall, CQI increased by 10% and 6% for L and M, respectively, compared to S across both depths. The BQI decreased significantly at 0–10 cm soil depth (L < M < S; Fig. 2c), being classified as high under S, medium under M, and low under L, corresponding to a 23% reduction for L, compared to S. This reduction was gradual and only observed during the L duration. At the subsurface, the BQI was medium under S and M, but decreased to low under L, and MP amount and type did not significantly impact BQI, as opposed to PQI and CQI. The durations of MP significantly reduced SQI ($p < 0.01$) across soil depths (L < M < S; Fig. 3). At the surface, SQI declined progressively with longer MP duration, being classified as medium under all treatments, with 9% and 17% decreases for M and L, respectively, compared to S. At the subsurface, SQI remained medium under S and M but decreased to low under L, corresponding to a 12% reduction relative to S.

3.3 Microplastic Abundance and Polymer Identification

Across all samples, 259 of 283 particles (91.5%) were confirmed as MPs. Surface MP abundance decreased with increasing mulching duration (S > M > L, $p < 0.05$), whereas subsurface abundance showed the opposite trend (L > M > S, $p < 0.05$; Fig. 4a). The distribution of MPs differed significantly between soil depths and mulching durations ($p < 0.05$; Fig. 4b). ATR-FTIR analysis identified two polymer types:

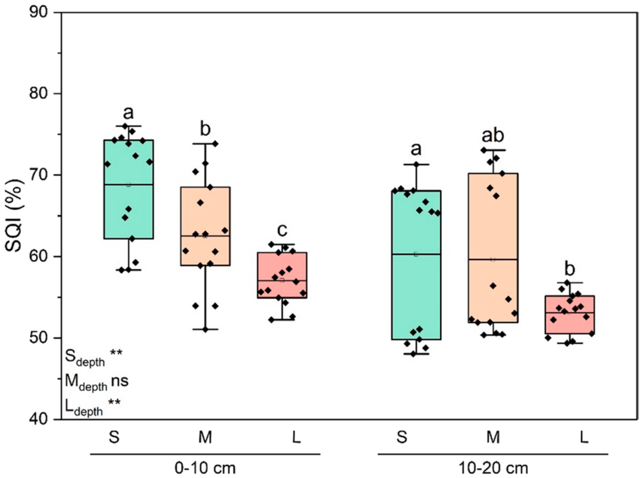

Fig. 3 Soil quality index. Different letters represent significant differences among different treatments (p<0.05). S, short-duration mulching; M, medium-duration mulching; L, long-duration mulching; Bars represents standard errors; * indicates significant differences between different depths within the same duration time at p<0.05 level according to paired sample t-test; ** indicates significant differences between different depths within the same duration time at p<0.01 level according to paired sample t-test; ns, non-significant; SQI, soil quality index

polyethylene (98.6%) and polypropylene (1.4%) (Fig. S2b, Fig. S2d).

3.4 PCA of Soil Quality Indices and their Relationship with Microplastic Abundance

In surface soil, PC1 and PC2 explained 51.24% and 19.74% of the total variance (70.97%; Fig. 5a). SQI and BQI had the highest positive loadings on PC1, while CQI loaded negatively. For PC2, CQI had the highest positive loading, and PQI had the highest negative loading. In subsurface soil, PC1 and PC2 accounted for 57.22% and 19.72% of the variance (76.93%; Fig. 5b). SQI and BQI again had the strongest positive loadings on PC1, while CQI and MP count loaded negatively. For PC2, CQI had the main positive loading, and PQI had the main negative loading.

4 Discussion

MP accumulation under different mulching durations creates progressive alterations in soil functioning and understanding how these changes evolve over time is essential for interpreting soil quality responses. Our results showed that MP abundance at the soil surface decreased with duration, while subsurface accumulation increased, suggesting that MPs gradually migrate downward through tillage, irrigation, and fragmentation processes (Akça et al. 2022; He et al. 2018; Rillig et al. 2017b). This depth-dependent shift indicates that MP effects are not static but transition from surface-driven interactions to deeper soil layers over the long term, reflecting the cumulative nature of plastic aging and redistribution. This redistribution forms the foundation for understanding how MPs influence physical, chemical, and biological indicators of soil quality across different mulching durations. In addition to downward transport processes, the observed temporal decrease in MP abundance at the soil surface may also be partially explained by wind-driven horizontal redistribution (Ockelford et al. 2025). Recent experimental and field-based studies have demonstrated that MPs, particularly low-density polymers and fibrous morphologies, are highly susceptible to aeolian entrainment and can be preferentially transported relative to mineral soil particles

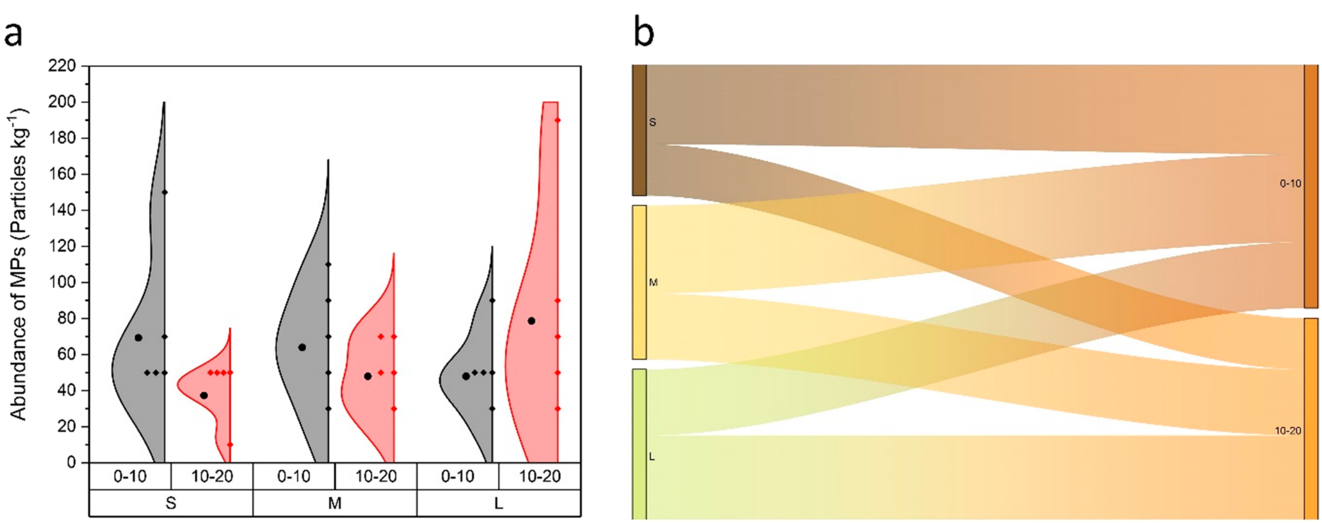

Fig. 4 (**a**)Violin plot showing total abundance of MPs in soils of different soil depths and mulch application duration (**b**) Sankey diagram showing the distribution of soil MPs between soil depths and mulch application duration (left side; short: S, medium: M, and long duration: L) and soil depth (right side; in cm)

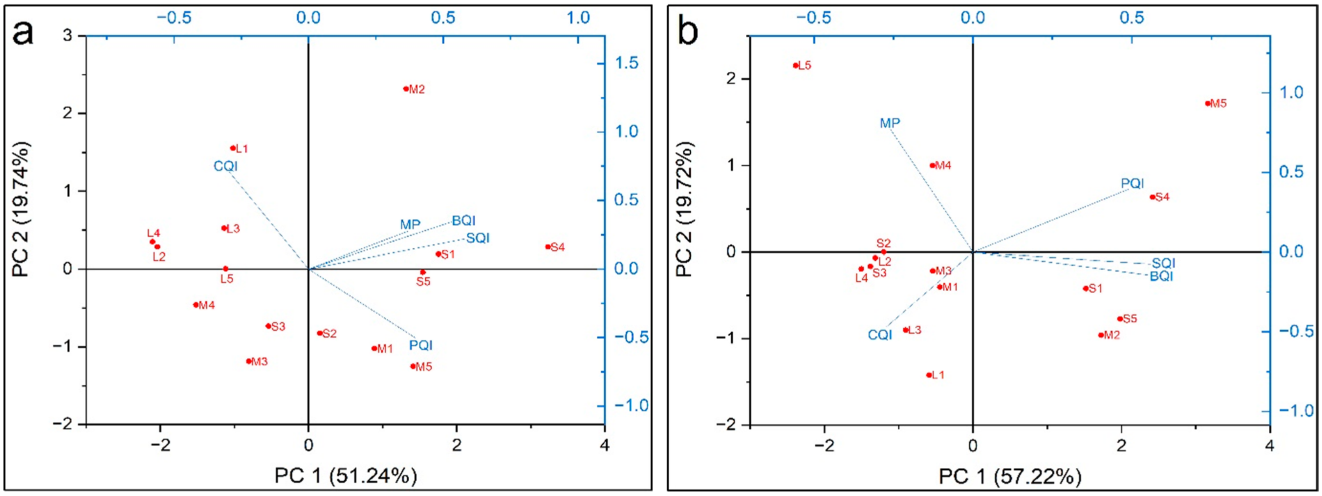

Fig. 5 Principal component analysis of soil quality indices and MP abundance at different soil depths: (**a**) 0-10 cm and (**b**) 10-20 cm. SQI, soil quality index; PQI, physical quality index; CQI, chemical quality index; BQI, biological quality index; MP, microplastic abundance

(Tian et al. 2022; Bullard et al. 2021; Rezaei et al. 2019, 2022). Wind tunnel experiments and natural storm observations have shown that MP concentrations in wind-eroded sediments can be enriched by one to two orders of magnitude compared to source soils, indicating efficient removal of MPs from surface layers (Rezael et al., 2019; Ockelford et al. 2025). Agricultural soils, especially under dry and disturbed conditions, have therefore been identified as dynamic secondary sources of airborne MPs rather than passive sinks (Rezaei et al. 2022). Moreover, fibers have consistently been reported as the dominant MP shape in wind-blown dust, reflecting their high aerodynamic mobility and propensity for long-range transport. In this context, the reduction of surface MP abundance observed in the present study cannot be attributed solely to vertical migration into deeper soil horizons but may also reflect lateral export from sampling sites via wind erosion and atmospheric transport. This highlights the need to consider aeolian processes alongside tillage and irrigation when interpreting temporal changes in surface soil MP inventories (Tian et al. 2022; Bullard et al. 2021; Rezaei et al. 2019, 2022; Ockelford et al. 2025).

The impact of MPs on soil physical conditions changed substantially over time, reflecting the combined influence of particle size, density, and degree of aging. In the medium mulching duration, MPs reduced available water content (AWC) at the soil surface by increasing bulk density (BD) and decreasing aggregate stability (AS), soil organic matter (SOM), and water-filled pore space (WFPS) (Wang et al. 2024). As MPs migrated from the 0–10 cm layer to the 10–20 cm layer and fragmented into smaller particles (Akca et al. 2024; Akça et al. 2022; Rillig et al. 2017b;), their influence on available water content (AWC) became less pronounced over time. Larger MP fragments can create desiccation cracks in the soil (Wan et al. 2019) and increase hydrophobicity (Wang et al. 2023), generating preferential flow pathways that reduce available water content (AWC). In contrast, finely milled or strongly aged MP particles may increase soil water content by occupying micropores (de Souza Machado et al. 2019). At short durations, large undegraded MP fragments may occupy soil macropores and dilute the soil matrix due to their lower density (de Souza Machado et al. 2019), while increasing MP content generally increases bulk density (BD) (de Souza Machado et al. 2019; Lozano et al. 2021; Wang et al. 2022). With longer exposure, MPs also contributed to the breakdown of water-stable aggregates (WSA) (Boots et al. 2019), fracturing of microaggregates (Lozano et al. 2021; Zhao et al. 2021), and reductions in pore space, resulting in greater bulk density (BD) (Acar et al. 2025b). Although agricultural cultivation alone may increase bulk density (BD) (de Souza Machado et al. 2019), conservation practices such as no-till systems, cover cropping, residue retention, and organic amendments can counteract this effect (Diacono and Montemurro 2011; Díaz-Zorita et al. 2004; Gao et al. 2016; Yost et al. 2022). Over time, MP particles became embedded within soil pores, disrupting aggregate formation processes (de Souza Machado et al. 2019) and altering the microbial aggregate interactions that support soil structural stability (Rillig et al. 2017b). Improvements in water-filled pore space (WFPS) observed during the medium duration reflect temporary clogging and sieving of MP particles within macropores (Dong et al. 2022; Guo et al. 2022; Rillig et al. 2017a; Zhao et al. 2021). However, as particles became smaller and entered micropores, water-filled pore space (WFPS) declined in the long term (Boots et al. 2019; Shafea et al. 2023; Wang et al. 2024). Altogether, these results illustrate that MPs progressively alter soil physical structure, initially affecting macropores and later micropores as fragmentation advances.

Chemical indicators demonstrated less pronounced but still meaningful responses to increasing MP exposure. Baseline soil pH values were unavailable, yet slight increases across mulching durations are consistent with earlier reports showing that MPs can raise pH through cation exchange alterations, proton consumption during denitrification, or additive leaching (Li and Liu 2022; Liu et al. 2024; Medynska-Juraszek and Jadhav 2022; Qi et al. 2020a; Zhao et al. 2021). Electrical conductivity (EC) increased at the soil surface where MP concentrations were highest, consistent with observations that moderate-to-high MP content increases electrical conductivity (EC) (Qi et al. 2020c), although lower concentrations or advanced microplastic aging may moderate this response (Peng et al. 2025; Zhang et al. 2024). Available phosphorus (P) exhibited complex patterns: while some studies report that phosphorus availability is not highly reactive to MP levels (Li et al. 2021), soil quality assessments revealed a decline associated with increased MP abundance, likely driven by changes in acid phosphatase activity (Fei et al. 2020; Liu et al. 2023; Yi et al. 2021). MP composition and soil type may also influence whether phosphorus availability increases or decreases (Liu et al. 2017, 2024; Yan et al. 2021). Exchangeable potassium (K) increased at the soil surface, consistent with the leaching effects observed at greater MP concentrations (Yan et al. 2023), while MPs embedded within aggregates may reduce the dispersion of potassium (K) within soil pores (Peng et al. 2025). Overall, these chemical responses indicate that MPs primarily exert indirect effects, mediated through structural alteration, changes in enzyme activity, and polymer-specific chemical interactions.

Biological responses to MP accumulation further highlight disruptions to fundamental soil processes. Elevated MP levels at the soil surface may alter the C: N ratio and limit microbial capacity to degrade MPs (Rillig et al. 2021). The higher soil organic carbon (SOC) observed at subsurface depths may be linked to MP degradation by-products or the increased stabilization of carbon within aggregates (Meng et al. 2022; Zhang et al. 2022). MPs can increase soil organic carbon (SOC) at low to moderate concentrations (Liu et al. 2024; Meng et al. 2022), although polymer composition plays a key role in determining whether organic carbon accumulates or declines (Peng et al. 2025; Zhang et al. 2024). Microbial biomass carbon (MBC) showed divergent responses: short-term increases (Liu et al. 2024) followed by declines after longer exposure, reflecting habitat degradation and shifts in substrate availability. Medium-duration increases in microbial biomass carbon (MBC) may result from electron-shuttling mechanisms associated with MP aging (Rillig et al. 2021; Zhang et al. 2022). The effects of MPs on biological processes may also vary across land-use types (Zhang et al. 2024). Potentially mineralizable nitrogen (PMN) decreased under high MP exposure (Meng et al. 2022), likely because MP surfaces provide immobilization sites that limit nitrogen transformations (Li and Liu 2022) and because PE mulch contributes limited nutrient value (Liu et al. 2024). As MPs continue to age, shifts in microbial community structure may help restore nitrogen mineralization potential (Zhang et al. 2022). Together, these patterns demonstrate that MPs alter both carbon and nitrogen cycling pathways, reshaping soil biological functioning over time.

Integrating these physical, chemical, and biological responses through the Soil Quality Index revealed that physical indicators particularly bulk density (BD) and available water content (AWC) carry the greatest importance due to their direct roles in regulating soil structure, aeration, hydraulic conductivity, and root growth (Acar et al. 2025a; Dengiz 2020üş et al. 2022). Chemical indicators showed lower weighting because they respond rapidly to management and environmental conditions (Dengiz et al. 2020; Kaya and Dengiz 2024), whereas pH was the principal indicator shaping chemical quality (Gozukara et al. 2022; Karaca et al. 2021; Ozlu et al. 2022). The Soil Quality Index clearly declined under medium and long mulching durations, reflecting reductions in water-filled pore space (WFPS), aggregate stability (AS), and laminar water flow (Jiang et al. 2017; Qi et al. 2020b). The convergence of all soil quality components into a narrow value range under long-term mulching suggests that persistent MP accumulation exerts a strong cumulative degradation effect.

Long-term mulching resulted in greater MP abundance in both soil layers, consistent with findings that extended exposure increases the number of film-derived particles without accelerating the rate of degradation (He et al. 2018; Huang et al. 2020; Zhang et al. 2016). Vertical translocation of MPs became more significant under long-duration mulching, likely enhanced by tillage (Liu et al. 2018) and irrigation (He et al. 2018). Within this context, PE was the dominant polymer detected, reflecting prevailing mulch usage. Although PP was not used as a mulching film in the study area, both PE and PP negatively affect soil structure, hydrology, microbial activity, and nutrient cycling (Chen et al. 2013; Khoironi et al. 2024; Liu et al., 2022; Meyer et al. 2021; Srinidhi et al. 2023; Tudor et al. 2019; Xie et al. 2025; Zhang et al. 2020). Overall, these results highlight the importance of considering polymer-specific properties when evaluating the ecological risks of MP accumulation in agricultural soils.

5 Conclusion

This study demonstrates that long-term use of plastic mulch initiates a series of mechanistic changes within the soil system by altering the movement, fragmentation, and incorporation of plastic residues into soil pore networks and aggregates. As these fragments migrate downward and interact with soil structure, they disrupt water retention, reduce aggregate stability, and interfere with microbial processes that regulate carbon and nitrogen dynamics. Consistent with these mechanistic disruptions, the findings showed a significant decline in soil quality, with the Soil Quality Index decreasing by approximately 9% under medium-duration mulching and 17% under long-duration mulching, as calculated using the Soil Management Assessment Framework model. These pathways explain the observed decline in soil functionality under prolonged mulching conditions. The findings confirm the initial hypothesis that continuous exposure to plastic mulch leads to cumulative and adverse impacts on soil health. By showing that plastic fragments persist, age, and redistribute through the soil profile, this research reveals that their influence extends beyond surface layers and intensifies over time. This mechanistic understanding highlights the long-term ecological risks associated with plastic residues in agricultural soils. The novelty of this work lies in linking field-based plastic accumulation patterns with a multidimensional assessment of soil physical, chemical, and biological responses under real agricultural conditions. These results provide practical guidance for farmers and agricultural advisers by emphasizing the importance of plastic residue management and highlighting the potential benefits of alternative mulching materials that reduce long-term plastic buildup. The main limitation of this study is that it was conducted under real field conditions, where environmental and management factors could not be fully controlled. To address this limitation, future research should focus on controlled experiments and long-term monitoring to further clarify how plastic residues interact with soil processes under different environmental and management scenarios. Such work is essential for improving remediation strategies and for supporting policy decisions aimed at reducing plastic pollution while maintaining sustainable agricultural production systems.

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Data Availability The datasets generated during the current study are available from the corresponding author on reasonable request.

Declarations

Conflict of interest The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

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