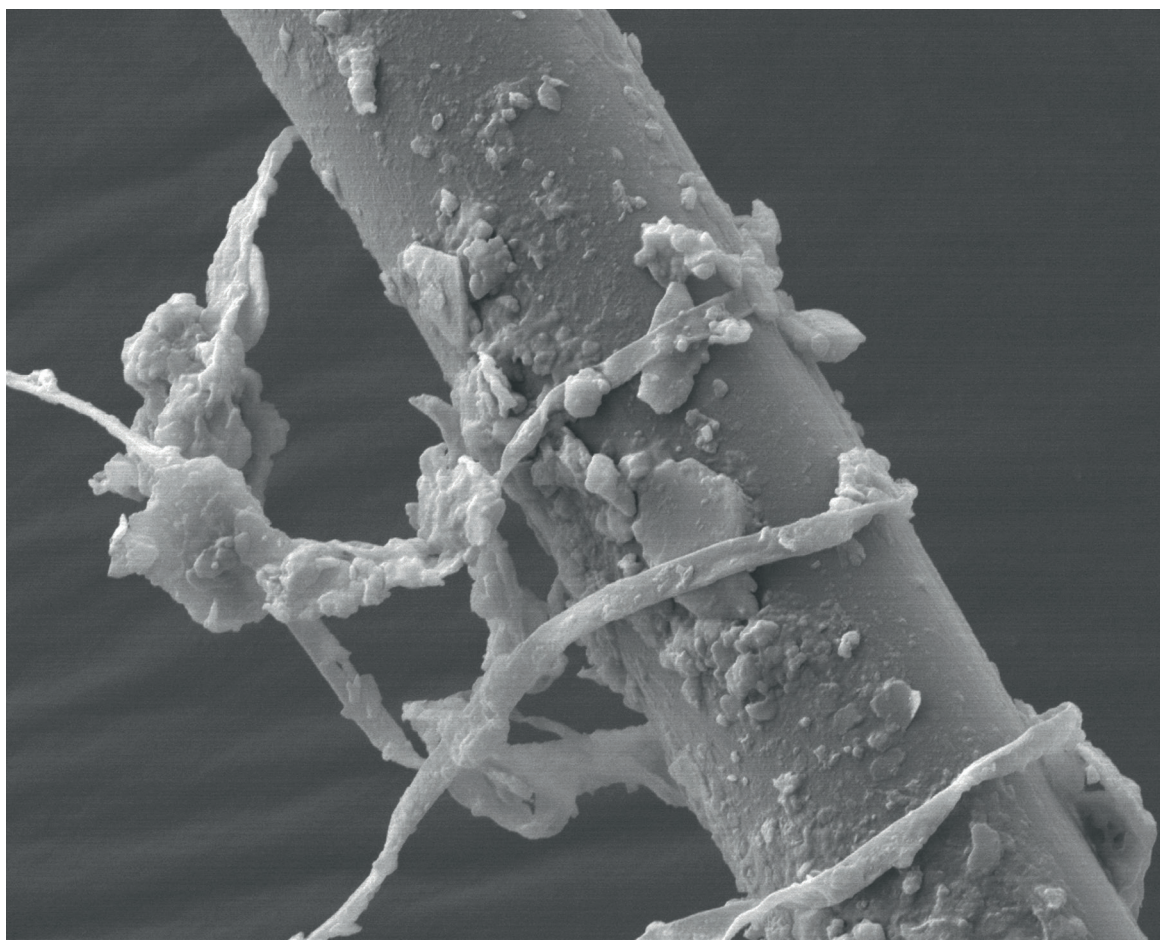




Effects of Microplastics on Carbon and Nitrogen Turnover Processes in the Rhizosphere of Crops

Katharina Julia Neubert

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which permits unrestricted use, distribution, and reproduction in any medium, provided the original work is properly cited.

Die heutige Mikroplastik-Belastung wird unter dem Überbegriff „anthropologischer Einfluss“ geführt. Mit anderen Worten: dieses Problem ist menschengemacht. Dazu möchte ich aus dem Kinderbuch *Die sieben Türen* von Adrian Draschoff ein Zitat anbringen, welches verdeutlicht, dass die Verantwortung zur Gestaltung unserer Zukunft und damit auch unserer Umwelt bei jedem Einzelnen liegt:

„[...] ‚Dazu möchte ich dir einen Rat mit auf den Weg geben‘, sagte sie einfühlsam.
‚Akzeptiere die Vergangenheit und umarme die Zukunft. Du musst nicht akzeptieren, was
geschehen ist. Nur, dass es geschehen ist.‘
‚Aber wie kann ich die Zukunft umarmen? Ich kenne sie doch noch gar nicht.‘
‚Nun ja, jeder kann auf gewisse Art die Zukunft gestalten. Jedenfalls wenn es darum geht,
was für ein Mensch man sein will.‘ [...]“

(Draschoff, 2024)

Today's microplastic pollution is classified under the overarching term "anthropogenic influence." In other words, this problem is man-made. In this context, I would like to quote a passage from the children's book *Die sieben Türen* (*The Seven Doors*) by Adrian Draschoff, which illustrates that the responsibility for shaping our future, and therefore also our environment, lies with each and every one of us:

"[...] 'I would like to give you a piece of advice,' she said gently.
'Accept the past and embrace the future. You do not have to accept what has happened – only
that it has happened.'
'But how can I embrace the future? I don't even know it yet.'
'Well, everyone can shape the future in their own way – at least when it comes to what kind of
person you want to be.' [...]"

Original text: Adrian Draschoff, *Die sieben Türen*
(translated from German by the author with the assistance of DeepL)

Danksagung

Zuallererst möchte ich mich bei *Herrn Prof. Dr. Nicolas Brüggemann*, meinem Doktorvater, Erstkorrektor und Teamleiter, bedanken. Inmitten vielfältiger Herausforderungen, Unwägbarkeiten und Problemstellungen war er für mich stets eine zuverlässige Stütze und Motivator, die mir neue Orientierung, Inspiration und Zuversicht verlieh.

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Ganz besonders danke ich unserem Team, das ich in vielerlei Hinsicht als große Bereicherung empfand. Die interdisziplinäre wie multikulturelle Zusammensetzung verschaffte mir wertvolle neue Perspektiven und Herangehensweisen. Vor allem blicke ich mit großer Freude auf die gemeinsamen Erlebnisse innerhalb und außerhalb des Instituts zurück.

Meiner Familie, vor allem meinen Eltern und Schwiegereltern gebührt mein außerordentlicher Dank. Ohne ihre unablässige Unterstützung und Kinderbetreuung wäre diese Arbeit nicht möglich gewesen.

Zu guter Letzt danke ich meinem Ehemann und Partner für seine bedingungslose Liebe und den beständigen Rückhalt, die mir stets Auftrieb verliehen haben.

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Abstract

Microplastics (MPs) are increasingly recognized as emerging contaminants in agricultural soils, where they alter soil structure, hydraulic properties, microbial communities, and plant performance. This work investigates the influence of soil texture and MP type on soil hydraulic properties, early winter wheat growth, and MP mineralization dynamics. In Chapter 2, the effects of four MP types (LDPE, PP, PS, and PES microfibers) on the hydraulic properties of three soils with contrasting textures (Luvisol, Albic Luvisol, and Chernozem) were evaluated at a concentration of 0.4% w/w. Results demonstrated that MP impacts on water retention and hydraulic conductivity varied with soil type, moisture conditions, and MP properties. With PES microfibers generally enhancing water retention in finer soils but reducing it in sandy soils, while PS and PP particles tended to reduce soil porosity and block pores, hindering water flow. In contrast, LDPE did not show significant influence on soil water hydraulics.

Chapter 3 assessed the effects of MPs on soil-plant-microbe interactions in early winter wheat. Results demonstrate that MPs primarily influenced belowground plant performance, while aboveground plant biomass remained largely unaffected. PES and PP significantly reduced root length and biomass, effects that coincided with altered nutrient dynamics, including elevated ammonium and nitrate concentrations. Microbial co-occurrence network analysis further revealed that MPs reshaped rhizosphere microbial community structures in distinct ways. LDPE exposure increased network complexity, enhancing microbial connectivity and centrality of key taxa such as *Vicinamibacteraceae*, *Pseudolabrys*, and *Nitrospira japonica*. In contrast, PES and PP reduced microbial clustering and average node degree, while PS exhibited no effects. Collectively, these findings indicate that MPs alter soil nutrient availability and rhizosphere microbial networks, with polymer-specific consequences for root development and community-level microbial organization.

Chapter 4 explored the interaction between MPs and soil texture on winter wheat performance in an early growth stage. While LDPE and PES MPs reduced root growth and nutrient uptake in loamy silt and sandy soils (Luvisol and Albic Luvisol) by increasing macroporosity and promoting water loss, LDPE improved root development in silty clay loam (Chernozem) by enhancing soil aeration. These findings highlight the complexity of MP-soil-plant interactions, strongly mediated by soil texture.

Finally, Chapter 5 addressed MP mineralization via Fenton reactions, revealing that mineralization rates vary with polymer type. PES microfibers exhibited the highest mineralization rate, followed by PP, while LDPE and PS were more resistant. Increased H₂O₂ concentrations improved mineralization of PES and PP but had limited effects on LDPE and PS. Although CO₂ release confirmed mineralization, scanning electron microscopy (SEM) imaging showed no visible surface changes, emphasizing the need for comprehensive analytical approaches to track MP breakdown.

Collectively, this research underscores the critical role of MP type and soil texture in shaping the environmental behavior of MPs, affecting soil functionality, plant performance, and mineralization potential. The findings call for more systematic, long-term studies to better understand and mitigate the impacts of MPs on agroecosystems.

Zusammenfassung

Mikroplastik (MP) wird zunehmend als neue Verunreinigung in landwirtschaftlichen Böden erkannt, wo es die Bodenstruktur, die hydraulischen Eigenschaften, die mikrobiellen Gemeinschaften und die Pflanzenleistung verändert. In dieser Arbeit wird der Einfluss von unterschiedlichen MP-Typen und der Bodentextur auf die hydraulischen Eigenschaften des Bodens, das Wachstum von Winterweizen im Frühstadium und die Dynamik des MP-Abbaus untersucht. In Kapitel 2 wurden die Auswirkungen von vier MP-Typen (LDPE-, PP-, PS-Partikel und PES-Mikrofasern) auf die hydraulischen Eigenschaften von drei unterschiedlichen Böden mit unterschiedlicher Textur (Luvisol, Albic Luvisol und Chernozem) in einer Konzentration von 0,4 % w/w untersucht. Die Ergebnisse zeigten, dass die Auswirkungen von MP auf das Wasserrückhaltevermögen und die hydraulische Leitfähigkeit je nach Bodentyp, Feuchtigkeitsbedingungen und MP-Eigenschaften variierten. Dabei verbesserten die PES-Mikrofasern im Allgemeinen das Wasserrückhaltevermögen in feineren Böden, verringerten es jedoch in sandigen Böden, während PS- und PP-Partikel dazu neigten, die Porosität im Boden zu verringern, Bodenporen zu blockieren und den Wasserfluss zu behindern. Im Gegensatz dazu zeigten LDPE-Partikel keinen signifikanten Einfluss auf den Wasserfluss.

In Kapitel 3 wurde untersucht, wie sich MP auf Winterweizen im frühen Wachstumsstadium auswirkt. Die Ergebnisse zeigten, dass MP vor allem die unterirdische Pflanzenleistung beeinflusst, während die oberirdische Biomasse weitgehend unverändert blieb. Insbesondere PES- und PP-MP führten zu einer Verringerung der Wurzellänge und der Wurzelbiomasse. Diese Effekte gingen mit Veränderungen der Nährstoffdynamik im Boden einher, vor allem mit erhöhten Ammonium- und Nitratkonzentrationen. Darüber hinaus ergab die Analyse der mikrobiellen Kookurrenznetzwerke, dass MP die Struktur der mikrobiellen Gemeinschaften in der Rhizosphäre polymerspezifisch verändert. Eine Exposition gegenüber LDPE erhöhte die Komplexität der mikrobiellen Netzwerke und verbesserte die Konnektivität sowie die Zentralität wichtiger Taxa wie *Vicinamibacteraceae*, *Pseudolabrys* und *Nitrospira japonica*. Im Gegensatz dazu führten PES- und PP-MP zu einer Reduktion der mikrobiellen Clusterbildung und des durchschnittlichen Knotengrads, während die PS-Exposition keine signifikanten Effekte zeigte. Zusammenfassend verdeutlichen diese Ergebnisse, dass MP sowohl die Nährstoffverfügbarkeit im Boden als auch die mikrobiellen Netzwerke in der Rhizosphäre beeinflusst.

Die Auswirkungen sind dabei stark polymerspezifisch und zeigen sich insbesondere in der Wurzelentwicklung, sowie in der Organisation der mikrobiellen Gemeinschaften.

Kapitel 4 untersuchte die Wechselwirkung zwischen MP und Bodentextur auf die Wuchsleistung von jungem Winterweizen. Während LDPE und PES MPs das Wurzelwachstum und die Nährstoffaufnahme in lehmigen Schluff- und Sandböden (Luvisol und Albic Luvisol) durch Erhöhung der Makroporosität und Förderung des Wasserverlustes verringerten, verbesserte LDPE die Wurzelentwicklung in lehmigen Chernozem durch eine Verbesserung der Bodenbelüftung. Diese Ergebnisse verdeutlichen die Komplexität der Wechselwirkungen zwischen MP, Boden und Pflanzen, die stark von der Bodenbeschaffenheit abhängen.

Kapitel 5 schließlich befasste sich mit dem MP-Abbau durch die Fenton-Reaktion und zeigte, dass die Abbaugeschwindigkeit je nach Polymertyp variiert. PES-Mikrofasern wiesen die höchste Abbaurate auf, gefolgt von PP, während LDPE und PS widerstandsfähiger waren. Erhöhte H_2O_2 -Konzentrationen erhöhten die Abbaurate bei PES und PP, hatten aber nur begrenzte Auswirkungen auf LDPE und PS. Obwohl die CO_2 -Freisetzung den Abbau bestätigte, zeigte die SEM-Bildgebung keine sichtbaren Oberflächenveränderungen, was die Notwendigkeit umfassender analytischer Ansätze zur vollständigen Bewertung des MP-Abbaus verdeutlicht.

Zusammenfassend unterstreicht diese Arbeit die zentrale Rolle von Bodentextur und MP-Typ hinsichtlich des Umweltverhaltens von MP und seinen Auswirkungen auf Bodenfunktionen, Pflanzenleistung und Abbaupotenziale. Die Ergebnisse verdeutlichen die Notwendigkeit weiterer systematischer Langzeitstudien, um die Auswirkungen von MP auf Agrarökosysteme besser verstehen und minimieren zu können.

Table of Contents

Danksagung	i
Abstract	ii
Zusammenfassung	iii
Table of Contents	v
List of Figures	ix
List of Tables	xviii
List of Abbreviations	xxi
CHAPTER 1 – Introduction	1
1.1. Background and rationale	1
1.2. State of the art	3
1.2.1. Overview of microplastics in the environment	3
1.2.2. Existing knowledge on microplastic-soil interactions	10
1.2.3. Emerging insights into microplastic-plant interactions	14
1.2.4. Current understanding of microplastic degradation and mineralization in soil	20
1.2.5. Research gaps and challenges	25
1.3. Objectives and research questions	27
CHAPTER 2 – Soil texture governs the influence of microplastics on soil hydraulic properties	30
Abstract	30
2.1. Introduction	31
2.2. Materials and methods	34
2.2.1. Soils	34
2.2.2. Microplastics	35
2.2.3. Soil preparation	36
2.2.4. Soil hydraulic characterization	37
2.2.5. Data analysis	39
2.3. Results	40
2.3.1. Saturated hydraulic conductivity (K_{sat})	40
2.3.2. Near-saturated hydraulic conductivity (K_{ns})	41

2.3.3.	K_{ns} -to- K_{sat} ratio	42
2.3.4.	Effect sizes of soil type vs. microplastic type	43
2.3.5.	Soil water retention and other soil characteristics	44
2.3.6.	Water-filled pore space (<i>WFPS</i>) and pore volume	49
2.4.	Discussion	52
2.4.1.	Microplastic effects in the Luvisol	52
2.4.2.	Microplastic effects in the Albic Luvisol	54
2.4.3.	Microplastic effects in the Chernozem.....	55
2.5.	Conclusion	56
	Acknowledgements	57
CHAPTER 3 – Microplastics affect belowground performance of winter wheat and microbial networks in the rhizosphere		58
	Abstract	58
3.1.	Introduction.....	59
3.2.	Materials and methods	61
3.2.1.	Soil	61
3.2.2.	Microplastics	61
3.2.3.	Experimental design	61
3.2.4.	Soil analysis.....	62
3.2.5.	Root analysis via WinRHIZO.....	63
3.2.6.	Leaf pigment analysis	63
3.2.7.	Microbiome analysis.....	63
3.2.8.	Statistical analysis of the soil and plant data	65
3.3.	Results	65
3.3.1.	Root analysis.....	65
3.3.2.	Soil nutrient analysis	66
3.3.3.	Microbiome analysis.....	69
3.3.4.	Combined effects of the different microplastics on microbiome and soil nutrients and root biomass	72

3.4. Discussion	73
3.5. Conclusion	76
CHAPTER 4 – Soil texture modifies the impact of microplastics on winter wheat growth.....	78
Abstract	78
4.1. Introduction.....	79
4.2. Materials and methods	81
4.2.1. Soils.....	81
4.2.2. Microplastics	82
4.2.3. Experimental design	82
4.2.4. Soil analysis.....	84
4.2.5. Root analysis.....	84
4.2.6. Leaf analysis.....	84
4.3. Results	85
4.3.1. Root analysis.....	85
4.3.2. Soil analysis.....	88
4.3.3. Leaf analysis.....	92
4.3.4. Plant biomass and root-to-shoot ratio	94
4.4. Discussion	97
4.4.1. Microplastic effects in loamy silt (Luvisol)	97
4.4.2. Microplastic effects in loamy sand (Albic Luvisol).....	99
4.4.3. Microplastic effects in silty clay loam (Chernozem).....	101
4.5. Conclusion	104
Acknowledgements	104
CHAPTER 5 – Microplastic mineralization rate in Fenton reactions depends on polymer type	105
Abstract	105
5.1. Introduction.....	106
5.2. Materials and methods	107
5.2.1. Microplastics	107
5.2.2. Fenton’s reagent	108

5.2.3.	Experimental set-up	108
5.2.4.	Curve fitting and calculations	109
5.2.5.	Scanning-electron microscopic (SEM) images.....	111
5.2.6.	Statistical evaluation	111
5.3.	Results	111
5.3.1.	CO ₂ emission patterns during the Fenton-mediated mineralization of the different microplastics	111
5.3.2.	Mineralization kinetics	113
5.3.3.	Surface structure after the radical attack	116
5.4.	Discussion	117
5.5.	Conclusion	122
CHAPTER 6 – Synopsis		123
6.1.	Summary.....	123
6.2.	Synthesis.....	127
6.3.	Conclusions and outlook	134
Appendices		138
Appendix A: Supplementary material of Chapter 2		138
Appendix B: Supplementary material of Chapter 3.....		147
Appendix C: Supplementary material of Chapter 4.....		160
Appendix D: Supplementary material of Chapter 5		164
Literature.....		168

List of Figures

Fig. 1.1. Concept graphic about the effects of microplastics (MPs) on plant performance. MPs can impede seed germination by obstructing testa pores and infiltrating seed tissues. Furthermore, they can influence aboveground plants’ vegetative or reproductive growth. Additionally, MPs have the potential to alter root traits and rhizosphere, thereby affecting root elongation. Taken from Chen et al. (2024), with friendly permission by Prof. Dr. Ke Sun, School of Environment, Beijing Normal University, Beijing..... 20

Fig. 1.2. Scanning electron microscopy (SEM) images of fungal hyphae attached to different microplastic (MP) types, which had been incubated in different soils for different time periods. A) Polypropylene (PP) particle incubated for 24 months in Albic Luvisol (AL) soil type, b) PP particle incubated for 12 months in AL soil type, c) low-density polyethylene (LDPE) particle incubated for 6 months in AL soil type, d) LDPE particle incubated for 3 months in Luvisol soil type, e) polystyrene (PS) particle incubated for 12 months in AL soil type, f) polyester microfiber incubated for 3 months in AL soil type. White arrows are pointing towards fungal hyphae..... 24

Fig. 2.1. Boxplot diagram of saturated hydraulic conductivity (K_{sat} , cm day⁻¹) for a) the Luvisol, b) Albic Luvisol, and c) Chernozem and near-saturated hydraulic conductivity (K_{ns} , cm day⁻¹) for d) the Luvisol, e) Albic Luvisol, and f) Chernozem mixed with the four different microplastic (MP) types at 0.4% w/w, and the respective soil types without the addition of MP. Boxes are the interquartile range (IQR) from the 25th to 75th percentile, lines inside boxes indicate the median, open squares indicate the mean values, upper and lower whiskers are 1.5 times the IQR, and markers are individual data points. Asterisks (*) mark significant differences in K_{sat} or K_{ns} between MP treatment and control, which was tested via one-sided, heteroscedastic t-test ($p \leq 0.05$). Double asterisks (**) mark highly significant differences at $p \leq 0.001$. Note, that the y-axis are scaled differently for better visualization. 41

Fig. 2.2. Size of the effect (η^2) of soil and microplastic (MP) type on a) K_{sat} (cm d⁻¹), and b) K_{ns} (cm d⁻¹). $\eta^2(A)$ is the effect size of soil type, $\eta^2(B)$ of MP type, $\eta^2(AB)$ of the interaction of soil and MP type, and $\eta^2(E)$ the error. LDPE is low-density polyethylene, PP is polypropylene, PS is polystyrene, and PES polyester microfibers made of polyethylene terephthalate..... 43

Fig. 2.3. Mean soil water retention curves for the a) Luvisol, b) the Albic Luvisol, and c) the Chernozem mixed with the four different microplastic (MP) amendments at 0.4% (w/w). LDPE is low-density polyethylene, PP is polypropylene, PS is polystyrene, and PES is polyester microfiber comprised of

polyethylene terephthalate. The control is lacking the addition of MP. Significant differences in water content at given pressure heads from the control are marked with an asterisk (*) and were tested via one-sided, heteroscedastic *t*-test ($p \leq 0.05$)..... 48

Fig. 2.4. Differences between the water-filled pore space (*WFPS*, %) calculated between volumetric water contents at saturation and given pressure heads ($\log_{10} h = 0, 0.6, 1.2, 1.8, 2.4, 3, 3.6, 4.2, 4.8, 5.4, \text{ and } 6 \text{ -cm}$) for the control and the microplastic amended a) Luvisol, b) Albic Luvisol, and c) Chernozem. LDPE is low-density polyethylene, PP is polypropylene, PS is polystyrene, and PES is polyester microfiber comprised of polyethylene terephthalate. Solid line marks zero, representing the control. Significant difference ($p \leq 0.5$) from the control is marked with an asterisk and was tested via one-sided, heteroscedastic *t*-test. Note that the y-axis are scaled differently for better visualization. 50

Fig. 2.5. Total porosity (assuming that total porosity equals θ_s), macroporosity, air capacity (all in %), and plant-available water ($\text{cm}^3 \text{ cm}^{-3}$) without (control) and with the addition of different microplastic types in a) the Luvisol, b) Albic Luvisol, and c) the Chernozem. LDPE is low-density polyethylene, PP is polypropylene, PS is polystyrene, and PES is polyester microfibers comprised of polyethylene terephthalate. Significant differences from the control are marked with an asterisk (*) and were tested via one-sided, heteroscedastic *t*-test ($p \leq 0.05$)..... 51

Fig. 3.1. Model of two rhizotrons used as plant growth chambers. The left rhizotron shows the control (0soil) without the addition of microplastic (MP) and the right one shows an MP treatment with 0.4% (w/w) of MP addition in soil (+soil). 62

Fig. 3.2. Root length of winter wheat plants grown in rhizotrons for 28 days measured via WinRHIZO. “+MP” indicates microplastic (MP) addition to soil, while no MP was added to control. The following MP types were added: LDPE = low-density polyethylene, PP = polypropylene, PS = polystyrene, PES = polyester microfiber. Hatched areas of the bars represent the portion of primary roots (0.4-1 mm in width), and unhatched areas the portion of lateral roots (0-0.4 mm in width). Whiskers represent the standard deviation of five replicates. The asterisk indicates significant difference ($p \leq 0.05$), while the asterisk in parentheses shows a weakly significant difference ($0.05 \leq p \leq 0.1$) to the control, which was tested with a one-sided, heteroscedastic *t*-test..... 66

Fig. 3.3. Nutrients per root surface area (RSA) (in $\mu\text{g cm}^{-2}$) of plant-available a) nitrate (NO_3^-), b) ammonium (NH_4^+), c) nitrite (NO_2^-), d) potassium (K_{av}) and e) phosphate (P_{av}) in rhizosphere (rhizo) and

bulk soil after winter wheat was grown for 28 days in rhizotrons. “+MP” indicates microplastic (MP) addition to soil, while the control represents soil without MP addition. The following MP types were added to the soil: LDPE = low-density polyethylene particles, PP = polypropylene particles, PS = polystyrene particles, PES = polyester microfibers. Boxes are the interquartile range (IQR) from the 25th to 75th percentile, lines inside boxes indicate the median, white squares indicate the mean values, upper and lower whiskers are 1.5 times the IQR and diamond-shaped markers are outliers (n = 5). Asterisks show significant differences ($p \leq 0.05$) and asterisks in parentheses show weak significant differences ($0.05 \leq p \leq 0.1$) to the control, tested with one-sided, heteroscedastic *t*-test. 67

Fig. 3.4. Plant-available potassium (K_{av}) versus plant-available phosphate (P_{av}) content in soil per root surface area (RSA) (in $\mu\text{g cm}^{-2}$) in microplastic-amended soil (+MP, dark cyan triangles) and the control (black squares) without MP. Soil nutrients were analyzed after winter wheat was grown for 28 days in rhizotrons. 68

Fig. 3.5. Community structure of bacteria colonizing bulk soil (bulk) and rhizosphere (rhizo); a) Box plot of ASV richness in bulk soil and rhizosphere. Statistical analysis was performed using Kruskal-Wallis and Wilcoxon Rank-Sum test with Benjamini-Hochberg correction for multiple comparisons; b) PCoA plot of weighted UniFrac distances including all MP treatments. Statistical analysis was performed using PERMANOVA with Benjamini-Hochberg correction for multiple comparisons. 69

Fig. 3.6. Venn diagram of shared ASVs between MP materials and MP-treated soil and rhizosphere. All ASVs with at least 0.1% reads in 75% of the samples per group were considered for core microbiome analysis. 70

Fig. 3.7. Microbial co-occurrence networks based on Pearson correlations for untreated (control) and MP-treated (LDPE, PES, PS, PP) rhizosphere samples. Each node represents an amplicon sequence variant (ASV), with node size proportional to its degree (number of connections). Node colors (purple and turquoise) indicate network modules (communities) identified by correlation-based clustering, representing groups of ASVs that are more strongly co-occurring with each other. All networks are compartmentalized, consisting of two hubs of microbial taxa that frequently interact and/or respond similarly to environmental cues. Edges represent significant correlations between ASVs (green = positive; red = negative). The networks include core ASVs, defined as those present in at least four out of five replicates. Displayed are the 100 most abundant ASVs. 72

Fig. 3.8. Principal coordinate analysis (PCoA) of rhizosphere bacterial communities associated with winter wheat after 28 days of growth, exposed to different MP treatments (Control, LDPE, PES, PP, PS). The ordination is based on Bray-Curtis dissimilarities of the microbial community composition. Red arrows indicate significant ($p < 0.05$) plant-available nutrient parameters fitted by *envfit*, showing correlations with community structure: plant available ammonium (NH_4^+), nitrite (NO_2^-), and phosphate (P_{av}). The direction and length of the arrows show the gradient and strength of the correlation, respectively. The plot illustrates that variations in microbial communities are partly explained by the availability of plant nutrients in the rhizosphere. 73

Fig. 4.1. Mean root lengths (in m) of winter wheat harvested from 75 cm long PVC tubes and grown in a) the loamy silt Luvisol, b) the loamy sand Albic Luvisol, and c) the silty clay loam Chernozem soil for 42 days. Plants were grown with the additions of the microplastic types low-density polyethylene (LDPE) particles and polyester microfibers (PES) at 0.4% w/w in dry soil. Control is the respective soil type without MP addition. Laterals indicate lateral roots with a width of 0-0.4 mm and Primaries (crosshatched) indicate primary roots with a width of 0.4-1 mm. Bars show the mean values of 4 replicates with whiskers representing the standard deviation. Asterisks show significant difference to the control ($p \leq 0.05$), while asterisks in parentheses show weak significance to the control ($p \leq 0.1$), tested via heteroscedastic, paired *t*-test..... 86

Fig. 4.2. Nutrient content including potassium (K), phosphate (P) and total nitrogen (N_t), in dry pulverized roots of winter wheat grown in a) the loamy silty Luvisol, b) the loamy sandy Albic Luvisol, and c) the silty clayey loamy Chernozem soil for 42 days. Plants were grown with the additions of the microplastic types low-density polyethylene (LDPE) particles and polyester microfibers (PES) at 0.4% w/w in dry soil. Control is the respective soil type without MP addition. Boxes are the interquartile range (IQR) from the 25th to 75th percentile, lines inside boxes indicate the median, white squares indicate the mean values, upper and lower whiskers are 1.5 times the IQR, and markers are individual data points. Asterisks show significant difference to the control ($p \leq 0.05$), while asterisks in parentheses show weak significance to the control ($p \leq 0.1$), tested via heteroscedastic, paired *t*-test. 87

Fig. 4.3. Soil nutrients including plant-available nitrogen compounds (NH_4^+ = ammonia, NO_2^- = nitrite, NO_3^- = nitrate) and plant-available potassium (K_{av}) and phosphate (P_{av}) content in a) the loamy silt Luvisol, b) the loamy sand Albic Luvisol and c) a silty clay loam Chernozem after the winter wheat growth period of 42 days. Initial refers to the respective nutrient concentration before plants were seeded. Microplastic (MP) types mixed at 0.4% w/w in soil were low-density polyethylene (LDPE) and

polyester (PES) microfibers. Control is s the respective oil type without MP addition. It was differentiated between rhizosphere soil (rhizo), which was shaken off the roots, and bulk soil not attached to the roots. Boxes are the interquartile range (IQR) from the 25th to 75th percentile, lines inside boxes indicate the median, white squares indicate the mean values, upper and lower whiskers are 1.5 times the IQR, and markers are individual data points. Asterisks show significant differences to the control ($p \leq 0.05$), while asterisks in parentheses show weak significance ($p \leq 0.1$), tested with heteroscedastic, paired *t*-test. 89

Fig. 4.4. Soil pH measured in calcium chloride extractions of a) the loamy silt Luvisol, b) the loamy sand Albic Luvisol, and c) the silty clay loam Chernozem soil measured after the winter wheat growth period of 42 days. Plants were grown with the addition of the microplastic types low-density polyethylene (LDPE) particles and polyester (PES) microfibers at 0.4% w/w in dry soil. Control is soil type without MP addition. It was differentiated between rhizosphere soil (rhizo), which was shaken off the roots, and bulk soil not attached to the roots. Boxes are the interquartile range (IQR) from the 25th to 75th percentile, lines inside boxes indicate the median, white squares indicate the mean values, upper and lower whiskers are 1.5 times the IQR, and markers are individual data points. Asterisks show significant difference to the control ($p \leq 0.05$), while asterisks in parentheses show weak significance ($p \leq 0.1$), tested with a heteroscedastic, paired *t*-test. 90

Fig. 4.5. Soil water content measured gravimetrically in a) the loamy silt Luvisol, b) the loamy sand Albic Luvisol, and c) the silty clay loam Chernozem soil measured after the winter wheat growth period of 42 days. Plants were grown with the additions of the microplastic types low-density polyethylene (LDPE) particles and polyester (PES) microfibers at 0.4% w/w in dry soil. Control is the respective soil type without MP addition. It was differentiated between rhizosphere soil (rhizo), which was shaken off the roots, and bulk soil not attached to the roots. Boxes are the interquartile range (IQR) from the 25th to 75th percentile, lines inside boxes indicate the median, white squares indicate the mean values, upper and lower whiskers are 1.5 times the IQR, and markers are individual data points. Asterisks show significant difference to the control ($p \leq 0.05$), while asterisks in parentheses show weak significance ($p \leq 0.1$), tested with a heteroscedastic, paired *t*-test. 91

Fig. 4.6. Chlorophyll content, including chlorophyll-a (*chl_a*) and chlorophyll-b (*chl_b*), and carotenoid (*car*) content in total leaf dry mass of winter wheat, grown for 42 days in a) the loamy silt Luvisol, b) the loamy sand Albic Luvisol, and c) the silty clay loam Chernozem, respectively. LDPE refers to plants grown in 0.4% w/w low-density polyethylene particles and PES refers to plants grown in 0.4% w/w polyester microfibers. Control is the respective soil type without MP addition. Boxes are the

interquartile range (IQR) from the 25th to 75th percentile, lines inside boxes indicate the median, white squares indicate the mean values, upper and lower whiskers are 1.5 times the IQR, and markers are individual data points. Asterisks show significant difference to the control ($p \leq 0.05$), while asterisks in parentheses show weak significance ($p \leq 0.1$), tested with a heteroscedastic, paired *t*-test. 92

Fig. 4.7. Nutrient content in total leaf dry mass of winter wheat harvested after 42 days and grown in a) the loamy silt Luvisol, b) the loamy sand Albic Luvisol, and c) the silty clay loam Chernozem, respectively. LDPE refers to plants grown in 0.4% w/w low-density polyethylene particles and PES refers to plants grown in 0.4% w/w polyester microfibers. Control is the respective soil type without MP addition. Boxes are the interquartile range (IQR) from the 25th to 75th percentile, lines inside boxes indicate the median, white squares indicate the mean values, upper and lower whiskers are 1.5 times the IQR, and markers are individual data points. Asterisks show significant differences to the control ($p \leq 0.05$), while asterisks in parentheses show weak significance to the control ($p \leq 0.1$), tested with a heteroscedastic, paired *t*-test. 94

Fig. 5.1. Net release of microplastic (MP)-derived carbon emitted as carbon dioxide (net CO₂-C) per MP mass during the mineralization of different MP types: a) LDPE = low-density polyethylene particles, b) PP = polypropylene particles, c) PS = polystyrene particles, and d) PES = polyester microfibers, using Fenton's reagent of different molar concentrations of hydrogen peroxide (H₂O₂) in aqueous solution: F0.2 = 0.2 M H₂O₂, F0.5 = 0.5 M H₂O₂ and F1 = 1 M H₂O₂. H₂O₂ was added in combination with Fe²⁺ in a molar ratio of H₂O₂:Fe²⁺ = 100:1. The second y-axis shows the amount of carbon (C) being released from the respective MP. All curves show blank-subtracted values. 112

Fig. 5.2. Change in CO₂ mixing ratio (in ppm) over time (in min) (d(CO₂)/dt) for the mineralization of different microplastic types: a) LDPE = low-density polyethylene particles, b) PP = polypropylene particles, c) PS = polystyrene particles, and d) PES = polyester microfibers, with Fenton's reagent of different molar concentrations of hydrogen peroxide (H₂O₂): F0.2 = 0.2 M H₂O₂, F0.5 = 0.5 M H₂O₂ and F1 = 1 M H₂O₂. H₂O₂ was added in combination with Fe²⁺ at a molar ratio of H₂O₂:Fe²⁺ = 100:1. 115

Fig. 5.3. Scanning electron microscopy (SEM) images of microplastic particles (LDPE = low-density polyethylene, PP = polypropylene, PS = polystyrene) and polyester (PES) microfibers before and after the treatment with Fenton's reagent of 1 M hydrogen peroxide (H₂O₂). 117

Fig. A.1. Blendtec blender for the production of MP from plastic beads (3-5 mm). 138

Fig. A.2. Particle size distribution of used microplastic (MP) particles. LDPE is low-density polyethylene, PP is polypropylene, PS is polystyrene. Each MP powder was measured three times and mean particle sizes were: LDPE = $444.6 \pm 1.1 \mu\text{m}$, PS = $532.3 \pm 4.7 \mu\text{m}$, PP = $548.0 \pm 4.7 \mu\text{m}$	139
Fig. A.3. Raman spectra of a) polyethylene terephthalate (PET) reference material and b) polyester (PES) fibers. Spectrum (b) is comparable to spectrum (a) which indicates that the polymeric structure of PES fibers is identical with PET.....	139
Fig. A.4. Mini-disk infiltrometer (MDI, Meter Group AG, Munich, Germany) filled with blue colored water placed on a soil-filled PVC column (10 x 10 cm).	140
Fig. A.5. Measured and fitted soil hydraulic characteristics for the Albic Luvisol (AL). From top downwards: control, LDPE, PP, PS, PES.....	142
Fig. A.6. Measured and fitted soil hydraulic characteristics for the Luvisol. From top downwards: control, LDPE, PP, PS, PES.....	144
Fig. A.7. Measured and fitted soil hydraulic characteristics for the Chernozem. From top downwards: control, LDPE, PP, PS, PES.....	146
Fig. B.1. Stereomicroscopic pictures of winter wheat roots harvested after a growth period of 28 days. a) Primary root (width: 0.58-0.6 cm) and lateral root (width: 0.37-0.39 cm), b) Lateral root (width: 0.16-0.17 cm), c) Primary root (width: 0.81-0.94 cm).	148
Fig. B.2. Shoot a) dry mass (in mg), b) length (in cm), c) nitrogen (N) concentration in leaves (in mg g^{-1}) and d) total leaf N content (mg) of winter wheat plants after the growth period of 28 days in rhizotrons. "+MP" represents microplastic (MP) addition to soil, while no MP was added to the soil of the control. The following MP types were added: LDPE = low-density polyethylene, PP = polypropylene, PS = polystyrene, PES = polyester fibers. Boxes are the interquartile range (IQR) from the 25 th to 75 th percentile, lines inside boxes indicate the median, white squares indicate the mean values and upper and lower whiskers are 1.5 times the IQR (n = 5).	150
Fig. B.3. Chlorophyll, a) chlorophyll-a (chl_a) and b) chlorophyll-b (chl_b), and c) carotenoid (car) content (in $\mu\text{g mg}^{-1}$) of winter wheat plants grown for 28 days in rhizotrons. "+MP" represents microplastic (MP) addition to soil, while no MP was added to the soil of the control. The following MP types were	

added: LDPE = low-density polyethylene, PP = polypropylene, PS = polystyrene, PES = polyester fiber. Boxes are the interquartile range (IQR) from the 25th to 75th percentile, lines inside boxes indicate the median, white squares indicate the mean values and upper and lower whiskers are 1.5 times the IQR (n = 5)..... 151

Fig. B.4. Shoot length (in cm) development over the growth period of winter wheat plants in rhizotrons for 28 days. Winter wheat plants grew without (control, in black squares) and with the addition of different microplastic (MP) types in soil (0.4% w/w). The following MP types were added: LDPE = low-density polyethylene particles in blue circles, PP = polypropylene particles in purple triangles, PS = polystyrene particles in green triangles, PES = polyester fibers in orange diamonds. 152

Fig. B.5. Soil nutrient analysis of plant-available nitrate (NO_3^-), ammonia (NH_4^+), nitrite (NO_2^-), potassium (K_{av}) and phosphate (P_{av}) (in $\mu\text{g g}^{-1}$) in rhizosphere (rhizo) and bulk soil before ("initial" in grey) and after the winter wheat growth period of 28 days in rhizotrons. The following microplastic (MP) types were added in soil (in dark cyan): LDPE = low-density polyethylene (in cyan), PP = polypropylene, PS = polystyrene, PES = polyester fiber. Control (in white) resembles no MP addition in soil. Boxes are the interquartile range (IQR) from the 25th to 75th percentile, lines inside boxes indicate the median, white squares indicate the mean values, upper and lower whiskers are 1.5 times the IQR and diamond-shaped markers are outliers (n = 5). The asterisk show significant difference ($p \leq 0.05$) towards the control and asterisks in parentheses show weak significant differences ($0.05 \leq p \leq 0.1$) to the control, tested with one-sided, heteroscedastic t-test..... 153

Fig. B.6. Plot of plant-available nitrite (NO_2^-) versus ammonium (NH_4^+) content of the soil per root surface area (RSA). 155

Fig. B.7. Plot of plant-available nitrate (NO_3^-) versus ammonium (NH_4^+) content of the soil per root surface area (RSA). 155

Fig. B.8. Rarefaction curve of amplicon sequence variants (ASVs) identified via 16S rRNA gene amplicon sequencing after subsampling to 44,905 reads per sample. The number of sequences ASVs reached a plateau for all samples, indicating a sufficient sequencing depth. 156

Fig. C.1. Particle size distribution of used low-density polyethylene (LDPE) microplastic particles, measured with a laser diffraction particle size analyzer (HORIBA LA-950). Particles were dispersed in ethanol and no ultrasound was used, n = 3. The mean particle diameter size was $444.6 \pm 1.1 \mu\text{m}$. . 161

Fig. C.2. Lysimeter space filled with sewage pipes used as plant pots (a). Each pipe contains one treatment. Full layout of treatments and arrangement in all of the four lysimeter spaces (b); the controls of the respective soil type without microplastic (MP) are bordered in grey, LDPE is low-density polyethylene (blue bordered), PES is polyester microfiber (orange bordered); LU is silty loamy Luvisol soil (yellow filled), AL is loamy sandy Albic Luvisol soil (green filled), CH is silty clayey loamy Chernozem soil (purple filled)..... 161

Fig. C.3. Weather data (temperature, in °C, black and precipitation, in mm, red) from 1st August 2023 to 30th September 2023. Data are taken from the weather station on the campus of Forschungszentrum Jülich. 161

Fig. C.4. Temperature conditions for roots in the lysimeter spaces. The mean temperature (T_{mean}) for all four lysimeter spaces was 18.7 ± 0.18 °C. Maximum (T_{max}) and minimum (T_{min}) temperatures were: $T_{\text{max}} = 23.11 \pm 0.91$ °C and $T_{\text{min}} = 13.03 \pm 0.13$ °C. 162

Fig. D.1. Particle size distribution of polystyrene (PS) in black, low-density polyethylene (LDPE) in grey and polypropylene (PP) in light grey for gas emission measurement. Mean particle diameters were 283 ± 0.3 µm, 353 ± 0.8 µm, and 305 ± 1.6 µm for PS, LDPE, and PP, respectively. 164

Fig. D.2. Experimental set-up of gas emission measurements of different MP types treated with Fenton's reagent. 165

Fig. D.3. Curve fitting for experimental data of CO₂ emissions, in ppm, with standard deviation (whiskers) for the degradation of different microplastic types (LDPE = low-density polyethylene, PP = polypropylene, PS = polystyrene, PES = polyester microfiber) with Fenton's reagent of different hydrogen peroxide (H₂O₂) concentrations: F1 = 1 M H₂O₂ + Fe²⁺, F0.5 = 0.5 M H₂O₂ + Fe²⁺, F0.2 = 0.2 M H₂O₂ + Fe²⁺. F1 and F0.5 experimental data (dots) were fitted with an exponential fit (ExpDec2 in Origin, solid lines) and the F0.2 experimental data were fitted with a polynomial fit (solid lines). 165

Fig. D.4. Scanning electron microscopy (SEM) images of microplastic particles (LDPE = low-density polyethylene, PP = polypropylene, PS = polystyrene) and polyester (PES) microfibers before and after the treatment with Fenton's reagent of 2 M hydrogen peroxide (H₂O₂). 167

List of Tables

Table 1.1. Most commonly used plastics, their structure, application, densities and appearance as microplastic (MP) in soil.	7
Table 1.2. Selected examples (published 2020-2023) of the reported effects of different microplastics (MPs) on plant growth and development. Adapted from Jia et al. (2023). (Jia et al., 2023)	16
Table 2.1. Characteristics of experimental soils including organic carbon (C_{org}) content, plant-available mineral nitrogen content (N_{min}), plant-available phosphate (P_{av}) and potassium (K_{av}).	34
Table 2.2. Characteristics of the experimental microplastic.	36
Table 2.3. Van Genuchten parameters for the three soil types used according to the user's manual of the MDI (Decagon Devices, WA, USA) as well as calculated A values.	39
Table 2.4. Mean K_{ns} -to- K_{sat} ratio for the three soils mixed with low-density polyethylene (LDPE) particles, polypropylene (PP) particles, polystyrene (PS) particles, and polyester (PES) microfibers at 0.4% w/w or no microplastics (control).	42
Table 2.5. van Genuchten parameters fitted on the four replicates used in the evaporation method, and calculated water content at saturation (θ_s), which is equivalent to total porosity, residual water content (θ_r), tortuosity (λ) water content at field capacity (θ_{FC}), water content at wilting point (θ_{WP}), and plant-available water (θ_{PAW}) for the tree different soils and four different microplastics (LDPE = low-density polyethylene, PP = polypropylene, PS = polystyrene, and PES = polyester microfiber comprised of polyethylene terephthalate) added at 0.4% w/w.	45
Table 3.1. Characteristics of experimental soil, including soil total nitrogen (N_t), soil organic carbon (C_{org}), plant-available mineral nitrogen (N_{min}), phosphate (P_{av}), and potassium (K_{av}) content.	61
Table 4.1. Characteristics of experimental soils including organic carbon (C_{org}) content, plant-available mineral nitrogen content (N_{min}), plant-available phosphate (P_{av}) and potassium (K_{av}).	82
Table 4.2. Root and shoot biomass (in mg) and root-to-shoot ratio of winter wheat plants grown for 42 days in loamy silt Luvisol, loamy sand Albic Luvisol and silty clay loam Chernozem, in the absence of	

microplastics (control), and with amendment of either low-density polyethylene (LDPE) particles or polyester (PES) microfibers. 96

Table 5.1. Amounts of the different components of the Fenton's reagents contained in 2 mL FeSO₄ solution and 13 mL of H₂O₂ solution (molar ratio of H₂O₂/Fe²⁺ = 100:1 mol/mol) for gas emission measurements. Fenton's reagent was used in different molar concentrations of H₂O₂: F0.2 = 0.2 M H₂O₂, F0.5 = 0.5 M H₂O₂ and F1 = 1 M H₂O₂. The corresponding blank consisted of the same amounts and volumes, but without the addition of MP. 108

Table 5.2. Amount of microplastic (MP)-derived carbon released as CO₂ (in µg CO₂-C g⁻¹ MP and percent of original MP mass) after 240 min treatment of different MP types (PES = polyester microfibers, PP = polypropylene particles, PS = polystyrene particles, LDPE = low-density polyethylene particles) with Fenton's reagent at different molar concentrations of H₂O₂, 0.2 M H₂O₂ = F0.2, 0.5 M H₂O₂ = F0.5, and 1 M H₂O₂ = F1, amended with FeSO₄ at a molar H₂O₂:Fe²⁺ ratio of 100:1. 113

Table A.1. Relevant assignments of functional groups of polyethylene terephthalate in Raman spectra (Rebollar et al., 2014; Alexiou et al., 2020). 140

Table B.1. Parameter set for WinRHIZO analysis. 148

Table B.2. Two-way ANOVA testing the effect of root zone and microplastic (MP) type on the amount of soil nutrients (NO₃⁻ = nitrate, NH₄⁺ = ammonium, NO₂⁻ = nitrite, K_{av} = potassium, P_{av} = phosphate) per root surface area (µg cm⁻²), described by the test statistic (F) and critical test statistic (F_{crit.}) values, the p-value lower than 0.05, and the effect size eta-squared (η²). 154

Table B.3. Statistical analysis of different alpha diversity measures. Statistical analysis was performed using Kruskal-Wallis and the Wilcoxon Rank-Sum test with Benjamini-Hochberg correction for multiple comparisons. 156

Table B.4. Global network properties of the microbial co-occurrence network based on Pearson correlation for untreated (control) and low-density polyethylene (LDPE) microplastics treated rhizosphere samples. Both treatments formed single-component networks with high clustering coefficients and edge densities. LDPE reduced vertex and edge connectivity by 50% and slightly lowered modularity. Average dissimilarity and path length remained comparable. 157

Table B.5. Global network properties of the microbial co-occurrence network based on Pearson correlation for untreated (control) and polyester (PES) fiber microplastics treated rhizosphere samples. Both treatments formed single-component networks with high clustering coefficients and edge densities. LDPE reduced vertex and edge connectivity by 50% and slightly lowered modularity. Average dissimilarity and path length remained comparable.	158
Table B.6. Global network properties of the microbial co-occurrence network based on Pearson correlation for untreated (Control) and polypropylene (PP) microplastics treated rhizosphere samples. Both treatments formed single-component networks. The PES network showed reduced clustering coefficient, edge density, connectivity, and natural connectivity, indicating decreased network complexity and cohesion.....	159
Table B.7. Global network properties of the microbial co-occurrence network based on Pearson correlation for untreated (control) and polystyrene (PS) microplastics treated rhizosphere samples. Both treatments formed single-component networks. The PP network showed a lower clustering coefficient, lower edge density, and natural connectivity compared to control. Vertex and edge connectivity increased under PP.	160
Table C.1. Parameter set for WinRHIZO analysis of root characteristics.	163
Table D.1. Variable definition for graph fitting.	166
Table D.2. Orders of reaction kinetics (n) of the degradation of different microplastic types (PES = polyester microfibers, PP = polypropylene particles, LDPE = low-density polyethylene particles) using Fenton's reagent. n was determined by using x, the factor by which the reaction rate was enhanced from 0.5 M to 1 M of H ₂ O ₂	166

List of Abbreviations

+MP	microplastic added to soil
AC	air capacity index
AL	Albic Luvisol
CaCl ₂	calcium chloride
CAL	calcium-acetate lactate solution
<i>car</i>	carotenoid
Cd	cadmium
CH	Chernozem
<i>chl_a</i>	chlorophyll-a
<i>chl_b</i>	chlorophyll-b
C _{org}	organic carbon
ECM	ectomycorrhizal fungi
FC	field capacity
H ₂ O ₂	hydrogen peroxide
HDPE	high-density polyethylene
K _{av}	plant-available potassium
K _{ns}	near-saturated hydraulic conductivity
K _{sat}	saturated hydraulic conductivity
LDPE	low-density polyethylene
LU	Luvisol
MDI	Mini-Disk Infiltrrometer
MP	microplastic
NH ₄ ⁺	ammonia
N _{min}	plant-available mineral nitrogen
NO ₂ ⁻	nitrite
NO ₃ ⁻	nitrate
N _t	total nitrogen
OH•	hydroxyl radical
PA fibers	polyacrylic fibers
P _{av}	plant-available phosphate
PAW	plant-available water
PAWC	plant-available water capacity
Pb	lead

PE	polyethylene
PES	polyester
PET	polyethylene terephthalate
P_{mac}	macropore volume
PMMA	polymethyl methacrylate
PP	polypropylene
PS	polystyrene
PU	polyurethane
PVC	polyvinyl chloride
RHZ	single-compartment rhizotron
ROS	reactive oxygen species
RSA	root surface area
SEM	scanning electron microscopy
SWRC	soil water retention curves
UV	ultraviolet
$WFPS$	water-filled pore space
WHC	water-holding capacity
WP	wilting point
Zn	zinc
θ	volumetric water content
θ_r	volumetric residual water content
θ_s	volumetric water content at full saturation
λ	tortuosity

CHAPTER 1 – Introduction

1.1. Background and rationale

Microplastic (MP) pollution, once largely overlooked, has been recognized as a growing environmental concern in recent years. While much attention has been directed towards the impact of MPs in aquatic ecosystems, their presence in terrestrial environments, particularly soil, has been comparatively under-recognized and under-researched. Current estimates of the stock of MPs in agricultural soils amount to 1.5 to 6.6 million tons (Kedzierski et al., 2023), which could be potentially more than the quantity of MPs present on the surface of the oceans (Eriksen et al., 2014). Therefore, soil is likely the largest reservoir of MP accumulation, as plastics from various sources – including agricultural practices, industrial waste, and atmospheric deposition – find their way into the soil.

MPs, defined as plastic particles and fibers smaller than 5 mm, were often thought to be relatively inert in soil systems and therefore harmless (Rillig, 2012), particularly when compared to more prominent forms of pollution, like heavy metals, pesticides or industrial chemicals. However, growing evidence suggests that these small particles are capable of interacting with soil ecosystems in complex and potentially harmful ways, affecting soil health, biodiversity, and even human food systems (Yu et al., 2022; Zhang et al., 2022a; Li et al., 2023c). Despite the growing body of research on MPs in soil, and the growing scientific consensus that MP pollution in soil is a significant environmental concern, significant uncertainties remain about the full extent of their impact on terrestrial ecosystems.

One reason for these uncertainties is that soil is a dynamic system with complex interactions between plants, microorganisms, and various physical as well as chemical processes (Turner, 2021). It consists of various components such as organic matter, minerals, water, air, and living organisms, all interacting in intricate ways (Keefer, 1999). Moreover, soils vary greatly in texture, pH, nutrient levels, moisture content, and microbial communities. This diversity makes it difficult to generalize how MPs will behave or affect different types of soil environments.

In this complex environment, MPs may influence soil structure, water retention, and nutrient availability, indirectly impacting plant-soil interactions. However, the effects of MPs on plant growth and performance cannot be generalized, as root systems vary widely not only by plant species but also by the type of soil in which they grow (Helliwell et al., 2019). Different plants in different environments have unique rhizosphere compositions – the soil region directly affected by root secretions and microbial communities (Lei et al., 2019; Ling et al., 2022). These differences in root architecture, exudates, and associated microorganisms can also shape the effects of MPs on soil nutrient availability

and water absorption (Chen et al., 2024). Consequently, the impact of MPs on plant health reflects a strong interconnection between root systems, soil types, and MPs, requiring a case-by-case approach.

Another level of complexity is added by the diversity of the structure and chemical composition of MPs, despite the fact that they can be generally defined as long-chained polymers that are highly persistent in the environment. This is because they encompass a wide variety of shapes, sizes, polymer compositions, and chemical structures. Different types of MPs come from different sources, including synthetic fibers, fragments from larger plastic debris, and degraded products of consumer goods, each with unique physical and chemical properties (Hoang et al., 2024). As a result, their environmental behavior, degradation and mineralization rates, and potential impacts differ significantly, demonstrating that these differences need to be considered when studying or addressing MP effects in soil. Furthermore, MPs can act as vectors for other pollutants, such as heavy metals or persistent organic pollutants, increasing their environmental risks (Iqbal et al., 2022).

Moreover, MPs have been long seen as less harmful due to their relatively high chemical inertness. However, several processes in soil and the broader environment foster their degradation, releasing harmful additives and thereby causing long-term effects on soil health that are difficult to predict (Du et al., 2020). The degradation of MPs, driven by factors such as weathering (e.g. UV radiation, temperature fluctuations) (Jansen et al., 2024), mechanical abrasion (Bullard et al., 2023), and chemical reactions involving microorganisms or fungi (Cai et al., 2023; Xiang et al., 2023), occurs over decades. Both the effects of weathered MPs in soil and the mechanisms by which MPs degrade are still areas of active research.

To address these uncertainties, this dissertation focuses on two key aspects of MPs in soil. The first aspect explores whether and how MPs affect essential physical and biogeochemical processes in soil. Specifically, it investigates how different MP types affect soil physical properties across various soil textures, examines the specific impacts of different MP types on plant growth and nutrient uptake, and analyzes how these effects vary with soil texture. The second aspect examines the contribution of soil-related reactions, such as Fenton reactions, to MP mineralization, with the goal of proving and characterizing these reaction mechanisms as a foundation for future mineralization studies.

1.2. State of the art

1.2.1. Overview of microplastics in the environment

1.2.1.1. Definition, properties and classification of microplastics

The widely accepted definition of MPs describes them as small plastic particles or fibers ranging in size from 1 micron to 5 millimeters. They form either through the fragmentation of larger plastic debris or are intentionally manufactured for use in a variety of consumer products. MPs are made up of synthetic hydrocarbons, mainly derived from fossil fuels, which is what makes them so widespread and persistent in the environment. The general material properties of plastics – such as their durability, resistance to degradation and mineralization, and versatility – are the reason why they are widely used in many industries and how these same properties contribute to their persistence in the environment, including the formation and long-term presence of MPs.

The unique material properties of plastics, often surpassing those of other materials, offer a lot of advantages across a wide range of applications. Their resistance to various factors, such as corrosion, chemicals, or moisture, makes them reliable materials in environments where others might degrade. Furthermore, the light weight and malleability of plastics saves transportation costs and allows for the creation of complex shapes and designs which would be challenging with materials like metal or glass. The ability to engineer plastics by adding various additives to create a range of properties, such as flexibility, rigidity, and thermal resistance, allows for the easy fulfilment of specific functional requirements. This versatility of plastics has made them essential to industries such as automotive, electronics, healthcare, packaging, and construction, leading to their widespread use around the world.

The most common plastic types that are produced globally are polyethylene (PE), polypropylene (PP), polyvinyl chloride (PVC), polyethylene terephthalate (PET), and polystyrene (PS) (Table 1.1). PE, both in high-density (HDPE) and low-density (LDPE) forms, is extensively used in packaging, including plastic bags and containers, due to its flexibility and durability. PP is highly valued for its strength and heat resistance, making it ideal for automotive parts, textiles, and reusable containers. PVC is widely used in construction for pipes, window frames, and flooring because of its robustness and longevity. PET, commonly found in beverage bottles and food packaging, is appreciated for its transparency, strength, and recyclability, and it is particularly useful in producing synthetic fibers. PS is predominantly used for disposable cutlery, packaging, and insulation. These plastics have been central to Germany's production, driven by strong demand across industries such as packaging, automotive, and construction.

While the above-mentioned advantages of plastics have made them an integral part of modern life, the same properties have also led to significant environmental problems. It is projected that global synthetic plastic production will reach around 1,800 million tons by 2050 (Gallo et al., 2020) with an estimated 12,000 million metric tons of plastic waste accumulating in the environment by that time (Geyer et al., 2017). Notably, approximately 90% of plastic products are single-use items (NatureCommunications, 2018). However, their disposal and littering have made them a global concern because plastics are highly resistant to degradation and mineralization and persist in the environment for a long time.

The widespread use, unauthorized disposal and low recycling rates of plastics, coupled with their resistance to degradation, has led to an exceptional accumulation of plastic waste in the environment. Under the influence of ultraviolet radiation, soil erosion, biological activity or hydrolysis, large plastic waste fragments into MPs. Terrestrial ecosystems are increasingly impacted by plastic contamination as well as oceans and waterways, but have only recently received much attention (Rillig & Lehmann, 2020).

They can be broadly classified into two main categories: primary MPs and secondary MPs. Primary MPs are plastics manufactured at micro scale sizes for specific applications (Boucher & Friot, 2017). These include microbeads used in personal care products like exfoliates and toothpaste, as well as industrial abrasives and plastic pellets (Cole et al., 2011; Rochman et al., 2015). These particles are directly released into the environment in their micro-sized form, through either domestic wastewater (Browne et al., 2011), industrial processes (Galafassi et al., 2019), or accidental spills during transportation (Karlsson et al., 2018). Secondary MPs originate from the breakdown of larger plastics or synthetic fibers, such as plastic bags or bottles, which gradually fragment into smaller pieces due to physical, chemical, and biological processes (Barnes et al., 2009). Environmental factors such as UV radiation, mechanical abrasion, and microbial activity contribute to their breakdown into MPs (Ter Halle et al., 2016). Depending on the environmental conditions and the chemical properties of the plastic, this process can take place over a period of months to decades (Chamas et al., 2020).

MPs can also be categorized based on their shape, polymer type, and origin. Common shapes include fibers, fragments, films, spheres, and foams. With regard to the shape of plastic particles several studies found fragments and films to be the most prominent shapes found in different German agricultural fields (Piehl et al., 2018; Weber et al., 2022), beside other plastic formations like cable pieces, ties, filaments, pellets and styrofoam. Each of the different MP shapes result from different sources or processes. Nylon, the first synthetic fiber made entirely from petrochemicals, remains widely used in products like hosiery, outerwear, swimwear, and industrial items such as tire cords and parachutes (Bashyal, 2024). However, nowadays polyester (PES) is one of the most widely produced

synthetic fiber globally (TextileExchange, 2024). PES is commonly used in clothing, home furnishings (bed sheets, blankets, and upholstery), and industrial applications such as ropes and conveyor belts. PES fibers are typically made from PET, which is produced through the esterification of terephthalic acid and ethylene glycol, followed by polycondensation to form PET polymer chains. Another widely used fiber is acrylic, made from the polymerization of acrylonitrile monomers to create polyacrylonitrile (PA). PA fibers mimic wool in texture but are more durable and are mainly used in apparel (especially sweaters and socks), home furnishings (carpets, rugs), and outdoor items like tents. Film MPs found in soil are primarily composed of PP and LDPE. LDPE, a branched form of PE with ethylene as its basic monomer and various chain branching, is commonly used in plastic wraps and agricultural films. PP, a more linear polymer made of repeating propylene units, is often found in soil derived from packaging materials, food containers, and agricultural films. Synthetic foams found as MPs in soil are usually composed of PS or polyurethane (PU). PS has a hydrocarbon backbone with phenyl groups attached to every other carbon atom, while PU is produced by the reaction of isocyanates with polyols, leading to either rigid or flexible foams. These foams are commonly used in automotive interiors, furniture, and insulation.

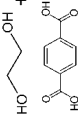
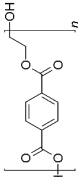
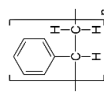
The diverse shapes, compositions and chemical structures of MPs result in a wide range of physical characteristics. Functional groups within the polymer chains contribute to surface roughness (Mincheva & Raquez, 2019), which affects how MPs interact with their surroundings and pollutants. This surface roughness can increase the adsorption of contaminants and promote biofilm formation, thereby enhancing the persistence and toxicity of MPs in the environment (Burrows et al., 2020). The density of MPs influences their buoyancy and distribution. For example, low-density MPs like polyethylene (PE) and polypropylene (PP) tend to float on water, while higher-density MPs such as polyvinyl chloride (PVC) can sink and accumulate in sediments. This buoyancy affects their movement and dispersion in aquatic (Wang et al., 2016) as well as in soil environments (Molazadeh et al., 2023). Additionally, the degree of crystallinity in MPs affects their mechanical strength and resistance towards degradation. Highly crystalline polymers, such as polyethylene terephthalate (PET), are more resistant to environmental degradation and mineralization than their amorphous counterparts, like polymethyl methacrylate (PMMA or Plexiglas) (Andrady, 2011). The type of polymer also impacts how MPs interact with environmental factors like UV radiation, temperature, and microbial activity, which further influence their breakdown and longevity in the environment (Lambert et al., 2017).

Additives are crucial in plastic production to enhance their functionality. Examples of these additives include plasticizers such as phthalates, which are used to increase flexibility in products like cables and flooring made from PET (Björkner, 2000). Stabilizers, including antioxidants and UV stabilizers, protect plastics from environmental degradation (Pfaendner, 2006). Other additives

include flame retardants to reduce flammability (Beach et al., 2010), colorants for aesthetic purposes (Al-Malaika et al., 2017), fillers like calcium carbonate to improve mechanical properties (Coleman, 2017), antistatic agents to minimize static buildup (Hahladakis et al., 2018), and blowing agents like azodicarbonamide to create foam structures (Al-Malaika et al., 2017). Many of these additives can contain significant amounts of heavy metals, such as cadmium (Cd), lead (Pb), and zinc (Zn), which can remain within MPs even after production. This can influence the adsorption of external heavy metals from soil onto MPs (Shirkhorshidi et al., 2023). Additionally, since most additives do not chemically bond with plastic polymers, they are more likely to be released into the environment during use, disposal, and other processes (Teuten et al., 2009; Kwan & Takada, 2019; Gunaalan et al., 2020). As plastics fragment and degrade, the release of these heavy metals and harmful additives into soil ecosystems can pose risks to soil health.

In conclusion, MPs are small plastic particles or fibers that result from the breakdown of larger plastics or are intentionally produced for specific uses. While the durability, lightweight nature, and versatility of plastics have made them indispensable across industries, these same qualities have led to significant environmental challenges. MPs persist in the environment, particularly in soils, where they accumulate and threaten ecosystems. Their diverse shapes, chemical compositions, and physical properties, such as surface roughness, density, and crystallinity, affect their environmental behavior, including their transport, persistence, and toxicity. As MPs continue to accumulate, understanding their environmental behavior is vital for addressing the long-term consequences of plastic pollution and developing strategies to mitigate its impact on ecosystems.

Table 1.1. Most commonly used plastics, their structure, application, densities and appearance as microplastic (MP) in soil.

Name	Monomer name	Monomeric structure ^a	Polymeric structure ^a	Application ^b	Density g cm ⁻³	Appearance as MP in soil
Polyethylene (PE)	Ethene			Plastic bags, bottles, food containers, mulching films	0.89-0.98	Fragments (Yu et al., 2021), film (Feng et al., 2021), pellets (Duan et al., 2020)
Polyethylene terephthalate (PET)	Ethylene terephthalate			Beverage bottles, food packaging, textiles	0.96-1.45	Fibers, fragments (Zubris & Richards, 2005)
Polypropylene (PP)	Propene			Food containers, auto parts, textiles, mulching films	0.83-0.92	Fragments (Yu et al., 2021), film (Feng et al., 2021), fibers (Ding et al., 2021)
Polyvinyl chloride (PVC)	Vinyl chloride			Pipes, window frames, flooring	1.3-1.4	Fragments, film (Briassoulis, 2023)
Polystyrene (PS)	Styrene			Disposable cutlery, packaging, insulation	1.04-1.1	Particles (Kukharchyk & Chernyk, 2022), foam (Helmberger et al., 2022)

^aFigures of monomeric and polymeric structures taken from www.wikipedia.com; ^bplastic applications taken from Heinrich-Böll-Stiftung, B. (2019).

1.2.1.2. Sources and pathways of MPs into the soil

MPs have become a widespread contaminant in terrestrial environments, particularly within soil ecosystems. Understanding the sources and pathways through which MPs enter the soil is critical for assessing their environmental impact and developing strategies for mitigation. Agricultural soils are particularly prone to higher levels of MP pollution compared to other soil ecosystems. Globally, the concentration of MPs in agricultural soils varies significantly, ranging from 0.34 to 1.27×10^7 items kg^{-1} (Shi et al., 2024).

However, difficulties to determine real MP concentrations in soil prevail, due to the complex nature of the soil matrix, variability in plastic types, sizes, and shapes, as well as the limitations of current methodologies. Separating MPs from these soil components is challenging, as MPs can be entangled in organic matter or adhere to mineral surfaces. This makes extraction and identification of MPs from soil labor-intensive, often requiring chemical digestion or density separation techniques, which may still not be completely effective (Zhang & Liu, 2018). Furthermore, MPs in soil can vary greatly in size, shapes (fibers, fragments, films, spheres), and densities. This diversity complicates their detection, as different techniques are required depending on the characteristics of the MPs. For instance, smaller MPs, particularly those in the nanometer range, are harder to detect with standard optical or microscopic techniques, necessitating more advanced instruments like spectroscopic methods (Rillig & Lehmann, 2020). Quantifying MPs in soil samples is also challenging due to variations in sampling methods and laboratory procedures. There is a lack of standardized protocols for sampling soil MPs, leading to variability in reported concentrations across studies. Even the units of measurement (e.g., particles/kg, mg/kg, and items/ m^3) differ widely in studies of MPs sampling. Furthermore, even after extraction, quantifying MPs by weight or particle number does not always account for the different sizes and shapes of the particles, which might affect their environmental behavior and potential impacts.

Despite the indecisiveness regarding real MP concentrations in soil, there are well-defined primary sources and multiple pathways through which they enter soil ecosystems. These pathways are diverse and can be attributed to several main sources (i-v):

i) Agricultural practices

Agricultural activities are among the most significant contributors to MP pollution in soils (Büks & Kaupenjohann, 2020; Tian et al., 2022). The use of plastic products, such as mulching films, greenhouses, and irrigation pipes, introduces large amounts of plastic materials into agricultural fields. Over time, these plastics fragment into MPs due to exposure to sunlight, weathering, and mechanical actions like tilling. Additionally, the application of sewage sludge as fertilizer, which often contains

MPs from household and industrial wastewater, directly deposits MPs into the soil (Tadsuwan & Babel, 2021). It has been shown that soils treated with sewage sludge, which contained approx. 97.66 MP particles g^{-1} , exhibited significantly higher levels of MP contamination compared to soils where sludge was not applied (Büks & Kaupenjohann, 2020; Weber et al., 2022), up to a 256% higher MP load in treated than in non-treated soils (van den Berg et al., 2020). Furthermore, it was shown that agricultural fertilizers, both organic and inorganic, significantly contribute to MP concentrations and accumulations in agricultural soils (Cusworth et al., 2024). In addition, compost, rich in essential nutrients and humus, has long been used as an organic fertilizer in agriculture. However, compost derived from organic waste is increasingly contaminated with plastic due to improper disposal and inadequate waste separation (UBA, 2022). Similarly, municipal biosolids are frequently applied to land as fertilizers and soil conditioners (Roychand et al., 2021), with application rates based on the nitrogen requirements of crops. A study has found that biosolids contain an average of 9.1 ± 1.7 MP particles per gram of dry weight, consisting mainly of fragments (44%) and fibers (39%) (Naderi Beni et al., 2023).

ii) Urban runoff and stormwater

Urban environments contribute to soil contamination through runoff and stormwater, which carry MPs from streets, sidewalks, and other surfaces into nearby soils (Cho et al., 2023). These MPs originate from various sources, including tire wear particles, synthetic fibers from clothing, and degraded plastic litter (Qiu et al., 2020). During rainfall, these particles are washed off into soil adjacent to urban areas, where they accumulate.

iii) Atmospheric deposition

MPs are also increasingly recognized as airborne pollutants. A study detected airborne MP particles, including PS, PE, PP, and poly-silicone compounds, over terrestrial, coastal, and remote marine locations (Trainic et al., 2020). The study highlights the presence of these particles in the atmosphere, suggesting that they are emitted from seawater into the atmosphere through wind-related processes and sea spray aerosol formation. MPs can be transported through the atmosphere over long distances and eventually settle onto land surfaces, contributing to soil contamination. Atmospheric deposition of MPs has been observed in multiple studies, showing varying rates across locations. In an urban area of Central Germany, a study reported an average deposition of 17 ± 14 MP/ m^2 /day for particles sized 11-130 μm (Kernchen et al., 2024), while in Hamburg, the median was 275 MP/ m^2 /day (Klein & Fischer, 2019). Even remote areas like the French Pyrenees have recorded atmospheric MP deposition, with 249 fragments, 73 films, and 44 fibers per m^2 deposited from distances of up to 95 km (Allen et al., 2019). In Paris, fiber fallout ranged from 2 to 355 particles/ m^2 /day, with an average of 110 ± 96

particles/m²/day, indicating high variability (Dris et al., 2016). Concerning airborne MP identification, a study revealed that synthetic textile fibers, rubber tire particles, and household furniture are major contributors to both indoor and outdoor air pollution by MPs and nanoplastics (NPs, particles <1 µm) (Facciola et al., 2021).

iv) Landfills and waste disposal sites:

Improperly managed landfills and waste disposal sites are hotspots for MP contamination in soil (Manikanda Bharath et al., 2023). The abundance of plastics and MPs, such as PE and PP, in landfill soil and landfill refuse is higher compared to aquatic sediments and agricultural soil (Pratiwi et al., 2024). As a result, landfill leachate, the liquid that drains or 'leaches' from a landfill, is not only a serious reservoir of MPs, but can also provide a pathway for MPs into the soil once released (Kabir et al., 2023). Additionally, wind can transport plastic debris from landfills or urban areas to adjacent lands, where it breaks down further into MPs (Dube & Okuthe, 2023).

v) Industrial activities:

Industrial processes that involve the production, handling, or disposal of plastic materials are another significant source of MPs in soils. Factories that manufacture or process plastics may release MP particles directly into the surrounding environment through spills, emissions, or improper waste management. In particular, industries located near agricultural or natural areas can contribute to the direct contamination of soils with MPs. Industries particularly prone to releasing MPs or plastic debris into the environment include textiles (due to synthetic fibers) (Zhou et al., 2020a; Chan et al., 2021), automotive (via tire wear), plastics manufacturing (e.g., production and processing of plastic materials) (Lechner & Ramler, 2015), and waste management (especially when handling plastic waste or through leaching from landfills) (Ponti et al., 2022). A study examining industrial effluents in Portugal found that PET, primarily associated with synthetic fibers, was the most prevalent MP type. However, it was also suggested that, contrary to expectations, the pharmaceutical and paint industries were responsible for higher MP releases than the textile sector (Magalhães et al., 2022). Additionally, the cleaning and maintenance of synthetic surfaces and consumer products like clothing, paints, and coatings also contribute to plastic pollution (van Wezel et al., 2016).

In conclusion, the sources and pathways of MPs into soil are diverse and complex, involving a range of human activities and environmental processes.

1.2.2. Existing knowledge on microplastic-soil interactions

The interactions between MPs and soil are complex, multifaceted, and have received considerable attention in recent years. This chapter aims to provide a comprehensive overview of the current state

of research in this field. It synthesizes findings from studies that explore the physical, chemical, and biological processes influencing MP behavior in soils, as well as how MPs can alter soil physical processes. Key focus areas include the mechanisms by which MPs are transported and accumulated in the soil matrix and their potential effects on soil health and function, emphasizing the need for a more nuanced understanding of MP-soil dynamics in the context of global environmental change.

The natural processes in the soil environment – physical, chemical, and biological – play a significant role in influencing the behavior, movement, and transport of MPs. These processes interact with several properties of MPs that dictate their mobility, including size, hydrophobicity, charge, density, and shape (e.g., fiber, bead, film, foam). Among these, size (Rillig et al., 2017b; Zhang & Liu, 2018) and hydrophobicity (Wan & Wilson, 1994; Yu et al., 2019) have been extensively studied and have provided direct evidence for their influence on MP transport. Understanding these mechanisms is crucial for predicting their behavior in the environment. Soil systems support both horizontal and vertical transport, which are governed by a combination of natural processes and the inherent properties of MPs.

Horizontal movement of MPs across soil surfaces is often carried out by heavy rainfall or irrigation events, where water flows carry MPs along with soil particles (Cho et al., 2023). Also, in areas prone to erosion, MPs can be detached from the soil surface and transported downslope. The detachment of MPs is often facilitated by the breakdown of soil aggregates during intense rainfall events, leading to the incorporation of MPs into sediment that is subsequently carried away by surface runoff (Laermanns et al., 2021). In addition, soil biota (mesofauna, fungi), harvesting, and plowing on agricultural fields contribute to particles being distributed horizontally (Rillig et al., 2017a).

Vertical transport, or leaching, refers to the downward movement of MPs through soil layers. This process is influenced by soil structure, water infiltration rates, and the characteristics of the MPs themselves, such as size, shape, and density (Gao et al., 2021). Soils with high porosity, such as sandy soils, facilitate faster water infiltration and thus greater potential for vertical transport of MPs (Gao et al., 2021). Conversely, clayey soils with smaller pore spaces may impede the downward movement of MPs, leading to their accumulation near the surface. In addition, it was shown that plant roots can have significant influence on PVC particle movement, enabling MP particles to penetrate deeper into the soil through fissures created by root elongation (Tumwet et al., 2024). Furthermore, MPs can adsorb to organic matter in the soil (Wang et al., 2019), affecting their mobility (Zhou et al., 2020b). The presence of high levels of organic matter can enhance the retention of MPs, particularly those made from polymers with hydrophobic properties. This adsorption can limit the transport of MPs, leading to their accumulation in organic-rich surface layers (Zhang & Liu, 2018). Moreover, the surface

charge of MPs can affect their adsorption to soil particles and therefore their mobility in soil (Rillig et al., 2019).

In terms of MP characteristics, smaller MPs are more likely to migrate vertically through soil profiles due to their small size and have a lower likelihood of being trapped in pore spaces. In contrast, larger MPs may be restricted in their vertical movement, especially in soils with fine textures. The density of MPs relative to soil particles and soil water influences their vertical distribution in the soil profile as well (Zhu et al., 2019a). Moreover, MPs with densities below 1 g cm^{-3} may be transported more readily in saturated soils due to their buoyant nature, compared to dry soils, where they may become trapped within aggregates or biofilms (Molazadeh et al., 2023). In addition, MPs may move vertically through preferential flow paths, such as cracks, root channels, or wormholes (Rillig et al., 2017a; Yu et al., 2019), whereby the shape of MP particles (e.g., fibers, beads, films, foams) significantly affects their movement through soil pores and their interaction with soil biota (Boots et al., 2019). These pathways allow MPs to bypass the bulk soil matrix, facilitating deeper infiltration into the soil profile.

Conversely, the effects of MPs on soil processes are influenced by soil properties such as organic matter content, biological activity, texture, and soil porosity. It is therefore necessary to consider both biotic and abiotic factors when assessing the impact of MPs on soil ecosystems. A study by Lehmann et al. (2019), examining how MP fibers affected soil aggregation and how it was influenced by abiotic and biotic factors, concluded that greater soil aggregation activity would lead to increased likelihood for MPs to interact with this biological process, potentially due to MP fibers becoming integrated into aggregates and subsequently destabilizing these structures through unknown mechanisms. Moreover, it was shown that the destabilizing effect of MPs on soil structure is shape- and polymer-dependent (Zhao et al., 2021). Such alterations in soil structure can affect soil porosity and water dynamics, which prompted researchers to explore the complex interactions between MPs and soil hydrology more closely.

The first relevant research on the effects of MPs on soil physical and hydrological properties was published by de Souza Machado et al. (2018). Their study observed alterations in key soil hydrological characteristics and the biophysical environment due to the presence of MPs (PES microfibers, PA microfibers, PE particles). They found that MPs altered important soil properties, including water-holding capacity (WHC), which is the maximum amount of water that can be held within the soil pore matrix against gravity, the soil bulk density and the functional relationship between microbial activity and water stable aggregates (de Souza Machado et al., 2018). These changes in the soil's biophysical properties directly affect its hydraulic characteristics. For example, particularly the addition of microfibers led to a modest increase in the soil's ability to retain water, which could potentially affect

soil moisture levels and evapotranspiration. In addition, PE particles, being lighter than the average soil components, caused a reduction in soil bulk density. While lower bulk density is often associated with better soil quality, it remained unclear in the study by de Souza Machado et al. (2018) whether the decrease in soil density would positively impact root growth, as it may not necessarily represent an increase in soil porosity. Moreover, a decrease in water-stable aggregates was observed, which could potentially lead to changes in soil structure, affecting water infiltration and retention. Also, soil aggregate size, porosity, and aeration were found to be increased. These changes can significantly influence water movement and retention in the soil. Furthermore, the functional relationship between microbial activity and water-stable aggregates was altered due to MP addition in soil, which is crucial for maintaining soil structure and function. The alterations in soil structure, porosity, and WHC can have cascading effects on various hydrological processes, including infiltration rates, water retention, and potentially even groundwater recharge.

In the following years, research on MP effects on soil hydrological properties was partly in line with the pioneering work of de Souza Machado et al. (2018) and partly not. While, for example, increased HDPE content in soil was found to generally lead to increased soil aggregate stability (Boots et al., 2019), in line with de Souza Machado et al. (2018), a study by Zhang et al. (2019) claimed that no detectable changes in soil bulk density could be observed (Zhang et al., 2019). In another study, PP particles in soil were found to decrease soil bulk density, whereas PP and PVC addition were found to have negligible effects on other soil physical properties (Li et al., 2023b). Moreover, positive effects of these two MP types on soil quality, such as increasing soil enzyme activity, were reported.

Given the divergent findings, it appears that the impact of MPs on soil physical properties might be more complex than initially thought, likely influenced by various factors such as MP type, concentration, soil type, and texture. As a result, recent studies have increasingly focused on testing different soil types and varying MP concentrations to better understand their effects (Guo et al., 2022; Ingraffia et al., 2022a; Shafea et al., 2023; Wang et al., 2023; Xie et al., 2023). For example, de Souza Machado et al. (2018) and Zhang et al. (2019) reported differing findings regarding soil bulk density, despite both testing PES microfibers. The studies used different soils – a loamy sandy Albic Luvisol (de Souza Machado et al., 2018) and a clay loamy Nitisol (Zhang et al., 2019) – suggesting that soil texture could be a crucial factor in evaluating the influence of MPs on soil properties. This assumption is further supported by Guo et al. (2022), who tested the effects of PP in various particle sizes and concentrations in three different soil textures (loamy, sandy, and clay) on soil hydraulic properties, reporting texture- and MP-size-dependent results. When assessing the saturated soil hydraulic conductivity (K_{sat}), which is the rate of vertical water flow through a water-saturated soil pore matrix, they found that increasing MP particle sizes led to increased K_{sat} values in loamy soil, while in clayey and sandy soils, K_{sat}

decreased. Additionally, MP particle size positively affected the volumetric water content at saturation (θ_s) and the residual water content (θ_r) in sandy soil, whereas in clayey and loamy soils, θ_r was negatively influenced by increased particle size.

These findings have sparked a debate as to whether the effects of MPs on soil hydraulic properties are due to widening or blocking of soil pores (Guo et al., 2022; Wang et al., 2023; Xie et al., 2023). It has been suggested that not only the blocking of macropores (Guo et al., 2022), the decrease in pore size and macropores (Xie et al., 2023), or the increase in macroporosity (Ingraffia et al., 2022a) play a role in MP effects on soil water flow, but also the connectivity of pores (Xie et al., 2023). The tortuosity value (λ), which represents the actual distance water travels through the soil structure, is directly influenced by pore connectivity. When MPs are incorporated into the soil, this leads to reduced pore sizes and fewer macropores, and a decrease in λ . Soil flow channels are significantly impacted by pore sizes and distribution, especially continuous macropores (Xie et al., 2023). A reduction in pore sizes and macropores contributes to a lower λ , indicating higher flow channel complexity. The addition of fine-sized MPs (150 and 50 μm) results in smaller pore sizes and more isolated pores, thereby decreasing pore connectivity. This reduction in pore sizes and connectivity contributes to higher tortuosity in the soils (Xie et al., 2023).

In summary, several open questions remain about the behavior and effects of MPs in soils. Uncertainty surrounds how natural processes and MP properties, such as size and shape, influence their transport through soil, as well as the roles of plant roots, soil organisms, and flow paths in this movement. Additionally, the destabilizing impact of MPs on soil structure, particularly soil aggregation and hydrology, is not fully understood. Conflicting studies suggest MPs may either increase or decrease soil bulk density, porosity, and water infiltration, depending on soil type and texture. Finally, how MPs affect soil pore structures, including pore connectivity and water flow, requires further investigation.

1.2.3. Emerging insights into microplastic-plant interactions

Research on the effects of MPs on soil and plants has significantly increased over the past 5-10 years, with many reviews citing literature from 2015 onward (Rillig et al., 2019; Ullah et al., 2021; Li et al., 2022b; Qiu et al., 2024). Early studies primarily focused on the presence and accumulation of MPs in soils before expanding to explore their effects on plants (Rocha-Santos & Duarte, 2015; Rillig et al., 2017a). More recent research reveals a complex interplay between MPs, soil, and plants (Mészáros et al., 2023). With emerging contradictory results – some studies suggesting MPs have detrimental effects on plant growth (Huang et al., 2022; Roy et al., 2022; Chen et al., 2023; Ikhajiagbe et al., 2023; Sun et al., 2023; Zhang et al., 2023), while others find neutral or even positive outcomes (Biba et al., 2023; Khan et al., 2023; Li et al., 2023a; Liu et al., 2023b) – studies in the field of MP-plant interactions

generally indicate that the effects of MPs on plants depend on several factors such as MP dose, MP type, plant species, and soil type. In fact, to fully understand the impact of MPs on plant growth, it is essential to consider the broader, more complex interactions occurring within the plant-soil system.

It is assumed that the rhizosphere is a hotspot for plant-soil interactions, a narrow region of soil surrounding the roots, by which plants engage in continuous exchange with their soil environment. It is characterized by a highly dynamic and biologically active environment, as plants release a variety of organic compounds, known as root exudates, into the rhizosphere. These exudates, which include sugars, amino acids, organic acids, and secondary metabolites, serve multiple purposes: they attract beneficial microorganisms, inhibit pathogens, and influence nutrient availability (Mendes et al., 2013). Some root exudates can even alter the pH of the surrounding soil, enhancing nutrient solubility and uptake by the plant (Richardson et al., 2009). The rhizosphere hosts a dense and diverse population of microorganisms, including bacteria, fungi, protozoa, and archaea, which play crucial roles in nutrient cycling, organic matter decomposition, and disease suppression (Ling et al., 2022). Some plants form symbiotic relationships with specific microbes, such as mycorrhizal fungi, which extend the root system's reach for nutrients like phosphorus (P) (Linderman, 1991), and nitrogen-fixing bacteria, which convert atmospheric nitrogen into plant-available forms like ammonia (NH_4^+) (Iggehon & Babalola, 2018). Additionally, root growth and rhizosphere processes contribute to the formation and stabilization of soil aggregates, improving soil structure and water retention (Bengough, 2012). However, rhizosphere interactions can be altered by environmental stresses, such as pollutants, drought, and nutrient deficiencies, leading to stress responses that directly affect plant performance and development (George et al., 2024).

So far, as outlined in Fig. 1.1, MPs have been shown to impact various aspects of plant performance, including seed germination (Sforzini et al., 2016; Bosker et al., 2019), vegetative and reproductive growth (Qi et al., 2018; Miao et al., 2023), root and leaf characteristics (de Souza Machado et al., 2019; Liu et al., 2022b), crop yield (Hegan et al., 2015; Chen et al., 2022a), as well as ecotoxicity and genotoxicity in plants (Jiang et al., 2019). Several additional implications of MP addition to soil, previously examined through plant experiments, are summarized in Table 1.2.

Table 1.2. Selected examples (published 2020-2023) of the reported effects of different microplastics (MPs) on plant growth and development. Adapted from Jia et al. (2023).

MP type and concentration	Plant species	Effects on plant growth and development	Reference
PC (0.1-10% w/w)	Garden cress	<ul style="list-style-type: none"> • MPs reduced seed germination • MPs decreased root and shoot length 	(Pflugmacher et al., 2020)
PE and PVC (1%, 5%, 10%, and 20% w/w)	Wheat (<i>Triticum aestivum</i> L.)	<ul style="list-style-type: none"> • MPs had a negative, dose-dependent impact on plant growth • MPs altered C flow through the plant-soil system and C-dependent soil functions. • MPs increased microbial biomass and changed the structure of the community • MPs stimulated microbial CUE but had little effect on exoenzyme activity. 	(Zang et al., 2020)
PS (1 mg/kg)	Soybean	<ul style="list-style-type: none"> • MPs induced oxidative stress • MPs altered gene expression 	(Xu et al., 2021a)
HDPE (3% w/w)	Rye grass (<i>Lolium multiflorum</i>)	<ul style="list-style-type: none"> • Regardless of the ages of HDPE particles, HDPE particles reduced seed germination and root-shoot length 	(Esterhuizen et al., 2022)
PET and PEF (0.5-2% w/w)	Lettuce	<ul style="list-style-type: none"> • MPs impaired lettuce growth • MPs enhanced ROS production • MPs reduced soluble sugar and nitrogen accumulation 	(Zhang et al., 2022b)
PP, HDPE, PS, PVC, PET, PUR (20% w/w)	Lotus (<i>Nelumbo nucifera</i>)	<ul style="list-style-type: none"> • Reduction of germination rate 	(Esterhuizen & Kim, 2022)
PE (0.001%, 0.01%, 0.1% w/w)	<i>Brassica napus</i> L.	<ul style="list-style-type: none"> • Reduction of the total chlorophyll content • MPs altered sugar metabolism • MPs enhanced bioaccumulation of lead 	(Jia et al., 2022)
Shoe sole fragments (0.1%, 1% w/w)	Mung bean	<ul style="list-style-type: none"> • MP fragments and leachates from shoe soles affected plant growth 	(Lee et al., 2022b)
PE (0.1% w/w)	Maize	<ul style="list-style-type: none"> • MPs reduced maize growth • MPs altered soil bacterial community • MPs affected gene expression of antioxidants 	(Fajardo et al., 2022)

PES (0.5% w/w)	Maize	<ul style="list-style-type: none"> • MPs reduced crop growth • MPs reduced N uptake and increased N loss via leaching • MPs reduced soil bulk density and microporosity 	(Ingraffia et al., 2022b)
LDPE (100, 1000 and 10000 kg ha ⁻¹ corresponding to 0.1%, 1% and 10% w/w)	Spring wheat (<i>Triticum aestivum</i> L., cv. Minaret)	<ul style="list-style-type: none"> • Pure MP loading had no effect on the soil microbial community • Wheat crop growth and yield were unaffected by MP loading compared to control • Nitrogen cycling (gene abundance and N₂O flux) showed little change • No MP dose-dependent impact on soil biological health 	(Brown et al., 2022)
PMFs (0.1% and 0.2% w/w)	Lettuce	<ul style="list-style-type: none"> • MPs reduced shoot lengths • MPs altered photosynthesis and chlorophyll contents • MPs altered carbohydrate and nitrogen metabolic pathways 	(Zeb et al., 2022)
PE and PVC (1% and 5% w/w)	Wheat (<i>Triticum aestivum</i> L.)	<ul style="list-style-type: none"> • Different root resource-acquisition strategies disturbed microbial nutrient turnover • Higher doses of PE (5% w/w) enabled roots to take up additional N • PVC did not lead to microbial SOM decomposition for N and P acquisition • PVC forced wheat to efficiently obtain available nutrients in the narrow root zone • Higher root and shoot biomass 	(Liu et al., 2022b)
PP and rubber crumb (1% w/w)	Peanut	<ul style="list-style-type: none"> • MPs inhibited vegetative growth and nitrogen content • MPs decreased root viability and number of xylem vessels • MPs altered soil microbial community and nitrogen cycling 	(Liu et al., 2023c)

While there is a general consensus that synthetic polymers do not belong in the environment and our soils, the question of whether MPs truly act as stressors for plants in all soils and environmental

conditions remains open and requires more comprehensive research. Moreover, the current findings are contradictory. For example, two studies on wheat (*Triticum aestivum* L.) development, both involving contamination of the soil with PE and PVC at 125 μm and 5% and 10% (w/w), yielded different conclusions. One study reported a dose-dependent negative impact on plant growth, highlighting that MPs negatively affected various carbon-dependent soil functions, such as β -glucosidase activity (Zang et al., 2020). In contrast, the other study found increased root and shoot biomasses, attributing this to altered root resource-acquisition strategies as microbial nutrient turnover was disrupted (Liu et al., 2022b). Despite working with the same plant species, MP types, concentrations, and sizes, the studies differed in the soil types used; Zang et al. (2020) used an Eutric Cambisol, while Liu et al. (2022) worked with an Aquatic Brown soil. This suggests that the effects of MPs on plant growth and microbial turnover processes may also depend on the surrounding soil environment.

An additional explanation for the observed effects of MPs on plant growth is their direct impact on soil microbial activity, which in turn affects plant growth and nutrient uptake. For example, MPs can physically disrupt soil structure, potentially limiting the availability of oxygen and nutrients essential for microbial growth and function (de Souza Machado et al., 2019). Additionally, MPs may leach toxic chemicals, such as plasticizers or other additives, into the soil, which can be harmful to soil microorganisms (Zhu et al., 2022). MPs can also alter the composition of microbial communities, favoring some microbes over others, which may reduce microbial diversity and impair soil ecosystem function (Rillig et al., 2019). More specifically, LDPE MPs, for example, altered the microbial community turnover in soil within 90 days at a concentration of 200 fragments/100 g soil (Wang et al., 2020c). This can have far-reaching effects on plant growth, because soil microbes play crucial roles in nutrient cycling, organic matter decomposition, and the formation of symbiotic relationships with plants. As an example, the diversity of nitrogen-fixing bacteria, providing nitrogen for plants, was influenced through the contamination with PE MPs at 1% w/w in soil, which potentially altered soil NH_4^+ levels and pH and thereby affected root growth and activity (Fei et al., 2020). Interestingly, one study claimed that the rough surface and the sharp edges of MPs might have caused physical damage to microorganisms, as well as the reduction of root length in plants exposed to these particles (Rozman et al., 2021).

What has to be seen critically though, is the report of negative effects while making use of unrealistically high MP concentrations, such as 28% w/w (Liu et al., 2017), 20% w/w (Zang et al., 2020; Esterhuizen et al., 2022) or 10% w/w (Pflugmacher et al., 2020; Wang et al., 2020b; Zang et al., 2020; Zhang et al., 2024) in soil. While this approach might seem scientifically unsound and deliberately aimed at highlighting adverse effects on plant performance, it can be justified to a certain extent. Testing very high MP loads allows researchers to identify threshold levels for specific effects, observe amplified interactions between MPs, soil, and plants that might be too subtle at lower concentrations,

and predict long-term ecological implications if MPs continue to accumulate in soils, thereby offering an anticipatory perspective.

On the contrary, not all studies claim negative effects of MPs on plant performance. For instance, it was found that certain types of MPs can promote shoot and root mass growth, depending on their shape (e.g. films, fragments and foams) and polymer type, suggesting that the physical characteristics of MPs might influence plant growth (Lozano & Rillig, 2022). Similarly, in a scientific review positive effects of MPs on the soil carbon pool and plant performance under specific conditions were reported, highlighting that MPs might interact with soil processes in ways that could benefit plants (Chen et al., 2024). Further, a study found that although MPs could be vertically transported by wheat roots, they did not observe significant negative effects on plant growth, pointing to a potential tolerance of certain plants to MPs in soil (Tumwet et al., 2024). There is a call to avoid generalizing the effects of MPs on plant growth, emphasizing the need to consider factors such as the type of MP, its concentration, the plant species involved, and the specific environmental context (Mészáros et al., 2023). These results indicate that the influence of MPs is not uniformly detrimental and can vary depending on specific circumstances.

In conclusion, research on the effects of MPs on plant growth has yielded a range of findings, e.g., on seed germination, vegetative and reproductive growth, crop yields, microbial communities, and root lengths, underscoring the need for a more nuanced understanding of their impact. While some studies highlight negative effects, including reduced biomass, altered microbial activity, and impaired nutrient uptake, others suggest that MPs may have neutral or even beneficial effects under certain conditions. The variation in results suggests that the impact of MPs is context-dependent, influenced by factors such as the type and concentration of MPs, plant species, soil characteristics, and potential co-contaminants. As research progresses, it is crucial to avoid overgeneralization and instead take a holistic approach that considers the complex interactions between MPs, plants, soil, and the surrounding environment.

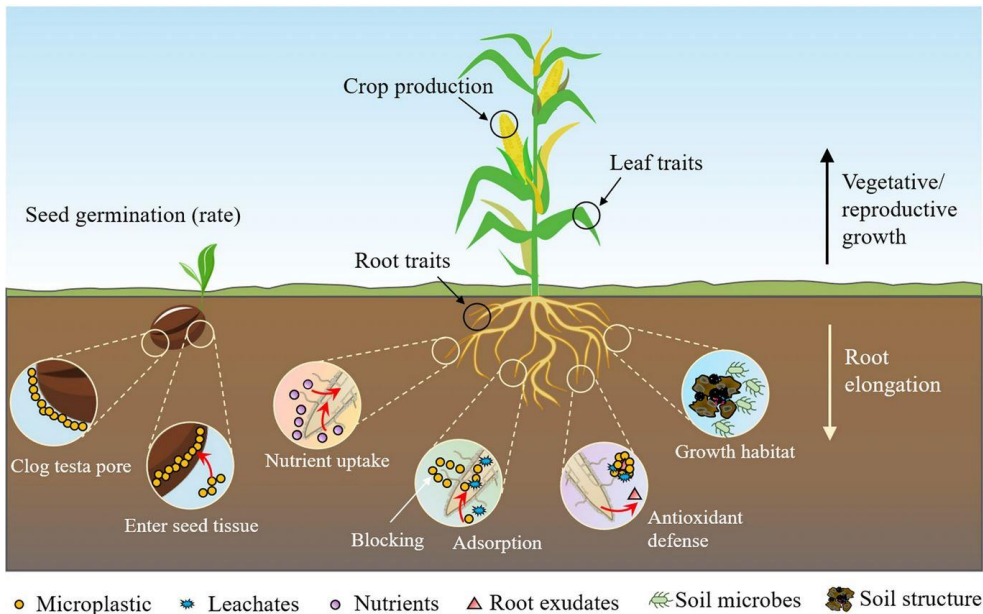


Fig. 1.1. Concept graphic about the effects of microplastics (MPs) on plant performance. MPs can impede seed germination by obstructing testa pores and infiltrating seed tissues. Furthermore, they can influence aboveground plants' vegetative or reproductive growth. Additionally, MPs have the potential to alter root traits and rhizosphere, thereby affecting root elongation. Taken from Chen et al. (2024), with friendly permission by Prof. Dr. Ke Sun, School of Environment, Beijing Normal University, Beijing.

1.2.4. Current understanding of microplastic degradation and mineralization in soil

Degradation refers to the process by which MPs are broken down through physical, chemical, and biological mechanisms into simpler constituents. This process may result in altered polymer structures or in end products such as organic acids or original monomers that can be metabolized by organisms. In contrast, mineralization describes the complete or breakdown of polymer chain sections into inorganic end products, specifically carbon dioxide (CO_2) and water (H_2O). Degradation and mineralization imply that the plastic no longer retains its polymeric structure and is no longer persistent in the environment. Fragmentation, on the other hand, is a physical process where plastics break into progressively smaller particles without any significant alteration of the chemical structure of the material. Current studies sometimes conflate fragmentation with degradation, and there is a need for precise methods to differentiate between the two processes. Degradation and mineralization occur through complex interactions between the soil environment and the plastic polymers, involving factors such as temperature, moisture, ultraviolet (UV) exposure, and microbial activity. Over time, these environmental factors lead to weathering of MPs – a process in which plastics undergo physical

and chemical changes that alter their structure and properties. Weathered MPs often have rougher surfaces, cracks, or fragmentation, which can increase their surface area and make them more prone to further degradation or mineralization. This weathering also changes the chemical composition of the plastic, potentially increasing the release of additives or absorbed contaminants into the soil. As a result, weathered MPs may interact differently with soil and organisms compared to their pristine counterparts, often making them more reactive and potentially more toxic. However, unlike in aquatic ecosystems, where MP degradation has been studied extensively, the mechanisms in soil remain less understood and are often much slower due to the comparably lower availability of oxygen, light, and heat.

At the molecular level, MP degradation involves the cleavage of the long polymer chains that constitute the structural backbone of plastic materials, whereas mineralization refers to the complete metabolic conversion or chemical breakdown of these organic degradation products into inorganic end products. Chemical processes such as photo-oxidation or hydrolysis, as well as biological processes due to microbial-induced reactions break the covalent bonds within the polymer, leading to chain-scissions and the formation of oligomers, dimers and monomers. In the environment, photodegradation is recognized as the most important process that initiates plastic degradation. UV light (especially high energy UV irradiation UV-B, 290-315 nm) provides energy exceeding that of the polymer's chemical bonds, causing them to break and to form free radicals (highly reactive molecules). These free radicals further react with oxygen, leading to oxidative degradation or mineralization. As a result, UV light often accelerates plastic degradation and mineralization in the presence of air. Furthermore, biodegradable plastics or polyesters, like PET, can undergo hydrolysis, which is the reaction of water with the ester bonds within the plastic, breaking the polymer chains into smaller units like alcohols and acids. Hydrolysis is typically slower in non-biodegradable plastics but can still occur in the presence of water in combination with high temperatures and/or catalysts over long periods (years to decades) (Abedsoltan, 2023). Additionally, microorganisms (like bacteria and fungi) present in the soil can play a role in MP degradation and mineralization by secreting enzymes that target certain types of plastics, contributing to biochemical degradation, though this process is often incomplete.

Enzymatic biodegradation induced by fungi might be the most promising biological method for breaking down MPs, particularly involving enzymes such as laccases, peroxidases, and cutinases. These enzymes, produced by certain fungi, have the ability to oxidize and break down the long polymer chains of MPs (Nasrabadi et al., 2023). Among these enzymes, laccase is particularly notable. The original purpose of this enzyme, which is secreted e.g. by the fungus *Pleurotus ostreatus*, is to break down lignin in wood, but was also found to catalyze the oxidation of plastic polymers, breaking down the chemical bonds of various plastic types, including PET (Tiso et al., 2021) and PE (Fujisawa et al., 2001).

Additionally, cutinases are capable of degrading polyesters like PET (Webb et al., 2012), and peroxidases further enhance the breakdown of complex polymers (Álvarez-Barragán et al., 2016; Gómez-Méndez et al., 2018). Fungi like *Aspergillus*, *Penicillium*, and white-rot fungi (e.g., *Phanerochaete chrysosporium*) have shown great potential in initiating these enzymatic reactions (Matúš et al., 2023), making them promising candidates for MP degradation and mineralization in soil.

Peroxidase enzymes, such as lignin peroxidase and manganese peroxidase, play a synergistic role in generating highly reactive oxygen species (ROS) through a process known as the Fenton reaction. In this reaction, hydroxyl radicals ($\text{OH}^\bullet$) are formed when hydrogen peroxide (H_2O_2) reacts with iron-II-ions (Fe^{2+}). These ROS are highly reactive and non-specific, initiating chain scissions by abstracting hydrogen atoms from hydrocarbon chains. While Fenton reactions have already been shown to effectively degrade soil organic carbon (Merino et al., 2020; Merino et al., 2021), researchers are now investigating their potential in degrading and mineralizing MPs as well (Laurent et al., 2012; Tagg et al., 2017; Hu et al., 2022; Ortiz et al., 2022; Xing et al., 2023). However, their effectiveness has only been demonstrated when MPs had been pre-treated with thermal or UV-irradiation.

The oxidation process generally begins at the surface of plastic particles and progresses towards the core, resulting in both physical and chemical alterations of the material. Fenton reactions can reduce the size of MPs through oxidation, eventually leading to their mineralization (Ortiz et al., 2022). MPs containing aromatic rings tend to exhibit higher oxidation yields compared to alkane-based MPs (Yang et al., 2022) because $\text{OH}^\bullet$ preferentially targets aromatic rings and double bonds rather than alkanes (Pignatello et al., 2006). This is consistent with findings by Lang et al. (2020), who observed an increase in oxygenated groups on the surface of PS MPs after Fenton treatment.

While the idea of using fungi to degrade plastics dates back to a 1968 study (Darby Richard & Kaplan Arthur, 1968), the growing concern over MP pollution in soil has driven more researchers in the last decade to explore the potential of various fungi in plastic biodegradation and mineralization. Recent studies have shown that different fungal species – often isolated from landfills – exhibit promising potential in biodegrading and mineralizing plastics such as LDPE (Zahra et al., 2010), PU (Khan et al., 2017), and poly- β -hydroxybutyric acid (PHB) (Gonda et al., 2000). However, it remains an ongoing area of research to determine whether fungi naturally present in soil can degrade or mineralize MPs, either by intentionally using them as a carbon source or through unintentional side reactions. What is already clear is that fungal hyphae naturally present in the soil readily attach to the surfaces of various MP types, as we could demonstrate in a two-year incubation experiment where MPs were exposed to outdoor soil conditions (Fig. 1.2).

In conclusion, research indicates that fungi and enzymes like laccases and cutinases show promise in degrading MPs. Additionally, Fenton reactions have potential for MP degradation and

mineralization but require specific pre-treatments. Although fungal hyphae have been observed attaching to MPs in natural soil conditions, further studies are needed to determine their effectiveness in MP degradation and the long-term impacts of weathered MPs on soil health and ecosystems.

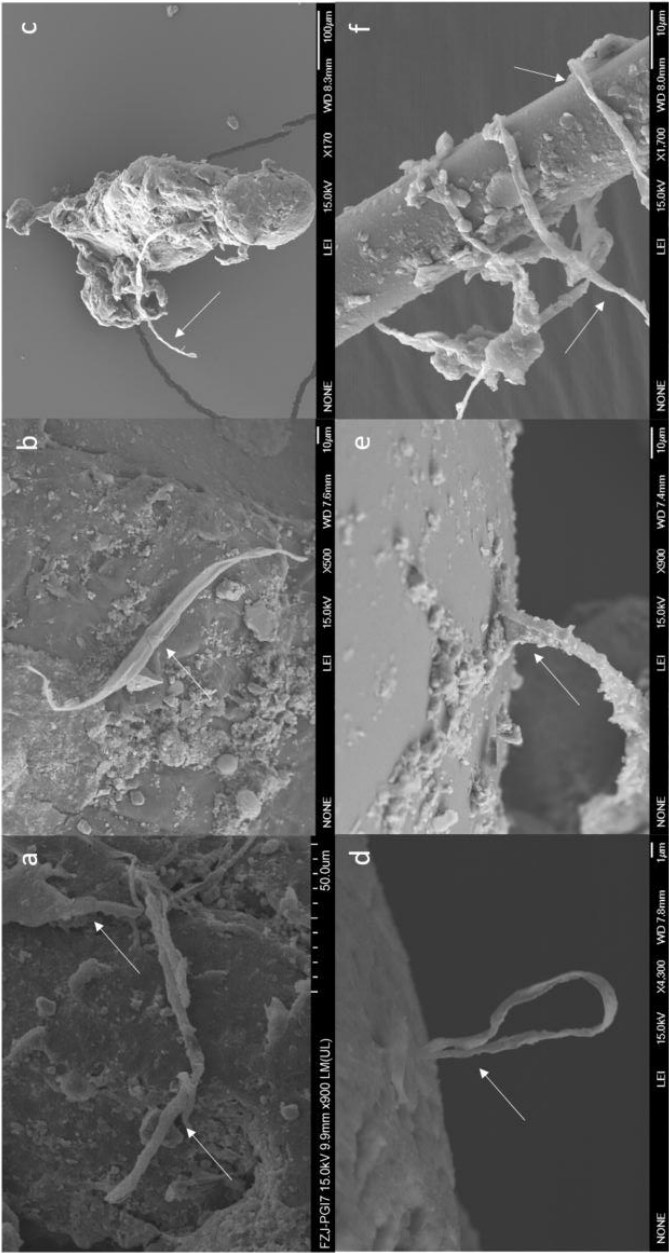


Fig. 1.2. Scanning electron microscopy (SEM) images of fungal hyphae attached to different microplastic (MP) types, which had been incubated in different soils for different time periods. A) Polypropylene (PP) particle incubated for 24 months in Albic Luvisol (AL) soil type, b) PP particle incubated for 12 months in AL soil type, c) low-density polyethylene (LDPE) particle incubated for 6 months in AL soil type, d) LDPE particle incubated for 3 months in Luvisol soil type, e) polystyrene (PS) particle incubated for 12 months in AL soil type, f) polyester microfiber incubated for 3 months in AL soil type. White arrows are pointing towards fungal hyphae.

1.2.5. Research gaps and challenges

Due to the seemingly unstoppable growth of plastic production and the lack of effective waste management measures (Sharma et al., 2020), the current pressure of MP contamination in soil is expected to increase for many years to come. To better understand the ecosystem effects and risks of MPs in the soil environment, there are critical unknowns that need to be addressed in future work. Therefore, this section provides insights into challenges associated with studying MPs in soil, including the limited and often inconclusive research on their interactions with plants and microorganisms, the variability among soil types, and the complex relationships between MPs, soil properties, and plant systems.

First of all, due to the complexity of soil ecosystems, the variability of MPs, the lack of long-term studies and challenges in detection, the real ecological and environmental risks of MPs are still being discussed (Li et al., 2023c; Chen et al., 2024). Studies examining the effects of MPs on plant-soil interactions often use MP concentrations far higher than those typically found in real-world environments (Liu et al., 2017; Pflugmacher et al., 2020; Wang et al., 2020b; Esterhuizen et al., 2022; Zhang et al., 2024). While these high concentrations can reveal potential disruptions in soil structure, nutrient cycling, water retention, and plant growth more quickly and visibly, they may also lead to overestimations of MP pollution's severity in most agricultural settings. This can distort scientific assessments and public perceptions of the environmental risks. However, such high-concentration studies can still be valuable, as they allow researchers to detect possible effects within a manageable timeframe.

Apart from this, one major research gap in the literature is the tendency to overlook the broader ecological implications of MPs on soil properties and plant growth. While recent studies have highlighted the importance of soil characteristics as a decisive factor (Guo et al., 2022; Krehl et al., 2022; Wang et al., 2023), it remains largely ignored in much of the research on MPs in soil. Much of the previous research has focused narrowly on MP accumulation and its direct toxicity (Zhang & Liu, 2018; Li et al., 2019; Xu et al., 2020), rather than on the potential of MPs to alter essential soil functions, such as soil structure, water dynamics, and plant-available water. Soil is functioning as a complex and dynamic system, characterized by a diverse array of physicochemical properties that vary significantly across different soil types. These properties – including pH, organic matter content, texture, and nutrient availability – play a critical role in determining how soils interact with MPs, and vice versa. It was already shown that MPs can alter these physicochemical properties (Zhang et al., 2019; Yu & Flury, 2022; Wang et al., 2023; Xie et al., 2023). For instance, the introduction of MPs may affect soil aggregation, water retention, and nutrient cycling. However, the effects of MPs cannot be generalized

across all soil types due to the inherent variability in soil composition and structure. Investigating how MPs can modify soil characteristics and impact plant growth through changes in water availability and retention should be of interest in future research.

There is also a lack of attention to how different types of MPs uniquely interact with soil and plants. Although some studies have compared multiple MP types (Lozano et al., 2021b; Esterhuizen & Kim, 2022), most research on MP effects on soil processes and plant growth involves only one or, at most, two types of polymers (Xu et al., 2021a; Esterhuizen et al., 2022; Fajardo et al., 2022; Liu et al., 2022b). The risk that arises when failing to consider the diversity in MP composition, e.g., polymer type, shape, and surface characteristics, is to overgeneralize the observed effects of one MP type to all MPs present in the environment. In fact, it was shown that especially the difference between fibrous and particulate MPs on plant growth is significant (de Souza Machado et al., 2018; de Souza Machado et al., 2019; Lehmann et al., 2021; Lozano et al., 2021a; Yu & Flury, 2022). However, observations on the direct and indirect effects of various MP types on plant growth and soil properties using a multi-MP-type approach are scarce. This would allow for a more accurate assessment of how various MP forms might differentially impact soil health, nutrient availability, and plant development. It is important to use varied and realistic MP types and soil conditions in experimental setups, moving beyond single-type models to capture the diversity of real-world MP contamination.

The relationship between MPs, soil texture, and soil structure is another underexplored area in the existing literature. Soil texture can significantly influence how MPs interact with soil particles, affect microbial activity, and alter water dynamics, yet many studies treat soil as a uniform medium without accounting for textural variation (Zang et al., 2020; Liu et al., 2022b). Exploring the unique and texture-dependent effects of MPs on soil structure, nutrient retention, and plant growth would highlight the role of soil texture and consider the inherent heterogeneity of soils in natural environments.

A further gap in the literature is the limited understanding of MP degradation and processes and mineralization within soil ecosystems. Although degradation mechanisms have been studied in the aquatic environment (Mishra et al., 2022; Priya et al., 2022), far fewer studies have examined how MPs are degraded and mineralized in soil, particularly in the presence of oxidative agents that mimic environmental conditions. Degradation and mineralization in soil is influenced by several factors, including microbial activity, soil mineral composition, and natural stressors. One of the strongest oxidants found in natural environments is Fenton's reagent, which showed potential to degrade and mineralize MPs via radical attack (Xing et al., 2023). Nevertheless, insights into the interaction of different MP types, radicals and degradation efficiency under mild reaction conditions remain unknown. While much attention has been given to MP degradation and mineralization with Fenton's

reagent under laboratory conditions with harsh reaction conditions (Hu et al., 2022; Ortiz et al., 2022), the relevance to realistic degradation and mineralization models under natural conditions is rarely demonstrated. Finally, it calls for the incorporation of realistic degradation processes in MP studies, using oxidative reagents and simulated environmental conditions, to better predict the fate of MPs in soil.

Soil type is also a critical factor influencing and impacting MP degradation and mineralization rates. However, the ways in which factors such as organic matter, nutrient availability and pH influence the breakdown of different MP types are still not well understood. Moreover, a key focus in the context of natural MP degradation and mineralization in soil should be on understanding the specific chemical and physical modifications that the persistent plastic particles and fibers undergo during the oxidation process, as well as the effects that weathered and partially degraded MPs have on the soil ecosystem. Beyond changes in morphology and particle size, these MPs also experience alterations in properties such as hydrophilicity/hydrophobicity and acidity/basicity until they are fully mineralized to CO₂. These surface changes could lead to unexpected interactions with soil particles, microbes, and plants, potentially influencing soil health in ways that are not yet fully understood.

In summary, the continuing increase in MP contamination in soil poses significant challenges to understanding its ecological and environmental impacts. The complexity of soils as dynamic systems, coupled with the variability of MP types and the limitations of current experimental methods, highlight the need for a multifaceted approach.

1.3. Objectives and research questions

The primary focus of this work was to investigate how different types of MP interact with distinct soil textures and explore their effects on early-stage plant development under controlled conditions. Additionally, the study examines whether MPs affect plant growth differently in different soils and whether soil texture plays a critical role in mediating these effects. A further aim was to assess whether MPs, often considered inert in soils, are susceptible to mineralization via soil-associated processes, such as Fenton reactions, and whether different polymeric compositions of MPs influence their susceptibility to radical attacks. The following research objectives are addressed in Chapters 2 to 5:

CHAPTER 2 investigates the broader ecological implications of MPs in soil, focusing on their potential to alter soil structure, water dynamics, and plant-available water, thereby highlighting their influence on plant growth. The specific research objectives in this context were to:

- i) Investigate the effects of MPs on soil water availability, thereby aiming to understand how different types of MP – LDPE, PP, PS particles, and PES microfibers – influence soil water retention and plant-available water across various soil textures.
- ii) Assess the impact of MPs on soil hydraulic properties, such as saturated and near-saturated hydraulic conductivity, across different soil textures.
- iii) Examine the interaction between the four distinct types of MP with three soils of different textures (loamy silt, loamy sand, and silty clayey loam) and in which way they alter the soils physical properties.

CHAPTER 3 addresses research questions related to plant growth through the use of rhizotrons. The aim was to understand both the direct and indirect effects of different MP types (PP, LDPE, PS particles, and PES microfibers) on soil properties, nutrient availability, and plant growth. Specifically, the research aimed to:

- i) Assess the impact of different MPs on soil nutrient concentrations and plant nutrient uptake, including concentrations of ammonium (NH_4^+), nitrate (NO_3^-), potassium (K), and P.
- ii) Evaluate the effects of MPs on the structure and connectivity of microbial communities within the rhizosphere.
- iii) Explore the broader impact of MPs on plant growth and vitality, namely whether the changes in soil conditions and microbial network caused by MPs had significant effects on above-ground plant development, including leaf growth, chlorophyll content, and overall plant health.

CHAPTER 4 of this work focuses on texture-dependent effects of MPs on soil and plant performance. The research objectives were to:

- i) Investigate the impact of two different MPs with distinct shapes – LDPE particles and PES microfibers – on plant growth in soils with different textures.
- ii) Explore how soil texture modulates the effects of MPs on soil structure, nutrient availability, and water retention, highlighting the differences in MP impact across silty loam, sandy loam, and silty clay loam.
- iii) Evaluate how LDPE and PES microfibers influence soil water content and nutrient dynamics (such as nitrogen levels) and how these changes affect plant access to water and nutrients.
- iv) Understand the interactions between MPs and soil porosity and how these structural changes influence root development, water movement, and nutrient availability.
- v) Contribute to the broader understanding of MP-soil-plant interactions and providing insights into the complex relationships between MPs, soil texture, and plant growth.

The research presented in *CHAPTER 5* aimed to understand the mineralization mechanisms of MPs in the presence of Fenton's reagent and the complex interaction between polymer types, radical dynamics, and reaction conditions. The following research objectives were pursued:

- i) Investigating the mineralization kinetics of different MP types (LDPE, PP, PS, and PES) using Fenton's reagent, focusing on the release of CO₂ as a marker of mineralization.
- ii) Evaluating the influence of varying concentrations of H₂O₂ affecting the efficiency and rate of MP mineralization.
- iii) Comparing the degree of mineralization of different polymer types and how the structural characteristics of each MP influence their susceptibility to Fenton-induced mineralization.
- iv) Visually examining surface changes (e.g., cracks) on MPs due to mineralization by using SEM, especially comparing more mineralizable polymers like PES to less responsive ones like LDPE.
- v) Understanding how factors such as OH[•] radical recombination, scavenging effects, and the complex multi-step kinetics of Fenton reactions influence the mineralization process across different MP types.

Through these objectives and research questions, this study aimed to contribute to the growing body of knowledge on MPs in terrestrial ecosystems by providing insights into their environmental behavior and potential ecological risks.

CHAPTER 2 – Soil texture governs the influence of microplastics on soil hydraulic properties

Based on:

Katharina J. Neubert, Lutz Weihermüller, Harry Vereecken, Nicolas Brüggemann (2025). Soil Texture Governs the Influence of Microplastics on Soil Hydraulic Properties. *Vadose Zone Journal*, 24(5), e70044. DOI: 10.1002/vzj2.70044.

Abstract

Adequate water availability in soil is crucial for optimal plant growth. However, even small amounts of microplastics (MPs) can disrupt the soil pore structure, impacting water retention and hydraulic conductivity, and consequently plant-available water. This study investigated the effects of various MPs on soil water availability and hydraulic properties in three soil types (Luvisol, Albic Luvisol, and Chernozem). Four polymer types – LDPE, PP, PS particles, and PES microfibers – were tested at a mass concentration of 0.4% w/w. Results showed that the impact of MPs varied with soil texture and saturated water content. In the Luvisol (loamy silt), PES microfibers enhanced porosity, creating more air-filled pore volume, macropores, and increased water retention and saturated hydraulic conductivity. Conversely, PS and PP particles reduced porosity in the Luvisol, enhancing near-saturated hydraulic conductivity, likely due to decreased tortuosity. In the Albic Luvisol (loamy sand), PES microfibers increased porosity and plant-available water, but both microfibers and particles negatively affected saturated and near-saturated hydraulic conductivity, with PS particles and PES microfibers potentially clogging soil pores. In the Chernozem (silty clay loam), PES microfibers significantly increased air-filled pore volume without changing overall porosity but negatively impacted water retention and plant-available water. The increase in saturated conductivity was likely due to preferential flow paths, while under dry conditions near-saturated hydraulic conductivity decreased, likely due to increased tortuosity.

2.1. Introduction

The presence of MPs in soil has emerged as a rising concern (Guo et al., 2020; Ya et al., 2021). These small plastic particles and fibers, which are less than 5 mm in size, have raised questions about their long-term effects on ecosystems and the hydrological and biogeochemical processes within them (Bian et al., 2022; Wang et al., 2022a). Although our understanding of the full scope of these impacts is still incomplete, it is evident that MPs are steadily accumulating in the environment. In agricultural soils, MPs enter through various pathways, such as plastic mulching (Liu et al., 2018; Huang et al., 2020), sewage sludge (Weber et al., 2022), road and tire wear (Rødland et al., 2022), litter, and improper waste disposal, to name just a few. Research has demonstrated that in agricultural fields in Germany, fertilization with sewage sludge and wastewater discharges have particularly served as significant gateways for MP accumulation (Brandes et al., 2021). Shredded synthetic microfibers released from washing machines (Carney Almroth et al., 2018) and MP particles from cleaning and personal care products (Estahbanati & Fahrenfeld, 2016) are the main source in the sewage sludge applied. Even 34 years after the end of sewage sludge application on a German agricultural field, macroplastics at a density of 637.12 items per hectare and up to 56 MP particles per kilogram of dry soil were still present (Weber et al., 2022), and Heinze et al. (2024) found even more than 4 mg MP per kg soil in a Swedish agricultural field.

The composition and prevalence of MPs in agricultural soils have attracted considerable attention in recent studies. A study conducted in northern Germany found that nearly 87% of the MPs identified in agricultural fields were PE, a material commonly used in various plastic products (Harms et al., 2021). This finding is consistent with the substantial production of PE in Germany, which accounts for approximately 30% of the country's total plastics production. Polyethylene, in both low-density and high-density forms, is widely used in items such as plastic films due to its ductility, flexibility, and transparency (Heinrich-Böll-Stiftung, 2019). In addition to PE, PP is emerging as another type of plastic prevalent in agriculture (Liu et al., 2018; Piehl et al., 2018). The high melting point and chemical resistance of PP make it ideal for applications such as bulk bags for seed and fertilizer transportation, as well as soil cover for protection against environmental factors and erosion (Sharma et al., 2020). PS and PES microfibers have also been identified in agricultural soils (Piehl et al., 2018), with PS accounting for 14% of all particles found, as it is commonly used in packaging and food containers, while PES microfibers, derived from PET, are prevalent due to their use as synthetic fibers (Heinrich-Böll-Stiftung, 2019). In particular, the presence of PES microfibers in agricultural soils is often associated with sewage sludge fertilization (Büks & Kaupenjohann, 2020). Around 80-99% of the microfibers contained in wastewater are retained in sewage treatment plants and end up in sewage sludge, resulting in concentrations ranging from 1,000-56,400 particles kg⁻¹ dry sewage sludge (Zubris & Richards, 2005;

Mahon et al., 2017; Mintenig et al., 2017). This highlights the complex pathways through which MPs enter agricultural ecosystems.

Research has demonstrated that MP accumulation in agricultural ecosystems is associated with changes in various soil parameters. These changes include alterations in pore structure, which lead to shifts in hydraulic properties (Zhang et al., 2019; Guo et al., 2022; Wang et al., 2023), bulk density (Ingraffia et al., 2022a), as well as aggregate distribution and water-holding capacity (de Souza Machado et al., 2019; Ingraffia et al., 2022a). Especially microfibers, due to their linear shape, their ability to entangle soil particles, their hydrophobic nature, and their flexibility have been shown to promote (Zheng et al., 2016; Lozano et al., 2021a) or reduce soil aggregation (Lozano et al., 2021b). Either way, this rearrangement of soil pores affects the soil-water characteristics, as enlarged pores allow for better drainage and small pores for better water retention. Since soil hydraulic characteristics can be indicators for soil health (Bagnall, 2022) and can have a direct influence on plant root growth (Ma et al., 2009), their precise determination is crucial.

Soil health is assessed, among other factors, using indicators for water movement and storage (Bagnall, 2022). For water movement, both saturated and near-saturated hydraulic conductivity (K_{sat} and K_{ns} , respectively) are important factors, but also the volumetric water content at field capacity (θ_{FC}), the permanent wilting point (θ_{WP}), and the plant-available water (PAW). Furthermore, the retention characteristics can be related to the soil pore size distribution (Liu et al., 2021). In general, total soil porosity can be differentiated and characterized by the macroporosity (P_{mac}), which indicates the volume of pores larger than 0.03 mm. This information is crucial as it defines the soil's capability to drain excess water (Reynolds et al., 2009). Secondly, the air capacity (AC) serves as a quantitative measure of the soil's capacity of both storing and providing soil gases (Topp et al., 1997) and the PAW quantifies the soil's ability to retain and provide plant-accessible water. All three parameters play a pivotal role in maintaining optimal physiological functions in plant roots.

The mentioned soil parameters can be significantly impacted by the presence of MPs in soil, although so far only demonstrated for concentrations well above the average occurrence of MPs in soil, as shown for instance in studies on the influence of PES microfibers on water-holding capacity, aggregate stability, and bulk density. For example, de Souza Machado et al. (2018) reported an increase in water-holding capacity and aggregate stability of a loamy-sandy soil due to elevated concentrations of PES microfibers (0.4% on a weight/weight basis, % w/w). Conversely, at a lower concentration of 0.1% w/w, no discernible effects were detected in the same study. Ingraffia et al. (2022a) demonstrated that the contamination with PES microfibers reduced aggregate formation by 32%, 47%, and 33% for a Vertisol, Entisol, and an Alfisol, respectively, accompanied with altered soil bulk density. In a complementary study, a notable reduction in the volume of meso- and micropores along with an

increase of macropores was observed at a concentration of 0.3% w/w of PES microfibers in clayey loam soils (Zhang et al., 2019). Furthermore, microfibers added at 0.1% and 0.3% w/w significantly increased water-stable macroaggregates (>2 mm). However, no discernible effects on soil bulk density or K_{sat} were found at the applied concentration of PES microfibers. This highlighted the strong dependence of the effect of PES microfibers on soil texture.

Another study established an inverse relationship between PP MP concentration and soil infiltration capacity (Guo et al., 2022). Addition of PP MPs to loamy, clayey, and sandy soils significantly reduced K_{sat} , whereas larger MP particles had a weaker effect on K_{sat} . Furthermore, Guo et al. (2022) reported that PP with increasing particle sizes (from 20 to 500 μm) reduced water retention, especially in clayey compared to sandy and loamy textured soils. This reduction affected both saturated (θ_s) and residual water content (θ_r). Finally, Xie et al. (2023) found that in compacted silty-sandy soil, different LDPE particle sizes had opposing effects on hydraulic properties. Particles of 500 μm increased K_{sat} , whereas particles of 50 and 150 μm reduced K_{sat} . In contrast, LDPE particles decreased θ_r of compacted silty sand soil independently of particle size or concentration added. This pattern of changes in K_{sat} and θ_r was explained by a reduction in pore size and number of macropores in case of K_{sat} changes.

As shown above, MPs can lead to changes in soil porosity and soil pore distribution, which in turn can have significant effects on soil hydraulic characteristics, such as water retention and conductivity. However, the mechanisms underlying the effects of MPs on soil hydraulic properties have not been systematically explored and are not yet fully understood. Moreover, ambiguous findings are reported in the literature. This ambiguity might stem from the study design itself, as either the same MP type is used in combination with different soils, or one soil is used in combination with different MP types. Focusing solely on the effects of MPs on one soil (or one soil texture) may result in an incomplete assessment, as the soil hydraulic properties are closely linked to the soil pore size distribution, and therefore, to soil texture (Ingraffia et al., 2022a; Wang et al., 2023). To gain a better understanding of the effects of MPs on soils, it is imperative to study the interplay between different MP particles and/or fibers (types, size, and concentrations) and different soils with different textures (de Souza Machado et al., 2018; Zhang et al., 2019; Xie et al., 2023).

In the present study, we conducted experiments with three soil types with contrasting texture (Luvisol: silty loam, Albic Luvisol: silty sand, and Chernozem: silty clay loam) and four different MPs (LDPE, PP, PS, and PES microfibers) at a fixed concentration (0.4% w/w). Furthermore, we measured the full soil water retention and hydraulic conductivity characteristics rather than only one soil characteristics such as K_{sat} . As MPs are known to be hydrophobic (Prajapati et al., 2022), we also measured the K_{ns} . We hypothesized that the MP types would affect the soil characteristics differently,

as they differed in size and shape. We also hypothesized that the different soil textures would be affected differently by the different MPs as the pore size distribution differed between the soils.

2.2. Materials and methods

2.2.1. Soils

Three soil types were used in the study: i) a Luvisol (LU) with a loamy silt texture, ii) an Albic Luvisol (AL) with a loamy sand texture, and iii) a Chernozem (CH) with a silty clay loam texture. The basic soil characteristics of the three different soils are listed in Table 4.1. The two soils CH and LU did not differ largely in soil texture but in their organic carbon (C_{org}) content (CH = 2.06%, LU = 1.01%). The LU soil was taken from arable land at the agricultural research station Campus Klein-Altendorf of the University of Bonn, Germany (50.613614°N, 7.000713°E). The AL soil was taken from an agricultural field of the Albrecht Daniel Thaer Institute for Agriculture and Horticulture Sciences of the Humboldt University of Berlin located in Thyrow, Germany (52.254674°N, 13.236030°E). The third soil type, CH, was collected from the agricultural research station of the Helmholtz Centre for Environmental Sciences (UFZ), located in Bad Lauchstädt, Germany (51.393447°N, 11.875048°E). All soil types were taken from 0-20 cm depth, air-dried and sieved to 2 mm. Soil texture was measured by wet sieving and the pipette method according to DIN ISO 11277 (2002). Organic carbon content (C_{org}) was measured by combustion (TOC-V CPH, Shimadzu, Kyoto, Japan) and pH was measured in $CaCl_2$ solution at a ratio of 5 g soil to 40 mL of $CaCl_2$.

Table 2.1. Characteristics of experimental soils including organic carbon (C_{org}) content, plant-available mineral nitrogen content (N_{min}), plant-available phosphate (P_{av}) and potassium (K_{av}).

Soil type	Abbr.	Texture	Sand 63-2000 μm %	Silt 2-63 μm %	Clay <2 μm %	pH ^a	C_{org} %
Luvisol	LU	Loamy silt	8	77	15	6.5	1.01
Albic Luvisol	AL	Loamy sand	87	10	3	5.1	0.50
Chernozem	CH	Silty clay loam	11	68	21	7.5	2.06

^apH determined in 0.01 M calcium chloride solution.

It must be acknowledged that it was impossible to definitively exclude the presence of MPs and/or nanoplastics in the soil types used, considering that the agricultural areas from which the soil was taken may have been subject to anthropogenic pollution. Nonetheless, these sites had no record of plastic mulching or sewage sludge fertilization. Any larger plastic debris found was removed during

sieving. In addition, the inclusion of control samples without added MP and the relatively high MP concentration (0.4% w/w) mitigated any potential bias from prior contamination.

2.2.2. Microplastics

Based on results on MP pollution reported in the literature (Piehl et al., 2018; de Souza Machado et al., 2019), the following MPs were selected: i) low-density polyethylene (LDPE) as powder with particle size 300-600 μm , (ii) polypropylene (PP) as 4 mm beads, (iii) polystyrene (PS) as 3-5 mm beads, and (iv) polyester (PES) microfibers used as pillow filling with 28.8 ± 0.17 mm in length and 0.74 ± 0.01 mm in diameter. Microfiber length was determined by manually measuring individual microfibers ($N = 50$), while the diameter was determined by analyzing scans (EPSON, Expression, 12000xl, Meerbusch, Germany) of microfibers via the software tool WinRHIZO (Regent Instruments Inc., Québec, Canada). The LDPE, PP, and PS was purchased from Goodfellow GmbH, Hamburg, Germany and the PES from JYSK Nordic A/S, Braband, Denmark.

PS and PP plastic beads were shredded in a conventional high-performance kitchen blender (BlendTec classic 575, Luba GmbH, Bad Homburg, Germany) and sieved to 100-1000 μm particle size with stainless steel analyzing sieves. The set-up of the blender is shown in the Appendix A, Fig. A.1. All particle types (LDPE, PP, and PS) were measured with a laser diffraction particles size analyzer (HORIBA LA-950, Irvine, CA) by dispersing the respective MP powder in ethanol (Appendix A: Fig. A.2). The LDPE powder had the narrowest particle size distribution with a mean particle diameter of 444.6 ± 1.1 μm , followed by PS (532.3 ± 4.7 μm) and PP (548.0 ± 4.7 μm).

The PES microfibers used consisted of PET, which was determined via Raman spectroscopy. A piece of a PET plastic bottle was cut out as a reference material for PES and its spectrum was compared with the spectrum of the microfibers (Appendix A: Fig A.3 and Table A.1). See further MP characteristics in Table 2.2.

Table 2.2. Characteristics of the experimental microplastic.

Microplastic	Abbr.	Mean size	Density
-	-	μm	g cm^{-3}
Low-density polyethylene	LDPE	444.0 ± 1.0	0.92^{a}
Polypropylene	PP	548.0 ± 4.7	0.90^{a}
Polystyrene	PS	532.3 ± 4.7	1.05^{a}
Polyester microfibers made of polyethylene-terephthalate	PES	2880 ± 170	1.38^{b}

^aDensity values were according to the technical datasheet from the supplier (Goodfellow GmbH, Hamburg, Germany).

^bDensity value according to the technical data from Reichelt-Chemietechnik GmbH + Co, Heidelberg, Germany.

2.2.3. Soil preparation

For the experiments, the four MP types were added separately to each air-dry soil type at 0.4% (w/w) and mixed thoroughly. According to de Souza Machado et al. (2018), the applied mass concentration was the upper limit of MP concentration at which the MP-amended soils show only minor total volume changes. The LDPE, PP, and PS particles were mixed with the soil manually in closed plastic bottles. The control was treated in the same way, but without MP addition. Since the synthetic microfibers were strongly entangled, the PES microfibers were torn apart by hand and added to the soil step by step while shaking the soil. As all experiments were performed in quadruplicate, 60 samples (3 soils x (4 MP + 1 control) x 4 replicates) were prepared for the measurements of the soil hydraulic characteristics (water retention and K_{sat}), and another set of 60 samples for the infiltration measurements (K_{ns}).

For the K_{sat} measurements and the evaporation method used to determine water retention and hydraulic characteristics, the air-dry samples were packed into stainless steel cylinders of 250 cm³ volume (height = 5 cm and diameter = 8 cm) at predefined bulk density (Luvisol = 1.2 g cm⁻³, Albic Luvisol = 1.6 g cm⁻³, and Chernozem = 1.4 g cm⁻³). The bulk density was adjusted according to the bulk densities of the different soils observed in the field. For the infiltration measurements (K_{ns}), the samples were packed into PVC columns of 10 cm height and 10 cm diameter at the same bulk densities as in the 250 cm³ cylinders. The PVC columns were sealed with a permeable cloth and left on a sand bed. Here, it has to be noted that the raw density of the MPs differs from those of the soil leading to very small differences in theoretical porosity compared to the control of the individual soils.

2.2.4. Soil hydraulic characterization

2.2.4.1. Saturated hydraulic conductivity (K_{sat})

The filled 250 cm³ cylinders were placed in boxes and saturated stepwise from the bottom for seven days using tap water. The KSAT device (Meter Group AG, Munich, Germany) was used to measure K_{sat} using the falling head method. K_{sat} values were normalized to 20°C. Each sample was measured three times, and the mean as well as the standard deviation of K_{sat} was calculated.

2.2.4.2. Soil water retention and other soil characteristics

After the K_{sat} measurements, the soil water characteristics were determined by the evaporation method using the HYPROP® system (Meter Group, München, Germany) as described by Schindler et al. (2010) in combination with the WP4® Dewpoint Potentiometer (Decagon Devices, Pullman, WA, USA). Finally, the Mualem-van Genuchten (van Genuchten, 1980) model was fitted to the HYPROP data (retention and conductivity data), where the actual water content θ can be expressed as a function of pressure head h (-cm) by:

$$\theta(h) = \frac{(\theta_s - \theta_r)}{(1 + (\alpha \cdot |h|)^n)^m} + \theta_r \quad (2.1)$$

with θ_s and θ_r as the saturated and residual volumetric water contents (cm³ cm⁻³), and n and α (-) as shape parameters, where n (-) can be related to the pore size distribution and α (cm⁻¹) is a scaling factor related to the air entry value. Van Genuchten (1980) related m is defined as $m = 1 - 1/n$.

The hydraulic conductivity function was described by:

$$K_r(h) = \left[\frac{[1 - (\alpha h)^{mn} [1 + (\alpha h)^n]^{-m}]^2}{[1 + (\alpha |h|)^n]^{m\lambda}} \right] \quad (2.2)$$

where $K_r(h)$ (-) is the relative hydraulic conductivity at a given pressure head h (-cm), and λ (-) is the tortuosity factor. To obtain the actual hydraulic conductivity, $K_r(h)$ has to be multiplied with the saturated hydraulic conductivity K_{sat} . It has to be noted that K_{sat} was not fitted and set as a matching point during the fitting process.

The four replicates were measured individually by the evaporation method and were also fitted individually. For the mean van Genuchten parameters of one soil/MP combination, the individual data points from the evaporation method were combined and simultaneously fitted. The corresponding van Genuchten parameters as well as further soil characteristics are listed in Table 2.3. For analyzing the

effect of MPs on the soil hydraulic characteristics the fitted van Genuchten soil hydraulic parameters (α , n , θ_s , and θ_r) were used. Additionally, the water content at field capacity (θ_{FC} , $\text{cm}^3 \text{cm}^{-3}$ measured at $\log_{10} h = 1.8$ or $h = -63$ cm), water content at wilting point at $\log_{10} h = 4.2$ ($h = -15,848$ cm) (θ_{WP} , $\text{cm}^3 \text{cm}^{-3}$) and the plant-available water (θ_{PAW} , $\text{cm}^3 \text{cm}^{-3}$) were calculated based on the soil hydraulic parameters.

Finally, some basic soil characteristic, i.e., the macroporosity (P_{mac} , $\text{cm}^3 \text{cm}^{-3}$, Eqn. 2.3) according to Reynolds et al. (2009), air capacity (AC , $\text{cm}^3 \text{cm}^{-3}$, Eqn. 2.4) according to Topp et al. (1997) and water-filled pore space ($WFPS$, %, Eqn. 2.5) were calculated.

$$P_{mac} = \theta_s - \theta_{@h=10cm} \quad (2.3)$$

$$AC = \theta_s - \theta_{@h=100cm} \quad (2.4)$$

$$WFPS = \frac{\theta_{@given\ h}}{\theta_s} \cdot 100 \quad (2.5)$$

Thereby, the ratio between the $WFPS$ for the control and that of a soil/MP mixture was calculated for predefined pressure heads (here at $\log_{10} h = 0, 0.6, 1.2, 1.8, 2.4, 3, 3.6, 4.2, 4.8, 5.4$, and 6). Negative differences mean that the $WFPS$ was reduced compared to the control, whereas positive differences indicate that the MP amendment increased the $WFPS$ with potential positive impacts on water availability for plant growth.

2.2.4.3. Near-saturated hydraulic conductivity (K_{ns})

The Mini-disk Infiltrometer (MDI, Meter Group AG, Munich, Germany) with a disk diameter of 4.5 cm was used to measure the water infiltration into dry soils. The MDI has a small water reservoir, from which the water can infiltrate into the soil, and a Mariotte compartment, which controls the suction of the infiltration. For the infiltration, a suction of $h = -2$ cm was set and the water reservoir was filled to its maximum volume of 95 mL with brilliant blue FCF (E133) stained tap water to facilitate the reading of the reservoir level. For the measurements, the MDI was placed on the soil surface of the soil-packed PVC columns (10 height, 10 cm diameter). During the infiltration process, a webcam installed in front of the infiltrometer recorded pictures of the scene every 10 s, which was facilitated by a custom-made software. The photos were stored as JPEG files along with a timestamp and were used to record the infiltrating water after the infiltration had been finished (Appendix A: Fig. A.3). Each soil column was measured three times after being dried for seven days at 60°C to ensure reproducible starting conditions in water content.

To calculate K_{ns} from the raw infiltration data (Eqn. 2.6) was fitted to the infiltration data, where C_1 (cm s^{-1}) is related to the hydraulic conductivity, and C_2 ($\text{cm s}^{-1/2}$) is related to the soil-specific

absorption/desorption process (Rojas et al., 2022). For the estimation of the hydraulic conductivity the method proposed by Zhang (1997) was used, where C_1 reflects the slope of the cumulative infiltration curve versus the square root of time.

$$I = C_1 t + C_2 \sqrt{t} \quad (2.6)$$

In a second step, K_{ns} can be calculated by:

$$K_{ns} = \frac{C_1}{A} \quad (2.7)$$

where A is related to the suction h (cm) imposed during infiltration by:

$$A = \frac{11.65 \cdot (n^{0.1} - 1) \cdot e^{[a(n-1.9)\alpha h]}}{(\alpha r_o)^{0.91}} \quad (2.8)$$

where r_o is the radius of the MDI (here $r_o = 2.25$ cm), and α and n are the van Genuchten parameters α and n . Parameter a is variable depending on the van Genuchten n : a is equal to 2.92 for $n \geq 1.9$, and equal to 7.5 for $n < 1.9$. The corresponding van Genuchten α and n values were selected from a look-up table provided by the user's manual (version 3) of the MDI (Decagon Devices, WA, USA) and are listed for the three soil types used (Table 2.3). K_{ns} measurements were performed on all four sample replicates three times to be in-line with K_{sat} measurements.

Table 2.3. Van Genuchten parameters for the three soil types used according to the user's manual of the MDI (Decagon Devices, WA, USA) as well as calculated A values.

Soil type	WRB soil	α	n	A @ $h = -2$ cm
-	-	cm ⁻¹	-	-
Luvisol	Silty loam	0.02	1.41	7.9299
Albic Luvisol	Silty and	0.12	2.28	2.4286
Chernozem	Silty clay loam	0.01	1.23	8.5112

2.2.5. Data analysis

To assess the contribution of both the independent variables of soil texture and MP type on the dependent variables, such as K_{sat} and K_{ns} , the effect size η^2 (%) was calculated (Richardson, 2011). The η^2 for each main effect, i.e., the effects of soil type ($\eta^2(A)$) and MP type ($\eta^2(B)$) on the soil hydraulic characteristics, as well as the interaction of both ($\eta^2(AB)$), were calculated in a two-way ANOVA with

replication using Eqs. (2.9) to (2.11). The effect size for the error ($\eta^2(E)$), which resembles the unknown variance, was calculated according to Eqn. (2.12).

$$\eta^2(A) = \frac{SSA}{SST} \cdot 100 \quad (2.9)$$

$$\eta^2(B) = \frac{SSB}{SST} \cdot 100 \quad (2.10)$$

$$\eta^2(AB) = \frac{SSAB}{SST} \cdot 100 \quad (2.11)$$

$$\eta^2(E) = \frac{SSE}{SST} \cdot 100 \quad (2.12)$$

where SSA is the sum of squares for soil type, SSB is the sum of squares for MP type, SSAB is the sum of squares for the interaction between soil type (A) and MP type (B), SST is the total sum of squares, and SSE is the sum of squares for the error.

For the K_{ns} and K_{sat} measurements, the standard deviation (SD) was calculated for the four sample replicates, with an error propagation according to Eqn. (2.13).

$$SD = \sqrt{\frac{1}{4}(SD_1^2 + SD_2^2 + SD_3^2 + SD_4^2)} \quad (2.13)$$

To test for significant differences between the different soils mixed with MP and the corresponding control, one-sided heteroscedastic t-tests were performed at a significance level of $p \leq 0.05$, indicating a significant MP effect, and $p \leq 0.001$, indicating a highly significant MP effect.

2.3. Results

2.3.1. Saturated hydraulic conductivity (K_{sat})

K_{sat} greatly differed between the soils (Fig. 2.1a-c) and was lowest for the Chernozem (mainly below 14 cm d⁻¹), slightly higher for the Luvisol (10-50 cm d⁻¹) and highest for the Albic Luvisol (250-730 cm d⁻¹). For both finer textured soils, the addition of PES microfibers significantly increased K_{sat} by 91% ± 47% ($p = 0.02$) in Luvisol and by 67% ± 43% ($p = 0.03$) in the Chernozem compared to the control. In contrast, in the coarse textured Albic Luvisol the addition of PES microfibers resulted in a highly significant reduction of K_{sat} by 58% ± 2% ($p < 0.001$). In this Albic Luvisol soil not only the PES microfiber amendment, but also the addition of PP and PS resulted in a significant reduction of K_{sat} compared to the control soil, with 25% ± 11% ($p = 0.007$) lower K_{sat} for PP-amended soil and 34% ± 13% ($p = 0.002$) lower K_{sat} values for PS-amended soil (Fig. 2.1a-c).

2.3.2. Near-saturated hydraulic conductivity (K_{ns})

The results of the K_{ns} measured at $h = -2$ cm in air-dried soil (Fig. 2.1d-f) revealed in general a different pattern as found for K_{sat} , with the lowest K_{ns} for the Luvisol (below 16 cm d⁻¹), higher K_{ns} for Chernozem (14-70 cm d⁻¹), and much higher K_{ns} for the Albic Luvisol (45-230 cm d⁻¹). The same trend of reduced K_{ns} as for K_{sat} was found for the MP-amended Albic Luvisol, where LDPE, PS and PES reduced K_{ns} significantly by 32% ± 13% ($p = 0.01$), 44% ± 23% ($p = 0.005$), and 62% ± 4% ($p < 0.001$), respectively, compared to the control. Surprisingly, in the Chernozem the same relative change in K_{ns} was observed with addition of LDPE, PS, and PES, significantly reducing K_{ns} by 32% ± 14% ($p = 0.02$), 44% ± 24% ($p = 0.006$) and 62% ± 4% ($p = 0.01$). In contrast, the addition of LDPE, PP, and PS in Luvisol significantly increased K_{ns} by 120% ± 73% ($p = 0.01$), 40% ± 17% ($p = 0.001$), and 27% ± 13% ($p = 0.04$), respectively. However, PES in Luvisol did not significantly change K_{ns} compared to the control.

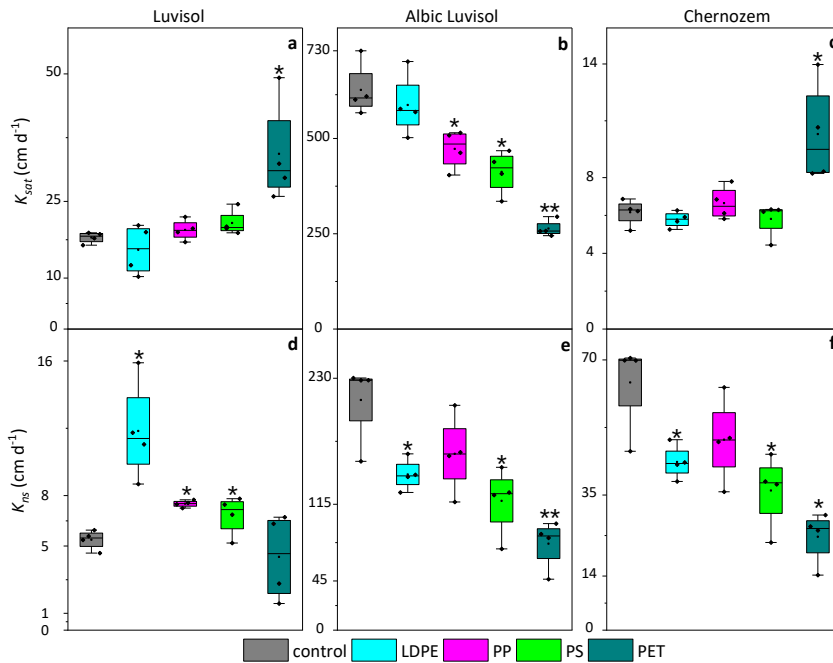


Fig. 2.1. Boxplot diagram of saturated hydraulic conductivity (K_{sat} , cm day⁻¹) for a) the Luvisol, b) Albic Luvisol, and c) Chernozem and near-saturated hydraulic conductivity (K_{ns} , cm day⁻¹) for d) the Luvisol, e) Albic Luvisol, and f) Chernozem mixed with the four different microplastic (MP) types at 0.4% w/w, and the respective soil types without the addition of MP. Boxes are the interquartile range (IQR) from the 25th to 75th percentile, lines inside boxes indicate the median, open squares indicate the mean values, upper and lower whiskers are 1.5 times the IQR, and markers are individual data points. Asterisks (*) mark significant differences in K_{sat} or K_{ns} between MP treatment and control, which was tested via one-sided, heteroscedastic t-test ($p \leq 0.05$). Double asterisks (**) mark highly significant differences at $p \leq 0.001$. Note, that the y-axis are scaled differently for better visualization.

2.3.3. K_{ns} -to- K_{sat} ratio

To obtain deeper insights into the relative differences between K_{ns} and K_{sat} across the different soil types in response to MP additions, the mean K_{ns} -to- K_{sat} ratio (K_{ns}/K_{sat}) for all MP amendments was calculated (Table 2.4). For the Luvisol and Albic Luvisol controls, K_{ns} was approximately one third of K_{sat} , whereas in the Chernozem soil K_{ns} was ten times higher than K_{sat} without the addition of MP. However, these ratios were differently influenced due to different MP amendments.

The presence of LDPE in the Luvisol increased the K_{ns}/K_{sat} by 172% ($p = 0.02$), primarily due to a significant increase in K_{ns} without altering K_{sat} substantially. Furthermore, the addition of PP to the Luvisol resulted in a significant 30% increase in K_{ns}/K_{sat} ($p = 0.02$) caused by higher K_{ns} compared to the control and only smaller increase in K_{sat} , while the addition of PES decreased the ratio by 50% ($p = 0.03$), attributed to substantial increase in K_{sat} , while K_{ns} stays relatively unaffected.

In the coarser textured Albic Luvisol, only the introduction of LDPE significantly affected the K_{ns}/K_{sat} ratio with a decrease of 27% ($p = 0.03$) compared to the control, as LDPE notably reduced K_{ns} while leaving K_{sat} unchanged. As the other MP types show significant decreases in both K_{ns} and K_{sat} the ratio of both parameters is not significantly affected.

In the case of the fine textured Chernozem, both PS and PES reduced the K_{ns}/K_{sat} by 39% ($p = 0.02$) and 76% ($p = 0.002$), respectively. The PES microfibers in particular caused a significant shift in hydraulic conductivity, with a significant reduction in K_{ns} and increase in K_{sat} .

Table 2.4. Mean K_{ns} -to- K_{sat} ratio for the three soils mixed with low-density polyethylene (LDPE) particles, polypropylene (PP) particles, polystyrene (PS) particles, and polyester (PES) microfibers at 0.4% w/w or no microplastics (control).

Soil type	Control	LDPE	PP	PS	PES
Luvisol	0.30 ± 0.03	0.82 ± 0.28 ^a	0.39 ± 0.04 ^a	0.33 ± 0.05	0.15 ± 0.09 ^a
Albic Luvisol	0.34 ± 0.06	0.25 ± 0.04 ^a	0.34 ± 0.06	0.29 ± 0.09	0.30 ± 0.07
Chernozem	10.56 ± 2.24	7.59 ± 1.05	7.63 ± 2.05	6.39 ± 1.80 ^a	2.57 ± 0.98 ^a

^aSignificantly different from control ($p \leq 0.05$).

2.3.4. Effect sizes of soil type vs. microplastic type

The results of the ANOVA indicated that the differences for both K_{sat} and K_{ns} were mainly caused by the differences in soil type and not by MP type, whereby the effect of soil type was more pronounced for K_{sat} (Fig. 2.2a) than for K_{ns} (Fig. 2.2b). The calculated $\eta^2(A)$ for K_{sat} was 77-97%, while for K_{ns} it was slightly lower (52-84%). The soil type effect was generally less pronounced in the K_{ns} measurements and also when PES microfibers were present.

The results for the calculated $\eta^2(B)$ showed that the MP influence was much more pronounced for the K_{ns} measurements based on infiltration into the dry soil, with the highest effect for PS (13%) and PES (28%), whereas for K_{sat} in the saturated soil no (0% for LDPE) to only a minor MP effect (6% for PES) was observed.

The interactive effect of soil type and PES on K_{sat} was 2.5 times greater than that of the microfibers alone ($\eta^2(AB) = 15\%$ and $\eta^2(B) = 6\%$). In contrast, the interactive effect soil type and PES on K_{ns} was less than half as high ($\eta^2(AB) = 13\%$ and $\eta^2(B) = 28\%$). Notably, PS particles had a greater effect on K_{ns} than the interactive effect ($\eta^2(B) = 13\%$, $\eta^2(AB) = 8\%$), but this effect was relatively small.

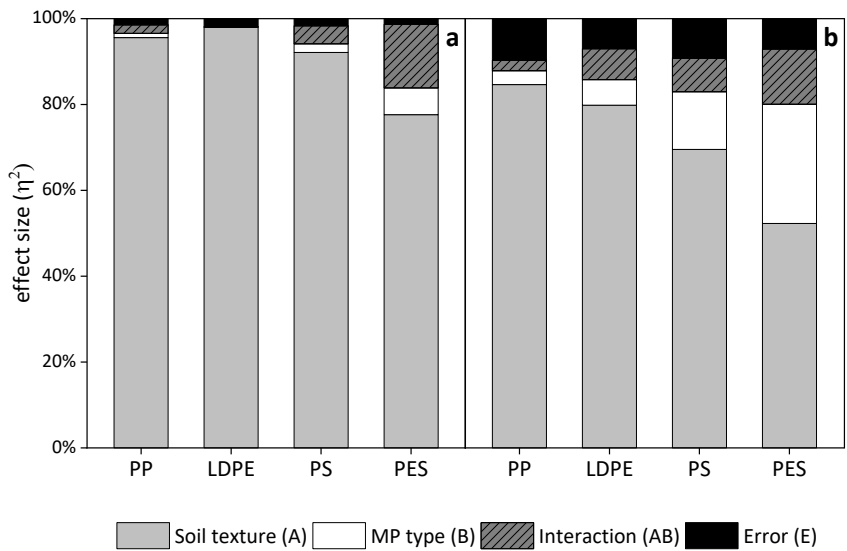


Fig. 2.2. Size of the effect (η^2) of soil and microplastic (MP) type on a) K_{sat} (cm d^{-1}), and b) K_{ns} (cm d^{-1}). $\eta^2(A)$ is the effect size of soil type, $\eta^2(B)$ of MP type, $\eta^2(AB)$ of the interaction of soil and MP type, and $\eta^2(E)$ the error. LDPE is low-density polyethylene, PP is polypropylene, PS is polystyrene, and PES polyester microfibers made of polyethylene terephthalate.

2.3.5. Soil water retention and other soil characteristics

Using the van Genuchten parameters (α , n , θ_s , and θ_r) obtained by fitting the van Genuchten function to the retention and hydraulic conductivity data from the evaporation experiment, it was possible to infer how the tested MPs affected the soil water characteristics. The measured soil retention data of the different soils including the replicates as well as the fitted van Genuchten model are shown in Fig. A.4 to A.6 in Appendix A. Additional characteristics, such as the water content at field capacity at $\log_{10} h = 1.8$ (θ_{FC} , $\text{cm}^3 \text{cm}^{-3}$), water content at the permanent wilting point at $\log_{10} h = 4.2$ (θ_{WP} , $\text{cm}^3 \text{cm}^{-3}$), and the plant-available water (θ_{PAW} , $\text{cm}^3 \text{cm}^{-3}$) were also calculated and are shown in Table 2.5. Based on the fitted mean soil hydraulic parameters (based on the four replicates) for the different soil/MP combinations, the retention curves were plotted in Fig. 2.3. For better visualization, the near-saturated part (0-10 cm suction) was plotted along the entire curve. As can be seen, the mean soil water retention curves showed in general only minor differences for the MP amendments, and significant differences to the control were detectable only at pressure heads $< \log_{10} h = 1$ (Fig. 2.3).

For the Luvisol at full saturation, θ_s ranked PES > control > LDPE > PP > PS, with the PES-amended Luvisol ($\theta_s = 0.513 \text{ cm}^3 \text{cm}^{-3}$) able to store $3\% \pm 1.5\%$ more and significantly the most water among the treatments ($p = 0.02$). In contrast, the Luvisol mixed with PP and PS showed significantly decreased θ_s ($p = 0.05$ for PP, and $p = 0.002$ for PS), meaning that the porosity of the Luvisol was significantly enhanced due to addition of microfibers, whereas due to addition of PP and PS particles the soil porosity was significantly decreased. It should be noted, though, that the fitted θ_s values are uncertain because the evaporation method fails to provide data in the near-saturated region ($h > -10 \text{ cm}$), and therefore the interpretation of the θ_s values should be done with caution.

Table 2.5. van Genuchten parameters fitted on the four replicates used in the evaporation method, and calculated water content at saturation (θ_s), which is equivalent to total porosity, residual water content (θ_r), tortuosity (λ) water content at field capacity (θ_{FC}), water content at wilting point (θ_{WP}), and plant-available water (θ_{PAW}) for the three different soils and four different microplastics (LDPE = low-density polyethylene, PP = polypropylene, PS = polystyrene, and PES = polyester microfiber comprised of polyethylene terephthalate) added at 0.4% w/w.

Soil	Treatment	α	n	θ_s	θ_r	λ	θ_{FC}	θ_{WP}	θ_{PAW}
-	-	kPa ⁻¹	-	cm ³ cm ⁻³	cm ³ cm ⁻³	-	cm ³ cm ⁻³	cm ³ cm ⁻³	cm ³ cm ⁻³
Luvisol	Control	0.023	1.229	0.499 ± 0.007	0.004 ± 0.003	-3.351	0.400 ± 0.012	0.086 ± 0.003	0.314 ± 0.011
	LDPE	0.018	1.300	0.493 ± 0.012	0.007 ± 0.006	-2.351	0.409 ± 0.021	0.089 ± 0.008	0.320 ± 0.017
	PP	0.024	1.287	0.492 ± 0.005 ^a	0.002 ± 0.003	-5.204	0.394 ± 0.006	0.090 ± 0.007	0.304 ± 0.011
	PS	0.022	1.289	0.481 ± 0.007 ^a	0.005 ± 0.005	-0.171	0.394 ± 0.011	0.087 ± 0.005	0.307 ± 0.009
	PES	0.029	1.280	0.513 ± 0.010 ^a	0.001 ± 0.002	-4.239	0.399 ± 0.010	0.092 ± 0.004 ^a	0.307 ± 0.009
	Control	0.024	2.894	0.385 ± 0.009	0.034 ± 0.014	0.414	0.193 ± 0.012	0.034 ± 0.014	0.160 ± 0.018
	LDPE	0.025	2.453	0.392 ± 0.002	0.036 ± 0.006	0.959	0.190 ± 0.010	0.036 ± 0.006	0.154 ± 0.015
	PP	0.025	2.060	0.378 ± 0.009	0.035 ± 0.020	2.451	0.204 ± 0.013	0.037 ± 0.017	0.167 ± 0.029
	PS	0.021	2.739	0.374 ± 0.008	0.041 ± 0.005	0.455	0.202 ± 0.002	0.041 ± 0.005	0.161 ± 0.007
	PES	0.022	2.388	0.401 ± 0.007 ^a	0.039 ± 0.009	0.678	0.223 ± 0.007 ^a	0.039 ± 0.009	0.184 ± 0.013
Chernozem	Control	0.005	1.541	0.517 ± 0.002	0.056 ± 0.010	-0.138	0.497 ± 0.003	0.087 ± 0.012	0.409 ± 0.015
	LDPE	0.004	1.638	0.509 ± 0.007	0.059 ± 0.006	-0.133	0.490 ± 0.006	0.089 ± 0.006	0.401

								± 0.009
PP	0.004	1.668	0.501 ± 0.007 ^a	0.064 ± 0.007	0.197	0.481 ± 0.008 ^a	0.090 ± 0.003	0.391 ± 0.008
PS	0.006	1.410	0.505 ± 0.013	0.035 ± 0.024	-0.139	0.479 ± 0.010 ^a	0.099 ± 0.021	0.380 ± 0.026
PES	0.008	1.428	0.520 ± 0.007	0.026 ± 0.013 ^a	-1.600	0.470 ± 0.005 ^a	0.102 ± 0.012	0.368 ± 0.014 ^a

^aSignificantly different from control ($p \leq 0.05$).

In the range of the steep decline of the retention curve, where soil drainage shifted from large pores to medium and small pores, no influence of MP particles on the retention curve could be detected. Only the Luvisol with added PES microfibers led to significantly elevated water contents at the wilting point at $\log_{10} h = 4.2$ ($+7\% \pm 4\%$, $p = 0.02$). However, the water content at field capacity and the plant-available water were not significantly affected by MPs at all. Furthermore, the PES-amended Luvisol exhibited $76\% \pm 62\%$ less, and therefore, the smallest θ_r among the treatments, statistically slightly above the critical p -value of 0.05 ($p = 0.056$).

The retention curve of the Albic Luvisol mixtures showed a later air entry compared to the loamy silt, and also a steeper slope in the mid-pressure head range ($\log_{10} h = 1.2$ -2.4), which can be also seen in the larger van Genuchten n values for the Albic Luvisol (mean n value for the control = 2.894) compared to the Luvisol (mean n value for the control = 1.229).

The water stored in Albic Luvisol therefore drained across a very narrow range of pressure heads, due to the uniform pore size distribution in this sandy soil. The mixture of sandy soil with microfibers enabled a higher water storage, characterized by a significantly increased porosity ($+4\% \pm 2\%$, $p = 0.03$), increased water content at high pressure heads ($<\log_{10} h = 1$), and at field capacity at $\log_{10} h = 1.8$ ($+15\% \pm 4\%$, $p = 0.008$), in combination with a positive effect on the plant-available water ($+15\% \pm 8\%$, $p = 0.05$).

The retention curve of the Chernozem exhibited a smoother curve progression, generally indicating higher θ_s (0.50 - $0.52 \text{ cm}^3 \text{ cm}^{-3}$), higher θ_{FC} (0.47 - $0.50 \text{ cm}^3 \text{ cm}^{-3}$), and higher θ_r (0.03 - $0.07 \text{ cm}^3 \text{ cm}^{-3}$) than the other two soil types. This suggests greater water retention compared to the Luvisol and Albic Luvisol, due to a higher proportion of fine particles. The retention characteristics of the various MP-Chernozem mixtures diverged at high pressure heads up to $\log_{10} h = 2.4$, after which they converged to a similar curve progression. The saturated water content of the Chernozem control soil ($0.52 \text{ cm}^3 \text{ cm}^{-3}$), and at the same time the soil porosity, was slightly reduced by $2\% \pm 1\%$ due to the

addition of LDPE and PS, and significantly reduced by $3\% \pm 1\%$ by the addition of PP ($p = 0.01$). Furthermore, the PP-amended Chernozem contained less water at suctions $< \log_{10} h = 1$. This can be also seen in the lower water content at field capacity ($-3\% \pm 1\%$, $p = 0.01$) and a slight reduction in plant-available water by $5\% \pm 2\%$ ($p = 0.07$). Additionally, in the Chernozem the PS particles affected θ_{FC} ($-4\% \pm 2\%$, $p = 0.02$) and plant-available water ($-7\% \pm 6\%$, $p = 0.08$) similarly negative. The addition of PES had no effect on θ_s of the Chernozem and therefore, on its porosity. However, PES microfibers in the Chernozem markedly reduced soil water retention at FC but also at the dry end. As a result, this adversely impacted θ_{FC} ($-5\% \pm 1\%$, $p = 0.009$) and θ_{PAW} ($-10\% \pm 3\%$, $p = 0.04$). Moreover, at the very dry end θ_r was $54\% \pm 23\%$ lower compared to the control ($p = 0.03$).

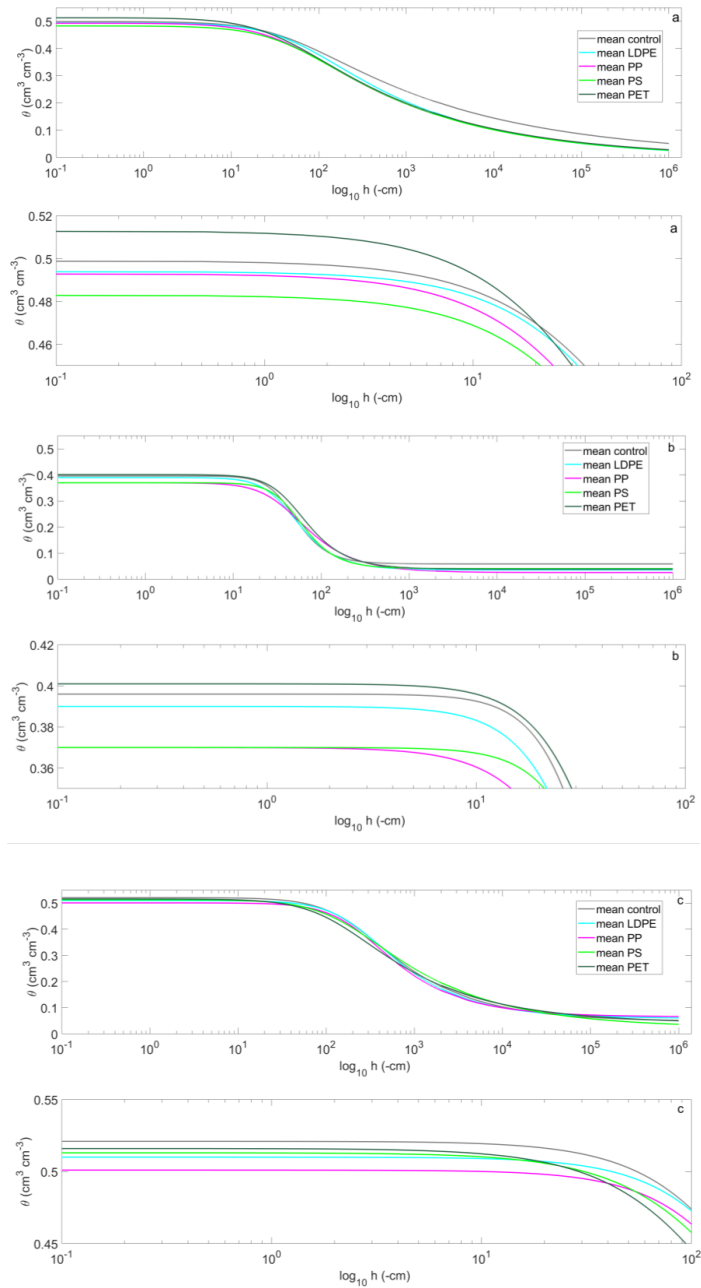


Fig. 2.3. Mean soil water retention curves for the a) Luvisol, b) the Albic Luvisol, and c) the Chernozem mixed with the four different microplastic (MP) amendments at 0.4% (w/w). LDPE is low-density polyethylene, PP is polypropylene, PS is polystyrene, and PES is polyester microfiber comprised of polyethylene terephthalate. The control is lacking the addition of MP. Significant differences in water content at given pressure heads from the control are marked with an asterisk (*) and were tested via one-sided, heteroscedastic t -test ($p \leq 0.05$).

2.3.6. Water-filled pore space (*WFPS*) and pore volume

To analyze the effect of the different MPs on the *WFPS*, the difference between the *WFPS* of the control and of the MP-amended soil was calculated (Fig. 2.4). As can be seen from Fig. 2.4, the largest relative difference in *WFPS* in the Luvisol was not found at full saturation or at low pressure heads, but in the range of field capacity at $\log_{10} h = 1.2$ to 2.4. Here, the *WFPS* for the PES and PS treatments deviated the most from the control, whereas PS showed significantly higher ($p = 0.05$) and PES non-significantly lower values compared to the control. Most strikingly, the course of *WFPS* over the entire pressure head range differed due to the addition of PES and PS in opposite ways. Parallel to the increase in the percentage of *WFPS* in the PS treatment, the proportion of *WFPS* in the PES treatment decreased, while both curves approached each other, converged in the area of the wilting point at $\log_{10} h = 4.2$ and then followed a similar course up to $\log_{10} h = 6$. At lower pressure heads, e.g., at $\log_{10} h = 4.8$ and 5.4, PES-amended soil had a significantly increased *WFPS* ($p = 0.05$). In contrast, LDPE and PP-amended soil showed no statistical differences to the *WFPS* in the control.

Unlike the declining trend of *WFPS* observed in PES-amended Luvisol, as described above, the addition of PES to the Albic Luvisol notably increased *WFPS* at field capacity ($\log_{10} h = 1.8$) by $11\% \pm 3\%$ ($p = 0.008$), reaching its peak. Similarly, PS in the Albic Luvisol led to a similar trend in *WFPS*, with significantly increased *WFPS* compared to the control ($+8\% \pm 2\%$, $p = 0.02$), and reaching its peak at field capacity. Interestingly, these two MP types evoked a similar trend in *WFPS* as a function of pressure head, unlike that of the Luvisol. Particularly, in the drier regions at lower pressure heads, PS in the Albic Luvisol effected a more pronounced increase in *WFPS* compared to the other MP species.

The *WFPS* in the Chernozem was most strongly influenced by the presence of PES microfibers. Compared to the control, *WFPS* was strongly reduced with decreasing pressure head up to $\log_{10} h = 2.4$, with significantly lower ($-16\% \pm 2\%$) *WFPS* at field capacity ($p = 0.03$). Above $\log_{10} h = 2.4$, *WFPS* in the Chernozem containing PES increased rapidly with decreasing pressure head, reaching its maximum at $\log_{10} h = 3.6$, thereafter dropping again and showing comparable values to the control around the wilting point at $\log_{10} h = 4.2$. In contrast, addition of the other MP species did not change the *WFPS* significantly.

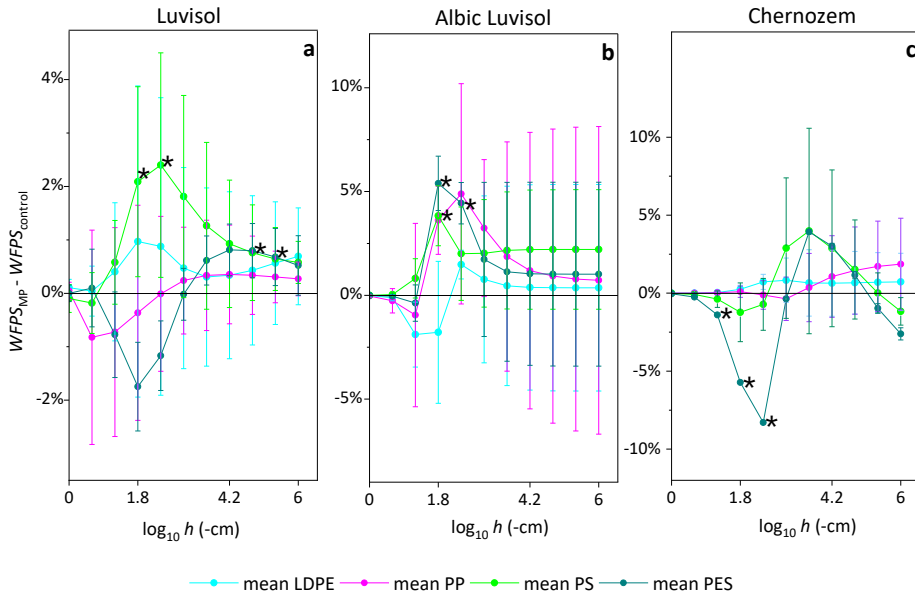


Fig. 2.4. Differences between the water-filled pore space ($WFPS$, %) calculated between volumetric water contents at saturation and given pressure heads ($\log_{10} h = 0, 0.6, 1.2, 1.8, 2.4, 3, 3.6, 4.2, 4.8, 5.4$, and 6 -cm) for the control and the microplastic amended a) Luvisol, b) Albic Luvisol, and c) Chernozem. LDPE is low-density polyethylene, PP is polypropylene, PS is polystyrene, and PES is polyester microfiber comprised of polyethylene terephthalate. Solid line marks zero, representing the control. Significant difference ($p \leq 0.5$) from the control is marked with an asterisk and was tested via one-sided, heteroscedastic t -test. Note that the y-axis are scaled differently for better visualization.

Total porosity, macroporosity, air capacity, and the plant-available water were calculated for the three soil types tested. The air capacity constituted 10-16%, 22-28% and 7-10% of the soil volume in the Luvisol, Albic Luvisol, and Chernozem, respectively, while the macroporosity ranged from 0.3-2.8% in the Luvisol, 0.2-2% in the Albic Luvisol, and 0.4-0.8% in the Chernozem. The plant-available water was the highest in the Chernozem (0.34 - 0.39 $\text{cm}^3 \text{ cm}^{-3}$), followed by the Luvisol (0.27 - 0.28 $\text{cm}^3 \text{ cm}^{-3}$), and the lowest in the Albic Luvisol (0.09 - 0.11 $\text{cm}^3 \text{ cm}^{-3}$). Adding the different MP species to the individual soil types significantly affected pore volumes with consequences for soil porosity, air capacity, and plant-available water (Fig. 2.5).

PS and PP particles significantly decreased the total porosity of the Luvisol by $4\% \pm 1\%$ ($p = 0.002$) and $1.4\% \pm 0.9\%$ ($p = 0.05$), respectively. Additionally, PS in the Luvisol reduced the air capacity ($-10\% \pm 9\%$, $p = 0.04$). Addition of LDPE particles led to a slight reduction of both air capacity and macroporosity. Oppositely, PES microfibers in the Luvisol significantly enhanced total porosity ($+3\% \pm$

1.5%, $p = 0.02$), air capacity (+11% $\pm$ 9%, $p = 0.02$), and macroporosity (+33% $\pm$ 23%, $p = 0.01$). Nonetheless, none of the MP types tested significantly affected plant-available water.

Similarly, PES microfibers led to a significant enhancement of total porosity in the Albic Luvisol (+4% $\pm$ 2%, $p = 0.03$), paired with a significantly reduced air capacity (-5% $\pm$ 2%, $p = 0.04$) and significantly higher plant-available water (+29% $\pm$ 11%, $p = 0.04$). PS amendment also led to significantly reduced air capacity (-7% $\pm$ 3%, $p = 0.02$), but did not affect total porosity.

Particularly the PES microfibers in the Chernozem significantly enhanced air capacity by 100% $\pm$ 33% ($p = 0.02$), which significantly reduced plant-available water by 13% $\pm$ 4% ($p = 0.03$). However, the increase in macroporosity by 384% $\pm$ 202% was only slightly significant due to the high variability between replicates ($p = 0.06$). Moreover, a reduced total porosity due to the presence of PP particles in the Chernozem was observed (-3% $\pm$ 1%, $p = 0.01$), while macroporosity or air capacity were not significantly affected. Nevertheless, PP in the Chernozem showed a slightly significantly reduced plant-available water by 5% $\pm$ 2% ($p = 0.08$).

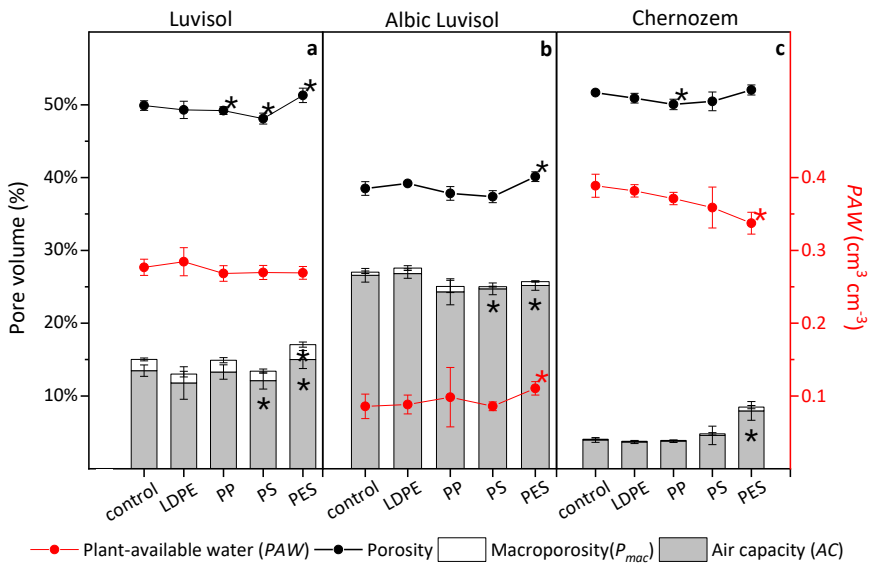


Fig. 2.5. Total porosity (assuming that total porosity equals θ_s), macroporosity, air capacity (all in %), and plant-available water ($\text{cm}^3 \text{ cm}^{-3}$) without (control) and with the addition of different microplastic types in a) the Luvisol, b) Albic Luvisol, and c) the Chernozem. LDPE is low-density polyethylene, PP is polypropylene, PS is polystyrene, and PES is polyester microfibers comprised of polyethylene terephthalate. Significant differences from the control are marked with an asterisk (*) and were tested via one-sided, heteroscedastic t-test ($p \leq 0.05$).

2.4. Discussion

We found that at the given mass concentration of 0.4% (w/w) our tested MPs significantly affected the hydraulic conductivity of the three soil types, not only as a function of soil type itself but also depending on whether the soils were initially dry, as characterized by K_{ns} , or saturated, as indicated by K_{sat} . Through the calculation of the effect size η^2 , the diverse effects on water flow arising from either soil type, MP type, or the interaction between MP and soil type could be discerned and ranked according to their extent in each soil condition, whether dry or saturated.

In general, the effect size η^2 clearly showed that soil type had the greatest influence on K_{sat} and K_{ns} . This is in line with the findings of Wang et al. (2023), who found that the impact of soil texture on hydraulic properties was significantly higher than that of MP concentrations or sizes. However, the soil type effect was generally smaller for K_{ns} than K_{sat} (Fig. 2.2). The relatively stronger MP-effect on K_{ns} was likely because the hydrophobicity of the dry MP particles and fibers was higher than that of the wet MP particles and fibers, and the greater interaction effect of PES microfibers and soil type on K_{sat} suggests that the wet microfibers alter water flow by interacting with the particular soil structure. These marked differences in the influence of MPs, in particular of PS and PES, on soil pore structure and thus on hydraulic properties of the investigated soils are discussed in more detail for each soil type in the following sections.

2.4.1. Microplastic effects in the Luvisol

In the Luvisol, the addition of PS particles and PES microfibers seemed to have antagonistic effects on soil hydraulic characteristics. While the addition of PES microfibers increased the porosity of the Luvisol to hold significantly more water and increased the percentage of air-filled pores, which are critical for root gas exchange, the addition of PS particles reduced the Luvisol's field capacity and the available air-filled pore volume (Fig. 2.5a).

The fact that PES microfibers at a concentration of 0.4% (w/w) would increase the water-holding capacity of a soil was already demonstrated by de Souza Machado et al. (2019). Presumably, the loamy silt texture of the dry Luvisol prevented a thorough compaction of the microfiber-soil mixture, consequently leading to better aeration, water infiltration, and increased water-holding capacity. The widening of the fine pore structure likely led to the creation of new pathways for the water through the saturated soil matrix, positively affecting K_{sat} (Fig. 2.1a). As reported previously, PES microfibers in soil led to altered physical soil properties (de Souza Machado et al., 2019; Ingraffia et al., 2022a). In these two studies, it was shown that the incorporation of PES microfibers led to an increased formation of newly formed aggregates. While Ingraffia et al. (2022a) reported this for a clay-rich Vertisol (-32%),

a loamy Entisol (-47%), and an Alfisol (-33%), in de Souza Machado et al. (2019) it remained unclear which soil type was used. As a result, the addition of PES microfibers to soil could promote the aggregation of soil particles into larger aggregates with larger pores between them. This increases the overall permeability of the soil, allowing water to percolate faster through the saturated soil matrix, which is in alignment with the increased K_{sat} values of the Luvisol in our experiment. However, it should be noted that this effect is most probably caused by the soil type-specific pore structure, as this phenomenon was not observed in our coarse-textured sandy Albic Luvisol (Fig. 2.1b). In addition, the Luvisol amended with PES microfibers held more water at the permanent wilting point ($\log_{10} h = 4.2$), but at complete dryness, the PES-treated Luvisol contained significantly less residual water within its pore space compared to all other MP-treatments and the untreated control Luvisol (Table 2.5). This also provides evidence for a more permeable soil structure in which water can be released more easily from soil pores at higher suction pressures.

In contrast, the presence of PS particles in the Luvisol reduced its porosity and thus its water-holding capacity and air-filled pore space (Fig. 2.5a), most probably due to blocking pores within the loamy silt soil structure, as also stated by Tötze et al. (2024). These authors used X-ray tomography and neutron imaging to show that clogging macropores or blocking effective water pathways with film fragments can significantly affect water transfer in the soil, resulting in uneven water distribution. By blocking pores, the soil's ability to absorb and to drain water is impaired, as reflected in the significantly enhanced proportion of water-filled soil pores in PS-amended Luvisol (Fig. 2.4a). The fact that PS in the Luvisol markedly decreased the soil porosity may have exerted the dominant influence on water infiltration into the dry Luvisol, as porosity and soil particle arrangement are critical factors affecting the interconnectedness of pores, determining the flow path of water, known as tortuosity (Gan et al., 2022). With decreased porosity, the impact of tortuosity on water flow becomes more pronounced (Gan et al., 2022). As porosity decreases with PS particles in the Luvisol, the pore clogging effect of PS in the Luvisol may have positively affected tortuosity, resulting in reduced water pathways and enhanced near-saturated conductivity (Fig. 2.1d). Additionally, LDPE and PP particles in the Luvisol also increased K_{ns} (Fig. 2.1d), while also reducing the Luvisol's porosity compared to the control (Fig. 2.5a).

This antagonistic behavior of MP particles and microfibers affecting the hydraulic conductivity and consequently the hydraulic properties of the loamy silt is also reflected in the K_{ns} -to- K_{sat} ratio. While the PES microfibers significantly reduced the K_{ns} -to- K_{sat} ratio, MP particles significantly enhanced it (Table 2.4). As MP particles in the Luvisol tended to block the soil pores, their effects were most pronounced in dry soil, while the pore-expanding properties of the microfibers were only effective under saturated conditions.

2.4.2. Microplastic effects in the Albic Luvisol

The presence of PS particles and PES microfibers in the loamy sand Albic Luvisol increased the water-filled pore space but reduced the air capacity. However, the considerable enhancement in porosity induced by the PES microfibers contrasts with the marked reduction caused by PS particles.

In the case of PES microfiber-amended Albic Luvisol, the increased porosity likely increased water storage within the pore volume, associated with increased water retention and plant-available water. The PES microfibers added to the Albic Luvisol obviously altered the soil structure in a beneficial way in terms of plant-available water. Conversely, the reduction in air-filled pore volume indicates a decrease in the available volume for root gas exchange and gas diffusion, which would negatively influence potential root respiration and soil microbial activity in the rhizosphere.

Interestingly, the presence of PES microfibers in the saturated Albic Luvisol affected K_{sat} negatively (Fig. 2.1b), completely opposite to the effect observed in the finer-textured Luvisol and Chernozem (Fig. 2.1a, and 2.1c). Moreover, K_{ns} was impaired as well in the PES microfiber-amended Albic Luvisol (Fig. 2.1e). The effect size η^2 revealed that when interacting with soil texture at water saturation, PES microfibers had the largest impact on K_{sat} (Fig. 2.2a). Conversely, in dry soils, the presence of PES microfibers alone exerted the greatest influence on K_{ns} (Fig. 2.2b).

On the one hand, it is conceivable that the PES microfibers interact with the soil structure, and therefore, with soil particles, likely due to soil aggregation (Lozano et al., 2021a), resulting in the formation of soil clods and impacting soil structure (Maddela et al., 2023). This would promote the upward movement of water during saturation, as reflected in the significantly increased water-filled pore space (Fig. 2.4b) and increased water-holding capacity (Fig. 2.3b). Moreover, it seems that due to addition of PES microfibers and subsequent soil aggregation, pores not only hold more water but also reduce water flow along the gravitational potential within the saturated soil pores by blocking potential pathways. On the other hand, the intertwining of hydrophobic microfibers in the expanded pore space could lead to a finer subdivision of the pores, increasing capillary forces in the dry Albic Luvisol. As a result, capillary rise increases, which counteracts the gravitational flow of water (Mantoglou & Gelhar, 1987), possibly leading to lower K_{ns} values for the Albic Luvisol.

In contrast, the Albic Luvisol containing PS particles exhibited reduced water-holding capacity compared to both the PES microfiber-amended and control Albic Luvisol, but also showing reduced drainage, as demonstrated by the increased residual water content (Table 2.5) and enhanced water-filled pore space (Fig. 2.4b). This phenomenon is likely attributed to obstruction of water flow paths by the MP particles, impeding water flow by acting as physical barriers both affecting water storage and drainage. The reduced water storage also suggests reduced soil porosity, i.e., reduced pore volume

and therefore enhanced attractive forces between water molecules and pore walls, thereby reducing K_{sat} , as observed for PS and PP.

However, PS particles in sandy soil not only reduced the K_{sat} , but also K_{ns} (Fig. 2.1b). The effect size η^2 revealed that the effect of PS on K_{ns} ranked highest amongst the three MP particle types (Fig. 2.2b). PS differed from the other particle types due to its higher density than water (1.05 g cm^{-3} , Table 2.2), which probably makes this MP type less buoyant during water infiltration compared to the other particle types (Table 2.2). This density variance could have allowed PS acting even more as a physical barrier against gravitational water flow.

In the sandy Albic Luvisol, only the LDPE particles influenced and reduced the K_{ns} -to- K_{sat} ratio (Table 2.4). This effect can be attributed to the fact that LDPE was the only particle type that did not impede water flow through the water-saturated soil matrix, probably due to its relatively narrow particle size distribution, which had no influence in the coarse texture. In addition, the Albic Luvisol amended with LDPE exhibited greater water-holding capacity compared to the control or the PS-amended Albic Luvisol (Fig. 2.3b), which reduced water storage and porosity.

2.4.3. Microplastic effects in the Chernozem

Only in the fine textured silty loam Chernozem, an increase in macropore volume with the incorporation of PES microfibers was detectable. Many other studies refer the effects of MP on hydraulic properties to the increase in macropores (Guo et al., 2022; Shafea et al., 2023). While this might be true for some textures, as seen for our Chernozem, this statement cannot be generalized, as no significant changes within our Luvisol or Albic Luvisol were found. The enhanced macroporosity within the microfiber-amended Chernozem was accompanied by an overall increased air-filled pore volume compared to the control (Fig. 2.5c). Although no difference in porosity was determined, the microfibers within the fine-structured Chernozem facilitated an increased drainage with significant negative consequences for plant-available water. Microfibers notably enhanced the permeability of the Chernozem, with pore water being bound less effectively to the pore walls, underscored by the significantly reduced residual water content (Fig. 2.3c) and water-filled pore space at field capacity (Fig. 2.4c).

Consequently, the presence of PES microfibers likely rearranged soil aggregates, resulting in locally increased pore sizes without affecting overall porosity. As proposed by Nimmo et al. (1997), structural alterations of a soil significantly changes not only its hydraulic conductivity, but also its water retention. As opposed to smaller pores, enlarged pores have altered physical forces, e.g., water surface tension, adhesion, capillary tension and meniscus formation, governing capillarity, and therefore,

water retention. Nonetheless, the alteration of soil structure positively affected K_{sat} due to the enhanced water permeability (Fig. 2.1c). However, water flow through the dry soil matrix was negatively affected (Fig. 2.1f). As K_{ns} also depends on the structure and connectivity of the pore space within the soil matrix (Daneshian et al., 2021), potential explanations must reflect on how the PES microfibers potentially rearranged the structure of the Chernozem, leading to an impaired water flow within the dry soil. In this context, it is plausible that the PES microfibers hindered the formation of coherent pore clusters, which could have increased the tortuosity in the Chernozem with the effect of extended flow paths for water and reduced K_{ns} .

The MP particles tested in this study, in particular PP, influenced the Chernozem's total porosity negatively without affecting the air-filled pore space or the macroporosity (Fig. 2.5c). As a result, the PP- and PS-amended Chernozem could store less water, accompanied by slightly reduced plant-available water. The MP particles in the Chernozem most probably blocked the pores, preventing water from entering the pore space, especially in the dry matrix. Furthermore, the Chernozem had a relatively high organic carbon content (2.06%), twice that of the Luvisol. As demonstrated by Ivanic et al. (2023), attractive hydrophobic interactions between hydrophobic PS particles (1 μm) and soil organic matter could have induced a more static behavior of MP particles in soil, leading to pore blockage and impeding water flow.

2.5. Conclusion

The findings pertaining to the impact of various MP types on the hydraulic characteristics of different soil types with various soil textures have provided valuable insights into the diverse effects of MPs on key hydraulic characteristics in the different soils. It is evident that the hydraulic properties of a given soil are not solely influenced by the surface properties of the MP type, but also by factors like its shape (fiber or particle), size and the interaction of the MPs with the surrounding soil structure. Our study highlights the complex interplay between MPs and soil characteristics, with the ecological implications (e.g., for root water uptake and crop growth) to be assessed. However, other MP concentrations and soil textures are likely to have different effects, which could be positive, negative, or insignificant.

To provide more accurate insights and future projections regarding the impact of specific MPs on the hydraulic properties of soils with different textures, we recommend focusing on intact (undisturbed) soils. Regarding the assessment of ecological impacts of MPs, it should be noted that MP concentrations of 0.4% (w/w) or similar used in our and many other studies are extraordinarily high, as much lower MP concentrations have been reported for today's agricultural soils, e.g., <0.002% (w/w) for arable land treated with biosolids for 10 years (Corradini et al., 2019). However, higher MP

concentrations might be realistic for heavily contaminated soils near disposal sites or at individual hotspots in the soil.

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CHAPTER 3 – Microplastics affect belowground performance of winter wheat and microbial networks in the rhizosphere

Based on:

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Abstract

Microplastics (MPs) are increasingly recognized as environmental pollutants affecting soil ecosystems and plant development, but their specific impacts on plant-soil-microbiome interactions remain unclear. This study investigated the effects of different MP types, including particulate PP, LDPE, PS, and PES, individually on early-stage winter wheat root growth, nutrient concentrations, soil conditions and the rhizosphere microbiome using rhizotrons. Our findings revealed that PP and PES MPs significantly reduced root length. We observed increased soil ammonium concentrations with the addition of all MPs, likely due to reduced nitrifying bacterial activity, while PS MP enhanced soil nitrate concentration, potentially reflecting impaired root absorption. No significant effects of MPs on plant potassium (K) and phosphorus (P) uptake were observed. Despite changes in soil nutrient levels, we found no significant alterations in leaf growth, nutrient uptake, or chlorophyll content, indicating that root systems, though affected by MPs, remained sufficiently functional to support shoot development. Microbial co-occurrence analysis revealed two distinct microbial clusters in the rhizosphere, confirming modular microbial structure. Among all treatments, LDPE exposure induced the most pronounced increase in network complexity, as reflected by higher edge density, clustering coefficient, and natural connectivity compared to the control. Many amplicon sequence variants (ASVs), e.g., members of *Vicinamibacteraceae*, *Pseudolabrys* and *Nitrospira japonica*, showed an increased degree and eigenvector centrality in the LDPE treatment, suggesting enhanced co-occurrence and higher influence within the microbial community structure. In contrast, the PES and PP treatments resulted in lower average node degree and reduced clustering. The PS treatment showed low effects on the microbial co-occurrence networks. Overall, our data indicate that MP-induced stress can alter microbial network topology, reduce community integration, and drive shifts in ecologically important keystone taxa in the rhizosphere, which are linked to changed pattern of the availability of plant derived nutrients. Our findings highlight the complex and variable influence of MPs on soil-plant interactions.

3.1. Introduction

The presence of MPs, i.e., plastic particles and fibers smaller than 5 mm, in soil has raised serious concerns due to their persistence and tendency to accumulate in agricultural systems. Agricultural soils, in particular, act as a major sink for MPs, with concentrations having increased markedly between 1966 and 2022 (Cusworth et al., 2024). The application of both organic (e.g., farmyard manure and compost) and inorganic fertilizers further elevates MP loads in comparison to unamended soils (Rana et al., 2023).

Although numerous studies have reported that MPs influence plant performance, the mechanisms underlying these effects remain poorly understood. Plants have been shown to respond to MP contamination with both growth inhibition and stimulation, depending on the context. For instance, some studies observed reduced biomass or nutrient uptake, whereas others found enhanced root and shoot development under specific conditions, such as drought (Lozano & Rillig, 2020; Liu et al., 2022b; Neubert & Brüggemann, 2025). However, the reasons for such divergent outcomes remain elusive because the processes linking MPs to plant responses occur through multiple, often interacting pathways within the soil environment, particularly in the rhizosphere.

From a physical perspective, MPs can modify soil structural and hydraulic properties, including porosity, aggregation, and water retention capacity, thereby influencing root penetration and water availability (de Souza Machado et al., 2018; Ingraffia et al., 2022a; Ingraffia et al., 2022b). These structural changes can either hinder or facilitate plant growth depending on soil texture, MP type, and environmental conditions.

From a chemical viewpoint, MPs have been reported to affect nutrient dynamics in the rhizosphere. For example, polyvinylchloride (PVC) particles may physically restrict root access to nutrients, leading to lower nitrogen (N) and phosphorus (P) uptake (Liu et al., 2022b). Conversely, improved aeration and nutrient mobility resulting from MP fiber incorporation have been linked to enhanced plant performance under drought stress (Lozano & Rillig, 2020). These findings suggest that MPs can alter nutrient availability both directly and indirectly through changes in soil structure and chemistry.

Equally important are the biological effects of MPs on soil microbial communities, which play a central role in nutrient cycling and plant health. Strong MP-induced shifts in microbial diversity and community composition have been reported, including alterations in bacterial and fungal functional groups that may feed back on plant performance (Zhou et al., 2021; Kublik et al., 2022; Aralappanavar et al., 2024; Lozano et al., 2024). Recent syntheses indicate that many of the observed MP effects on

plants likely stem from context-dependent processes in the rhizosphere – such as changes in microbial composition, enzyme activities, or symbiotic interaction – which are triggered or modulated by MPs (Chen et al., 2025b). In addition, MP incorporation alters not only bacterial communities (Zhou et al., 2021), but also fungal functional groups such as pathogens, saprotrophs, and mutualists (Lozano et al., 2024).

However, these fragmented approaches focussing either on chemical, physical or biological properties limit our ability to determine the dominant pathways through which MPs ultimately affect plant growth and health. To overcome this limitation and directly link an altered plant performance with MP-induced changes in the soil chemistry, physics and biology in the rhizosphere, experimental systems are needed that allow plants to be grown under controlled conditions with direct observation of root development, while also enabling easy access to rhizosphere soil sampling. These requirements can be met with plant root observation chambers, called rhizotrons. Rhizotron experiments offer several advantages, such as i) non-destructive in situ observation and monitoring of root growth and development over time, ii) control of factors like soil moisture, nutrient availability, and compaction to study their effects on root growth, iii) study of root-soil interactions, such as root penetration of different soil layers, rhizosphere processes, and the formation of root-soil mucilage complexes (Lärm et al., 2023), and lastly iv) validation of models where data obtained from rhizotron experiments can be used to validate and improve root growth and soil-plant-atmosphere continuum models, enhancing our understanding of root system functioning and its implications for crop productivity (Peters et al., 2023). While rhizotrons have limitations in terms of scale and potential artefacts due to the narrow width of the growth compartment, they remain a valuable tool for studying root traits and processes that are difficult to observe under field conditions.

Here, we investigated MP effects on soil physical (pH, water availability), chemical (nutrient availability) and biological (bacterial diversity) parameters, and linked those data to plant growth responses of winter wheat at an early growth stage in a rhizotron experiment to better pinpoint MP-induced plant-soil-microbiome interactions in the rhizosphere affecting plant performance. We tested four different MP types – LDPE particles, PP particles, PS particles, and PES microfibers – separately, to disentangle material-dependent effects on the microbiome.

3.2. Materials and methods

3.2.1. Soil

The soil used for the experiment was a Luvisol characterized by a loamy silt texture. The Luvisol soil was taken in June 2021 from an agricultural field at the agricultural research station Campus Klein-Altendorf of the University of Bonn, Germany (50.613614°N, 7.000713°E) from 0-20 cm depth. The soil was air-dried and sieved to 2 mm. The basic soil characteristics of the soil are listed in Table 3.1. The method for the determination of the volumetric water content at saturation is shown in the Appendix B.

Table 3.1. Characteristics of experimental soil, including soil total nitrogen (N_t), soil organic carbon (C_{org}), plant-available mineral nitrogen (N_{min}), phosphate (P_{av}), and potassium (K_{av}) content.

Soil type	Texture	Sand 50-2000 μm	Silt 2-50 μm	Clay <2 μm	pH $CaCl_2^a$	N_t	C_{org}	N_{min}	P_{av}	K_{av}	VWC ^b
-	-	%	%	%	-	%	%	$\mu g\ g^{-1}$	$\mu g\ g^{-1}$	$\mu g\ g^{-1}$	$cm^3\ cm^{-3}$
Luvisol	Loamy silt	8	77	15	6.5	0.15	1.00	168	20	116	0.50

^a $CaCl_2$ = calcium chloride solution, ^bVWC = volumetric water content at saturation.

3.2.2. Microplastics

Based on results on MP pollution of German agricultural soils from the literature (Piehl et al., 2018; de Souza Machado et al., 2019), the following types of MP were considered significant for further investigation: LDPE, PP and PS particles, and PES microfibers. Detailed information about the MPs used in the experiment, including their particle sizes, microfiber lengths and characterization methods, can be found in Chapter 2, Section 2.2.2.

3.2.3. Experimental design

The rhizotrons had a size of 45 cm x 30 cm x 3.5 cm (height x length x width) and were covered with a transparent Plexiglas lid on one side, enabling root growth observation (Fig. 3.1). Before filling the rhizotrons, air-dried Luvisol soil was mixed with the respective MP type at 0.4% w/w (Rillig et al., 2019; Rillig & Lehmann, 2020). The control soil without addition of MP was processed in the same way. All rhizotrons were gradually filled with soil and stepwise compacted to reach a dry soil bulk density of $1.2\ g\ cm^{-3}$. Five replicates were prepared for each MP treatment and the control.

After the soil-filled rhizotrons had been watered and pre-incubated in a climate chamber at controlled conditions for one week (day/night, 16/8 h, $T = 22/10^\circ C$; humidity = 60%), seeds of winter

wheat (*Triticum aestivum*, var. Nordkap, SAATEN-UNION GmbH, Isernhagen, Germany) were germinated in the dark at 10°C on a moist paper towel for 24 h and then placed 2 cm deep into the soil exactly in the center of each rhizotron. The plants were grown for 28 days, reaching principal growth stage 2 (tillering) according to the BBCH scale (Meier, 2018), with 9 to 15 visible tillers. During the growth period, the rhizotrons' Plexiglas lids were covered with customized windshield covers for cars and the rhizotrons were inclined by 45° to let roots grow in the dark close to the lids. While no fertilization of the soil was applied, the plants were allowed to grow under well-watered conditions. To compensate for soil water loss, rhizotrons were irrigated twice per week from above with irrigation water to maintain the soil water content at 25% w/w gravimetric water content (= 60% gravimetric WHC), as determined by weighing the rhizotrons. In addition, shoot length was recorded over the course of growth period.

Soil samples were taken from each rhizotron before and after the growth period of 28 days. Since sampling required the opening of the lids, potentially destroying roots directly adhered to the lids, soil sampling during the growth period was avoided. Soil sampling was achieved by using stain-less steel cylinders (10 cm x 1 cm length x diameter) inserted horizontally along the whole length of the root system. It was thereby differentiated between soil samples in direct vicinity of roots (rhizosphere soil) and free of roots (bulk soil). Soil and rhizosphere samples were stored at -20°C for soil chemical and for microbiome analysis.

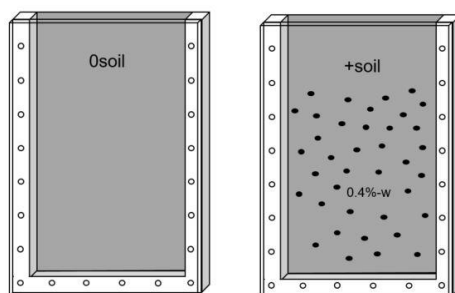


Fig. 3.1. Model of two rhizotrons used as plant growth chambers. The left rhizotron shows the control (0soil) without the addition of microplastic (MP) and the right one shows an MP treatment with 0.4% (w/w) of MP addition in soil (+soil).

3.2.4. Soil analysis

For each soil sample, two extractions were performed: 1.) with 0.01 M CaCl_2 solution for plant-available mineral N content ($N_{\min}$) analysis (VDLUFA, 2016), and 2.) with calcium acetate/calcium lactate solution (CAL) for analysis of plant-available potassium (K_{av}) and plant-available phosphate (P_{av}) based on VDLUFA (VDLUFA, 2016). The respective pH values of soil samples were measured in the 0.01 M CaCl_2

extracts for 5 min each with a pH meter (Multi 3630 IDS, Xylem Analytics Germany GmbH, Weilheim, Germany). The method for the determination of the volumetric water content at saturation as well a detailed description of the above-mentioned methods is shown in Appendix B.

3.2.5. Root analysis via WinRHIZO

Root systems, together with the surrounding soil, were removed from the rhizotrons and placed in plastic bags until further processing. Root fragmentation during transfer was considered negligible, as root length was determined by scanning root images using Epson Perfection LA2400 scanner and the WinRHIZO image analyzing system (Regent Instruments Inc., Québec, Canada), quantifying individual fragments, which were summed up to total root length. All root fragments were washed from soil with a shower head over a 2-mm sieve. Finally, roots were collected from the sieve and were stored in 50% v/v ethanol/water until roots were scanned (EPSON, Expression, 12000xl, Meerbusch, Germany) and analyzed. A batch analysis of all scans was accomplished with the parameters defined in Appendix B Table B.1. The diameter range of harvested roots was determined from stereomicroscopic images (Appendix B: Fig. B.1), ranging from 0-0.4 mm for lateral roots, and from 0.4-1 mm for primary roots. Total root surface area was used to calculate soil nutrient content per root surface area (RSA).

3.2.6. Leaf pigment analysis

Leaf pigments from 28 days old winter wheat shoots were analyzed as suggested by Lichtenthaler & Wellburn (1983). Briefly, portions of 20 mg of dried (at 60°C, 7 d) and ground (ball mill, model type MM200, Retsch GmbH, Haan, Germany) winter wheat leaves were weighed into 5-mL Eppendorf vials in triplicate and mixed with 5 mL of an acetone-water mixture (4:1 v/v) (acetone: analysis grade $\geq 99.8\%$, Honeywell International Inc., Offenbach, Germany). Vials were vortexed for 5 s and shaken for 90 min at 200 rpm in the dark, before they were centrifuged at $1823 \times g$ (2800 rpm) for 15 min. Each of the supernatants (2 mL) were transferred into cuvettes and measured photometrically (Thermoscientific Genesys 50, Thermofischer Scientific, Darmstadt, Germany) at 470, 646 and 663 nm against a blank of the prepared extraction medium. Equations to calculate the amount of chlorophyll-a (*chl_a*), chlorophyll-b (*chl_b*) and the carotenoid (*car*) content of the extracts are shown in Appendix B.

3.2.7. Microbiome analysis

3.2.7.1. DNA extraction and library preparation

Genomic DNA was extracted from frozen rhizosphere soil using the NucleoSpin® Soil Kit (Macherey-Nagel, Düren, Germany) with SL1 buffer for lysis. An extraction without soil was included as negative

control. DNA quantity was measured using the Quant-iT PicoGreen dsDNA Assay Kit (Thermo Fisher Scientific, Darmstadt, Germany). Bacterial communities were assessed following the “16S Metagenomics Sequencing Library Preparation” protocol (Illumina, San Diego, CA, USA) and quality guidelines (Schöler et al., 2017). The v4 region of the 16S rRNA gene was amplified using primers 515F and 806R in 25 µl reactions containing 10 ng DNA, 0.5 µl of each primer (10 pmol), 2.5 µl 3% BSA, 12.5 µl NEBNext High-Fidelity 2x PCR MasterMix (New England Biolabs, Frankfurt am Main, Germany), and DEPC-treated water. Products were purified using MagSi NGSprep Plus beads (Steinbrenner, Germany) and quantified with a Fragment Analyzer 5200 (Agilent Technologies, Santa Clara, CA, USA) using the NGS Fragment Kit (1-6000 bp). Indexing PCR (25 µl) contained 10 ng purified amplicon, 2.5 µl of each indexing primer (Nextera® XT Index Kit v2 Set C or D; Illumina, San Diego, CA, USA), 12.5 µl NEBNext High-Fidelity 2x PCR MasterMix, and 6.5 µl DEPC-treated water. Indexed amplicons were purified, quantified, diluted to 4 nM, pooled equimolarly, and sequenced paired-end on a MiSeq® using the MiSeq® Reagent kit v3 (600 cycles).

3.2.7.2. Processing and statistical analysis of the sequencing data

Sequences were processed on Galaxy (Afgan et al., 2016). Reads were trimmed to ≥ 50 bp with Cutadapt (Martin, 2011), quality checked with FastQC (Andrews, 2010), and denoised with DADA2 (Galaxy Version 1.30.0) (Callahan et al., 2016) using: trim = 20 bp from 5' end, truncLen = 240/200 (F/R), max EE = 3/4. Taxonomy was assigned using SILVA v138.1 (99%) (Quast et al., 2013). Amplicon sequence variants (ASVs) present in controls, unassigned, mitochondrial, chloroplast, or singleton reads were removed. The raw sequence data obtained in this study are deposited in the short read archive of NCBI under accession number PRJNA1422079. Analyses were conducted in R 4.3.1 (R Core Team, 2023). Alpha diversity was calculated using species richness based on ASV number. Beta diversity was analyzed via unweighted and weighted UniFrac distance matrix. For statistical purposes, the Kruskal-Wallis test, the Wilcoxon-rank sum test, and PERMANOVA with Benjamini-Hochberg *p*-value correction for multiple comparisons were used. Plots were created in R using ggplot2, ggpubr and metacoder. Venn diagrams for core microbiome analysis were made using Venny (Oliveros, 2007) with a minimum ASV threshold of 75% (ASV must be present in 75% of the samples to be considered for the group). Microbial co-occurrence networks were inferred using Pearson correlations in NetCoMi v1.1.0 (Peschel et al., 2021), retaining ASVs present in $\geq 4/5$ replicates per treatment (control 1628 ASVs, LDPE: 1544 ASVs, PES: 1596 ASVs, PP: 1852 ASVs, PS: 1582 ASVs). Data were clr-transformed, zeros replaced via “multRepl”, sparsified at 0.3 threshold, and converted to dissimilarities using the “signed” method. Network modules were identified with the cluster_fast_greedy algorithm. Visualizations were limited to the top 100 ASVs. To relate microbial community composition to plant function the metadata were merged and analyzed in R using the packages phyloseq, vegan, tidyverse

and ggplot2. An environmental fitting analysis (*envfit*) was applied to test the correlation between microbiome and plant-available nutrients. Significant variables ($p < 0.05$) were visualized as scaled vectors in a PCoA biplot.

3.2.8. Statistical analysis of the soil and plant data

All plant growth parameters, soil nutrient composition, and leaf characteristics were tested for statistical differences between wheat plants exposed to different MP types and the control by conducting two-way ANOVA and paired, heteroscedastic t-test.

3.3. Results

3.3.1. Root analysis

The roots harvested from the individual rhizotrons (Fig. 3.2) ranged between 1200 and 1700 cm total length. Of these, lateral roots accounted for 87-92%. Root growth significantly decreased by $31\% \pm 20\%$ ($p = 0.02$) in PP-amended soil. Soil mixed with PES microfibers resulted in a weakly significant reduction in root growth of $29\% \pm 35\%$ ($p = 0.09$) compared to the control.

Gravimetric (Fig. B.2a) and length measurements of shoots (Fig. B.2b), nitrogen content in leaves (Fig. B.2c and B.2d), leaf pigment analysis (Fig. B.3), and shoot growth over time (Fig. B.4) showed no significant differences between +MP treatments and the control.

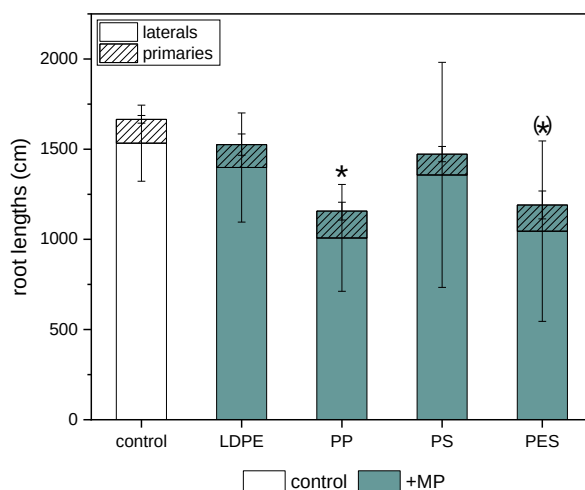


Fig. 3.2. Root length of winter wheat plants grown in rhizotrons for 28 days measured via WinRHIZO. “+MP” indicates microplastic (MP) addition to soil, while no MP was added to control. The following MP types were added: LDPE = low-density polyethylene, PP = polypropylene, PS = polystyrene, PES = polyester microfiber. Hatched areas of the bars represent the portion of primary roots (0.4-1 mm in width), and unhatched areas the portion of lateral roots (0-0.4 mm in width). Whiskers represent the standard deviation of five replicates. The asterisk indicates significant difference ($p \leq 0.05$), while the asterisk in parentheses shows a weakly significant difference ($0.05 \leq p \leq 0.1$) to the control, which was tested with a one-sided, heteroscedastic *t*-test.

3.3.2. Soil nutrient analysis

3.3.2.1. Root surface area-based soil nutrient availability

Soil nutrient concentrations, i.e., nitrate (NO_3^-), nitrite (NO_2^-), ammonium (NH_4^+), K_{av} , and P_{av} , on a weight basis (in $\mu\text{g g}^{-1}$) are shown in Appendix B, Fig. B.5. They were used to calculate plant-available nutrients per RSA (Fig. 3.3). This provided a clearer picture of nutrient availability to individual root systems, which varied in length and surface area between different treatments and replicates.

For NO_3^- per RSA, only soil containing PS particles showed a weakly significant increase compared to the control ($+96\% \pm 110\%$, $p = 0.09$). The NH_4^+ per RSA in bulk soil was significantly higher in the presence of PP ($+71\% \pm 65\%$, $p = 0.04$), PS particles ($+93\% \pm 67\%$, $p = 0.02$), and PES microfibers ($+126\% \pm 80\%$, $p = 0.01$), with a weakly significant increase observed in soil containing LDPE particles ($+42\% \pm 42\%$, $p = 0.05$). In the rhizosphere soil, NH_4^+ per RSA was weakly significantly enhanced in the presence of PS particles ($+111\% \pm 105\%$, $p = 0.06$). The NO_2^- per RSA showed a weakly significant increase with PP particles in the rhizosphere soil ($+101\% \pm 143\%$, $p = 0.1$) and with PES microfibers in the bulk soil

(+167% $\pm$ 221%, $p = 0.09$). Additionally, K_{av} per RSA increased in both rhizosphere and bulk soil with PP particles by 98% $\pm$ 132% ($p = 0.09$) and 74% $\pm$ 97% ($p = 0.09$), respectively, and in the rhizosphere soil with PES microfibers (+91% $\pm$ 107%, $p = 0.07$). The P_{av} amount per RSA showed no significant difference among the treatments towards the control.

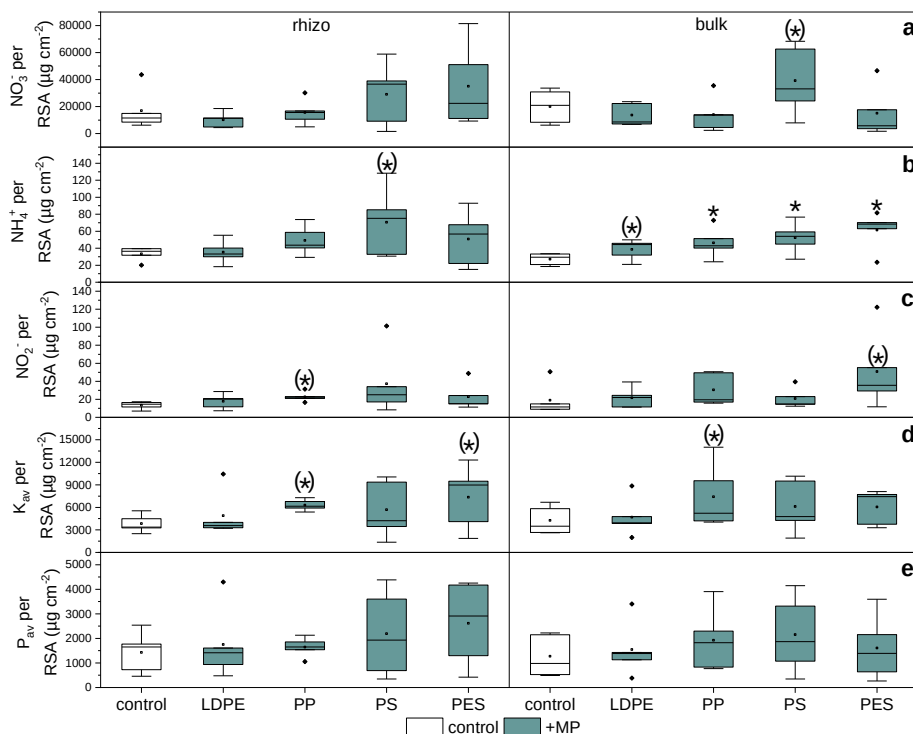


Fig. 3.3. Nutrients per root surface area (RSA) (in $\mu\text{g cm}^{-2}$) of plant-available a) nitrate (NO_3^-), b) ammonium (NH_4^+), c) nitrite (NO_2^-), d) potassium (K_{av}) and e) phosphate (P_{av}) in rhizosphere (rhizo) and bulk soil after winter wheat was grown for 28 days in rhizotrons. “+MP” indicates microplastic (MP) addition to soil, while the control represents soil without MP addition. The following MP types were added to the soil: LDPE = low-density polyethylene particles, PP = polypropylene particles, PS = polystyrene particles, PES = polyester microfibers. Boxes are the interquartile range (IQR) from the 25th to 75th percentile, lines inside boxes indicate the median, white squares indicate the mean values, upper and lower whiskers are 1.5 times the IQR and diamond-shaped markers are outliers ($n = 5$). Asterisks show significant differences ($p \leq 0.05$) and asterisks in parentheses show weak significant differences ($0.05 \leq p \leq 0.1$) to the control, tested with one-sided, heteroscedastic t -test.

3.3.2.2. Statistical evaluation of soil nutrient availability and root uptake

A two-way ANOVA was conducted to examine the effects of the root zone (rhizosphere soil) and the type of MP on soil nutrient availability for roots (Appendix B: Table B.3). Among the tested nutrients

per RSA, significant effects were only found for NO_3^- and NH_4^+ per RSA. The type of MP affected the amount of NO_3^- per RSA, with an effect size of 20% ($F(4, 40) = 2.76, p = 0.04$). The highest amount of NO_3^- per RSA in soil was observed for the treatment with PS particles (39 mg cm^{-2}), while for all other MP (LDPE, PP and PES) the soil NO_3^- availability was not significantly different from the control.

NH_4^+ availability was significantly affected by MP addition. The effect size of MP on the NH_4^+ content was 25% ($F(4, 40) = 3.65, p = 0.01$). In the control, the rhizosphere soil had 23% more NH_4^+ per RSA compared to bulk soil. Significantly enhanced NH_4^+ concentrations (42-126%) were measured in the bulk soil of all MP treatments compared to the control.

Plotting K_{av} versus P_{av} revealed a high correlation for the +MP treatments ($R^2 = 0.77$) and for the control ($R^2 = 0.83$) (Fig. 3.4). These high correlations suggest simultaneous root uptake of both nutrients (P_{av} and K_{av}). The linear fit for the +MP treatments had a steeper slope compared to the control, indicating higher nutrient availability per RSA and presumably less uptake of P_{av} and K_{av} . In contrast, the plot of NO_2^- versus NH_4^+ showed weak correlations for both +MP ($R^2 = 0.43$) and the control ($R^2 = 0.05$) (Appendix B: Fig. B.3), as well as the plot of NO_3^- vs. NH_4^+ (R^2 for +MP = 0.08 and R^2 for control = 0.54) (Appendix B: Fig. B.4), indicating a weak relationship between these parameters. Soil nutrient analysis also revealed a significant increase in NH_4^+ in the bulk soil of +PP, +PS, and +PES (Fig. 3.2b), likely due to low nitrification rates to NO_2^- , which is further supported by the low correlation between the two N_{min} species.

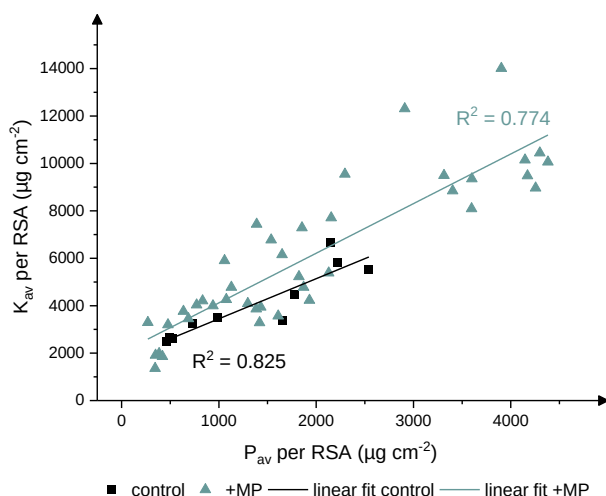


Fig. 3.4. Plant-available potassium (K_{av}) versus plant-available phosphate (P_{av}) content in soil per root surface area (RSA) (in $\mu\text{g cm}^{-2}$) in microplastic-amended soil (+MP, dark cyan triangles) and the control (black squares) without MP. Soil nutrients were analyzed after winter wheat was grown for 28 days in rhizotrons.

3.3.3. Microbiome analysis

3.3.3.1. Sequencing quality and diversity analysis

Sequencing of the generated amplicons resulted in 44,905 to 143,559 reads per sample after quality filtering. To compare samples without statistical bias, a subsampling strategy of 44,905 reads per sample was performed, reflecting the lowest observed read number. In total 15853 ASVs were detected. Rarefaction analysis indicated that the resulting sampling depth was sufficient for further analysis of all samples (Fig. B.8). Alpha diversity analysis revealed no significant differences between bulk soil and rhizosphere samples ($p = 0.652$), nor between the different MP treatments ($p = 0.721$) (Fig. 3.5a, Appendix B: Table B.3). These results were confirmed by Beta diversity analysis (Fig. 3.5b).

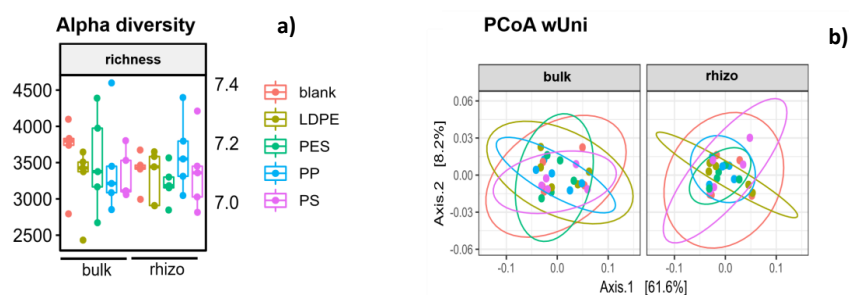


Fig. 3.5. Community structure of bacteria colonizing bulk soil (bulk) and rhizosphere (rhizo); a) Box plot of ASV richness in bulk soil and rhizosphere. Statistical analysis was performed using Kruskal-Wallis and Wilcoxon Rank-Sum test with Benjamini-Hochberg correction for multiple comparisons; b) PCoA plot of weighted UniFrac distances including all MP treatments. Statistical analysis was performed using PERMANOVA with Benjamini-Hochberg correction for multiple comparisons.

3.3.3.2. Comparative analysis of major ASVs across treatments

For the core microbiome analysis, all ASVs with at least 0.1% reads in 75% of the samples per group were considered. This revealed 92 ASVs in total, with 50 ASVs shared by all groups, representing the core microbiome (Fig. 3.6). Core ASVs primarily belong to the phyla Crenarchaeota, Actinobacteria, Acidobacteria, and Proteobacteria. 27 ASVs were exclusively detected in MP-treated samples, e.g. ASVs typical for *Lysobacter* and *Sphingomonas*. Unique ASVs were detected for PES, PP, and PS. For PES-treated samples, two ASVs of *RB41* (Acidobacteriota), an unclassified member of Vicinamibacteriales, *Ellin6067* (Proteobacteria), and an unclassified ASV of KD4-96 (Chloroflexi) were detected exclusively. *Chryseolinea* was detected exclusively in PP-treated samples. And a member of *Thermomonas* for PS-treated samples. An ASV identified as *Acidibacter*, *Intrasporangiaceae*, and

importance under LDPE treatment. Although natural connectivity was marginally higher in LDPE, implying greater robustness to random loss, vertex connectivity was lower, suggesting reduced resistance to fragmentation.

In contrast, PES exposure reduced overall network connectivity and resilience. The PES network showed lower average node degree, closeness, and eigenvector centrality than the control, but higher betweenness centrality, indicating a more fragmented and reorganized structure. Local clustering and edge density decreased, and both natural and vertex connectivity declined, reflecting diminished structural robustness. Keystone taxa such as *Vicinamibacterales* (ASV01002), *Oryzihumus terrae* (ASV01604), and *Thermomonas* (ASV13638) retained central positions but their influence weakened under PES, pointing to reduced network cohesion and stability.

Similarly, PP treatment resulted in moderate declines in connectivity and centrality relative to the control. The PP network exhibited lower node degree, eigenvector centrality, clustering, and natural connectivity, accompanied by a small reduction in edge density. While the influence of major keystone taxa (*Vicinamibacterales*, *Oryzihumus terrae*, *Thermomonas*) decreased, new taxa such as *JGI 0001001-H03* (ASV00384) and *mle 1-27* (ASV08274) gained importance through higher betweenness centrality, indicating a redistribution of structural roles within the network.

The PS network remained fully connected and comparable in size and density to its control, yet showed subtle shifts in node-level properties. Central taxa such as *Candidatus Nitrosocosmicus* (ASV00005), *Blastocellaceae* (ASV00411), and *Vicinamibacterales* (ASV01002) decreased in centrality, while *Thermomonas* (ASV13638) and *Nitrospira japonica* (ASV08768) gained influence. Clustering and natural connectivity declined slightly, whereas vertex connectivity increased marginally, indicating minor structural reorganization but preserved overall network integrity.

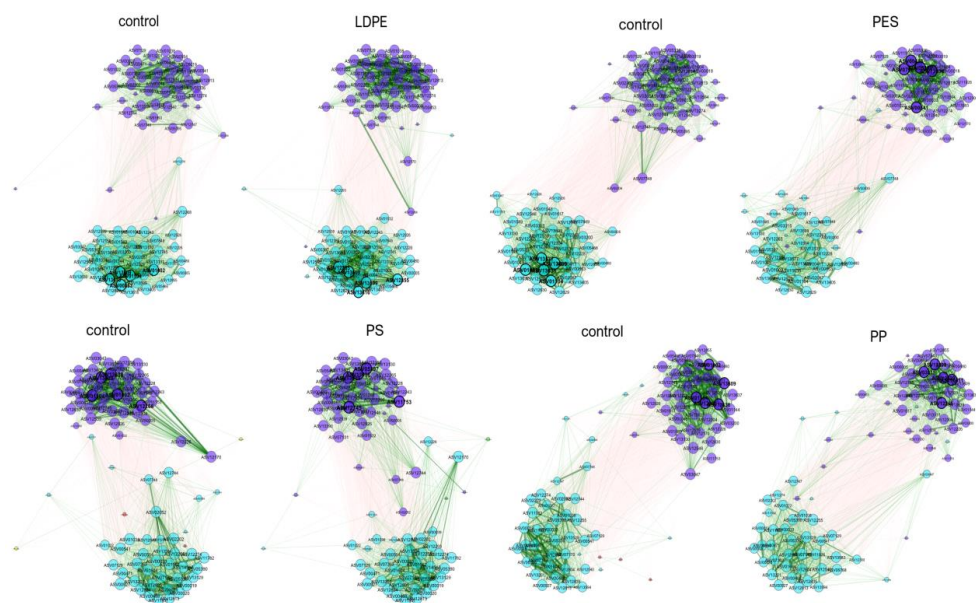


Fig. 3.7. Microbial co-occurrence networks based on Pearson correlations for untreated (control) and MP-treated (LDPE, PES, PS, PP) rhizosphere samples. Each node represents an amplicon sequence variant (ASV), with node size proportional to its degree (number of connections). Node colors (purple and turquoise) indicate network modules (communities) identified by correlation-based clustering, representing groups of ASVs that are more strongly co-occurring with each other. All networks are compartmentalized, consisting of two hubs of microbial taxa that frequently interact and/or respond similarly to environmental cues. Edges represent significant correlations between ASVs (green = positive; red = negative). The networks include core ASVs, defined as those present in at least four out of five replicates. Displayed are the 100 most abundant ASVs.

3.3.4. Combined effects of the different microplastics on microbiome and soil nutrients and root biomass

A principal coordinate analysis (PCoA) based on the microbiome and plant performance data (normalized root surface area), coupled with an *envfit* analysis, identified three plant nutrients that were significantly linked to shifts in microbial community structure (Fig. 3.8), including NH_4^+ ($p = 0.0451$), NO_2^- ($p = 0.0111$), and P_{av} ($p = 0.0008$). The direction and length of the vectors represent the gradient and strength of the association between nutrient concentrations and community dissimilarity in the ordination space. In the ordination space PS and PP- treated samples were positioned in the direction of NO_2^- and NH_4^+ vectors, showing that variation in community composition along the PCoA axes was associated with increasing concentrations of those nitrogen forms. In contrast, the P_{av} vector pointed in the opposite direction, suggesting that microbial communities from the control and, to a lower extent, PES treatment were associated with higher phosphate availability.

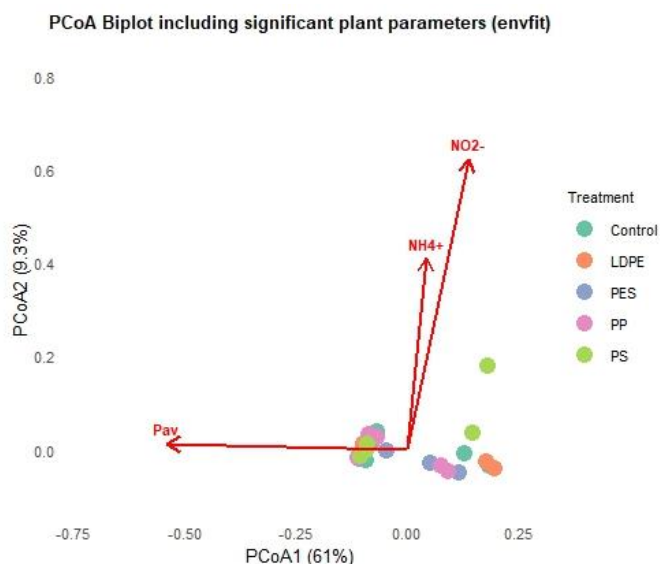


Fig. 3.8. Principal coordinate analysis (PCoA) of rhizosphere bacterial community composition associated with winter wheat after 28 days of growth, exposed to different MP treatments (Control, LDPE, PES, PP, PS). The ordination is based on Bray-Curtis dissimilarities of the microbial community composition. Red arrows indicate significant ($p < 0.05$) plant-available nutrient parameters fitted by *envfit*, showing correlations with community structure: plant available ammonium (NH_4^+), nitrite (NO_2^-), and phosphate (P_{av}). The direction and length of the arrows show the gradient and strength of the correlation, respectively. The plot illustrates that variations in microbial communities are partly explained by the availability of plant nutrients in the rhizosphere.

3.4. Discussion

Our results demonstrate that MP effects on early winter wheat performance are strongly polymer-type specific, likely interdependent, and closely linked to shifts in rhizosphere microbial network structure and function.

3.4.1. Microplastic type-driven alterations in root development, microbial networks, and nutrient cycling

Among the tested MPs, PP particles and PES microfibers consistently reduced root length, with effects comparable to those reported in previous studies. For instance, PP MPs at a concentration of 0.1% w/w have been shown to impair root growth, germination, and biomass, while also modifying soil enzyme activities – such as reduced phytase and increased dehydrogenase activity – and disrupting rhizosphere bacterial connections (i.e., nutrient exchange, signaling, cooperative metabolism) (Lian et

al., 2024). Similarly, PES microfibers at 0.5% w/w in soil have been reported to reduce both root length and biomass (~30%) in maize after 34 days (Ingraffia et al., 2021; Ingraffia et al., 2022b). This reduction in root length suggests that the presence of PES microfibers in the soil adversely affects root development, which can have implications for overall plant health and nutrient uptake. Unlike our results, these studies also highlighted detrimental effects on both shoot and root biomass, indicating that MP effects vary across plant and soil types (Ingraffia et al., 2021; Ingraffia et al., 2022b). In our study, the negative impacts of PP and PES on root development coincided with fragmentation of the microbial co-occurrence network under PES treatment, indicated by lower node degree, clustering coefficients, eigenvector and closeness centrality. This network simplification points to weakened microbial cooperation and reduced functional integration, which can limit plant-beneficial services such as nutrient mineralization, phytohormone production, and pathogen suppression (Huang et al., 2019a; Hernandez et al., 2021).

Responses on nutrient turnover processes further support this link. Across all MP treatments, bulk soil NH_4^+ concentrations increased significantly (Fig. 3.3b), with the largest effect sizes observed for PP and PES ($\eta^2 = 25\%$; Appendix B: Table B.3). A similar NH_4^+ accumulation has been reported for PP MP pollution (Li & Liu, 2022). Such increases can result from reduced microbial nitrification activity (e.g., inhibition of nitrifying bacteria) (Ayiti & Babalola, 2022), enhanced mineralization of organic matter, or decreased nutrient uptake by roots and/or microbes. MPs can also create novel microbial habitats (Kublik et al., 2022), potentially altering nutrient cycling. In our study, the NH_4^+ accumulation especially in PP and PES treatments could potentially be explained by observed microbial network fragmentation, lower edge density and reduced influence of formerly central taxa (e.g., *Thermomonas* and *Sphingomonas*).

Soil NO_3^- dynamics were more material-specific. The ANOVA showed a large η^2 (20%) effect of MP type on soil NO_3^- concentration (Appendix B: Table B.3). PS particles increased NO_3^- availability per RSA, while LDPE, PP, and PES had no significant effect. Elevated NO_3^- levels can arise from reduced plant uptake, increased nitrification, reduced denitrification, or decreased microbial uptake. For example, Rillig et al. (2021) found that MP fibers at 0.4% w/w altered N mineralization and nitrification, increasing NO_3^- due to enhanced soil aeration, which stimulated nitrification and inhibited denitrification. Similarly, de Souza Machado et al. (2019) reported that PS particles (2% w/w) altered soil structure in ways that could indirectly affect nutrient availability. In our study, PS-treated soils showed increased influence of nitrifying taxa such as *Nitrospira japonica* (Lee et al., 2022a), suggesting functional reorganization toward enhanced nitrification.

Relationships between other nutrients were less affected by MP type. We found a significant correlation between soil P_{av} and K_{av} concentrations, in line with previous work showing simultaneous

uptake of these nutrients by wheat roots (Römer et al., 1988; AdAlsteinsson et al., 1994; Erdei et al., 2006). This relationship was not disrupted by MP addition, or was affected similarly across treatments. Literature reports on MP effects on K dynamics vary: PE MPs at 0.5% w/w reduced K leaching (Yan et al., 2023), but plastic film mulching had no effect on soil K content (Jin et al., 2024). Similarly, we did not see any effect on soil K content in our study.

MP effects on soil pH also varied in previous studies. Zhao et al. (2021) observed pH fluctuations over time after MP addition (initial increase, then decrease, then increase again), while Yin et al. (2007) found no effect of PP groundcover sheets. Our findings of no significant pH change support the latter.

Soil water dynamics showed MP-type dependent responses. Jazaei et al. (2022) reported that 3 mm polyacrylic microfibers at 0.4-0.6% w/w increased WHC, while acrylonitrile butadiene styrene pellets decreased WHC. Wang et al. (2023) found that small PE MPs (150 μm , 2% w/w) increased WHC in loamy soil by raising porosity. In addition, Neubert et al. (2025) showed that effects of MPs on soil hydraulic properties depended strongly on soil texture. PET fibers at 0.4% w/w generally increased porosity and air-filled pore volume, whereas PS and PP particles (0.4% w/w) reduced porosity and altered hydraulic conductivity, enhancing water flow in finer soils but clogging pores in sandy soils. In our study, LDPE particles caused the largest increase in soil water content, while PES microfibers caused the lowest. These differences likely reflect MP size, shape, and polymer properties, as our LDPE particles were one-tenth the size of those used by Jazaei et al. (2022) and Rillig et al. (2021).

3.4.2. Microbiome network responses and functional links to plant performance

Microbial network analysis revealed that LDPE promoted the most substantial increase in rhizosphere microbial co-occurrence complexity, with higher edge density, clustering coefficient, and natural connectivity. Such patterns suggest increased microbial connectivity, potentially due to the surface properties of LDPE, which promote microbial colonization and biofilm formation (Huang et al., 2019b; Rong et al., 2021; Li et al., 2022a; Ya et al., 2022) or even its potential use as a carbon-rich substrate (Gong et al., 2019; Chen et al., 2022b). New keystone taxa emerged under LDPE treatment, including *Vicinamibacteraceae* (ASV01049) and *Pseudolabrys* (ASV11937), both potentially linked to plant-associated processes and organic matter degradation (Gonçalves et al., 2024; Arunrat et al., 2025). The absence of root length reductions under LDPE suggests that these microbial changes may strengthen locally resilient microbial communities that maintain or even promote functional interactions, possibly through enhanced water retention or by serving as surfaces for microbial colonization and biofilm formation (Gong et al., 2019; Huang et al., 2019b). The increased connectivity and emergence of new keystone species could further support nutrient cycling and potentially buffer environmental stress.

By contrast, PES and PP reduced network cohesion, with lower node degree, clustering coefficients, and centrality measures. This fragmentation suggests reduced microbial cooperation and stability (Griffiths & Philippot, 2013), consistent with the observed negative effects on root length. Loss of keystone taxa could weaken nutrient cycling, pathogen suppression, and other plant-beneficial functions. Network robustness also declined under PES, PP, and PS, although slight increases in vertex and edge connectivity for PP and PS may indicate compensatory adaptations (Berry & Widder, 2014).

PS caused only minor changes in network structure, e.g., slightly increased edge density without major connectivity loss, but altered the identity of central taxa. Gains in influence of nitrifiers such as *Nitrospira japonica* likely explain the moderate NO_3^- increase in PS-treated soils, suggesting targeted functional shifts rather than broad community disruption (Lee et al., 2022a). The limited effect on root and shoot metrics under PS further supports a more functionally resilient microbial community.

Taxonomic analyses of rhizosphere microbial networks indicated that generalist taxa, such as *Vicinamibacteraceae* and *Pseudolabrys*, maintained high centrality across all treatments, reflecting network stability and functional redundancy in nutrient cycling and organic matter decomposition. In contrast, specialist taxa, including *Nitrospira japonica* (nitrification), *Oryzihumus terrae* (organic matter decomposition), and *Thermomonas* (nutrient mineralization), were more sensitive to PES and PP exposure, showing reduced centrality and influence on network structure. Opportunistic taxa, like *JG1/0001001-H03* and *mle 1-27*, gained prominence under PP, suggesting a redistribution of structural roles. These findings support the ecological theory that generalists provide resilience under disturbance, whereas specialists perform niche-specific functions that are more vulnerable to environmental stressors (Clavel et al., 2011).

3.5. Conclusion

Our findings demonstrate that MP effects on early wheat growth, nutrient dynamics, and rhizosphere microbial interactions are polymer-specific and interconnected. At 0.4% w/w, PP and PES significantly altered soil properties and reduced root growth, effects linked to fragmented microbial networks, loss of keystone taxa, and potential suppression of nitrifying activity. In contrast, LDPE enhanced microbial network complexity and soil moisture, suggesting improved microbial integration and possible stress buffering. PS induced only minor structural changes, with subtle shifts favoring nitrification but without major network disruption.

Despite MP-induced changes in soil moisture and NH_4^+ concentrations, leaf growth, nutrient uptake, and chlorophyll content remained unaffected, indicating that early wheat root systems maintained sufficient shoot support at this developmental stage. Network analyses further revealed that LDPE strengthened microbial connectivity, while PP and PES weakened it, and PS caused only

limited reorganization. These results highlight that physicochemical properties of MPs – such as polymer type and shape – govern their impact on plant-soil-microbiome interactions.

Overall, our study emphasizes the importance of rhizosphere microbiome robustness in mediating plant responses to MP stress and underlines the need for future research addressing longer growth periods, diverse soil types, and field conditions. Such work is essential to fully understand how MPs influence soil functionality, microbial resilience, and crop productivity.

CHAPTER 4 – Soil texture modifies the impact of microplastics on winter wheat growth

Based on:

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Abstract

Research on the impact of microplastics (MPs) on plant performance has primarily focused on MP type or concentration, often neglecting the role of soil texture. In this study, a 42-day experiment was conducted in which winter wheat was grown in three soils of different textures, contaminated with two types of MPs: LDPE particles and PES microfibers at 0.4% w/w concentration. The effects on soil water content, nutrient levels, and plant growth were examined. In silty loam (Luvisol), LDPE reduced root length and biomass, likely due to altered soil texture, which created more macropores and reduced water and nutrient availability. PES microfibers had similar effects, indicating that changes in soil porosity impacted root access to resources. In sandy loam (Albic Luvisol), both MP types reduced root growth, with PES microfibers causing a significant 85% reduction in root length and decreasing nitrogen content, suggesting impaired nutrient availability due to reduced nitrification. Conversely, in silty clay loam (Chernozem), LDPE increased root length by 4.6 times, likely due to enhanced water movement pathways, although it also increased water loss. PES microfibers showed minimal positive effects on root growth but reduced nutrient content. Overall, soil texture had a significant impact on how MP affected plant growth, as the two types of MP had different effects on different soil textures. LDPE increased macroporosity in fine soils, promoting root growth, but reduced nutrient uptake in coarse soils. PES microfibers influenced soil structure, affecting water retention and nutrient availability differently in different soil types. The study highlights the complexity of MP–soil–plant interactions. Moreover, it also calls attention to rethinking soil management in the future, such as using biodegradable alternatives, applying biochar or avoiding plastic-coated controlled-release fertilizers.

4.1. Introduction

Plastic waste, a ubiquitous characteristic of the Anthropocene, has pervaded natural environments, with MP pollution emerging as a global environmental challenge. MPs, defined as plastic particles or synthetic microfibers smaller than 5 millimeters, raise concerns about their long-term impact on ecosystems and associated hydrological and biogeochemical processes (Bian et al., 2022; Wang et al., 2022a). While our understanding of these impacts is evolving, it is clear that MPs are increasingly accumulating in the environment.

Terrestrial ecosystems and soils have become significant reservoirs for MP accumulation, with approximately 80% of marine plastic waste originating from land sources (Andrady, 2011; Jambeck et al., 2015). Various activities contribute to MP pollution, including wastewater irrigation, sewage sludge utilization (Corradini et al., 2019), atmospheric deposition (Klein & Fischer, 2019), and plastic film mulching (Wang et al., 2022c). In agricultural soils, PE and PES fibers are commonly found plastic types, which is plausible, as the sum of LDPE and HDPE accounts for almost 30% of the total plastic production in Germany (Heinrich-Böll-Stiftung, 2019). LDPE, for example, is ductile, flexible and transparent, and is therefore used in films. PES fibers, which are synthesized from petrochemical-based PET, accounted for the highest share of synthetic fiber production in 2017, at 80% (Heinrich-Böll-Stiftung, 2019). They have been detected in the soil of agricultural fields, mostly due to the application of sewage sludge (Büks & Kaupenjohann, 2020). Around 80-99% of the fibers contained in wastewater are retained in sewage treatment plants and end up in sewage sludge, resulting in concentrations ranging from 1,000-56,400 particles kg⁻¹ dry sewage sludge (Zubris & Richards, 2005; Mahon et al., 2017; Mintenig et al., 2017). Global trends that contribute to plastic contamination, particularly in agriculture, include the use of plastic seedling trays, protective meshes or plastic irrigation tubing, as well as some fertilizers and pesticides which are encapsulated in plastic coatings. According to Meizoso-Regueira et al. (2024), fertilized soils have a 0.47% higher growth rate of MPs.

The concerning prevalence of MPs in agricultural fields has prompted a series of research efforts aimed at understanding and addressing their impact. On the one hand, scientists have focused on mitigation strategies to reduce MP contamination and prevent its accumulation in soil. These include more targeted use of fertilizers and pesticides with the use of GPS and sensor-based equipment (Nakachew et al., 2024) or the development of biodegradable fertilizer which embeds nutrient granules in naturally generated, biodegradable plastic (Witt et al., 2024). On the other hand, it is equally important to evaluate whether MP exposure has specific effects on plant growth and physiology, particularly in wheat as one of the world's major staple crops (Wang et al., 2020a; Zang et al., 2020; Gkoutselis et al., 2021; Zhou et al., 2021; Liu et al., 2022b; Wang et al., 2022c). However,

results pertaining to the effects of MP contamination in agricultural soil on wheat root and shoot growth have been controversial, reporting increase (Liu et al., 2022b), dose-dependent decrease (Zang et al., 2020), or no discernible changes in wheat growth (Lozano et al., 2021a; Mészáros et al., 2023). As an example, one study reported an increase in biomass of both wheat roots and shoots of up to 200% (Liu et al., 2022b), while another study claimed a reduction of wheat and shoot biomass by 13% to 53%, despite using PE and PVC in the same concentrations of 1% and 5% in soil (Zang et al., 2020). Liu et al. (2022) assumed that these differences could have been caused by the different soil types used. Likewise, while it was shown that PES in a loamy sand decreased soil bulk density, which affects root growth (de Souza Machado et al., 2019), another study using a clay loam could not confirm these findings although they used the same type and concentration of MP fibers (Zhang et al., 2019). These contradictions underscore the need to study the influence of soil type on MP effects on plant growth, in addition to the type, size, and concentration of MPs used.

Nevertheless, awareness has increased that soil characteristics can have a potential impact on MP effects, thereby influencing root growth indirectly (Krehl et al., 2022). On the one hand, an altered soil structure due to MP contamination can impact soil hydraulic properties, such as water-holding capacity (Guo et al., 2022), water flow in the soil pore space (Hangele et al., 2020) or field capacity (Qi et al., 2020b). Especially fibers in soil have been shown to have a more pronounced effect on soil hydraulic properties compared to granular MPs (de Souza Machado et al., 2019; Lehmann et al., 2021; Lozano et al., 2021a; Ingraffia et al., 2022b; Yu & Flury, 2022). This is because the fibrous, flexible nature of MP microfibers can rearrange soil structure more extensively than particles and can lead to increased or reduced aggregation (Lozano et al., 2021b), depending on soil type. Especially in the presence of soil biota, microfibers reduced soil aggregate stability (Lehmann et al., 2019). However, some studies have found that PES microfibers can actually increase the formation and stability of soil macroaggregates, particularly in clay-rich soils (Lehmann et al., 2021; Ingraffia et al., 2022a). This suggests that the effects of PES microfibers on soil aggregation are complex and depend on factors like soil texture, mineralogy, and the presence of a microbial community.

On the other hand, the changes in soil properties can also create selection pressure, driving changes in soil microbial communities, their habitats and functions, and further impacting plant performance. For example, the increase in porosity by MP in soil was found to increase air flow and to be beneficial for the growth of aerobic microbes (Wang et al., 2023), but can also affect the performance of wheat seedlings negatively (Zhang et al., 2024). Due to these indirect alterations in microbial communities in soil, nutrient dynamics and enzyme activities are changed or even impaired (Chen et al., 2022a; Liu et al., 2023c). It has also been reported that the hydrophobic surface of MPs is a suitable niche for a wide variety of microorganisms (Miao et al., 2019). As a result, the definition of the *plastisphere*, a term which was introduced in the marine literature (Zettler et al., 2013), has been

extended to the terrestrial realm and defined as the soil volume immediately influenced by the plastic particles (Rillig et al., 2024).

Another important factor determining plant performance is the unrestricted functioning of the rhizosphere, characterized as the soil region influenced by plant roots, and playing an important role in nutrient dynamics, such as mineralization or denitrification and nutrient acquisition (Richardson et al., 2009; Pii et al., 2015). In this context, an increased ammonium (NH_4^+) and nitrate (NO_3^-) consumption by wheat roots and subsequent disturbance of the rhizospheric microbial community were detected after the addition of PE in soil at 5%, using *in situ* soil zymography techniques (Liu et al., 2022b). Moreover, it was found that the addition of 1% w/w LDPE as MP significantly altered the bacterial community composition in the rhizosphere of wheat compared to the control (Qi et al., 2020b) by disrupting beneficial plant-microbe interactions in the rhizosphere. The variations in volatile compounds and microbial communities caused by LDPE residues may account for the observed negative effects on wheat growth.

These findings underscore the multifaceted impacts of MPs on soil-plant interactions and highlight the need for further investigation. To this end, we addressed the current knowledge gaps by introducing LDPE particles and PES microfibers, varying in their shape (particle vs. fiber), into three different soil types. We hypothesized that the impacts of MPs on plant performance would vary not only as a function of polymer type and morphology, but would also depend on the respective soil's inherent properties, and primarily on its texture.

4.2. Materials and methods

4.2.1. Soils

Three soil types were used in the study: i) a Luvisol with a loamy silt texture, ii) an Albic Luvisol with a loamy sand texture, and iii) a Chernozem with a silty clay loam texture. The basic soil characteristics of the three different soils are listed in Table 4.1. A detailed description of the soils used in this experiment, including the sampling location, is provided Chapter 2, Section 2.2.1.

Table 4.1. Characteristics of experimental soils including organic carbon (C_{org}) content, plant-available mineral nitrogen content (N_{min}), plant-available phosphate (P_{av}) and potassium (K_{av}).

Soil type	Texture	Sand 63-2000 µm	Silt 2-63 µm	Clay <2 µm	pH ^a	C _{org}	N _{min}	P _{av}	K _{av}
		%	%	%		%	µg g ⁻¹	µg g ⁻¹	µg g ⁻¹
Luvisol	Loamy silt	8	77	15	6.5	1.01	168	20	116
Albic Luvisol	Loamy sand	87	10	3	5.1	0.50	37	22	112
Chernozem	Silty clay loam	11	68	21	7.5	2.06	21	12	51

^apH determined in 0.01 M calcium chloride solution.

4.2.2. Microplastics

Based on results on MP pollution of German agricultural soils from literature (de Souza Machado et al., 2019; Piehl et al., 2018) and on results from previously conducted rhizotron plant experiments, LDPE particles and PES microfibers were considered significant for further investigation. Detailed information about the LDPE powder and PES microfibers used in the experiment, including the particle size, fiber lengths and characterization methods, can be found in Chapter 2, Section 2.2.2.

4.2.3. Experimental design

For the pot experiment, sewage pipes (10 cm diameter, 80 cm length) made of polypropylene (PP) were used. Potential contamination of the soil with PP plastic debris or additives from the PP pipes was considered negligible, as the 42-day exposure period was insufficient for significant weathering processes of the PP plastic due to UV radiation. To allow for optimal drainage and aeration, these pipes were equipped with perforated caps at the bottom, covered inside with a cloth, and supplemented with a layer of 2 cm quartz sand. To ensure stable temperature conditions in the root zone, the pipes were installed in empty lysimeter spaces on the campus of Forschungszentrum Jülich (50.908738°N, 6.403356°E), embedded in a custom-made polystyrene rack with only the top 10 cm of the PP pipes above ground.

Prior to filling the pipes, the air-dried soils were mixed with 0.4% w/w of the respective MP type by manually stirring the mixture in a box for 5 min. According to de Souza Machado et al. (2018a), the applied mass concentration of 0.4% w/w was the upper limit of MP concentration at which the MP-

amended soils show only minor changes in total volume. The control of each soil without the addition of MP followed the same procedure. Subsequently, the sewage pipes were then gradually filled with portions of approximately 300 g of soil, which were carefully compacted by hitting the pipes to the ground, achieving compaction densities of 1.5, 1.7, and 1.3 g cm⁻³ for dry Luvisol, Albic Luvisol and Chernozem, respectively, which corresponded to the bulk densities of these soils in the field. Final amounts of soil within the columns for Luvisol, Albic Luvisol and Chernozem soil, were 8.7 ± 0.3 kg, 9.8 ± 0.2 kg and 7.9 ± 0.2 kg, respectively. A total of nine treatments (three soils x three MP treatments including the control) were prepared, with each treatment replicated in quadruplicate. To minimize bias, one set of replicates was allocated to each of the four lysimeter spaces (Appendix C: Fig. C.2), arranged in squares of 1 m x 1 m, approximately 3 m apart from each other.

After assembling the soil-filled pipes in the lysimeter spaces, they were irrigated from above with irrigation water equivalent to rainwater to reach 60% of the soil type-specific gravimetric water-holding capacity (WHC) for Luvisol and Albic Luvisol soils, corresponding to a total water content of 20% w/w and 11% w/w of dry soil, respectively. For the CH soil, the target values was 70% WHC, corresponding to a total water content of 27% w/w of dry soil. The initial weight of the soil-filled and irrigated tubes served as reference. After a one-week equilibration period outdoors, two pre-germinated winter wheat seeds (*Triticum aestivum*, variety Nordkap, SAATEN-UNION GmbH, Isernhagen, Germany) of similar size were planted 2 cm deep into the soil in the center of each pipe. Germination had occurred in darkness under ambient conditions on moist paper towels for 24 hours. After one week, the less developed seedling was removed.

Plant growth continued for a period of 42 days. Each set of replicates was planted at staggered intervals of one week, thus the total growth period ranged from August until October 2023. Plants were exposed to outdoor weather conditions. The mean temperatures of August, September, and October 2023 were 17.97 ± 4.24 °C, 17.98 ± 4.75 °C, and 13.07 ± 4.25 °C, respectively, with total precipitation of 123.4 mm, 75.1 mm, and 95.0 mm during these months. The weather data were obtained from the weather station located on the campus of the Forschungszentrum Jülich, and exact temperature curves and precipitation are depicted in Appendix C, Fig. C.3. To ensure well-watered conditions, soil water content was maintained at the initial level by weighing each tube and replacing the water loss by irrigating every 3 to 4 days from above. Stable temperature conditions in the root zone were maintained throughout the experiment by both the insulating polystyrene rack and the positioning of the pipes below ground level in the empty lysimeter spaces (Appendix C: Fig. C.2a). A temperature probe positioned at half the depth of each of the lysimeter spaces indicated realistic soil temperature conditions (Appendix C: Fig. C.4). Alongside the continuous monitoring of water content and root zone temperature, regular assessments of shoot length were conducted.

It should be noted that the applied MP concentration of 0.4% w/w used in our study was much higher than reported for today's agricultural soils, e.g., <0.002% w/w for arable land treated with biosolids for 10 years (Corradini et al., 2019). However, the aim of our study was to investigate a future scenario of MP effects in different soils assuming a constant or even increasing MP input to agricultural soils, and to exclude bias from any potential prior MP content of the soils.

4.2.4. Soil analysis

On the final sampling day, the soil-filled pipes were cut horizontally with a saw into three compartments of approx. 2000 cm³ volume, representing soil depths of 0-25 cm, 25-50 cm, and 50-75 cm. The root-containing soil of the compartments was stored in plastic bags at -20°C until further analysis. Immediately before root sampling, soil samples were taken, differentiating between the soil attached to individual roots, labelled as "rhizo soil", and the surrounding soil without roots, referred to as "bulk soil". The rhizo soil was carefully brushed from individual roots, while particular attention was given to the bulk soil to ensure the complete absence of root material within the sample.

As described in Chapter 3, Section 3.2.5, soil samples were prepared by performing two extractions: one with 0.01 M CaCl₂ solution for N_{min} analysis and another with CAL solution for K_{av} and P_{av} analysis (VDLUFA, 2016). The detailed extraction procedures and pH measurements can be found in the aforementioned section.

4.2.5. Root analysis

Roots were cleaned from attached soil with a water showerhead over a 2-mm sieve. Finally, roots were collected from the sieve and were stored in 50% v/v ethanol until roots were scanned and quantified with the software WinRHIZO (Regent Instruments Inc., Québec, Canada). The exact method is explained in detail in Chapter 3.2.5. After root analysis using WinRHIZO, samples were oven-dried at 60 °C for 7 days and ground using a ball mill (MM200, Retsch GmbH, Haan, Germany). Portions of approx. 200-400 mg were then analyzed by an ICP-OES analysis system (iCAP 7600, Thermo Scientific, Waltham, MA). The exact method for sample digestion can be found in the Appendix C.

4.2.6. Leaf analysis

On the final harvest day, wheat shoots were harvested, cut, and oven-dried at 60 °C for 7 days. The dried material was subsequently analyzed for chlorophyll and nutrient content. The chlorophyll analysis procedure is described in detail in Chapter 3.2.6. Nutrient concentrations in the ground leaf material were determined by an external laboratory using an ICP-OES analysis system (iCAP 7600,

Thermo Scientific, Waltham, MA). The exact method for sample digestion can be found in the Appendix C.

4.3. Results

4.3.1. Root analysis

4.3.1.1. Root lengths

The roots harvested from a soil depth of 75 cm in the control soils were generally longest in the loamy sand Albic Luvisol, with on average 270 ± 60 m, slightly shorter in the loamy silt Luvisol with on average 200 ± 70 m, and shortest in the silty clay loam Chernozem with on average 36 ± 8 m (Fig. 4.1). In the Luvisol (Fig. 4.1a) and the Albic Luvisol (Fig. 4.1b), the same trend of significantly lower root growth in the presence of both MP types was observed. In the Luvisol, LDPE reduced the root length by more than half ($-57\% \pm 29\%$, $p = 0.04$), and PES even by two thirds ($-65\% \pm 12\%$, $p = 0.01$) of the length measured in the Luvisol control soil. In the sandy Albic Luvisol, LDPE also reduced the total length by half ($-49\% \pm 20\%$, $p = 0.01$), while PES reduced the root lengths by 85% $\pm 2\%$ ($p = 0.003$). The trend was exactly the opposite in the Chernozem (Fig. 4.1c), with greater root length in MP treatments compared to the control soil. In comparison to the control, LDPE in Chernozem increased root length by 462% ($\pm 100\%$, $p = 0.002$). The PES microfibers, in contrast, led only to a weak increase in root length ($+45\% \pm 38\%$, $p = 0.07$).

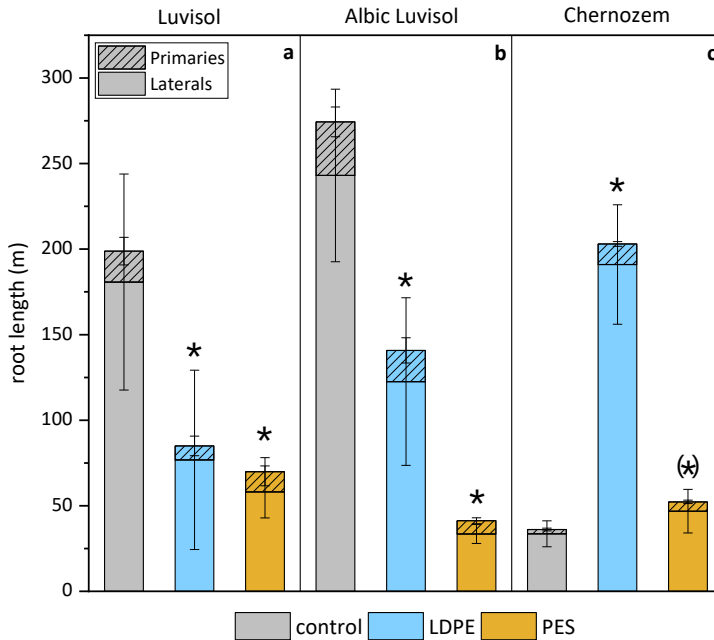


Fig. 4.1. Mean root lengths (in m) of winter wheat harvested from 75 cm long PVC tubes and grown in a) the loamy silt Luvisol, b) the loamy sand Albic Luvisol, and c) the silty clay loam Chernozem soil for 42 days. Plants were grown with the additions of the microplastic types low-density polyethylene (LDPE) particles and polyester microfibers (PES) at 0.4% w/w in dry soil. Control is the respective soil type without MP addition. Laterals indicate lateral roots with a width of 0-0.4 mm and Primaries (crosshatched) indicate primary roots with a width of 0.4-1 mm. Bars show the mean values of 4 replicates with whiskers representing the standard deviation. Asterisks show significant difference to the control ($p \leq 0.05$), while asterisks in parentheses show weak significance to the control ($p \leq 0.1$), tested via heteroscedastic, paired t -test.

4.3.1.2. Nutrient content in roots

Analysis of the nutrient content in dried winter wheat roots, including K, P and total nitrogen (N_t), revealed slight variations for the different soil types (Fig. 4.2). In particular, the K content in roots grown in all soil types remained below 2.8 mg g^{-1} , and the P content in all root samples was consistently below 1.5 mg g^{-1} . Nevertheless, the N_t content varied in particular between soil types, with the highest values in roots grown in the Luvisol (Fig. 4.2a) control, ranging from 9 to 13 mg g^{-1} . In contrast, roots grown in Albic Luvisol (Fig. 4.2b) and Chernozem (Fig. 4.2c) revealed N_t contents below 7 to 14 mg g^{-1} .

Significant variations in root nutrient content were observed between plants grown in MP-treated soil and control soil across different soil types. Roots in MP-treated Luvisol and Albic Luvisol consistently showed reduced K, P, and N_t contents. The most significant reduction was in N_t uptake,

particularly in Luvisol soil amended with LDPE particles and PES microfibers ($-92-93\% \pm 5\%$ less N_t , $p = 0.03$). Albic Luvisol soil showed less pronounced but still significant N_t reductions with LDPE ($-36\% \pm 19\%$ less, $p = 0.04$) and PES microfibers ($-75\% \pm 13\%$ less, $p = 0.003$). In Chernozem soil, only PES-treated roots exhibited significantly less N_t ($-64\% \pm 3\%$, $p = 0.03$), while LDPE-treated roots were similar to the control. Similarly, the P content of roots in LDPE-treated Chernozem was similar to the Chernozem control, whereas roots in PES-treated Chernozem contained not only significantly less P ($-64\% \pm 12\%$, $p = 0.04$) but also less K ($-64\% \pm 17\%$, $p = 0.02$). Roots grown in LDPE-treated Chernozem showed significantly higher K content ($+62\% \pm 24\%$, $p = 0.02$) compared to the control.

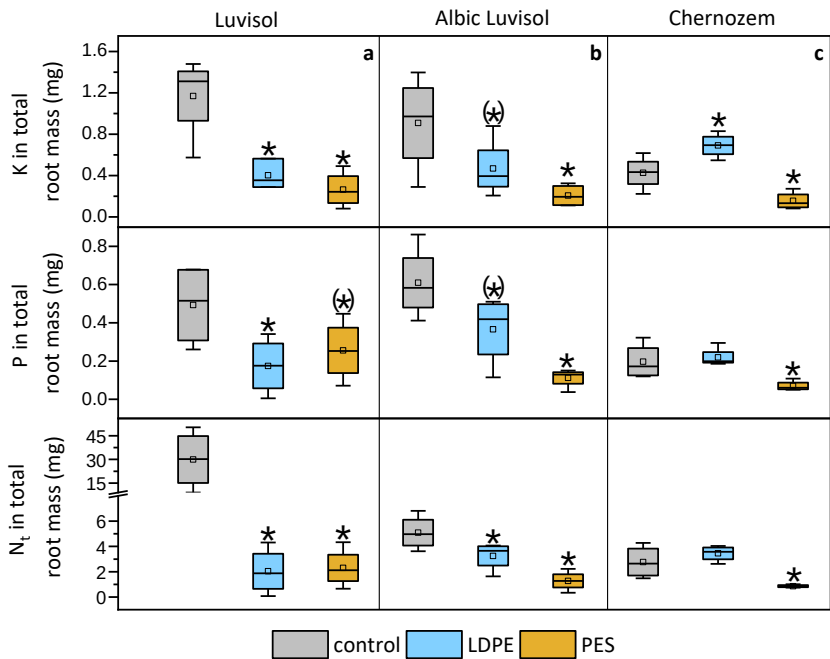


Fig. 4.2. Nutrient content including potassium (K), phosphate (P) and total nitrogen (N_t), in dry pulverized roots of winter wheat grown in a) the loamy silty Luvisol, b) the loamy sandy Albic Luvisol, and c) the silty clayey loamy Chernozem soil for 42 days. Plants were grown with the additions of the microplastic types low-density polyethylene (LDPE) particles and polyester microfibers (PES) at 0.4% w/w in dry soil. Control is the respective soil type without MP addition. Boxes are the interquartile range (IQR) from the 25th to 75th percentile, lines inside boxes indicate the median, white squares indicate the mean values, upper and lower whiskers are 1.5 times the IQR, and markers are individual data points. Asterisks show significant difference to the control ($p \leq 0.05$), while asterisks in parentheses show weak significance to the control ($p \leq 0.1$), tested via heteroscedastic, paired *t*-test.

4.3.2. Soil analysis

4.3.2.1. Nutrient content in soil

In general, the analysis showed that after the 42-day growth period of winter wheat plants, all soil types exhibited decreased nitrogen levels compared to the initial concentrations before planting (Fig. 4.3). The most significant reduction after the growth period was found for the NO_3^- content in Luvisol (Fig. 4.3a), with a decrease of approx. 97%, possibly due to heavy rainfall and leaching events. However, the P_{av} content increased during the growth period, particularly in the loamy silt Luvisol ($+49\% \pm 20\%$, $p = 0.02$) and the loamy sand Albic Luvisol ($+42\% \pm 27\%$, $p = 0.02$). Conversely, there was no increase in P_{av} content in the silty clay loam Chernozem after the growth period.

While in the Luvisol and Chernozem the contents of most of the N_{min} compounds (NO_3^- , NO_2^- and NH_4^+) and P_{av} and K_{av} remained stable despite the addition of MP, in the sandy Albic Luvisol, NO_2^- and P_{av} content was increased in the presence of PES microfibers compared to the control. Specifically, in the bulk soil of PES-treated AL, P_{av} levels were significantly more increased by $29\% \pm 16\%$ ($p = 0.04$) than in the corresponding rhizo soil, which was increased by $20\% \pm 17\%$ ($p = 0.09$). LDPE particles in the loamy silt Luvisol led to a slight increase in NO_2^- levels by $30\% \pm 22\%$ ($p = 0.07$). Conversely, NO_3^- concentrations decreased by approx. 70% ($p = 0.06$) compared to the control in both rhizo and bulk soil of Chernozem.

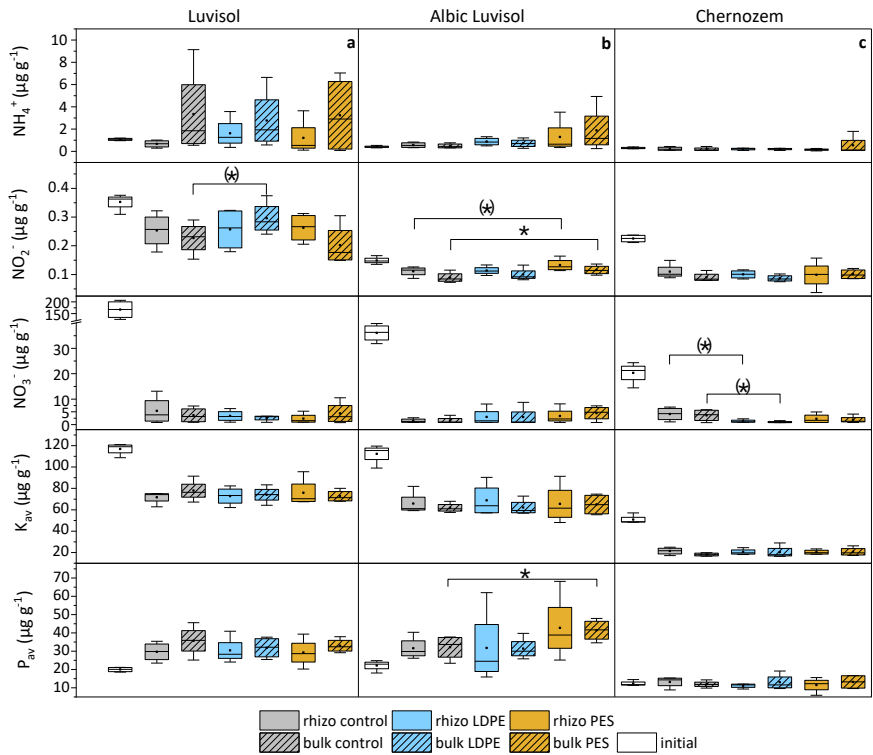


Fig. 4.3. Soil nutrients including plant-available nitrogen compounds (NH_4^+ = ammonia, NO_2^- = nitrite, NO_3^- = nitrate) and plant-available potassium (K_{av}) and phosphate (P_{av}) content in a) the loamy silt Luvisol, b) the loamy sand Albic Luvisol and c) a silty clay loam Chernozem after the winter wheat growth period of 42 days. Initial refers to the respective nutrient concentration before plants were seeded. Microplastic (MP) types mixed at 0.4% w/w in soil were low-density polyethylene (LDPE) and polyester (PES) microfibrers. Control is s the respective oil type without MP addition. It was differentiated between rhizosphere soil (rhizo), which was shaken off the roots, and bulk soil not attached to the roots. Boxes are the interquartile range (IQR) from the 25th to 75th percentile, lines inside boxes indicate the median, white squares indicate the mean values, upper and lower whiskers are 1.5 times the IQR, and markers are individual data points. Asterisks show significant differences to the control ($p \leq 0.05$), while asterisks in parentheses show weak significance ($p \leq 0.1$), tested with heteroscedastic, paired *t*-test.

4.3.2.2. pH and water content in soil

The pH values were lowest for the loamy sand Albic Luvisol (5.53 ± 0.15) (Fig. 4.4b), slightly higher for the loamy silt Luvisol (6.58 ± 0.16) (Fig. 4.4a) and the highest for the silty clay loam Chernozem (7.41 ± 0.04) (Fig. 4.4c). In the Luvisol, no significant differences in pH were found between bulk and rhizo soil, nor between the MP-treated and control soils. However, pH values were significantly enhanced in the

PES-amended Albic Luvisol, where the pH in the rhizo Albic Luvisol was elevated by 0.41 ± 0.23 units ($p = 0.02$) and in the bulk AL by 0.32 ± 0.14 units ($p = 0.02$). In addition, the pH of the LDPE-treated Albic Luvisol bulk soil was higher than that of the rhizo soil ($+0.13 \pm 0.04$ pH units, $p = 0.06$) and also slightly higher than the pH of the control soil. The pH of the PES-treated Chernozem soil was significantly lower than that of the control (-0.08 ± 0.04 pH units, $p = 0.03$), and slightly lower in the rhizo soil compared to the respective bulk soil (-0.04 ± 0.04 pH units, $p = 0.08$).

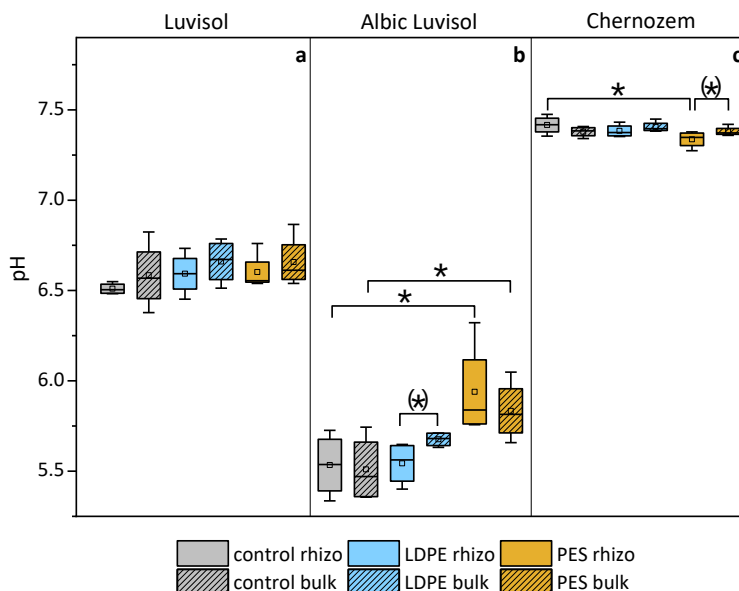


Fig. 4.4. Soil pH measured in calcium chloride extractions of a) the loamy silt Luvisol, b) the loamy sand Albic Luvisol, and c) the silty clay loam Chernozem soil measured after the winter wheat growth period of 42 days. Plants were grown with the addition of the microplastic types low-density polyethylene (LDPE) particles and polyester (PES) microfibers at 0.4% w/w in dry soil. Control is soil type without MP addition. It was differentiated between rhizosphere soil (rhizo), which was shaken off the roots, and bulk soil not attached to the roots. Boxes are the interquartile range (IQR) from the 25th to 75th percentile, lines inside boxes indicate the median, white squares indicate the mean values, upper and lower whiskers are 1.5 times the IQR, and markers are individual data points. Asterisks show significant difference to the control ($p \leq 0.05$), while asterisks in parentheses show weak significance ($p \leq 0.1$), tested with a heteroscedastic, paired *t*-test.

In general, the silty clay loam Chernozem had the highest water content, with average gravimetric water content in the Chernozem control of $31\% \pm 1\%$ in rhizo and $26\% \pm 4\%$ in bulk soil (Fig. 4.5c). Conversely, the silty loam Luvisol contained less water, with average values of $26\% \pm 2\%$ in rhizo and $24\% \pm 3\%$ in bulk soil (Fig. 4.5a), while the loamy sand AL had the lowest gravimetric water content,

averaging at $10\% \pm 2\%$ in both rhizo and bulk soil (Fig. 4.5b). Notably, PES microfibers within the rhizo soil of the finer-textured soil types Luvisol and Chernozem were associated with a considerable variability of soil water content, tending towards lower values compared to their respective controls. However, this trend was not observed in the PES bulk soil. Interestingly, in sandy Albic Luvisol rhizo soil, PES microfibers increased water content by $25\% \pm 16\%$, albeit only with marginal significance at $p = 0.09$.

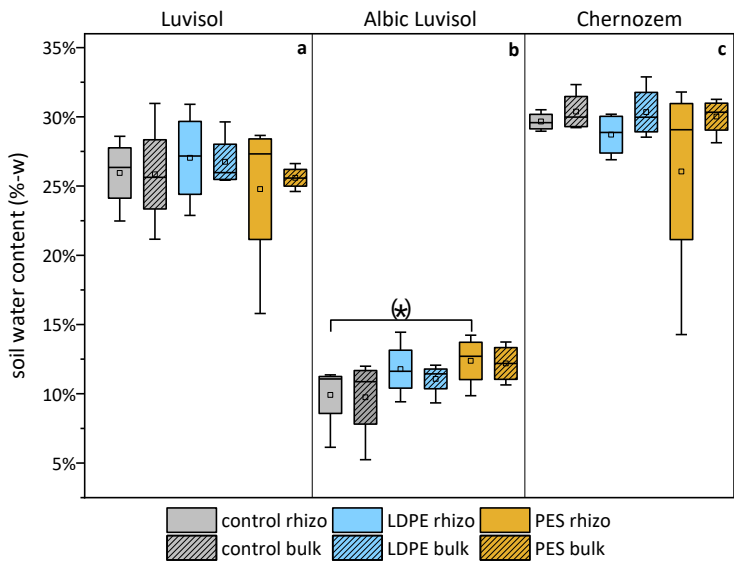


Fig. 4.5. Soil water content measured gravimetrically in a) the loamy silt Luvisol, b) the loamy sand Albic Luvisol, and c) the silty clay loam Chernozem soil measured after the winter wheat growth period of 42 days. Plants were grown with the additions of the microplastic types low-density polyethylene (LDPE) particles and polyester (PES) microfibers at 0.4% w/w in dry soil. Control is the respective soil type without MP addition. It was differentiated between rhizosphere soil (rhizo), which was shaken off the roots, and bulk soil not attached to the roots. Boxes are the interquartile range (IQR) from the 25th to 75th percentile, lines inside boxes indicate the median, white squares indicate the mean values, upper and lower whiskers are 1.5 times the IQR, and markers are individual data points. Asterisks show significant difference to the control ($p \leq 0.05$), while asterisks in parentheses show weak significance ($p \leq 0.1$), tested with a heteroscedastic, paired t -test.

4.3.3. Leaf analysis

4.3.3.1. Chlorophyll and carotenoid content in leaves

The presence of LDPE in Albic Luvisol consistently resulted in weakly significant reductions in leaf chl_a , chl_b , and car levels, with reductions of $37\% \pm 28\%$ ($p = 0.09$), $36\% \pm 29\%$ ($p = 0.09$), and $44\% \pm 25\%$ ($p = 0.09$), respectively (Fig. 4.6b). Conversely, LDPE in Chernozem was associated with significant increases in chl_a , chl_b and car leaf contents by $43\text{--}47\% \pm 25\%$ ($p = 0.05$) compared to the control (Fig. 4.6c). In contrast to Albic Luvisol and Chernozem, plants grown in MP-amended Luvisol showed no statistical differences in their chl and car contents compared to the control (Fig. 4.6a).

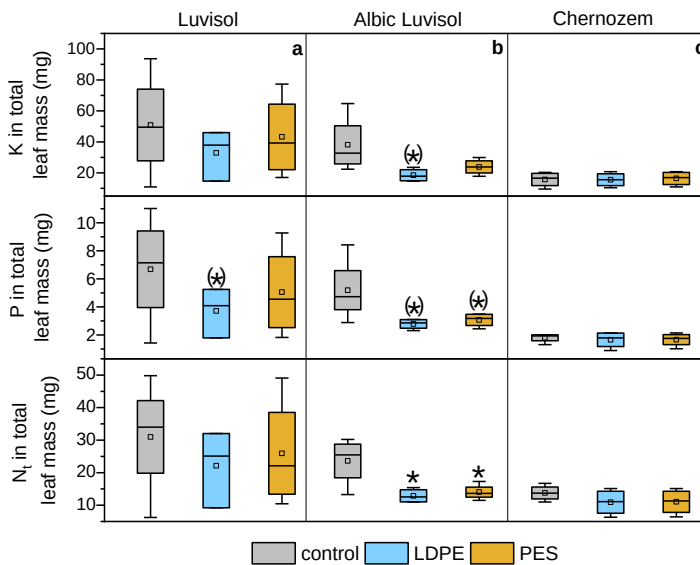


Fig. 4.6. Chlorophyll content, including chlorophyll-a (chl_a) and chlorophyll-b (chl_b), and carotenoid (car) content in total leaf dry mass of winter wheat, grown for 42 days in a) the loamy silt Luvisol, b) the loamy sand Albic Luvisol, and c) the silty clay loam Chernozem, respectively. LDPE refers to plants grown in 0.4% w/w low-density polyethylene particles and PES refers to plants grown in 0.4% w/w polyester microfibers. Control is the respective soil type without MP addition. Boxes are the interquartile range (IQR) from the 25th to 75th percentile, lines inside boxes indicate the median, white squares indicate the mean values, upper and lower whiskers are 1.5 times the IQR, and markers are individual data points. Asterisks show significant difference to the control ($p \leq 0.05$), while asterisks in parentheses show weak significance ($p \leq 0.1$), tested with a heteroscedastic, paired t -test.

4.3.3.2. Nutrient content in leaves

The analysis of K, P and N_t content in leaves revealed different patterns among different soil types (Fig. 4.7). Overall, leaves from plants grown in the loamy silt Luvisol had the highest nutrient contents (Fig. 4.7a), albeit with the highest variability between replicates. Conversely, leaves of plants grown in the loamy sand AL (Fig. 4.7b) showed slight reductions with less variability between replicates, while those of plants grown in the silty clay loam Chernozem (Fig. 4.7c) had the lowest values of all nutrients tested. Leaves of plants grown in LDPE-treated Albic Luvisol exhibited lower nutrient contents compared to the MP-free control, with a $51\% \pm 10\%$ reduction in K ($p = 0.06$), $46\% \pm 6\%$ reduction in P ($p = 0.06$), and $45\% \pm 8\%$ reduction in N_t contents ($p = 0.03$). The presence of PES in Albic Luvisol led to a slightly significant reduction in P content ($p = 0.08$) and a significant reduction in N_t content ($p = 0.04$), both decreased by $41\% \pm 9\%$ compared to the control. In Luvisol, only LDPE particles caused a slight reduction in P contents in leaves ($-44\% \pm 22\%$, $p = 0.08$). In contrast, plants grown in Chernozem soil did not exhibit significant changes in their leaf nutrient contents in response to the addition of the two MP types.

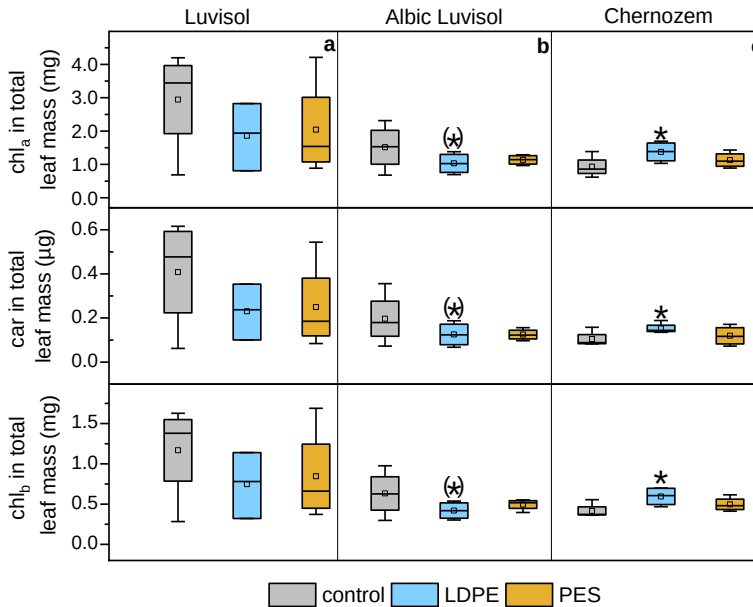


Fig. 4.7. Nutrient content in total leaf dry mass of winter wheat harvested after 42 days and grown in a) the loamy silt Luvisol, b) the loamy sand Albic Luvisol, and c) the silty clay loam Chernozem, respectively. LDPE refers to plants grown in 0.4% w/w low-density polyethylene particles and PES refers to plants grown in 0.4% w/w polyester microfibers. Control is the respective soil type without MP addition. Boxes are the interquartile range (IQR) from the 25th to 75th percentile, lines inside boxes indicate the median, white squares indicate the mean values, upper and lower whiskers are 1.5 times the IQR, and markers are individual data points. Asterisks show significant differences to the control ($p \leq 0.05$), while asterisks in parentheses show weak significance to the control ($p \leq 0.1$), tested with a heteroscedastic, paired *t*-test.

4.3.4. Plant biomass and root-to-shoot ratio

In the loamy silt Luvisol, the LDPE treatment significantly decreased root biomass ($p = 0.03$), although this did not translate into changes in shoot biomass or the root-to-shoot ratio (Table 4.2). Plants in the PES microfiber-amended Luvisol exhibited slightly reduced root biomass ($p = 0.07$) and a corresponding slight decrease in the root-to-shoot ratio ($p = 0.1$). In loamy sand Albic Luvisol, plants grown in both the LDPE and PES treatments showed significantly decreased root biomass ($p_{LDPE} = 0.03$, $p_{PES} = 0.009$) and slightly reduced shoot biomass ($p_{LDPE} = 0.05$, $p_{PES} = 0.1$). Among these, only plants grown in the PES-amended sandy AL soil showed a significant reduction in their root-to-shoot ratio ($p = 0.03$). Plants grown in the LDPE-amended Albic Luvisol did not differ statistically from the control, as both root and shoot biomass were equally reduced, leaving the ratio unaffected.

As expected from the root length data, in the silty clay loam Chernozem, the LDPE amendment led to slight increases in root biomass by $52\% \pm 23\%$ ($p = 0.07$) and significantly increased shoot biomass ($+65\% \pm 16\%$, $p = 0.003$). However, the PES amendments only led to significantly increased shoot biomass ($+43\% \pm 14\%$, $p = 0.02$), while root biomass remained unaffected. Nevertheless, the root-to-shoot ratio remained the same, supposedly due to the high variability of the control replicates.

Table 4.2. Root and shoot biomass (in mg) and root-to-shoot ratio of winter wheat plants grown for 42 days in loamy silt Luvisol, loamy sand Albic Luvisol and silty clay loam Chernozem, in the absence of microplastics (control), and with amendment of either low-density polyethylene (LDPE) particles or polyester (PES) microfibers.

	Root biomass (mg)			Shoot biomass (mg)			Root-to-shoot ratio (-)		
	control	LDPE	PES	control	LDPE	PES	control	LDPE	PES
Loamy silt	512 ± 157	213 ± 147 ^a	299 ± 147 ^b	1157 ± 665	633 ± 475	1109 ± 606	0.617 ± 0.351	0.717 ± 0.617	0.281 ± 0.057 ^b
Loamy sand	604 ± 154	322 ± 130 ^a	213 ± 43 ^a	875 ± 359	406 ± 78 ^b	528 ± 95 ^b	0.746 ± 0.198	0.844 ± 0.425	0.415 ± 0.107 ^a
Silty clay loam	270 ± 122	411 ± 62 ^b	184 ± 44	400 ± 86	662 ± 58 ^a	572 ± 56 ^a	0.765 ± 0.516	0.626 ± 0.115	0.324 ± 0.076

^aSignificant difference to the control ($p \leq 0.05$), ^bweak significant difference to the control ($p \leq 0.1$).

4.4. Discussion

The effect of LDPE MP particles and PES microfibers in soil on winter wheat growth and performance was investigated in three different soil types with distinct textures: loamy silt Luvisol, loamy sand Albic Luvisol and silty clay loam Chernozem. After a growth period of 42 days, soil N, P and K content, pH and water content, root length and nutrient content, and leaf chlorophyll, carotenoid and nutrient contents were analyzed. The results clearly show that LDPE particles and PES microfibers at a mass concentration of 0.4% w/w influenced the growth and nutrient uptake of winter wheat. However, the observed effects varied depending on the MP type and, more importantly, on the soil type and texture.

4.4.1. Microplastic effects in loamy silt (Luvisol)

Soil analysis reveals no significant change

At the end of the experiment, a decrease in NO_3^- in the Luvisol was observed, which could be linked to either increased nitrogen uptake or turnover processes, as well as to increased leaching possibly due to heavy rainfall (Appendix C: Fig. C.3). However, the presence of both MP types did not significantly affect soil NO_3^- content in our study. Moreover, the soil analysis in general did not reveal any significant statistical differences in nutrient content, pH or water content in the presence or absence of MP in the loamy silt Luvisol. Only a slight increase in NO_2^- levels in LDPE-treated bulk Luvisol soil was detectable (Fig. 4.3a).

LDPE effects on root growth, soil structure and water availability

In addition, LDPE in Luvisol significantly reduced wheat root length (Fig. 4.1a) and biomass (Table 4.2). Previous research has demonstrated that LDPE particles and fragments present in soil can impact soil structure, potentially reducing the stability of aggregates and fostering the formation of larger soil pores, known as macropores (Krehl et al., 2022). This alteration can accelerate drainage through macropores, as capillary forces are reduced, potentially leading to increased water loss. However, the extent of this effect varies depending on the concentration of LDPE particles and soil texture. For instance, it has been reported that LDPE fragments are capable of disturbing and expanding the soil structure in clay-rich soils with fine texture but not in sandy soils with coarse texture (Krehl et al., 2022). In our study, the loamy silt texture of the Luvisol was at the finer end of the soil texture spectrum. This suggests that LDPE particles at a concentration of 0.4% w/w likely widened the soil texture, creating additional macropores and potentially reduced water availability to roots. As proposed by Daneshian et al. (2021), water flow in soil depends on the structure and connectivity of the pore space within the soil matrix. With additional macropores and less connected pore space, LDPE

particles may have disrupted the continuous water phase within soil pores, which is essential for efficient water flow through the pore matrix. Roots growing in this altered loamy silt texture, characterized by increased macropores, may have more space for expansion but could have also faced reduced water and nutrient access compared to roots in the original texture, leading to impaired root development as observed in our results.

Nutrient reductions in LDPE-treated roots

Furthermore, the P, K and N_t content of the roots grown in LDPE-amended Luvisol were significantly reduced (Fig. 4.2a). The significant reduction in P content of the roots was mirrored in slightly reduced P contents of the leaves (Fig. 4.7a). This is plausible as there is a strong relationship between root and shoot P levels in winter wheat. The regulation of phosphate influx involves complex root-shoot interactions, suggesting that P levels in roots directly influence those in leaves (AdAlsteinsson et al., 1994). In contrast, there was no significant difference in the root-to-shoot ratio of the LDPE treatment compared to the control. Literature indicates that while many MP types negatively impact root biomass, shoot biomass may either increase or remain unchanged, depending on soil type and MP concentration (Chen et al., 2025a). Consequently, the root-to-shoot ratio is rarely significantly affected, as observed in our study. The fact that roots also showed reduced N_t contents at early growth stages of the plants suggests a reduced nitrogen availability, as also indicated by Farow et al. (2024).

Textural changes by PES microfiber addition and effects on root growth

In the presence of PES microfibers in loamy silt soil, almost the same results for root development and plant performance were obtained as in the presence of LDPE particles. This was reflected in reduced root elongation (Fig. 4.1a) and a slight reduction in root biomass (Table 4.2) as well as significantly reduced root nutrient contents (Fig. 4.2a), but without significant effects on leaf biomass or nutrient content (Table 4.2, Fig. 4.7a). Other studies also indicated a strong dependency of the MP effects, e.g., for PES microfibers, on soil physico-chemical properties, such as texture, clay mineralogy, organic matter content, as well as aggregation (Ingraffia et al., 2022a). For example, while in a clay-rich Vertisol PES microfibers increased macroporosity, no changes in the capacitive indicators of soil physical quality (including macroporosity) were observed in a loamy Entisol after the addition of PES microfibers (Ingraffia et al., 2022a).

However, our results indicate that the presence of PES microfibers in the loamy silt Luvisol led to changes in texture, potentially by restructuring the soil, enhancing the porosity and leading to accelerated water loss, through which the roots may have faced significantly lower water and nutrient availability. The reduction of soil aggregates, particularly in microfiber-amended soil, has already been demonstrated due to the microfibers' linear shape, hydrophobic nature and flexibility, which impedes

macroaggregate formation from microaggregates (Lozano et al., 2021b). This rearrangement of soil pores influences soil-water characteristics, as larger pores facilitate soil drainage.

4.4.2. Microplastic effects in loamy sand (Albic Luvisol)

Changes in soil nutrient content in PES microfiber-treated loamy sand

After adding PES microfibers, the loamy sand Albic Luvisol showed reduced NO_2^- and P_{av} levels (Fig. 4.3b). Moreover, plants grown in the loamy sand mixed with PES microfibers also had reduced N_t and P contents in their root biomass (Fig. 4.2b). The enhanced NO_2^- level in soil could be linked to an imbalanced nitrification process in soil, potentially reducing nitrogen availability for plants. NO_2^- is an intermediate product of the nitrification process, which produces NO_3^- as end product, the main form of nitrogen used by modern cereal cultivars (Lyu et al., 2022). Enhanced levels of this intermediate product could be linked to different causes, such as incomplete nitrification (i.e., inhibition of nitrite oxidizers), anaerobic ammonium oxidation to NO_2^- or increased denitrification rates, reducing NO_3^- to NO_2^- (Giles et al., 2012).

Impact of PES microfibers on oxygen availability for soil microorganisms and roots

Interestingly, soil water content and soil pH of the PES microfiber-treated Albic Luvisol were significantly increased compared to the control, indicating on the one hand less aeration due to more water-filled soil pores, and on the other hand changed environmental conditions for microorganisms. Sandy soils typically show good water drainage associated with comparably low water-holding capacity due to the texture-related macroporosity. Recent studies have shown that the addition of microfibers tends to increase water-holding capacity and water retention in sandy soils (de Souza Machado et al., 2018; de Souza Machado et al., 2019; Lozano & Rillig, 2020). As a result, air-filled pores are reduced, potentially limiting the oxygen availability for aerobic microbial processes and potentially promoting anaerobic microbial processes.

Considering that NO_2^- levels were significantly elevated in this sandy soil, it is likely that nitrite-oxidizing bacteria (NOB), which require oxygen as a terminal electron acceptor to oxidize NO_2^- to NO_3^- , were limited in their NO_2^- oxidation rates due to limited oxygen availability (Daims et al., 2016; Beman et al., 2021). These NOB prefer a neutral to alkaline pH between 7.0 and 7.8 (Daebeler et al., 2020). In our study, the pH of the PES microfiber-amended Albic Luvisol ranged between 5.7 to 6.0, i.e., far from the preferred pH range of NOB, but still significantly less acidic than the control soil, which had a mean pH of 5.5 (Fig. 4.4b). As a result, the significantly reduced N content of roots grown in Albic Luvisol with PES microfibers could be linked to a decreased nitrification rate due to a decreased NOB activity, providing less NO_3^- for the plant.

However, the reduced root growth (Fig. 4.1b) in the PES-amended Albic Luvisol can hardly be assigned to the factor of a decreased NOB activity alone. As the relationship between nitrogen content and root length in winter wheat is complex and depends on various factors (i.e., pH, interaction with other nutrients, feedback mechanisms; Shi et al., 2023), it can be assumed that also root gas exchange might have been impaired due to the reduced air-filled pores in the microfiber-amended Albic Luvisol, leading to adverse conditions for roots. Overall, the changed environmental conditions as stated above could have been the driving factors for this strong reduction in root growth by 85%.

Reduced phosphate and potassium contents in PES-treated plants

The notable increase in soil P_{av} content found in PES microfiber-treated AL might be attributed to a reduced P_{av} uptake efficiency by the roots, possibly due to the above-mentioned alteration of the soil water status. This assumption is further supported by the significant decrease (-64%) in root P content (Fig. 4.2b) and the marginal decline in leaf P content (Fig. 4.7b). However, the reduction in root K content did not correspond to a similar decrease in leaf K content. While speculative, it can be assumed that due to limited root uptake, plants may reallocate K from roots to younger, actively growing tissues like leaves to maintain adequate K levels and therefore crucial metabolic processes, such as photosynthesis (Sustr et al., 2019). This reallocation may have been the reason for reduced root development and K content, prioritizing leaf growth instead.

Reduced root-to-shoot ratio in PES microfiber-treated plants

The root-to-shoot ratio of plants provides insight into the plant's resource allocation and growth dynamics. The root-to-shoot ratio of winter wheat grown in Albic Luvisol mixed with PES microfibers was reduced by on average almost half (Table 4.2). That was due to the strong reduction in root biomass, but only low reduction in shoot biomass. In our study, all mean root-to-shoot ratios of winter wheat plants were below one. This trend towards lower root-to-shoot ratios in modern wheat cultivars likely reflects adaptation and selection under favorable conditions during breeding with sufficient water and nutrient supply (Zhu et al., 2019b). Nonetheless, we could show that plants grown in PES microfiber-amended sandy soil had an even lower root-to-shoot ratio, which indicates that plants invested even less in root growth than in shoot growth, although shoot growth was reduced compared to the control.

Reduced leaf nitrogen content in PES microfiber-treated plants

The roots in PES microfiber-treated Albic Luvisol may have experienced a reduced oxygen availability due to the increasing effect of the PES microfibers on soil water retention, which can affect root nutrient uptake (Pais et al., 2023). The significantly reduced leaf nitrogen content (Fig. 4.7b) supports this assumption, although the leaf chlorophyll and carotenoid contents were not significantly affected

(Fig. 4.6b), indicating that photosynthetic performance was most probably not affected despite the reduced nitrogen content.

LDPE effects on root growth and water availability

LDPE particles in loamy sand Albic Luvisol also reduced root length (Fig. 4.1b) and biomass (Table 4.2) by on average almost half. Moreover, roots showed slightly reduced P and K contents and were significantly reduced in their nitrogen content. A reason for the reduced root growth and impaired nutrient acquisition might be the direct interaction of LDPE particles with the sandy soil texture, thereby indirectly affecting water and root conditions in the soil. As already demonstrated previously, increasing concentrations of LDPE film fragments in sandy soils created larger pores between soil particles, thereby accelerating water loss (Krehl et al., 2022).

As this effect may be strongly correlated with the amount and material properties of the LDPE (film vs. particle), it could not be unambiguously revealed whether the LDPE particles used in our study led to comparable effects. Nevertheless, it is likely that LDPE particles interact directly with the soil matrix, blocking water pathways through the soil pore space and limiting nutrient diffusion within the water phase, resulting in a negative impact on root growth. This has already been demonstrated by Tötze et al (2024), where plastic film fragments in soil impeded effective water pathways and significantly affected water flow in the soil, confirmed by X-ray tomography and neutron imaging.

Implications of LDPE effects on plant nutrient uptake and biomass

As a result, shorter roots take up less nutrients and provide less nitrogen for chlorophyll synthesis than longer roots (Fig. 4.6b), which could impair photosynthesis. Impaired photosynthesis leads to reduced supply of photosynthates, reflected in reduced biomass of both roots and shoots (Table 4.2), as it was observed for plants grown in the LDPE-amended sandy soil.

4.4.3. Microplastic effects in silty clay loam (Chernozem)

Increased root length in LDPE-treated silty clay loam

In the silty clay loam Chernozem soil, it was particularly striking that the LDPE treatment resulted in greater root length compared to the control. In addition, due to significantly higher root K levels and increased *chl/car* content in the leaves, plants grown in the LDPE-amended Chernozem soil grew better than those in the control. We assume that the high water content of the silty clay loam Chernozem generally led to more anoxic or microoxic conditions for the roots compared to the other two soils, as reflected in the reduced root length of the control. Furthermore, it can be concluded that LDPE particles provided better aeration of the root zone compared to the control without LDPE by creating

macropores. This would explain why significantly higher root growth was observed in LDPE-treated Chernozem (Fig. 4.1c), but only marginal increases in root biomass (Table 4.2) and slightly reduced NO_3^- content of the soil. The reduced NO_3^- content of the soil could be linked to increased NO_3^- absorption due to the greater root length in this treatment compared to the control (Fig. 4.1c).

LDPE-induced reduction of waterlogging and positive effects for root growth

Previous studies have shown that LDPE particles in soil can decrease soil bulk density and reduce the stability of soil aggregates, particularly when concentrations exceed 0.2% w/w and when MP particles are larger than the soil particles, thereby disturbing aggregate formation (Joos & De Tender, 2022; Krehl et al., 2022). Notably, our LDPE particles, ranging from 300-600 μm , exceeded the size of soil particles found in the clay and silt fraction of the Chernozem, which typically range from 2-50 μm for silt and <2 μm for clay. Consequently, soil water could be transported along the plastic surfaces at a significantly faster rate than between soil pores. Thus, the presence of LDPE particles could have created pathways and channels in the soil, enhancing hydraulic conductivity (Wan et al., 2019) and consequently reducing the negative effects of waterlogging, such as anoxic conditions for the roots. Guo et al. (2022) and Shafea et al. (2023) both reported an increase in macropores and therefore a reduction in water retention due to PS (Shafea et al., 2023) and PP (Guo et al., 2022) addition to soil. Moreover, the impact of macropore addition on water retention would be more pronounced in clay-rich soils with higher carbon content, such as our silty clay loam Chernozem, compared to sandy or loamy soils (Guo et al., 2022). The potential creation of additional pathways could have stimulated roots to grow deeper into the soil compared to the control, as they might have faced better root conditions, such as better aeration of the subsoil. The alteration of the pore space both between and within aggregates, influencing water and air movement as well as root development, could explain the observed greater root length in the silty clay loam Chernozem containing LDPE particles.

Increased potassium absorption due to enhanced root system in LDPE-treated silty clay loam

The root K content was significantly higher in the LDPE treatment, potentially due to the larger root system, providing more surface area with more absorption sites for K^+ ions (Bell et al., 2021). Also, a higher number of root hairs and finer branching increase the effective surface area for nutrient absorption (Bell et al., 2021), which was only seen for K absorption in our study. Moreover, an extended root system could have released more exudates like organic acids, helping to solubilize K from mineral sources in the soil, thereby increasing the availability for absorption (Xu et al., 2021b). Consequently, the presence of LDPE particles likely mitigated the impairment of root absorption and the release of exudates near the root zone, compared to the control.

Resource allocation and plant growth in LDPE-treated silty clay loam

Although there was no indication of increased nitrogen uptake from the silty clay loam soil containing LDPE particles (Fig. 4.3c), nor of elevated leaf nitrogen content (Fig. 4.7c), the chlorophyll and carotenoid contents were increased (Fig. 4.6c). The increase in leaf biomass and a root-to-shoot ratio of 0.6 (Table 4.2) suggest that the plants primarily allocated their resources to aboveground tissue, indicating that they were adequately supplied with the required nutrients by the extended root system.

PES microfiber effects on soil structural changes in silty clay loam

For the PES treatment, alterations of the root systems were only minor, although a widening of the fine pore structure of the clay-rich soil by the PES microfibers can be assumed, thereby decreasing soil bulk density and increasing macroporosity, as it was already shown for a clay-rich Vertisol (Ingraffia et al., 2022a). Roots grown in PES microfiber-treated silty clay loam Chernozem were slightly longer than in the control (Fig. 4.1c), but with no significant difference in root biomass (Table 4.2) compared to the control. Interestingly, roots in the PES treatment showed significantly reduced K, P and N contents (Fig. 4.2c), completely opposite to the LDPE treatment. The increased macroporosity in the presence of PES microfibers could have resulted in a disruption of water phase continuity along the soil pore space, thereby reducing diffusion and nutrient exchange between roots and soil water. Although increased macroporosity and reduced bulk density have a positive effect on microbial communities, as they have been shown to lead to more diverse microhabitats and greater bacterial diversity (Li et al., 2002; Carson et al., 2010), the disruption of the continuous water phase by MP fibers is likely to have a greater negative impact on microbial diversity in the future. A discontinuous water phase could result in restricted nutrient and oxygen diffusion or a disruption in the exchange of genetic material within the microbial community.

PES microfibers and soil pH reduction

Finally, the soil pH was significantly lower after PES microfiber addition (Fig. 4.4c). Since this phenomenon is not reported in the literature, it can only be assumed that the pH changes are a secondary effect of the altered soil structure. This alteration potentially influences water retention, oxygen availability and nutrient cycling, leading to different pH conditions over time. Changes in soil pH caused by MP contamination have been shown to significantly affect key enzymes involved in nutrient cycling, reducing their activity. In particular, β -D-glucosidase, cellobiosidase, and N-acetyl- β -glucosaminidase activity were decreased in the presence of MP fibers, foams and films (Zhao et al., 2021), which was shown to negatively affect microbial-mediated nutrient cycling (Guo et al., 2024). In addition, it was also stated that biodegradable MP would impact microbial metabolic pathways more drastically than conventional MP (Guo et al., 2024).

4.5. Conclusion

While soil texture has a direct effect on plant growth because it determines water retention and soil pore size, and thus water and oxygen availability and space for roots, we were able to show that the addition of MP alters the effect of soil texture on plant growth. This is novel, since previous studies of the effects of MP on plant growth have often not included different soil textures but have transferred observed effects to any soil type. We propose that the soil type is critical in determining the effect of MPs on soil macroporosity, water and oxygen availability and plant growth. However, because the results in the literature vary depending on whether fine-textured soils with high clay content or coarse-textured soils with high sand content were studied (e.g., Guo et al., 2022; Ingraffia et al., 2022a; Krehl et al., 2022), a general assessment of MP effects and development of generic mitigation options for the wide range of agricultural soils is premature. Further research with soils of defined textures is needed to fully understand MP-soil interactions and their implications for plant health and ecosystem dynamics.

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CHAPTER 5 – Microplastic mineralization rate in Fenton reactions depends on polymer type

Based on:

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Abstract

Despite microplastics (MPs) being highly inert pollutants, Fenton-type reactions – using hydrogen peroxide (H_2O_2) and iron(II) ions (Fe^{2+}) – may effectively initiate chain cleavage and induce mineralization. However, mineralization rates and mechanisms for different MP types at varying Fenton reagent concentrations remain unclear. This study examined the mineralization of four MPs – low-density polyethylene (LDPE), polypropylene (PP), polystyrene (PS), and polyester (PES) – by measuring CO_2 release across varying H_2O_2 concentrations. Mineralization rates depended on both polymer type and H_2O_2 concentration. PES showed the highest rate of mineralization, followed by PP, while LDPE and PS exhibited the lowest rates. Increased H_2O_2 concentrations enhanced CO_2 release and accelerated reaction saturation, especially for PES and PP, suggesting efficient mineralization due to elevated hydroxyl radical ($\text{OH}^\bullet$) production. In contrast, PS and LDPE showed no significant increase in mineralization above certain H_2O_2 levels. Despite visible Fe-oxide precipitates, scanning electron microscopy did not provide evidence of surface changes associated with mineralization. Contrary to expectations, aromatic PS was less susceptible to Fenton mineralization than aliphatic PP, possibly due to structural factors. The study also emphasizes the importance of radical recombination and scavenging at high $\text{OH}^\bullet$ concentrations, which can lower mineralization efficiency. Non-integer reaction orders suggest a multi-step mineralization process influenced by both polymer structure and radical dynamics. These findings underscore the high environmental persistence of MPs, as natural mineralization, such as by fungi utilizing Fenton-like mechanisms, occurs at even slower rates than those observed under controlled lab conditions.

5.1. Introduction

Plastic debris smaller than 5 mm, known as MP, is increasingly entering the environment, particularly the oceans and soils. The widely used synthetic material has grown in production since the 1950s, with 400 million tons of plastic produced globally in 2022 (Garside, 2024). About 12.5 million tons of plastic is used in agriculture each year, of which a significant portion likely to fragment into MP in soil over time (Morrison, 2024). Most MP found in soil is composed of polyethylene (PE), polypropylene (PP), polyvinyl chloride (PVC), polyethylene terephthalate (PET), or polystyrene (PS) (Fan et al., 2023; Lehmann et al., 2021). They enter the soil through various pathways, e.g., sewage sludge and compost application, plastic mulching, littering, tire wear, and atmospheric deposition (Büks et al., 2020).

Despite their chemically inert character, microbial mineralization of MP has been found in soil. For instance, several bacterial genera (e.g., *Bacillus*, *Rhodococcus*, *Pseudomonas*) have shown the ability to mineralize PE or PS, though reported mineralization rates are typically low, with mass losses of 3% to 6% for conventional and up to 23% for modified plastics in soil within 30 days (Tribedi & Sil, 2013; Mohan et al., 2016; Auta et al., 2017). Similarly, fungi have been reported to modify synthetic polymer structures or to produce surface cracks on PE and polyurethane (Yamada-Onodera et al., 2001; Russell Jonathan et al., 2011; Chen et al., 2016; Sangale et al., 2019).

The ability of fungi to mineralize MPs has been demonstrated particularly through their utilization of the Fenton reaction (Krueger et al., 2015; Sánchez, 2020; Schlosser, 2021). This reaction was first discovered by H. J. H. Fenton in 1894, when he observed a significant enhancement in the oxidative potential of hydrogen peroxide (H_2O_2) in the presence of Fe^{2+} ions (Fenton, 1894). Later, in 1934, Haber and Weiss (1934) proposed the mechanism behind this process, now known as the Fenton reaction (Eqn. 1), due to its strong oxidative capabilities to generate hydroxyl radicals ($\text{OH}^\bullet$) (Haber and Weiss, 1934). Since then, the combination of H_2O_2 and Fe^{2+} in aqueous solution has been known as Fenton's reagent. Once $\text{OH}^\bullet$ radicals are produced, they undergo a series of propagation reactions, leading to the formation of additional reactive oxygen species (Eqns. 2-5). The Fenton reaction (Eqn. 1) and the regeneration of Fe^{3+} ions back to Fe^{2+} in the presence of superoxide radicals ($\text{O}_2^{\bullet-}$) (Eqn. 5) culminate in the Haber-Weiss reduction (Eqn. 6).

Wood-decaying brown-rot fungi, for example, naturally utilize the Fenton reaction to compensate for their lack of ligninolytic enzymes required to break down polysaccharides in wood (Hatakka, 2001; Shah et al., 2016). The generation of the highly reactive and non-specific $\text{OH}^\bullet$ radicals during Fenton reaction allow for the depolymerization of long-chain aliphatic structures by abstracting hydrogen atoms and can also mineralize aromatic hydrocarbons (Palmroth et al., 2006) and organochlorines (Laurent et al., 2012). Ectomycorrhizal fungi, such as *Paxillus involutus* switch to

protein decomposition as their nitrogen source, using the Fenton reaction to break down solid organic matter during ammonia depletion (Op De Beeck et al., 2018).

However, if fungal-induced Fenton reactions are to be used for the remediation of soils contaminated with MPs, two key limitations arise; (i) soil carbonates act as radical scavengers (Watts et al., 2005), and (ii) a continuous Fe^{3+} to Fe^{2+} reduction system is required, typically via extracellular metabolites such as hydroquinones, or reductive enzymes (Baldrian & Brabcová, 2008; Krueger et al., 2015). Furthermore, some fungi (e.g., *Pleurotus ostreatus*) only mineralize plastics effectively after pre-treatment such as UV or heat exposure (da Luz et al., 2013; da Luz et al., 2014; da Luz et al., 2015). Identifying effective fungal strains thus remains challenging.

In addition to biotic systems, the purely chemical Fenton reaction has been applied directly to mineralize MPs. For instance, thermal Fenton treatments mineralize cosmetic PE MPs in water (Hu et al., 2022), while Ortiz et al. (2022) reported weight losses of around 10% for PE, PET, PVC, PP, and expanded PS (EPS) after 7.5 hours of treatment. Significant surface changes due to the introduction of oxygenated groups made the MPs more acidic and hydrophilic. The study noted progressive size reduction and more effective oxidation of polymers with aromatic rings (EPS and PET) compared to PP and PVC. In contrast, Tagg et al. (2017) found no significant chemical or size alterations of PE, PP, PVC, and nylon MP, though they suggested that the Fenton reaction could be suitable as a pre-treatment method for MP analysis in wastewater. Interestingly, Ortiz et al. (2022) utilized harsh reaction conditions (80 °C) and $\text{Fe}(\text{NO}_3)_3 \cdot 9 \text{H}_2\text{O}$ as the iron source, while Tagg et al. (2017) used ambient conditions and FeSO_4 . Collectively, these studies demonstrate that while Fenton reaction can mineralize MPs, efficiency varies depending on factors such as polymer type, particle size, reaction conditions (temperature, pH, reagent concentrations), and treatment duration.

The aim of this study was to demonstrate that, while MP mineralization via the Fenton reaction under laboratory conditions can be monitored through a consistent release of CO_2 , with reaction kinetics influenced by H_2O_2 concentration and polymer structure, there was no clear evidence of significant $\text{OH}^\bullet$ radical surface mineralization on MPs.

5.2. Materials and methods

5.2.1. Microplastics

Based on literature data on MP contamination of German agricultural soils (Piehl et al., 2018; de Souza Machado et al., 2019), low-density PE (LDPE), PP, PS, and PES microfibers were considered significant for our study. Detailed information about the MPs used in the experiment, including their particle sizes,

microfiber lengths and characterization methods, can be found in Chapter 2, Section 2.2.2. LDPE, PP, and PS were further sieved to 100-355 μm . The particle size distributions of these granular MP types were measured in duplicate via a laser scattering particle size distribution analyzer (Partica LA-960, Horiba, Japan) by suspending the respective powder in ethanol. The mean diameter sizes for LDPE, PP, and PS were $283 \pm 0.3 \mu\text{m}$, $353 \pm 0.8 \mu\text{m}$, and $305 \pm 1.6 \mu\text{m}$, respectively (Appendix D: Fig. D.1).

5.2.2. Fenton's reagent

The Fenton's reagent consisted of H_2O_2 solution (30% w/w, Merck, Darmstadt, Germany) and a $\text{FeSO}_4 \cdot 7 \text{H}_2\text{O}$ solution (Merck, Darmstadt, Germany). All solutions were prepared with Milli-Q water.

Fifteen mL of Fenton's reagent was prepared, consisting of 2 mL of FeSO_4 solution and 13 mL of H_2O_2 solution of different concentrations (0.2, 0.5, and 1 M), keeping the molar ratio of $\text{H}_2\text{O}_2/\text{Fe}^{2+}$ (mol/mol) constant at 100:1. The FeSO_4 solution was prepared by dissolving the respective amount of FeSO_4 salt with Milli-Q water in a 2-mL Eppendorf vial with subsequent vortexing. The H_2O_2 solutions of different concentrations were prepared by diluting the respective volume of concentrated H_2O_2 (30% w/w) with Milli-Q water in a 20-mL volumetric flask. See exact amounts in Table 5.1. The pH of the resulting solutions was 2 to 3, as determined using pH indicator strips.

Table 5.1. Amounts of the different components of the Fenton's reagents contained in 2 mL FeSO_4 solution and 13 mL of H_2O_2 solution (molar ratio of $\text{H}_2\text{O}_2/\text{Fe}^{2+} = 100:1$ mol/mol) for gas emission measurements. Fenton's reagent was used in different molar concentrations of H_2O_2 : F0.2 = 0.2 M H_2O_2 , F0.5 = 0.5 M H_2O_2 and F1 = 1 M H_2O_2 . The corresponding blank consisted of the same amounts and volumes, but without the addition of MP.

Treatment	H_2O_2 mmol	FeSO_4 μmol	$\text{H}_2\text{O}_2/\text{MP}$ mg mg^{-1}
F0.2	2.6	26	1.77
F0.5	6.6	65	4.42
F1	13	130	8.85

5.2.3. Experimental set-up

The CO_2 evolution due to MP mineralization was followed online with an infrared laser absorption analyzer (model G2508, Picarro, Inc., Santa Clara, CA, USA), referred to as "Picarro" in the following, using a glass bowl filled with tap water and a 20-mL glass beaker. The gas inlet and outlet of the Picarro were connected with $\frac{1}{4}$ " PTFE tubing to a 10 cm long household sewage pipe (HTM DN50, inner diameter: 50 mm, inner volume: 17.5 mL), closed gastight with an end cap at the top, which served as measurement chamber. In addition, the measurement chamber also had a third piece of $\frac{1}{8}$ " PTFE tubing open to the environment for pressure equilibration. The beaker acted as the reactor, to which 50 mg of the respective MP type and 15 mL of the Fenton's reagent were added. Once the H_2O_2 and

the FeSO₄ solution had been added, the beaker was placed in the center of the glass bowl filled with tap water, and the measurement chamber was positioned over the beaker. The water in the glass bowl sealed the chamber, ensuring that the whole system was gastight (Appendix D: Fig. D.2). The oxidation reaction of the MP took place at room temperature in the dark. The CO₂ emissions were measured in duplicate over a period of 4 h from the start of the reaction.

5.2.4. Curve fitting and calculations

The CO₂ mixing ratios (in ppm) measured over time (in min) for each treatment were first adjusted to match the initial CO₂ mixing ratio of the blank (without MP addition) by adding or subtracting the offset between treatment and blank, and then the blank values were subtracted from the treatment values. The net CO₂ emissions over time for treatments F0.5 and F1 followed an exponential degradation curve, fitted with the “ExpDec2” function in OriginPro (version 2021b, OriginLab Corporation, Northampton, MA, USA) according to Eqn. (5.8). The CO₂ emission curves for treatment F0.2 were fitted using a polynomial fit in OriginPro, as described by Eqn. (5.9). All fitting curves are provided in Appendix D, Fig. D.3, and definitions of the variables for each treatment are listed in Appendix D, Table D.1.

$$y = A1 \cdot e^{\left(-\frac{x}{t}\right)} + A2 \cdot e^{\left(-\frac{x}{t^2}\right)} + y0 \quad (5.8)$$

$$y = B1 \cdot x + B2 \cdot x^2 + y0 \quad (5.9)$$

To calculate the net release of MP-related carbon I emitted as CO₂ per MP mass (“net(mCO₂-C)” in µg g⁻¹), first the change in CO₂ mixing ratio over time (in ppm min⁻¹) was calculated by dividing the difference in CO₂ concentration between two consecutive time points (in min) by the time interval between these points (Eqn. 5.10):

$$\frac{dc(CO_2)}{dt} \approx \frac{[CO_2]_{n+1} - [CO_2]_n}{t_{n+1} - t_n}, \quad (ppm \text{ min}^{-1}) \quad (5.10)$$

with [CO₂]_{n+1} = concentration of CO₂ (in ppm) at time point n+1 min, [CO₂]_n = concentration of CO₂ (in ppm) at time point n min, t_{n+1} = time (in min) at time point n plus 1 min, t_n = time (in min) at time point n.

Furthermore, the rate of change in CO₂ mixing ratio was used to calculate the amount of CO₂ (in mole) in the headspace (in L) per minute (Eqn. 5.11). From this, the CO₂ emission rate (“flux(CO₂-C)” in mol L⁻¹ min⁻¹) was calculated (Eqn. 5.12). Finally, the net(mCO₂-C), the blank-subtracted CO₂ emission

originating from the breakdown of polymer chains from the respective MP, was determined using the $\text{flux}(\text{CO}_2\text{-C})$ and the mass of MP initially added to the reaction (Eqn. 5.13).

$$\frac{dn(\text{CO}_2)}{dt} = \frac{\frac{dc(\text{CO}_2)}{dt} \cdot p_{\text{atm}}}{R \cdot (T_{\text{av}} + 273.15) \cdot 10^6}, \quad (\text{mol L}^{-1} \text{ min}^{-1}) \quad (5.11)$$

$$\text{flux}(\text{CO}_2 - \text{C}) = \frac{\frac{dn(\text{CO}_2)}{dt} \cdot V_{\text{hsp}} \cdot M(\text{C})}{m(\text{MP})} \cdot 10^6, \quad (\mu\text{g g}^{-1} \text{ min}^{-1}) \quad (5.12)$$

$$\begin{aligned} \text{net}(m\text{CO}_2 - \text{C}) \\ = \text{net}(m\text{CO}_2 - \text{C})_{n-1} + \frac{\text{flux}(\text{CO}_2 - \text{C})_{n-1} + \text{flux}(\text{CO}_2 - \text{C})_n}{2} \\ \cdot m(\text{MP}), \quad (\mu\text{g g}^{-1}) \end{aligned} \quad (5.13)$$

with $n(\text{CO}_2)$ = amount of CO_2 formed in the headspace (mol L^{-1}), t = time (min), $c(\text{CO}_2)$ = concentration in the headspace of CO_2 formed (ppm), p_{atm} = atmospheric pressure (in bar), R = ideal gas constant ($\text{L bar K}^{-1} \text{ mol}^{-1}$), T_{av} = average room temperature ($^{\circ}\text{C}$), $\text{flux}(\text{CO}_2\text{-C})$ = flow of CO_2 -derived carbon in the headspace per time ($\mu\text{g g}^{-1} \text{ min}^{-1}$), V_{hsp} = volume of headspace in tube (L), $M(\text{C})$ = molar mass of carbon (g mol^{-1}), $m(\text{MP})$ = mass of MP added (g), $\text{net}(m\text{CO}_2\text{-C})$ = blank-subtracted mass of CO_2 -derived carbon in the headspace per mass of added MP

The kinetic order can be determined by applying the generalized rate law for a reaction:

$$R = k[A]^n \quad (5.14)$$

where R is the reaction rate, k is the rate constant, $[A]$ is the concentration of reactant A, and n is the order of the reaction with respect to A.

When the concentration of reactant, in this case the concentration of H_2O_2 , is doubled (from 0.5 to 1 M), Eqn. (5.15) applies:

$$\frac{R_2}{R_1} = \frac{k(2[A])^n}{k[A]^n} = 2^n = x \quad (5.15)$$

where R_1 and R_2 are the reaction rates at lower and higher reagent concentration, and x is the factor by which the reaction rate is enhanced.

Solving Eqn. (5.15) for n is shown in Eqn. (5.16):

$$2^n = x \quad (5.16)$$

$$n \log 2 = \log x \Leftrightarrow n = \frac{\log x}{\log 2}$$

5.2.5. Scanning-electron microscopic (SEM) images

The microstructural analysis of the MPs was performed using scanning electron microscopy (SEM, Hitachi SU-8000) at 15 keV. The initial MPs (before treatment with Fenton's reagent) and after the Fenton's reaction were placed onto a double-sided C-tape and mounted onto a SEM sample holder. Before analysis, the samples were sputter-coated with a 10 nm thick gold layer to avoid charging effects. At higher magnification, secondary electron images predominately arising from the surface of the sample were used.

5.2.6. Statistical evaluation

To test for statistical differences between CO₂ emission curves between treatments, a one-way ANOVA was performed using Microsoft Excel (Microsoft Corporation, Version 2016, Redmond, WA, USA).

5.3. Results

5.3.1. CO₂ emission patterns during the Fenton-mediated mineralization of the different microplastics

The net release of CO₂ (measured CO₂ minus blank) confirmed the mineralization capability of the Fenton reaction (Eqn. 5.1). With a molar ratio of H₂O₂ to Fe²⁺ consistently maintained at 100 for all treatments, H₂O₂ was in excess relative to the MP species. For treatments F0.2, F0.5 and F1, this corresponded to 130, 260 and 520 mmol H₂O₂ per gram of MP species, respectively. The reactions were conducted in the dark to prevent the reduction of Fe³⁺ to Fe²⁺ by UV radiation (Yu & Kuzyakov, 2021) which could have reduced the reaction rate. The reaction ceased once H₂O₂ was depleted.

The curves obtained by plotting the net(mCO₂-C) per MP mass (in µg g⁻¹) versus time (in min) displayed a polynomial trend with monotonic increases for treatment F0.2, reaching 349 ± 26 µg g⁻¹, 210 ± 25 µg g⁻¹, 103 ± 13 µg g⁻¹, and 81 ± 35 µg g⁻¹ CO₂-C per MP mass after 240 min for the mineralization of PES, PP, PS, and LDPE, respectively (Fig. 5.1). Different from the mineralization curves of the particulate MP types (Fig. 5.1a-c), the mineralization curve of the fibrous PES (Fig. 5.1d) was notably steeper, suggesting a faster mineralization.

In contrast, the mineralization curves for treatments F0.5 and F1 (Fig. 5.1) exhibited an exponential decay pattern, with an initial rapid increase followed by an asymptotic flattening towards a threshold depending on the MP species and H₂O₂ concentration. Higher H₂O₂ concentrations resulted

in steeper initial slopes of the curves that flattened earlier than the curves recorded at lower concentration of the Fenton's reagent. In the F0.5 treatment, PES mineralization resulted in the highest CO₂-C release per MP mass with $804 \pm 207 \mu\text{g g}^{-1}$ (Fig. 5.1d), while PS, LDPE, and PP showed similar trends after 120 min, with total CO₂-C releases after 240 min of $501 \pm 77 \mu\text{g g}^{-1}$, $377 \pm 26 \mu\text{g g}^{-1}$, and $428 \pm 84 \mu\text{g g}^{-1}$, respectively. In treatment F1, only PP and PES showed significantly higher CO₂-C losses than in the F0.5 treatment. PP mineralization in F1 released $943 \pm 48 \mu\text{g g}^{-1}$ CO₂-C, while PES mineralization released even $1177 \pm 105 \mu\text{g g}^{-1}$ CO₂-C after 240 min. For LDPE, both F0.5 and F1 curves converged after 120 min, with the F1 curve further rising until the end of the measurement period ($437 \pm 9 \mu\text{g g}^{-1}$ CO₂-C), while the F0.5 curve stabilized at $377 \pm 26 \mu\text{g g}^{-1}$ CO₂-C after 240 min (Fig. 5.1a). In contrast, PS mineralization (Fig. 5.1c) in the F1 treatment produced less CO₂-C ($358 \pm 84 \mu\text{g g}^{-1}$) compared to 0.5 M H₂O₂. The F1 curve gradually reached $358 \pm 84 \mu\text{g g}^{-1}$ CO₂-C after 240 min, while the F0.5 curve stabilized at $501 \pm 77 \mu\text{g g}^{-1}$, indicating no further CO₂-C release or MP mineralization.

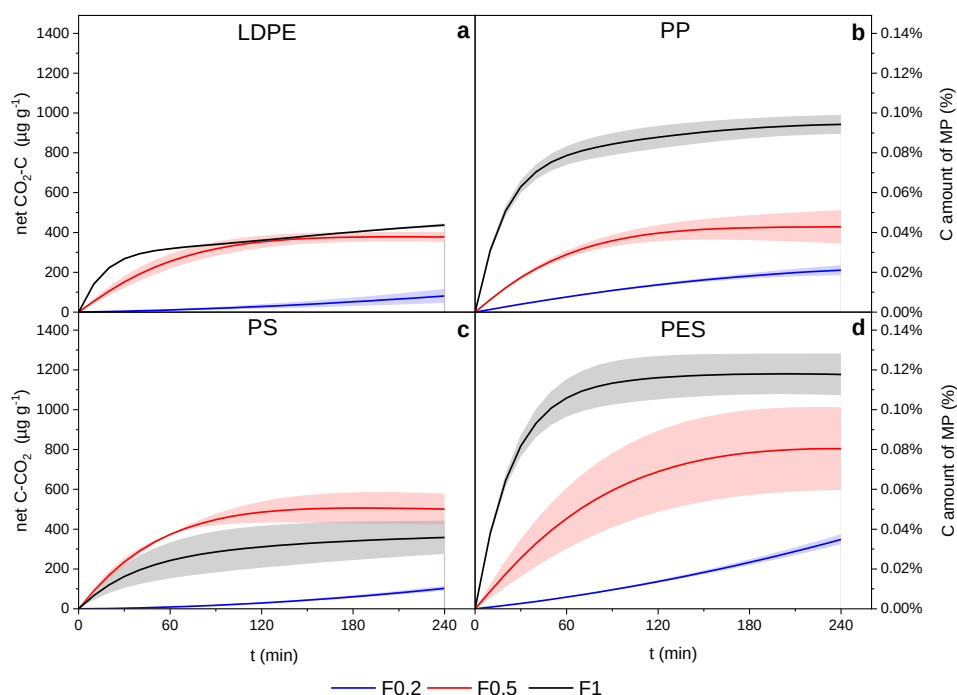


Fig. 5.1. Net release of microplastic (MP)-derived carbon emitted as carbon dioxide (net CO₂-C) per MP mass during the mineralization of different MP types: a) LDPE = low-density polyethylene particles, b) PP = polypropylene particles, c) PS = polystyrene particles, and d) PES = polyester microfibers, using Fenton's reagent of different molar concentrations of hydrogen peroxide (H₂O₂) in aqueous solution: F0.2 = 0.2 M H₂O₂, F0.5 = 0.5 M H₂O₂ and F1 = 1 M H₂O₂. H₂O₂ was added in combination with Fe²⁺ in a molar ratio of H₂O₂:Fe²⁺ = 100:1. The second y-axis shows the amount of carbon (C) being released from the respective MP. All curves show blank-subtracted values.

The comparison of the net CO₂-C release from MP in the different treatments (Table 5.2) revealed that not all MP types become more degradable with higher H₂O₂ concentrations. However, the amount of CO₂-C released from PES microfibers after 240 min of treatment with the Fenton's reagent increased progressively with higher H₂O₂ concentrations, rising from 0.03% to 0.08%, and then to 0.12% of the MP mass with 0.2 M, 0.5 M, and 1 M H₂O₂ in combination with Fe²⁺ (100:1), respectively. A similar pattern was observed for the mineralization of PP, where the release of CO₂-C increased progressively with higher H₂O₂ concentrations, starting at 0.02% of the MP mass with 0.2 M H₂O₂, then to 0.04% with 0.5 M, and reaching 0.09% with 1 M H₂O₂. In contrast, PS and LDPE did not show increased mineralization with higher H₂O₂ concentrations; the CO₂-C release for both types remained around 0.04% with 0.5 M H₂O₂, with no further CO₂-C release at 1 M H₂O₂.

Table 5.2. Amount of microplastic (MP)-derived carbon released as CO₂ (in µg CO₂-C g⁻¹ MP and percent of original MP mass) after 240 min treatment of different MP types (PES = polyester microfibers, PP = polypropylene particles, PS = polystyrene particles, LDPE = low-density polyethylene particles) with Fenton's reagent at different molar concentrations of H₂O₂, 0.2 M H₂O₂ = F0.2, 0.5 M H₂O₂ = F0.5, and 1 M H₂O₂ = F1, amended with FeSO₄ at a molar H₂O₂:Fe²⁺ ratio of 100:1.

Treatment	MP type	MP weight	CO ₂ release from MP	
		µg	µg CO ₂ -C g ⁻¹	% of original MP
F0.2	PES	518	349 ± 26	0.035 ± 0.003
	PP	530	210 ± 25	0.021 ± 0.002
	PS	540	103 ± 13	0.010 ± 0.001
	LDPE	500	81 ± 35	0.008 ± 0.004
F0.5	PES	518	804 ± 207	0.080 ± 0.021
	PP	498	428 ± 84	0.043 ± 0.008
	PS	505	501 ± 77	0.050 ± 0.008
	LDPE	516	392 ± 26	0.039 ± 0.003
F1	PES	560	1177 ± 105	0.117 ± 0.011
	PP	530	943 ± 48	0.094 ± 0.005
	PS	540	358 ± 84	0.036 ± 0.008
	LDPE	500	437 ± 9	0.044 ± 0.001

5.3.2. Mineralization kinetics

The rate of change of CO₂ concentration over time (d(CO₂)/dt) in ppm min⁻¹) was calculated from the net release of CO₂ (in ppm) (Fig. 5.2). For the F0.5 and F1 treatments, the change in CO₂ concentration was highest at the beginning of the reaction and gradually decreased over time. This trend indicates that the mineralization process was most active initially and slowed down as the reactants were consumed (Fig. 5.2). However, in the F0.2 treatment, the CO₂ concentration curves did not show any initial increase. The rate of CO₂ production remained constant close to zero, indicating minimal

mineralization activity under these conditions. This was particularly evident for the polymer types LDPE, PP, and PS (Fig. 5.2a-c), suggesting that the mineralization of these MPs was negligible with the F0.2 treatment. In contrast, PES exhibited a slight increase in the rate of CO₂ production over time. The CO₂ concentration for PES showed a gradual upward trend, reaching a maximum rate of 0.48 ± 0.07 ppm min⁻¹ over the measurement period of 240 min.

The F0.5 treatment showed a lower initial CO₂ production rate compared to the F1 treatment, reaching maximum rates of 2.76 ± 0.46 ppm min⁻¹, 1.55 ± 0.07 ppm min⁻¹, and 2.20 ± 1.33 ppm min⁻¹, respectively, for the mineralization of LDPE (Fig. 5.2a), PP (Fig. 5.2b), and PES (Fig. 5.2d), respectively. Only the mineralization of PS (Fig. 5.2c) in the F0.5 treatment showed a higher initial CO₂ production rate (2.31 ± 0.6 ppm min⁻¹) than in the F1 treatment. Furthermore, the rate of change in CO₂ production in the F0.5 treatment generally showed a less steep decline compared to the F1 treatment, indicating that the reaction was progressing less rapidly. The ANOVA revealed that the mineralization rates of LDPE with F1 and F0.5 was not statistically different ($p = 0.16$).

In the F1 treatment, the initial CO₂ production rates were 11.56 ± 0.09 ppm min⁻¹ for PES and 9.63 ± 0.83 ppm min⁻¹ for PP, whereas they were only 4.29 ± 0.45 ppm min⁻¹ for LDPE and 1.90 ± 0.83 ppm min⁻¹ for PS. Notably, the mineralization of PS in the F1 treatment was less efficient than in the F0.5 treatment, reflected in the lower rates of change in CO₂ mixing ratio (Fig. 5.2c).

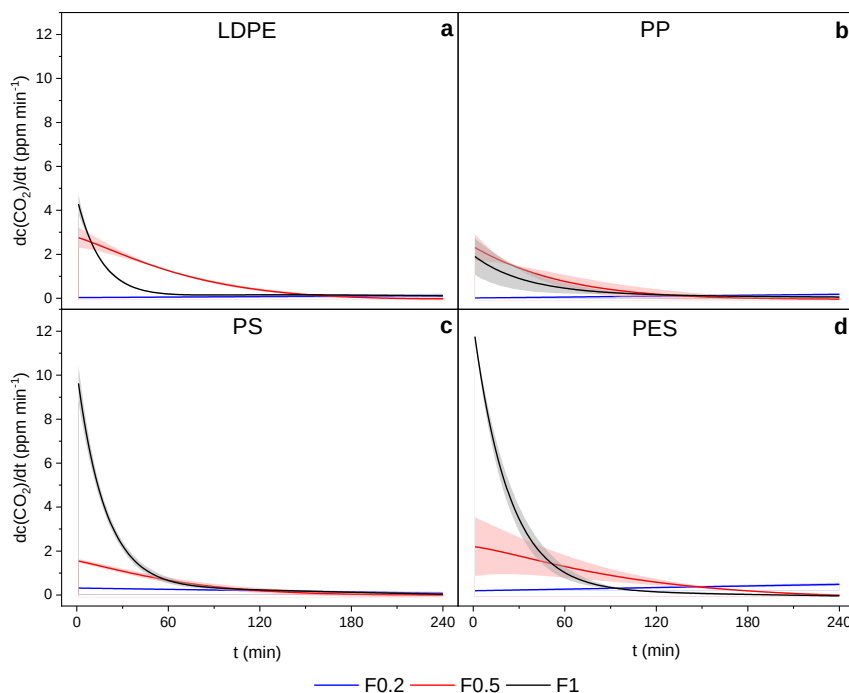


Fig. 5.2. Change in CO_2 mixing ratio (in ppm) over time (in min) ($d(\text{CO}_2)/dt$) for the mineralization of different microplastic types: a) LDPE = low-density polyethylene particles, b) PP = polypropylene particles, c) PS = polystyrene particles, and d) PES = polyester microfibers, with Fenton's reagent of different molar concentrations of hydrogen peroxide (H_2O_2): F0.2 = 0.2 M H_2O_2 , F0.5 = 0.5 M H_2O_2 and F1 = 1 M H_2O_2 . H_2O_2 was added in combination with Fe^{2+} at a molar ratio of $\text{H}_2\text{O}_2:\text{Fe}^{2+} = 100:1$.

In summary, the degradation rate for PP increased 4.9-fold from F0.2 to F0.5 and 6.2-fold from F0.5 to F1. For PES, the mineralization rate increased 11.2-fold from F0.2 to F0.5 and 5.4-fold from F0.5 to F1. In contrast, the mineralization rates of LDPE and PS increased from F0.2 to F0.5 22-fold for LDPE and 13-fold for PS. However, from F0.5 to F1, the mineralization rate for LDPE only increased by 1.6-fold, while the rate for PS decreased by 0.82-fold.

The reaction orders for the mineralization of PES, PP, and LDPE are listed in the Appendix D Table D.2 and were obtained by applying Eqns. (5.14-5.16). Only the factor by which the reaction rate was enhanced from F0.5 to F1 was considered usable, as the reaction curves of F0.2 are not comparable to

those of F0.5 and F1. While the reaction orders (n) for the mineralization of PES and PP was around 2.5 ($n_{\text{PES}} = 2.43$, $n_{\text{PP}} = 2.63$), n was for LDPE 0.64.

5.3.3. Surface structure after the radical attack

After the MP species had been treated with Fenton's reagent of 1 M H_2O_2 for 240 min, the particles and microfibers were analyzed with SEM (Fig. 5.3). None of the MP types exhibited typical signs of mineralization, such as cracks or holes on their surfaces. Small droplets were observed adhering to the surfaces of the LDPE particles, likely due to the precipitation of Fe^{3+} ions. The PS particles, on the other hand, displayed a roughened texture, probably resulting from Fe oxide deposits. Similarly, PES microfibers were coated with an additional layer that may have dried, leading to visible cracks. After the MP species had reacted with Fenton's reagent, the MP particles and microfibers were covered with a visible orange precipitate, which indicates $\text{Fe}(\text{OH})_3$ (Eqn. 5.7).

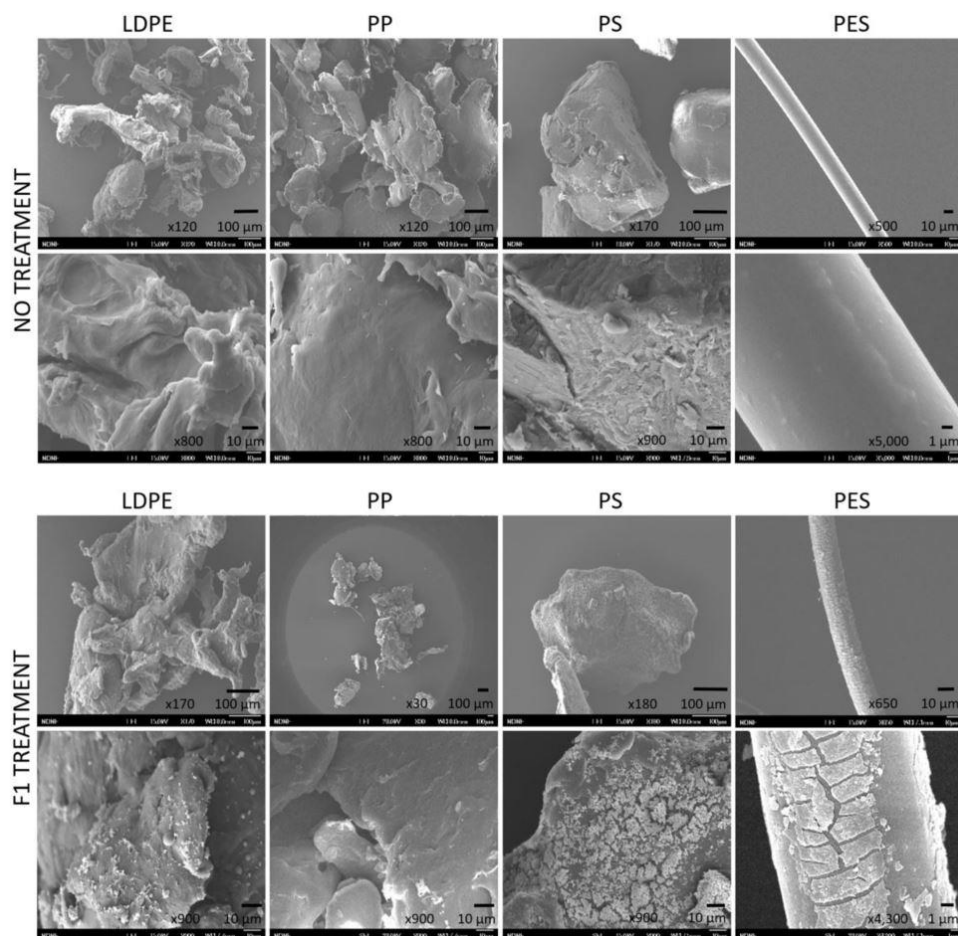


Fig. 5.3. Scanning electron microscopy (SEM) images of microplastic particles (LDPE = low-density polyethylene, PP = polypropylene, PS = polystyrene) and polyester (PES) microfibers before and after the treatment with Fenton's reagent of 1 M hydrogen peroxide (H_2O_2).

To corroborate these results, the MP particles and microfibers were also treated with a 2 M H_2O_2 solution mixed with FeSO_4 and analyzed with SEM (Appendix D: Fig. D.4). Again, no changes were observed on the polymer surfaces.

5.4. Discussion

This study investigated the mineralization of various MPs – LDPE, PP, PS, and PES – using Fenton's reagent with different concentrations of H_2O_2 . The results indicate that the Fenton reaction is effective in degrading MPs, but the efficiency varies depending on the type of polymer and the concentration of H_2O_2 used.

Degradability of MPs by the Fenton reaction

The mineralization of MPs was confirmed by the steady net release of CO₂ during the reaction. The CO₂ emission curves for F0.5 and F1 showed typical saturation curves and varying slopes of the curve depending on the H₂O₂ concentration. The net CO₂ release can be considered as the abstraction of carbon atoms from the polymer chains in the form of CO₂, possibly due to the generation of OH[•] radicals by the Fenton reaction. The temporal development of the CO₂ observed in the release curves indicated that not only more CO₂ was generated per MP species at higher H₂O₂ concentration, but that also the reaction saturated earlier. This is because the reaction between H₂O₂ and FeSO₄ is very fast (exponential increase) and the generation of OH[•] radicals is mainly limited by the depletion of H₂O₂ (saturation region) (Lin et al., 2021). Accordingly, the mineralization of the MP polymer chains and thus the CO₂ generation is strongly related to the concentration of the reactant H₂O₂ at a constant ratio of H₂O₂ and Fe²⁺ ions (in our case 100:1).

The calculated CO₂-C per MP mass varied significantly among the different treatments and types of MPs. PES showed the highest degradability in all treatments, followed by PP, whereas the mineralization of LDPE and PS with Fenton's reagent was less effective, reflected by the slower release of CO₂ compared to PES and PP. In line with this observation, the mineralization of PES and PP, as quantified by the release of CO₂, increased with higher H₂O₂ concentration, indicating more efficient mineralization at presumably higher OH[•] radical concentrations. In contrast, PS and LDPE did not show increased CO₂ release with higher H₂O₂ concentrations, suggesting that the mineralization efficiency for these polymers does not significantly increase beyond a certain H₂O₂ concentration threshold.

In previous studies it was reported that polymers with aromatic rings, such as PS or PET, were found to be more prone to oxidation by Fenton's reagent compared to polymers formed of alkane chains like PE or PP (Ortiz et al., 2022; Yang et al., 2022). In our case, we can confirm that the PES microfibers consisting of PET showed a good mineralization susceptibility, whereas LDPE had a poor susceptibility to mineralization by the Fenton's reagent. However, PS showed a rather low degradability in our study, while the alkane PP, contrary to the general assumption, showed a rather good degradability towards OH[•] radical attack.

In light of the unexpectedly low degradation efficiency of PS observed in our study, it is worth reconsidering whether measuring only CO₂ release is a sufficient indicator of PS mineralization. Other studies examining PS MP mineralization via the Fenton reaction have employed additional analytical methods, which may provide a more comprehensive picture of the mineralization process (Yang et al., 2022). For example, Yang et al. (2022) demonstrated PS degradation by evaluating weight loss, changes in the oxygen-to-carbon (O/C) ratio of treated PS, and identifying degradation products, such as

benzene ring derivatives. These methods offer direct evidence of structural changes and the breakdown of PS, which CO₂ measurement alone may fail to capture. Moreover, the study specifically showed that PS treated with a microbe-driven Fenton reaction had a significantly greater weight loss of $0.6\% \pm 0.1\%$ after 14 days compared to the radical-based Fenton reaction alone. This suggests that microbial activity facilitated more substantial alterations to the aromatic structure of PS, pointing to degradation beyond CO₂ release alone. Therefore, measuring CO₂ exclusively may not be the most effective approach to validate PS mineralization, as it could overlook other significant changes to the material's structure and composition that alternative methods would reveal.

The reason why LDPE turned out to be less degradable but PES very well degradable in our study can be explained by the chemical nature of their structures (e.g., alkane vs. polyester) and the reaction mechanism of OH[•] radical attack. Alkanes are saturated hydrocarbons with only single bonds between carbon atoms, making them chemically unreactive. They lack functional groups such as double bonds or aromatic rings that can be readily attacked by OH[•] radicals generated in the Fenton reaction. In contrast, polyesters contain ester functional groups (–COO–) that are more reactive towards oxidative attack (Zandieh et al., 2024). In addition, the OH[•] radical is more effective at attacking weaker bonds or bonds adjacent to electron-withdrawing groups, which are more common in polyesters. Polyesters can form oxygenated functional groups (e.g., hydroxyl, carbonyl) upon reaction with OH[•] radicals. These groups can further facilitate the breakdown of the polymer chain through oxidative cleavage (Zandieh et al., 2024). Alkanes, however, do not readily form such intermediates, making their degradation less efficient. Furthermore, polyesters, such as PET, contain aromatic rings that can stabilize radical intermediates formed during the Fenton reaction. This stabilization facilitates further oxidative reactions, leading to more efficient degradation (Ortiz et al., 2022; Zandieh et al., 2024). Alkanes lack such stabilizing structures, resulting in lower reactivity and slower degradation rates, as seen for LDPE in our study.

Despite this principle explanation, PP as an alkane showed a comparatively good degradation capacity towards OH[•] radical attack. OH[•] radicals typically initiate degradation by abstracting hydrogen atoms from the polymer chains (Calvert, 1986). Unlike LDPE, PP has methyl groups as side chains on every other carbon atom of its main hydrocarbon chain. This creates tertiary carbon sites, which are the most likely sites for hydrogen abstraction by OH[•] radical attack (Sirtori et al., 2014), as they form more stable radicals. In addition, LDPE has a high degree of both short- and long-chain branching (Malpass, 2010), whereas PP is arranged in a more linear structure. The branched structure in LDPE may hinder the mobility of formed radicals, potentially leading to more localized mineralization (Aldas et al., 2018). On the contrary, the more regular structure of PP might allow for greater radical mobility along the polymer chain (Kausch, 2005).

Fenton reaction kinetics

The rate of CO₂ production over time ($d(\text{CO}_2)/dt$) provided additional insights into the mineralization kinetics. For the F0.5 and F1 treatments, the CO₂ production rate was highest at the beginning and then gradually decreased, indicating rapid initial mineralization, followed by a slower phase as reactants were consumed. The F0.2 treatment, however, showed minimal mineralization activity, except for PES. Generally, higher initial CO₂ production rates indicate a rapid initial mineralization, likely due to a higher concentration of reactive species. A decrease in the change in CO₂ concentration over time (approaching zero) suggests that the reaction is reaching completion or a steady state where the rate of CO₂ production, and therefore mineralization, slows down.

The rate of change in CO₂ release generally declined less steeply in F0.5 than in F1, indicating a more gradual reaction progress, possibly due to fewer available OH[•] radicals compared to the F1 treatment. While the F0.5 treatment resulted in lower initial CO₂ production rates for LDPE, PP, and PES compared to F1, the mineralization of PS showed higher initial rates for F0.5 than for F1.

Although it may seem counterintuitive that a lower H₂O₂ concentration – and subsequently presumably a lower concentration of OH[•] radicals – can lead to more effective mineralization, several factors can explain this phenomenon. At high radical concentrations, radicals can react with each other instead of the target material, a process known as radical recombination, which reduces their effectiveness and decreases the overall mineralization rate (Chamas et al., 2020). In addition, scavenging effects could have occurred. Additives or impurities in the polymer matrix can act as radical scavengers, neutralizing radicals before they cause further mineralization (McNeill, 1997). These scavenging effects are less pronounced at lower radical concentrations, allowing for more effective mineralization, whereas at high radical concentrations, scavenging effects can become more pronounced, potentially slowing down the overall mineralization process. Radicals may also abstract hydrogen atoms from nearby polymer molecules, thereby stabilizing themselves but not contributing to further mineralization (Yousif & Haddad, 2013). The effectiveness of the mineralization process depends on the balance between productive radical reactions (those that effect mineralization) and nonproductive reactions like scavenging. This balance is influenced by factors such as the concentration of radicals, the presence of oxygen, and the polymer's physical state (Crincoli et al., 2020). High radical concentrations can lead to side reactions or the formation of less reactive intermediates, which may slow down the overall mineralization process (Behnami et al., 2022).

The non-integer reaction orders of 2.43 for PES, 2.63 for PP and 0.64 for LDPE (Appendix D: Table D.2) suggest a more complex reaction mechanism. In practice, such a result may indicate:

1. A reaction with multiple steps or competing pathways (Krenos, 2001);
2. The involvement of catalysts or inhibitors affecting the rate in a complex manner (Krenos, 2001);
3. A reaction that does not follow simple integer-order kinetics.

A multi-step reaction is often reflected by non-integer orders when intermediate species are involved. If the intermediate species influence the overall rate, they can cause the observed reaction rate to have a non-integer order. In our study, CO₂ was the final mineralization product due to the OH[•] radical attack on the polymer chains of the MPs. Initially, however, alkyl radicals are formed which propagate through further abstraction and recombination reactions to continue the breakdown of long polymer chains into smaller carbonyl compounds and ultimately into CO₂ (Speight, 2020). Therefore, in reaction media containing Fenton's reagent, other intermediates can be identified, such as aromatic compounds from PS degradation (Liu et al., 2022a) or oxygenated groups after the breakdown of PE (Hu et al., 2022). This was also demonstrated in a study that provided an in-depth analysis of the oxidizing intermediates in Fenton reactions and discussed how their formation and interactions lead to complex kinetics (Meyerstein, 2022).

For practical purposes, the reaction orders of the reaction of PES and PP with Fenton's reagent can be approximated to a reaction order of 2.5 or treated as a mixed-order reaction, while for LDPE a reaction order of 1 appeared to be the closest approximation. However, the exact values provide more precise information about the reaction kinetics and suggests that the reaction mechanism is more complex than a simple elementary reaction. The literature recognizes that kinetic modeling of Fenton and Fenton-like reactions is complex, as the reaction mechanism often does not fit neatly into integer reaction orders, or two different reaction orders are required to adequately describe the reaction kinetics (Guedes et al., 2003; Duesterberg & Waite, 2006; Liu et al., 2007; Wang, 2008). For example, the reaction kinetics of Fenton reactions for wastewater decolorization are described as a combined pseudo-first order process with strong dependence on the initial Fe²⁺ and Fe³⁺ concentrations (Wang, 2008). Another study concluded that the overall kinetics of the Fenton reaction in cork boiling can be described by a second-order followed by a zero-order rate equation (Guedes et al., 2003). In contrast, a fractional order kinetics was identified for a photocatalytic degradation reaction (Wang, 2018).

MP mineralization as indicated by SEM

The SEM images revealed no visible signs of mineralization. Instead, a fine precipitate – most likely iron oxide or iron hydroxide – was observed on the surfaces of the various MPs. This finding aligns with the study of Tagg et al. (2017), who also reported no significant surface changes in MPs following

treatment with Fenton's reagent under ambient conditions, using FeSO_4 as the iron source. In contrast, in another study, clear signs of mineralization, such as cracks on the surface of PS MPs, were visible with SEM imaging (Liu et al., 2022a), but only after intensive treatment involving four cycles of Fenton's reagent (30% H_2O_2 + 0.2 M Fe^{2+} solution) combined with peroxymonosulfate for 5 h in each cycle.

5.5. Conclusion

We were able to show that $\text{OH}^\bullet$ radicals interact uniquely with MPs of different polymeric structures, leading to different and complex mineralization kinetics as indicated by differences in CO_2 release. The efficiency of the Fenton's reagent in degrading various types of MPs was mainly influenced by polymer type and H_2O_2 concentration. PES and PP showed a significant increase in mineralization with increasing H_2O_2 concentrations, while the mineralization of PS and LDPE did not benefit from higher H_2O_2 levels beyond a certain point, in our case a H_2O_2 concentration of 0.5 M. We assume that lower H_2O_2 concentrations resulted in more effective mineralization of PS, possibly due to reduced radical recombination and scavenging effects. The non-integer reaction orders for PES, PP, and LDPE suggest complex reaction mechanisms. However, although a measurable amount of CO_2 was detected, the overall mineralization efficiency of MPs was extremely low, underscoring the strong persistence of MPs in the soil environment. If higher mineralization efficiency is desired, harsh reaction conditions, such as high temperatures (80°C) and an acidic pH around 3, have to be applied, as demonstrated in Ortiz et al. (2022).

Based on these results, we conclude that natural mineralization of MP particles and microfibers in soil, regardless of their polymeric structure, by Fenton-like reactions facilitated by soil fungi may be ineffective. Although Fenton's reagent can be present in soil, as fungal extracellular dehydrogenases release H_2O_2 , which further react with iron species present in the soil, this Fenton's reagent is not as concentrated under natural conditions as the solutions typically used in laboratory settings. Ultimately, the resulting insufficient concentration of reactive radicals leads to an ineffective mineralization of the polymer surfaces of MPs. Moreover, the complex nature of soils can act as a scavenger for $\text{OH}^\bullet$ radicals, further reducing the effectiveness of Fenton-based mineralization. Soil components, such as organic matter, minerals, and antioxidants (Rimmer, 2006), can rapidly react with and neutralize these radicals before they interact with the polymer surface, thereby limiting the mineralization process.

CHAPTER 6 – Synopsis

6.1. Summary

CHAPTER 2 – Soil texture governs the influence of microplastics on soil hydraulic properties:

MP accumulation in agricultural soils alters soil structure and hydraulic properties, including pore distribution, bulk density, and water retention. Different MP types, particularly microfibers, can either promote or hinder soil aggregation, affecting drainage and water availability. Given the strong link between soil hydraulic characteristics and soil health, precise measurements of water movement and retention are crucial. Previous research has shown that MP effects vary depending on soil texture, MP type, and concentration (Guo et al., 2022; Wang et al., 2023). However, inconsistent findings highlight the need for systematic investigations considering both soil type and MP properties.

This study examined the effects of four MP types (LDPE, PP, PS, and PES microfibers) on three soils with different textures (Luvisol: loamy silt, Albic Luvisol: loamy sand, and Chernozem: silty clay loam) at a fixed concentration (0.4% w/w). The study assessed soil water retention and hydraulic conductivity, measuring both K_{sat} and K_{ns} to account for MP hydrophobicity.

The results showed that MP addition significantly influenced soil hydraulic conductivity, with effects varying based on soil type, moisture condition, and MP properties. Soil type had the strongest influence on K_{sat} and K_{ns} , though MP effects were more pronounced in dry conditions due to hydrophobic interactions. PES microfibers generally increased porosity and WHC, enhancing K_{sat} in finer-textured soils but reducing it in coarser sands. PS particles tended to block pores, reducing porosity and water retention while increasing K_{ns} in some cases. In the Luvisol, PES microfibers increased water-holding capacity and aeration, enhancing K_{sat} , whereas PS particles reduced porosity and air-filled pore space, impeding water flow. The Albic Luvisol showed increased porosity and water retention with PES microfibers but decreased air capacity, while PS particles blocked flow paths, reducing both K_{sat} and K_{ns} . In the Chernozem, PES microfibers increased macroporosity and air-filled pore volume, while other MPs had minimal impact.

Overall, MP effects on soil hydraulic properties were highly dependent on interactions between MP type, soil texture, and moisture conditions. PES microfibers often increased permeability in finer soils but restricted flow in sandy soils, while MP particles generally reduced porosity and hydraulic conductivity. These findings emphasize the complexity of MP-soil interactions and the need for further studies to fully understand their impact on soil health.

CHAPTER 3 – Microplastics affect belowground performance in early winter wheat growth, leaving aboveground growth unaffected:

MPs in agricultural soils pose risks to soil structure, water retention, nutrient cycling, and microbial communities, with cascading effects on plant growth. Because root systems closely interact with soil microorganisms through root exudates, disruptions to the soil microbiome can directly affect nutrient availability and plant health. Winter wheat, a major global staple crop, may therefore experience altered root development and nutrient uptake when grown in MP-contaminated soils.

The study assessed the effects of LDPE, PP, PS particles and PES microfibers (at 0.4% w/w) on early winter wheat growth in a rhizotron experiment with a Luvisol soil characterized by a loamy silt texture. Results showed that MP effects on early wheat growth and rhizosphere processes are polymer-specific and mediated through changes in soil properties and microbial networks. PP and PES impaired root growth aligning with previous studies linking PP to decreased germination, biomass, and root system integrity (Esterhuizen & Kim, 2022; Liu et al., 2023c). Furthermore, they disrupted microbial connectivity, including the loss of keystone taxa, while LDPE enhanced microbial network complexity and soil moisture, suggesting greater microbial resilience. PS caused only minor microbial and functional shifts. Despite changes in soil moisture and nitrogen dynamics, aboveground growth and nutrient status were unaffected at this early stage, highlighting the buffering capacity of the rhizosphere microbiome.

Nutrient uptake patterns remained unchanged, with K and P absorption occurring simultaneously. MPs did not significantly alter soil pH. A previous study highlighted that changes in soil pH due to MP contamination depend on the type of MP and exposure duration (Zhao et al., 2021). This may explain the absence of detectable pH effects in this study.

Overall, the results underscore the role of MP physicochemical properties in shaping plant-soil-microbiome interactions and the need for longer-term, field-based studies to assess impacts on soil function and crop productivity.

CHAPTER 4 – Soil texture modifies the impact of microplastics on winter wheat growth:

Research on MP impacts on wheat growth has yielded conflicting results, with some studies reporting increased biomass, others showing reductions, and some finding no significant changes. Soil type appears to influence these outcomes, as MPs alter soil hydraulic properties, aggregation, and microbial dynamics. MPs can also affect the rhizosphere, disrupting nutrient uptake and microbial communities, ultimately influencing plant performance.

To address these uncertainties, this study investigated how LDPE particles and PES microfibers interact with different soil types (loamy silt Luvisol, loamy sand Albic Luvisol, and silty clay loam

Chernozem), at a constant concentration of 0.4% w/w, hypothesizing that MP effects on plant growth depend on polymer type, morphology, and soil texture.

The parameters (soil nutrient content, pH, water availability, root length, and leaf nutrient composition) analyzed after a growth phase of 42 days revealed varying effects on plant performance depending on soil type and MP characteristics. In loamy silt Luvisol, the presence of LDPE particles did not significantly alter soil nutrient content, pH, or water availability. However, LDPE significantly reduced wheat root length and biomass. This reduction may be linked to changes in soil structure, as LDPE particles can increase macroporosity, disrupting the soil's ability to retain water and nutrients. Consequently, roots in LDPE-treated Luvisol exhibited significantly lower P, K, and N content, which was also reflected in slightly reduced P levels in leaves. The presence of PES microfibers led to similar root growth inhibition and nutrient deficiencies, likely due to increased porosity and accelerated water loss. The structure of loamy silt soil may have been altered by PES microfibers, which prevented effective macroaggregate formation, leading to a disruption of the soil's water retention capacity and consequently impairing nutrient uptake.

In loamy sand Albic Luvisol, PES microfibers caused a reduction in NO_2^- and P_{av} levels, along with lower N and P content in root biomass. The elevated NO_2^- levels suggested an imbalance in nitrification, possibly due to limited oxygen availability for NOB. The soil water content and pH were significantly increased, which might have reduced aeration and altered microbial activity. Sandy soils typically drain well, but PES microfibers increased their WHC, which could have restricted root oxygen availability, thereby further reducing root growth by 85%. In addition, the reduced root P and K content, combined with minimal changes in leaf nutrient levels, suggested a reallocation of resources to maintain essential functions like photosynthesis. The root-to-shoot ratio was significantly reduced, implying a shift in biomass allocation toward shoots despite overall growth limitations. LDPE particles also negatively affected root growth and nutrient uptake in sandy soil. Their presence likely disrupted water flow and nutrient diffusion, leading to shorter roots and reduced N content. The impaired water pathways caused by LDPE particles may have hindered effective nutrient acquisition, contributing to reduced biomass in both roots and shoots.

In contrast to the other soil types, LDPE had a positive effect on root growth in silty clay loam Chernozem. Plants grown in LDPE-treated soil exhibited greater root length, higher K uptake, and improved leaf chlorophyll and carotenoid content compared to the control. It is hypothesized that due to the high water content of silty clay loam anoxic conditions that limit root development were created in the control, whereas the presence of LDPE particles may have increased soil aeration by forming macropores, thus mitigating waterlogging effects and improving root growth. Despite these benefits, N uptake did not increase significantly, although chlorophyll content was higher, indicating an

adequate nutrient supply. PES microfibers also induced structural changes in silty clay loam soil but to a lesser extent than LDPE. The potential increase in macroporosity caused by PES microfibers likely disrupted water phase continuity, limiting nutrient exchange between roots and soil water.

Overall, the effects of LDPE and PES MPs on winter wheat growth depended on the soil type and the way these MPs interacted with soil structure and water dynamics. In loamy silt and sandy soils, both MPs had predominantly negative effects, reducing root growth and nutrient uptake by increasing macroporosity and accelerating water loss. In silty clay loam, however, LDPE improved aeration and facilitated root expansion, while PES microfibers led to nutrient deficiencies due to disrupted water phase continuity. These findings highlight the complex interactions between MPs, soil texture, and plant performance, and contribute to the ongoing research in this field.

CHAPTER 5 – Microplastic mineralization rate in Fenton reactions depends on polymer type:

While MPs are considered chemically inert, studies have shown that certain bacteria (e.g., *Bacillus*, *Rhodococcus*) and fungi can degrade them. In particular, some fungi utilize the Fenton reaction – an oxidative process involving hydrogen peroxide and iron ions – to break down organic macromolecules, such as lignin. This mechanism, commonly used by wood-decaying and ectomycorrhizal fungi, has shown potential for MP mineralization. However, natural soil conditions, such as presence of carbonate and limited iron reduction, hinder its efficiency.

This study examined the mineralization of four types of MPs – LDPE, PP, PS particles, and PES microfibers – using Fenton's reagent at varying H_2O_2 concentrations. The mineralization process was assessed by monitoring CO_2 release, indicating that PES showed the highest mineralization rate, followed by PP, while LDPE and PS exhibited lower degradability. The mineralization rate was influenced by polymer structure, with polyesters (like PES) being more reactive due to their ester groups and aromatic rings.

Higher H_2O_2 concentrations led to increased CO_2 release for PES and PP but did not significantly increase mineralization of LDPE and PS, suggesting a threshold effect. Kinetic analysis revealed non-integer reaction orders, indicating complex reaction mechanisms influenced by radical recombination and scavenging effects.

Although CO_2 measurements confirmed mineralization, SEM imaging did not reveal any visible surface changes in MPs. This suggests the need for alternative analytical methods (e.g., weight loss, chemical composition changes) to fully assess MP mineralization.

6.2. Synthesis

The Anthropocene we live in presents us with numerous challenges. One of the most significant anthropogenic impacts that we and future generations must confront is the pollution of our planet with persistent materials and substances (i.e., per- and polyfluoroalkyl substances, radioactive contaminants, heavy metals, etc.). At the forefront of recent research is the topic of MP contamination in soil. For a long time, this issue went largely unnoticed because MP particles and fibers easily integrate into the soil matrix and are not as visually apparent as they are in our waters. However, recent scientific attention has raised critical questions about whether – and if so, to what extent – MP contamination in soils has short- and long-term consequences for soil and plant health.

This overarching theme also formed the core of this dissertation, which aimed to explore whether and how processes within the soil are altered by MP contamination and how this affects nutrient cycles and exchange processes – such as the carbon and nitrogen cycles – between the surrounding soil, plant roots, and microorganisms. To observe changes in plants, the rhizosphere plays a pivotal role, as it serves as the primary site for essential and life-sustaining exchange processes. Disturbances within this zone – whether detrimental or beneficial – can significantly influence plant growth, development, and health. Given the crucial role of plant water availability in maintaining optimal root exchange processes, this dissertation also investigated soil water characteristics under MP amendments in the absence of plants. Within the framework of carbon turnover processes, this research further addressed the potential mineralization of synthetic MPs. Such mineralization processes within the soil environment could increase carbon availability for surrounding microorganisms, potentially causing downstream effects on microbial activity and ecosystem function.

To identify potential processes and consequences, we opted for a relatively high MP concentration of 0.4% w/w, which is not representative for natural soil conditions. While some studies have reported very high MP contaminations of 0.03-6.7% (300-67,500 mg/kg) (Fuller & Gautam, 2016), these findings are limited to particularly polluted sites (e.g., industrial sites) and are not comparable to typical agricultural soils in rural regions. This specific mass concentration of 0.4% w/w was adopted from the Rillig research group at the University of Berlin, which reported that this concentration would be the upper limit at which soils experienced only minor changes in volume after the addition of linear MPs (de Souza Machado et al., 2018). The advantage of using such high concentrations is that processes, which might remain hidden or insignificant at lower concentrations, become detectable and measurable. Since the aim was not to explore concentration-dependent effects but to identify the key drivers behind the observed processes, this specific concentration was kept constant.

Furthermore, we chose to investigate individual, environmentally relevant MP types in isolation. Although this approach does not accurately represent natural conditions, where various MP types with differing shapes and material properties are heterogeneously dispersed in the soil, it was essential to identify individual effects of certain MP types. This approach enabled us to draw the conclusion whether potential effects varied according to shape (e.g., particle vs. fiber) or were instead associated with the polymeric structure and the resulting material properties. Our findings revealed that the addition of microfibers to soil had a more significant effect on soil water characteristics than particulate MPs. It is assumed that due to their linear structure and ability to entangle soil particles, microfibers in soil have higher potential to interact with the soil structure and impact soil aggregation (Lozano et al., 2021b), leading to more stable soil structure and better-defined macropores. In our case, MPs were widening the soil structure in finer soils (Luvisol and Chernozem) and blocking soil pores in coarse soil (Albic Luvisol).

Furthermore, it became apparent that the addition of particulate and fibrous MPs to the Luvisol resulted in contrasting effects on hydraulic conductivity and soil structure. Previous studies also highlighted the importance of MP shape as a key factor influencing soil physical properties (de Souza Machado et al., 2018; Lehmann et al., 2021; Zhao et al., 2021). In our study, the incorporation of PS and PP particles led to a reduction in total soil porosity, yet both enhanced K_{ns} . A CT-based study examining pore structure and connectivity found that the impact of adding particulate MPs (PVC) on pore connectivity was texture-dependent, while also reporting a general reduction in overall porosity (Wang et al., 2024). This suggests that, despite the overall decrease in pore volume, likely due to an altered spatial pore structure (Wang et al., 2024) or macropore blocking (Guo et al., 2022; Wang et al., 2023), the (re)arrangement of the remaining (meso- and micro-)pores may have improved connectivity and facilitated water flow under near-saturated conditions. In support of this, Wang et al. (2024) reported an increase in the number of connected pores in loam soil with rising particulate MP content (up to 4% w/w).

In contrast, the addition of PES microfibers resulted in an increase in both K_{sat} and porosity in the Luvisol, suggesting the formation or stabilization of macropores that possibly prevented pore collapse under wet conditions, helping to maintain open pathways for water movement. Although previous studies have shown that fibers can restructure the soil pore network and enhance porosity (Liang et al., 2021; Ingrassia et al., 2022a), the influence of the soil water status is often neglected. Our findings emphasize that fiber-induced changes are predominantly noticeable under saturated conditions. Overall, these results reveal not only the fundamental differences between rigid particles and flexible fibers in their interactions with the soil matrix, but also highlight the significance of water saturation state in evaluating the effects of MPs on soil pore structure.

To investigate whether this shape dependency on structural and hydraulic changes in soil translated into biological effects, the first of two plant experiments was conducted with the Luvisol and the same MP types (LDPE, PP and PS particles and PES microfibers) alone. Despite the particulate and fibrous MPs showed contradictory effects on soil structure and water dynamics in the Luvisol, their impact on plant performance showed surprising convergence. In particular, both PP particles and PES microfibers significantly reduced root length, while PS, PP, and PES all decreased NH_4^+ content in the soil. Nevertheless, the influence of MP morphology on plant development has been highlighted in a recent study, where fibers and particles, despite sharing the same polymer composition, elicited distinct biological responses (Chen et al., 2025a). Despite variation in soil physical changes, the uniformity in plant response we have found suggests that different types of MPs may exert similar stresses on plant development and nutrient dynamics through distinct pathways. In previous studies it was already concluded that MP effects on plant growth are highly variable and depend on MP shape, size, polymer type and additives (de Souza Machado et al., 2019; Lozano & Rillig, 2020).

One possible explanation for this convergence may be reflected in the behavioral differences between MP particles and fibers within the soil-root system. Unlike rigid particles, PES microfibers are flexible and more adaptable to their environment (de Souza Machado et al., 2018). In the presence of growing roots, these microfibers likely shift position within the soil, dynamically interacting with the root architecture and possibly altering the local soil structure in a way that mimics the disruptive effects of rigid particles. Previous studies have also shown that microfibers have similar disruptive effects as MP particles in plant experiments (Lozano et al., 2021a; Ingrassia et al., 2022b; Lozano & Rillig, 2022). This mobility and adaptability may explain why, despite exhibiting different effects on porosity and water flow in root-free conditions, PES microfibers ultimately led to similar negative effects on root growth and nitrogen availability as structurally more disruptive MPs, such as PS and PP particles. Thus, while the physical nature of MPs strongly influences soil structure, their biological impact may also be shaped by their interaction with dynamic processes such as root growth. These results further highlight the context-dependent nature of MP effects on soil functions.

Interestingly, the microbial co-occurrence network analysis of the rhizosphere soil helped to further explain the negative effects of PES and PP on root growth. Reduced interaction between clusters within the modular microbial network were found in both treatments, PES and PP. Reduced communication between microbial clusters results in lower exchange of metabolites and signaling molecules, with consequences for both the rhizosphere microbial community and the plant. Limited exchange can reduce the growth of microbial partner species and may even drive some to local extinction (Piccardi et al., 2024). For the plant, this disruption leads to less efficient nitrogen and phosphorus acquisition (Narula et al., 2009), reduced access to microbially produced hormones and

vitamins (Lareen et al., 2016), and impaired root development. Together, these effects can slow root growth, branching, and fine-root formation, ultimately reducing plant biomass, yield, and beneficial microbial diversity. Furthermore, sparser co-occurrence networks are associated with lower complexity and weaker resilience to disturbances such as drought or pollution (Wang et al., 2022b). Moreover, PS particles in soil were found to have no effect on the microbial network structure, which coincided with the absence of effects on root growth or plant performance. Although PS and PP particles were found to disrupt the soil structure of the Luvisol by reducing porosity (Chapter 2), this did not affect root growth. This suggests that, when evaluating the effects of MPs on plant performance, microbial disturbances in the rhizosphere dominate over physical changes, at least in the early stages of plant development. Another study could also show that the responses of plant root growth to MP contamination co-occurred with microbial changes (Qi et al., 2020c).

Despite the varied effects observed across the three contrasting soil types, the analysis of soil hydraulic properties indicated that soil type itself emerged as the dominant influencing factor. This aligns with previous findings in the literature, where effects of MPs in soil varied with soil texture (e.g., Ingraffia et al., 2022a; Wang et al., 2024; Guo et al., 2022). Additionally, our results revealed that soil water status also played a significant role. Especially under saturated conditions, the influence of soil type on soil hydraulic properties was strongest and the impact of MPs was less pronounced, as shown by the effect size η^2 (Fig. 2.2). This finding supports our hypothesis that MP effects in soil are not universal but are instead modulated by both water content and soil texture. Notably, the role of soil water status in shaping MP effects has not been previously reported in the literature. To investigate this further, a second plant pot experiment was conducted with the same three soil types and two MP forms: a particulate type (LDPE) and a fibrous type (PES).

The results demonstrated that the impact of PES and LDPE on root length varied with soil type and was not uniformly detrimental. In the Chernozem, characterized by its fine texture and strong water retention capacity, the addition of both PES and LDPE appeared to support root growth, unlike in the control where root development was impaired. This positive effect was likely due to LDPE particles and PES microfibers modifying soil structure, enhancing aeration and nutrient availability under saturated conditions. Specifically, the hypothesis that PES addition to Chernozem allowed for better drainage and thereby sustaining normal root growth is similarly supported by de Souza Machado et al. (2019), who proposed that PES microfibers have the ability to widen fine soil structures. This was supported by our K_{sat} measurements, which showed a significant increase in water flow in saturated Chernozem (Fig. 2.1). Although LDPE did not significantly alter K_{sat} , its irregular shape and variable particle size may have introduced small air-filled voids between soil aggregates, thereby enhancing gas diffusion and improving root zone aeration. In support of this assumption, Lozano et al. (2021a) found that PE particles favorably altered the pore size distribution in a way that slightly

increased lettuce biomass under PE MP contamination. Moreover, LDPE MPs could have improved the diffusion paths for dissolved nutrients. Thus, our findings suggest that the addition of both fibrous and particulate MPs can loosen fine, saturated soils and potentially benefit root development under certain conditions.

In contrast, in sandy Albic Luvisol, root development was impaired by MP contamination. As shown in Chapter 2 (Fig. 2.5), especially the addition of microfibers to sandy soil reduced the air-filled pore space, potentially creating anoxic conditions for roots. Similarly, previous studies have shown that microfibers tend to increase water-holding capacity and water retention in sandy soils (de Souza Machado et al., 2018; de Souza Machado et al., 2019; Lozano & Rillig, 2020), which ultimately reduces air-filled pore space. Moreover, the structural changes induced by MPs in this soil type may have hindered or altered water flow, as shown in Fig. 2.1, limiting also the availability of key nutrients such as K, P, and N for the rhizosphere. This was reflected in reduced nutrient contents in both roots (Fig. 4.2) and leaves of plants grown in Albic Luvisol (Fig. 4.7). In comparison, plants grown in Chernozem remained well-nourished, suggesting that under certain soil texture and moisture conditions, the presence of MPs may not disturb nutrient dynamics and plant growth.

Aligning with recent advances in MP research in soil environments, this study supports the growing consensus that the negative impacts of MPs on plant growth – particularly root development – are largely attributable to indirect effects on soil structure and function. Our findings demonstrate that MPs alter soil properties, including pore structure, aeration, and water transport, ultimately impairing the soil's capacity to deliver water and nutrients to plant roots. However, this explanation cannot be taken as the primary factor alone. Other studies have suggested that impaired root development may also result from mechanical damage to crop roots (Bosker et al., 2019) and root hairs (Urbina et al., 2020), caused by MP particles adhering to root surfaces. Additionally, shifts in microbial communities and changes in biological soil parameters (Li et al., 2023b) must also be considered as contributing factors.

Both the existing literature and our findings on plant performance and the rhizosphere microbiome (see Chapter 3) suggest that MPs influence more than just the soil's physical properties, but also induce shifts in the microbial community and nutrient cycling in a specific way. Such disruptions can negatively affect plant roots and nutrient availability. In Chapter 3, it was shown that especially under the influence of PP particles and PES fibers the reduced root growth of early winter wheat coincided with a fragmented microbial co-occurrence network. Such a fragmented microbial network indicates weakened microbial cooperation within the rhizosphere, thereby limiting its functions (e.g., nutrient mineralization, phytohormone production, and pathogen suppression) (Hernandez et al., 2021; R. Huang et al., 2019). However, the microbiome analysis also showed that

MP-induced changes of the rhizosphere microbiome are MP-dependent. For example, while PP and PES contamination weakened the microbial network, LDPE contamination enhanced the microbial connectivity of the early-stage winter wheat rhizosphere microbiome. A potential explanation could be the surface characteristics of the polymers, potentially providing better habitats for microbial colonization and biofilm formation. Furthermore, early winter wheat roots exhibited a functionally resilient microbial community, which was evident through the increase in nitrifiers and the moderate increase in NO_3^- in PS-treated soils.

A reduction in root length limits the extent of root-soil contact and consequently decreases the spatial range over which root exudates are released. This reduction likely leads to both lower quantities and a lower diversity of root-derived compounds (e.g., sugars, amino acids, organic acids), which typically serve as key substrates for microbial metabolism in the rhizosphere. As a result, microbial activity may decline, particularly among copiotrophic microbial taxa (r-strategists) that depend on easily accessible carbon sources (Ling et al., 2022). The shift in carbon availability may also drive changes in microbial community composition, favoring oligotrophic or dormant species over fast-growing r-strategists, thereby promoting a transition toward more resource-efficient K-strategists (Fierer et al., 2007). A dominance of K-strategists can accelerate the decomposition of soil organic matter (SOM) (Pan et al., 2022), leading to carbon loss over time. As the input of root-derived carbon declines, microbial communities may increasingly rely on existing SOM to meet their metabolic demands, potentially triggering a priming effect (Zhou et al., 2020c). This phenomenon could accelerate the decomposition of previously stable organic matter, ultimately contributing to greater carbon loss from the soil system.

However, the possibility of soil microorganisms potentially shifting toward alternative carbon sources raises an important question: Can MPs, as hydrocarbon-based materials, serve as a viable carbon source for microorganisms and thereby influence soil microbial diversity or alter long-term carbon cycling (Rillig, 2018; Xiang et al., 2024)? Such changes towards alternative carbon inputs could ultimately affect plant performance and crop productivity. In Chapter 5, we explored the MP mineralization efficiency by applying the Fenton reaction – naturally utilized by wood-decaying fungi – which generates $\text{OH}^\bullet$, which is among the most powerful oxidizing agents known. We tested whether this reaction could degrade various MP types at ambient temperature and high H_2O_2 concentrations under controlled laboratory conditions. Our findings revealed that a) MP mineralization was minimal even at high H_2O_2 concentrations (maximum 0.1% over 4 hours), and b) mineralization efficiency varied depending on the polymer type and H_2O_2 concentration. Finding a) aligns with Tagg et al. (2017), who also reported no significant surface change or weight reduction of MPs isolated from wastewater following Fenton treatment at ambient temperature. In contrast, studies employing a thermal Fenton

reaction or functionalized polymers reported clear signs of mineralization (Hu et al., 2022; Ortiz et al., 2022). Moreover, finding b) presents a novel insight, since the impact of polymer type and H_2O_2 on the mineralization efficiency of conventional MPs has not been reported previously. Consequently, these results are a fundamental step in further investigating MP mineralization mechanisms and their broader ecological implications.

Although some studies report the biodegradation of MPs in soil, closer examination reveals that these studies often involve biodegradable plastics (Allemann et al., 2024), pre-aged or heated MPs (Nowak et al., 2012; Rajandas et al., 2012; Jeon & Kim, 2013; Gong et al., 2023), or conditions resembling composting environments (Nowak et al., 2012; Skariyachan et al., 2018). In contrast, non-weathered MPs – such as those used in our Fenton experiment as well as in our plant pot experiments – are highly resistant to mineralization in natural soils. This is consistent with our Fenton experiment, which showed that, while MP mineralization to CO_2 is to some extent possible, the reaction is limited by the rapid saturation of H_2O_2 with Fe^{2+} (Fig. 5.1). Based on this observation, we conclude that under natural conditions, the mineralization of conventional, non-weathered plastics is likely negligible. As a result, given the 4- to 6-week exposure period in our plant pot experiments, it is unlikely that our MPs served as a microbial carbon source or released toxic leachates via extended biodegradation, making these factors unlikely causes for the observed impairments in plant performance.

Nevertheless, over prolonged periods of MP exposure to soil, landfill or composting environments, biodegradation of MPs could become a relevant factor. Not only specialized microbial communities in landfills or high temperatures in compost, but also natural weathering processes, such as photo-oxidation, pave the way for subsequent microbial-initiated biodegradation processes. Although photo-oxidation may take months to years due to limited UV penetration caused by soil particles and organic matter (Xu et al., 2024), it can induce surface oxygenation of plastics which increases the hydrophilicity of polymer chains (Gijsman et al., 1999) and therefore enables and enhances microbial biofilm formation on the plastic surface (Ali et al., 2021; Chen et al., 2022b; Liu et al., 2023a). These factors may collectively contribute to a gradual breakdown of MPs. In combination with Fenton-utilizing fungi, naturally producing H_2O_2 -generating enzymes (e.g., aryl-alcohol oxidases, glyoxal oxidases) (Daou & Faulds, 2017), a constant supply of $\text{OH}^\cdot$ necessary for polymer mineralization is guaranteed. This constant supply of H_2O_2 is essential, as our results show that due to its high reactivity with Fe^{2+} , H_2O_2 is quickly degraded, making $\text{OH}^\cdot$ production dependent on a constant supply. In light of this, the varying degradability of different MPs, as demonstrated in our study, is important for assessing long-term changes in soil microhabitats and chemistry.

Polymers which are more susceptible to oxidative degradation, such as PES, are likely to be degraded to intermediate products which can serve as carbon and energy sources for soil

microorganisms (Qi et al., 2021). This may promote microbial growth and activity locally, creating microhabitats with intensified biological processes around degrading MPs. Previous studies showed that the presence of MPs increased the abundance of plastic-degrading bacteria in soil (Aralappanavar et al., 2024). This is known to be the first step of microbial mineralization, followed by biofilm formation which can cause severe damage to the polymer surface (Gu, 2003). Especially fungi are able to invade the pores of the plastic surface using their mycelium, and induce the formation of cracks (Gong et al., 2023). This decreases the polymers' resistance and facilitates a partial degradation of the polymer chains into oligomers and smaller organic compounds (Bonhomme et al., 2003). Concerning the degradation of PES fibers made from PET, especially hydrolases have been shown to generate incomplete hydrolysis products, such as monohydroxyethyl terephthalate and Bis-(2-hydroxyethyl) terephthalate (Qi et al., 2021). Such degradation products may alter nutrient cycles and soil respiration rates, as well as the quality and accessibility of the soil organic carbon pool (Xiao et al., 2021).

In addition, the mineralization or degradation of MPs, irrespective of polymer type, can lead to the release of additives incorporated during manufacturing, such as plasticizers, UV stabilizers, and flame retardants (Sørensen et al., 2021; Xu et al., 2025). More rapidly degradable polymers may facilitate earlier and more concentrated additive release events, potentially resulting in acute toxicity episodes (Meng et al., 2021). Although persistent MPs (particularly aliphatic MPs) delay additive release, they pose a long-term risk through slow leaching. This maintains a continuous exposure and affects soil biogeochemistry and organism health over extended periods (Luo et al., 2022). Additionally, if degradation intermediates are toxic or poorly metabolized, they may exert stress on microbial communities or soil fauna.

6.3. Conclusions and outlook

This thesis provides a comprehensive analysis of MP-induced changes in soil structure and plant growth, and demonstrates the mineralization potential of MP through radical attack via the Fenton reaction. The findings emphasize the complex interactions between MPs, soil properties, and plant performance, demonstrating that MP effects are context-specific and influenced by both polymer type and soil texture. This final chapter critically reflects on the conclusions drawn from our research and proposes future study approaches to better understand MP effects on soil functions.

Overall, the cumulative findings from our studies show that the presence of MPs in soils modifies key factors such as water retention, porosity, and nutrient availability, which in turn, significantly influence plant growth, particularly root development and nutrient uptake. It was revealed that different soil textures (e.g., loamy silt, loamy sand, and silty clay loam) exhibited varying responses to MP exposure. This variation can partially be explained due to the unique ways in which MPs interact

with soil structure and water dynamics, consequently affecting microbial communities as well. Furthermore, the type and morphology of MPs themselves, such as particulate and fibrous MPs, govern the extent and direction of these interactions. However, it was shown that a dynamic soil system with actively growing roots influences the measurable effects of different MP types in the soil matrix. Moreover, biodegradation during our plant pot experiments can be excluded, while it was shown that conventional, non-weathered MPs possess different degradability efficiencies depending on their polymeric structure.

In general, it can be concluded that, although a MP concentration of 0.4% w/w at which we found substantial changes in soil water behavior and plant performance, is relatively high for current estimations, our results could soon provide a realistic picture due to the steady accumulation of MPs in soil, as well as their poor degradability and mineralization potential. This aligns with the consideration of an MP load of $\geq 2\%$ as a realistic, environmentally relevant range in the future (Barnes et al., 2009; Meixner et al., 2020; Piñon-Colin et al., 2020; Qi et al., 2020a).

Based on our findings, several areas that warrant further investigation regarding the effects of MPs on plants and soil processes can be highlighted. Our experiments on the hydraulic properties of various soils with incorporated MPs revealed changes that could be crucial for understanding broader impacts on plant growth. To better isolate and characterize these effects, it would be beneficial to use artificial soils with controlled and gradually varied textures. This would help determine threshold values – for example, the specific sand or clay content at which different types of MPs begin to significantly alter soil hydraulic behavior. Furthermore, as demonstrated by Saljnikov et al. (2025), who analyzed soil samples taken from greenhouses where plastic mulching was performed for many years, the cumulative accumulation of MPs in soil may lead to outcomes that differ significantly from those observed in soil directly added with freshly prepared MPs without long incubation periods. This would potentially provide a more realistic picture of MP accumulation for future perspectives (Saljnikov et al., 2025).

Furthermore, incorporating different MP types in combination in future studies could provide improved perspectives and better reflect realistic environmental conditions. In our work, we observed that fibers and particles had contrasting effects in the Luvisol, suggesting that MP morphology plays a significant role. It would be particularly interesting to investigate whether the distinct effects observed with individual MP types persist, are amplified, or potentially neutralize each other when fibers and particles are combined. In addition to the dependency of MP morphology, different sizes would be a significant factor for hydraulic conductivity measurements, as argued by Xie et al. (2023). Moreover, it was shown that the K_{sat} of soils would change in dependency of the quality of leachates from landfill cells of different ages running through MP-contaminated soil (Xie et al., 2023).

In addition to the measurement of physical soil properties, microbiological analyses of both rhizosphere and bulk soil should be conducted gradually during and after plant growth, potentially over extended crop rotations. Our microbiome analyses focused on early root development and nutrient availability, but the open question remains if the rhizosphere of crop plants, despite initial disturbances, can recover and adapt over time, or is subject to long-term damage from MP exposure. Addressing the resilience of the rhizosphere under MP influence is therefore a crucial next step in this field of research.

The different rates at which MPs undergo Fenton oxidation are relevant to soil-related processes. Faster mineralizing MPs introduce short-term, potentially intense ecological disturbances but may also allow for eventual ecosystem recovery. In contrast, poorly mineralizing MPs create a cumulative, long-lasting ecological burden, disrupting soil structure and chemistry and altering microbial communities. Understanding these dynamics is crucial for assessing the full impact of MPs on terrestrial ecosystems.

However, analytical confirmation of MP mineralization – particularly under the influence of Fenton-utilizing wood-decaying fungi – remains a significant challenge. This is especially true in soil environments, where background CO₂ emissions are naturally high due to soil respiration. This complicates the use of highly sensitive analysis systems, like isotope ratio mass spectrometry (IRMS), to attribute ¹³C-CO₂ evolution specifically to the mineralization of ¹³C-labelled MP. As a result, assessing MP mineralization based solely on CO₂ measurements in soil is challenging. For this reason, mineralization experiments conducted in soil should ideally combine gravimetric methods with surface analysis techniques such as SEM or spectroscopic techniques (Raman or infrared spectroscopy) to detect physical as well as structural changes on the MP surface and the potential modification of functional groups bonded to polymer chains. To test the hypothesis that wood-decaying fungi are capable of degrading and mineralizing MPs in soil systems, wood chips can be added to the soil as a lignin-rich carbon source to enhance the potential for co-metabolic degradation of synthetic polymers as a secondary effect of lignin degradation.

Moreover, we could show that non-weathered MPs are highly persistent even under accelerated mineralization conditions, which highlights the urgent need to reduce plastic pollution and explore alternative materials and processes. In addition to replacing plastic mulch films with straw in strawberry fields, for example, and reducing the use of slow-release MP-containing fertilizers, regulations governing the application of sewage sludge, which is considered one of the pathways with highest MP input to agricultural soils (Brandes et al., 2021), should be reevaluated with regard to MP contamination. Based on the directive 86/278/EEC, which was adopted in 1986 and adapted in 2019, the use of sewage sludge in European agriculture is strictly regulated due to concerns about its heavy metal content. Despite these concerns, the directive allows exceptions under which farmers may still

use sewage sludge as fertilizer. In some EU countries, untreated sludge may be applied to fields if it is promptly injected or incorporated into the soil (Union, 2020). Importantly, while the directive focuses on heavy metals, it does not address the presence of MPs in sewage sludge.

Although several studies report high MP loads in compost (Piehl et al., 2018; Brandes et al., 2021), making it a likely pathway for MP entry into soil when applied as fertilizer, compost fertilization remains a resource-efficient practice that supports the principles of a circular economy. It recycles organic waste into valuable soil amendments, which reduces the dependency on synthetic fertilizers. Recent findings suggest, however, that compost may not be the sole or even primary source of MP contamination in soils. For example, Braun et al. (2023) highlight that improper littering can override the contribution of compost-derived MPs, complicating efforts to trace back contamination sources. Furthermore, Edo et al. (2022) report that door-to-door collection of municipal biowaste results in lower plastic contamination, and that no biodegradable plastics were detected after the composting process, indicating the potential for cleaner compost when best practices are followed (Edo et al., 2022). To minimize MP inputs while maintaining the benefits of compost use, several measures should be considered. Households should be educated on proper waste separation, supported by clear labeling of compostable materials and local outreach programs promoting correct disposal habits. Composting facilities can improve their pre-treatment and post-processing steps to filter out plastic contaminants, while also implementing MP monitoring. On the regulatory side, stricter compost quality standards should be enforced, and non-compostable plastics in organic waste streams should be more controlled.

In conclusion, this thesis highlights the importance of incorporating more realistic environmental scenarios into study approaches, as the effects of MPs on soil and plant functions are highly context-dependent. Furthermore, addressing MP contamination requires an interdisciplinary approach that integrates soil science, agronomy, and environmental chemistry, as well as a supportive policy framework, to develop effective mitigation strategies for sustainable land management.

Appendices

Appendix A: Supplementary material of Chapter 2

Microplastic production and characterization

An amount of 100 g of plastic beads were filled into the belonging jar (1,360 mL, made of BPA-free copolyester), which was equipped with a blunt stainless-steel safety blade, and blended at average to high power for 45 s. After five times of repeating the blending process, the content had to be cooled down as the shearing forces led to a rapid heating of the plastic beads. After the content had cooled down, the shredded plastic beads were filled into a sieving cascade, comprised of metal analysis sieves ranking from a mesh size of 1000 μm to 50 μm . After the plastic beads were removed, the jar contained electrostatically adhered plastic powder, which was removed by filling the jar with tap water and pouring the water-powder mixture over a 50- μm -analysis-sieve. The powder on the metal sieve was dried in the oven at 60°C over night. An additional self-constructed device was built by the institute's workshop, which enabled constant discharging and collection of MP particles during the blending process (Fig. A.1).

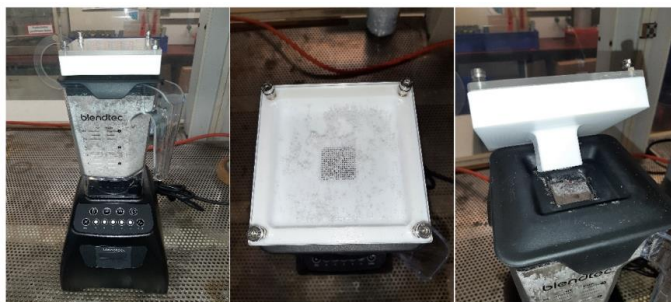


Fig. A.1. Blendtec blender for the production of microplastic from plastic beads (3-5 mm).

All particle types (LDPE, PP, and PS) were measured with a laser diffraction particles size analyzer (HORIBA LA-950, Irvine, CA) by dispersing the respective MP powder in ethanol.

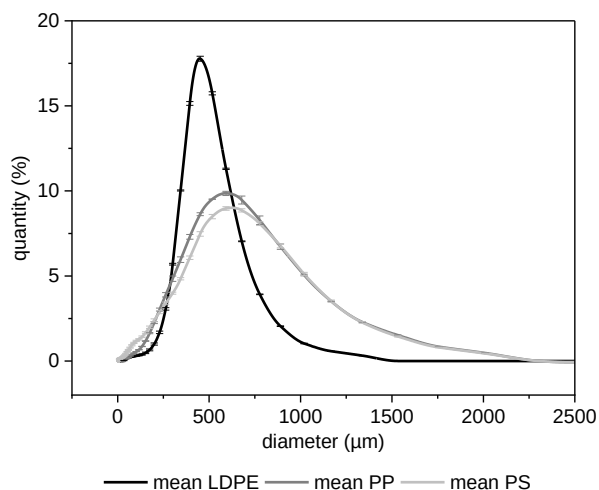


Fig. A.2. Particle size distribution of used microplastic (MP) particles. LDPE is low-density polyethylene, PP is polypropylene, PS is polystyrene. Each MP powder was measured three times and mean particle sizes were: LDPE = $444.6 \pm 1.1 \mu\text{m}$, PS = $532.3 \pm 4.7 \mu\text{m}$, PP = $548.0 \pm 4.7 \mu\text{m}$.

The used microfibers were analyzed for their polymer structure by applying Raman spectroscopy (Fig. A.2). Thereby the spectrum of the fibers were compared to a PET piece which was cut out from a plastic water bottle.

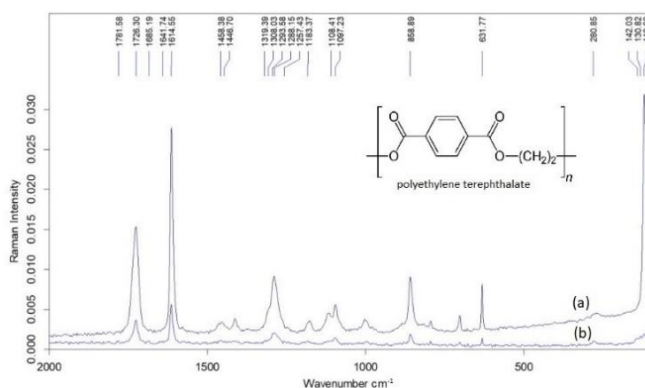


Fig. A.3. Raman spectra of a) polyethylene terephthalate (PET) reference material and b) polyester (PES) fibers. Spectrum (b) is comparable to spectrum (a) which indicates that the polymeric structure of PES fibers is identical with PET.

Relevant assignments of PET Raman spectra are shown in

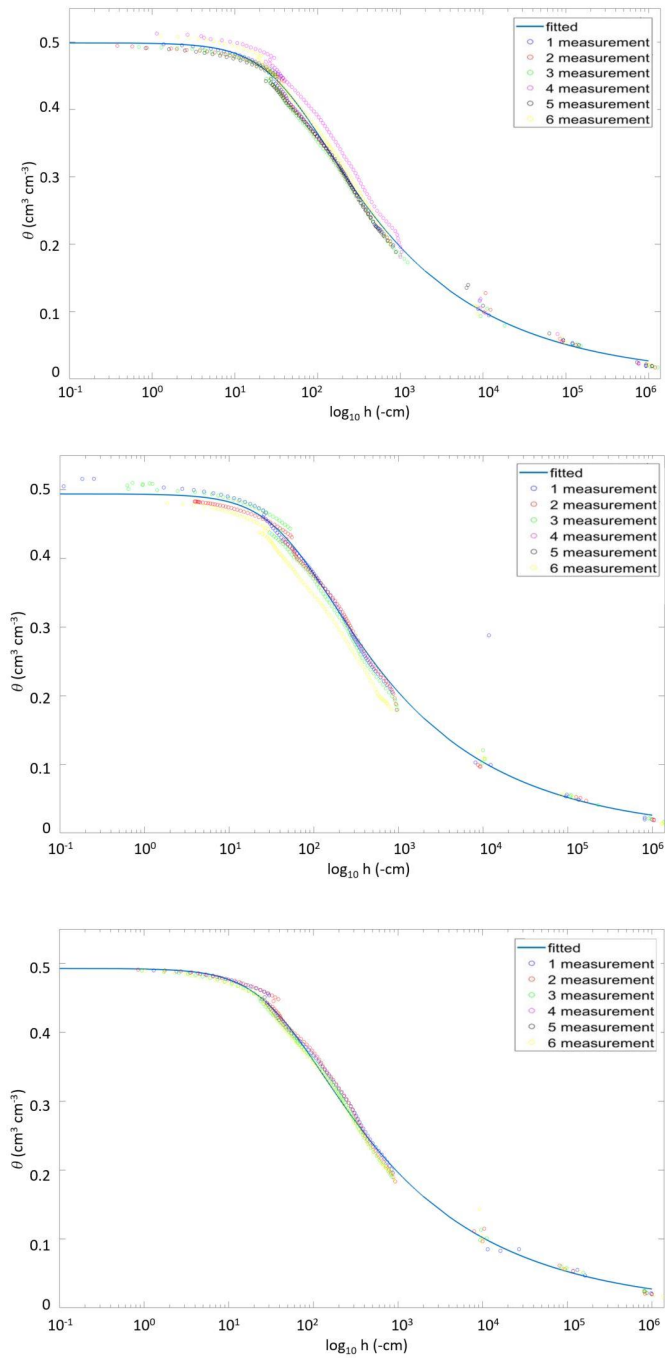
A.1.

Table A.1. Relevant assignments of functional groups of polyethylene terephthalate in Raman spectra (Rebollar et al., 2014; Alexiou et al., 2020).

Wavenumber cm ⁻¹	Assignment
1726	Stretching C=O vibration
1614	Ring mode 8a (in Wilson's notation)
1293	C(O)–O stretching
1183	Ring in plane C–H bond and C–C stretch
1096	a combination of C–O stretching, COC bending, CCO bending, and C–C stretching in the ethylene glycol segment

Experimental set-up**Fig. A.4. Mini-disk infiltrrometer (MDI, Meter Group AG, Munich, Germany) filled with blue colored water placed on a soil-filled PVC column (10 x 10 cm).**

Fitted soil hydraulic characteristics curves



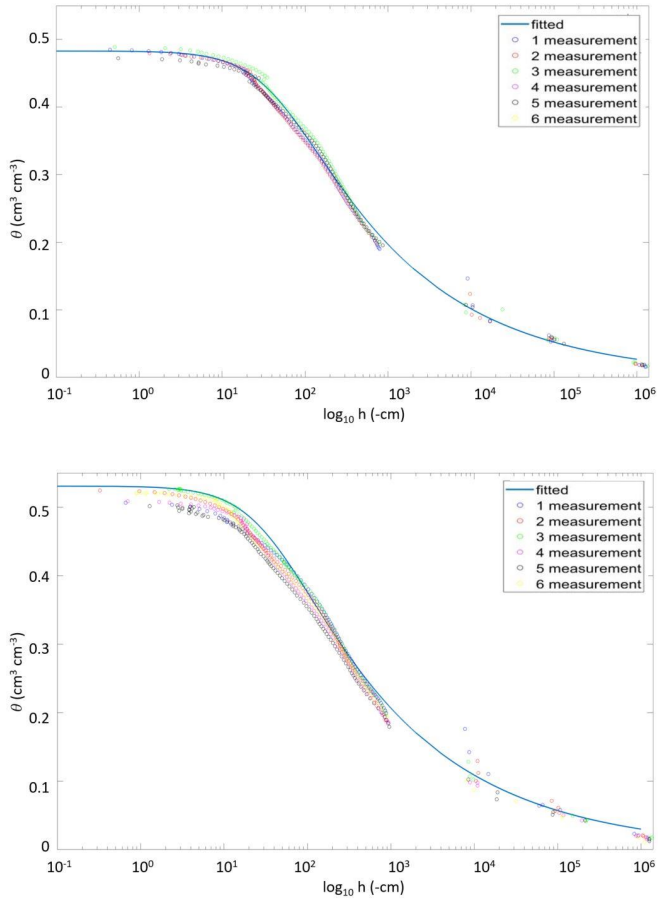
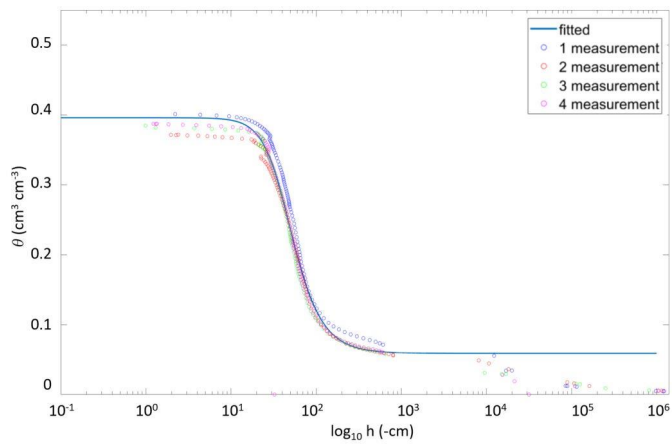
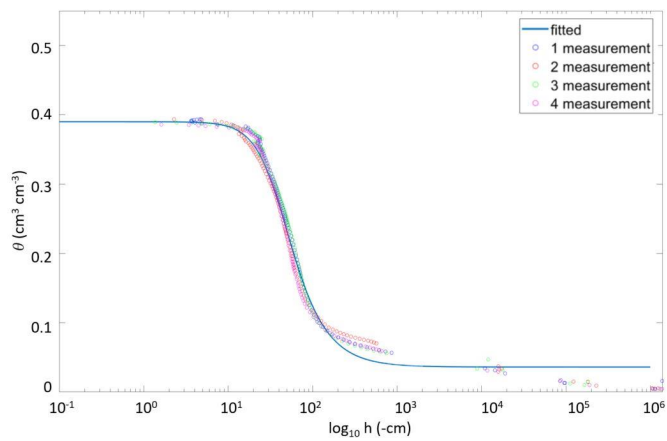
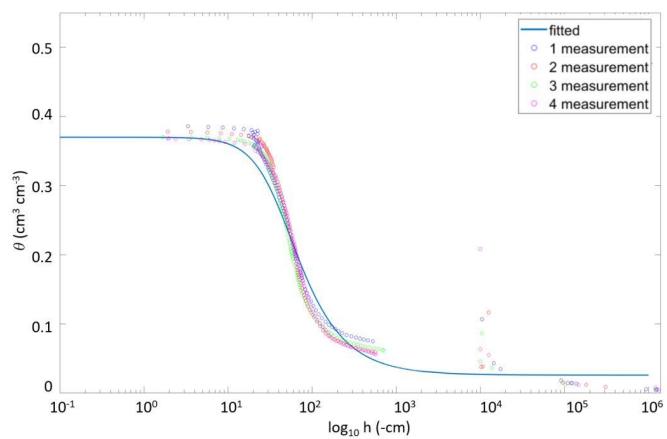


Fig. A.5. Measured and fitted soil hydraulic characteristics for the Albic Luvisol (AL). From top downwards: control, LDPE, PP, PS, PES.



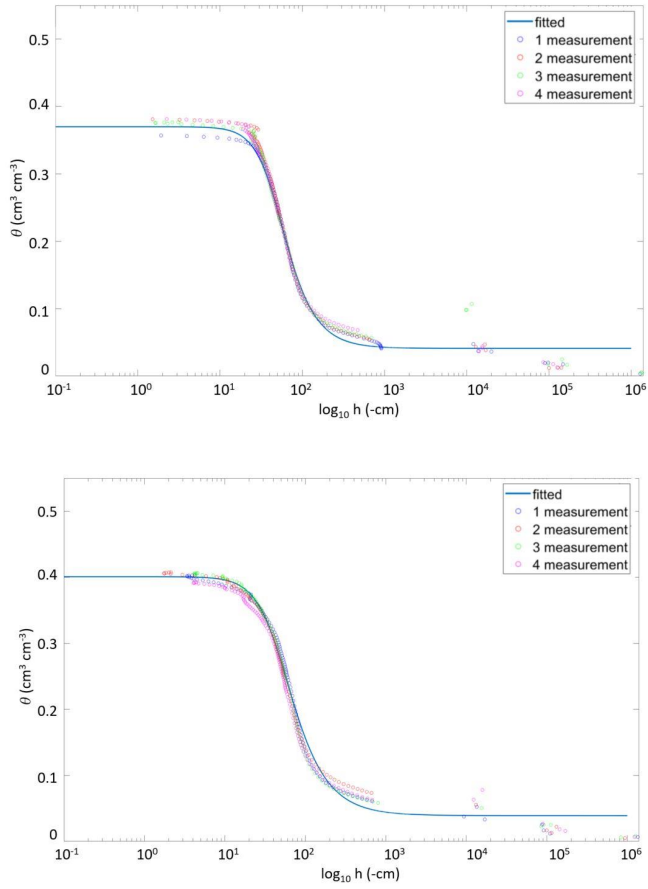
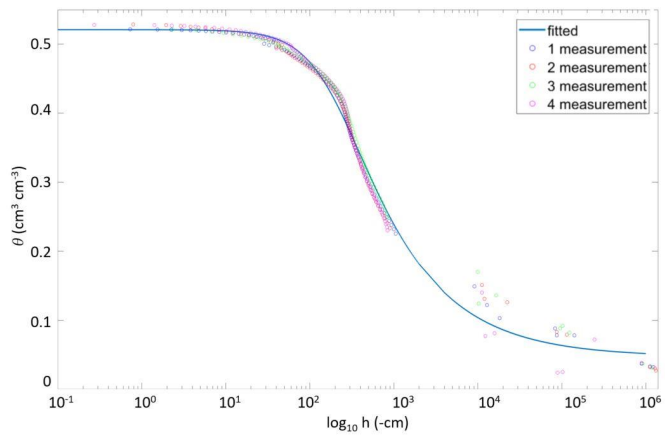
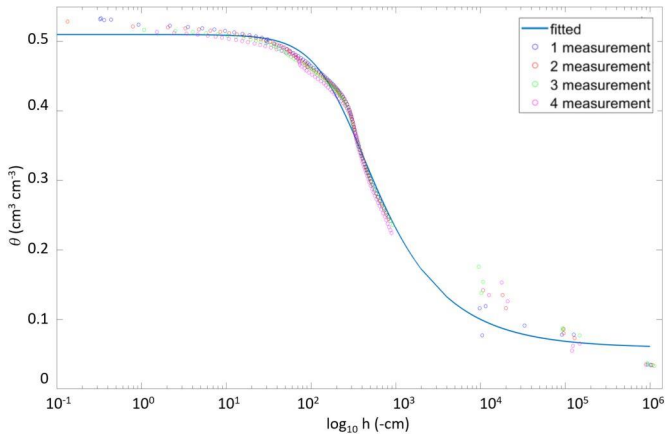
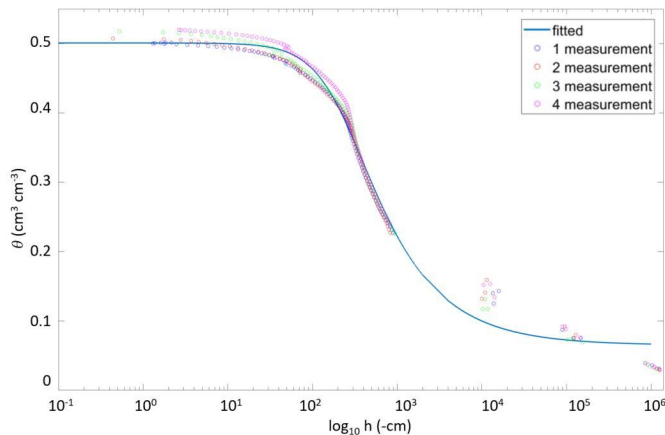


Fig. A.6. Measured and fitted soil hydraulic characteristics for the Luvisol. From top downwards: control, LDPE, PP, PS, PES.



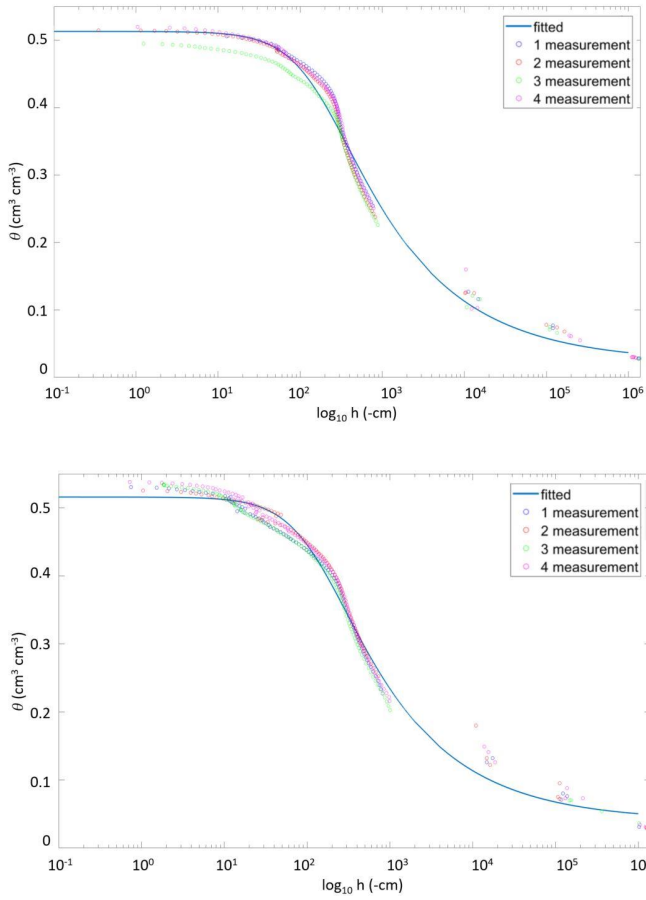


Fig. A.7. Measured and fitted soil hydraulic characteristics for the Chernozem. From top downwards: control, LDPE, PP, PS, PES.

Appendix B: Supplementary material of Chapter 3

The volumetric water content at saturation was $0.50 \text{ cm}^3 \text{ cm}^{-3}$, which is equivalent to a volumetric water-holding capacity (WHC) of 50% v/v, which was determined measuring soil water retention curves (SWRC). SWRCs of six soil samples, compacted to 1.2 g cm^{-3} , were recorded by using the evaporation method using a HYPROP® system (Meter Group, München, Germany) as described by Schindler et al. (2010) in combination with a dewpoint potentiometer (WP4®, Decagon Devices, Pullman, WA, USA). The volumetric WHC corresponded to a gravimetric WHC of 42% w/w (Eqn. B.1), which was used to calculate the amount of water added to the plants during the experiment.

$$\text{grav. WHC} = \frac{\text{vol. WHC}}{\rho_{\text{soil}}} \quad (\text{B.1})$$

With grav. WHC = gravimetric water-holding capacity (g g^{-1}), vol. WHC = volumetric water holding capacity ($\text{cm}^3 \text{ cm}^{-3}$), ρ_{soil} = soil bulk density (g cm^{-3}).

It is not possible to definitively exclude the presence of micro- and/or nanoplastics in the soil, since the agricultural area from which the soil was collected may have been exposed to anthropogenic plastic inputs. Nonetheless, the site has no record of plastic mulching or sewage sludge fertilization. Any larger plastic debris discovered was promptly removed. In addition, the inclusion of control samples without added MP and the relatively high MP concentration (0.4% w/w), a level not yet commonly observed in German farmland, mitigated any potential bias from prior contamination.

Extraction method and preparation of extraction media for soil analysis

For each extraction method, 5 g ($\pm 10\%$) of fresh frozen soil samples were weighed into 50-mL centrifuge vials (Eppendorf AG, Hamburg, Germany), mixed with 40 mL of the respective extraction medium and then shaken horizontally for 1 h at 200 rpm. After centrifugation for 15 min at 3500 rpm, a syringe was used to collect 20 mL of the supernatant. The first 5 mL were discarded through a $0.45 \mu\text{m}$ filter (Sartorius AG, Göttingen, Germany), and the remaining 15 mL were collected and stored at -20°C for further analysis. The resulting concentration of soil in extraction medium was 0.1 g mL^{-1} .

Extraction medium for mineral plant-available nitrogen (N_{min}) analysis:

A 0.01 M calcium chloride (CaCl_2) solution was used by dissolving 7.35 g of $\text{CaCl}_2 \times 2 \text{ H}_2\text{O}$ salt (Merck KGaA, Darmstadt, Germany) in 5 L deionized water.

Extraction medium for plant-available potassium (K_{av}) and phosphate (P_{av}) analysis:

The calcium acetate lactate (CAL) solution was prepared as 1 L stock solution by slowly dissolving 77.00 g of calcium lactate pentahydrate ($\text{C}_6\text{H}_{10}\text{CaO}_6 \times 5 \text{ H}_2\text{O}$) and 39.50 g of calcium acetate ($\text{Ca}(\text{CH}_3\text{COO})_2 \times \text{H}_2\text{O}$) respectively in 300 mL of hot deionized water while both solutions were stirred at 200 rpm. Both

solutions were filled into a 1 L volumetric flask and were adjusted to 1 L after 90 mL of conc. acetic acid were added. The final stock solution was diluted to 5 liters.

Root analysis with WinRHIZO

Table B.1. Parameter set for WinRHIZO analysis.

Criteria	Set parameter
Root and Background Detection	220
Debris Rough Edges	Area smaller than: 0.0025 cm ²
Rough Edges and Noise Removal	low

Diameter classes of winter wheat roots

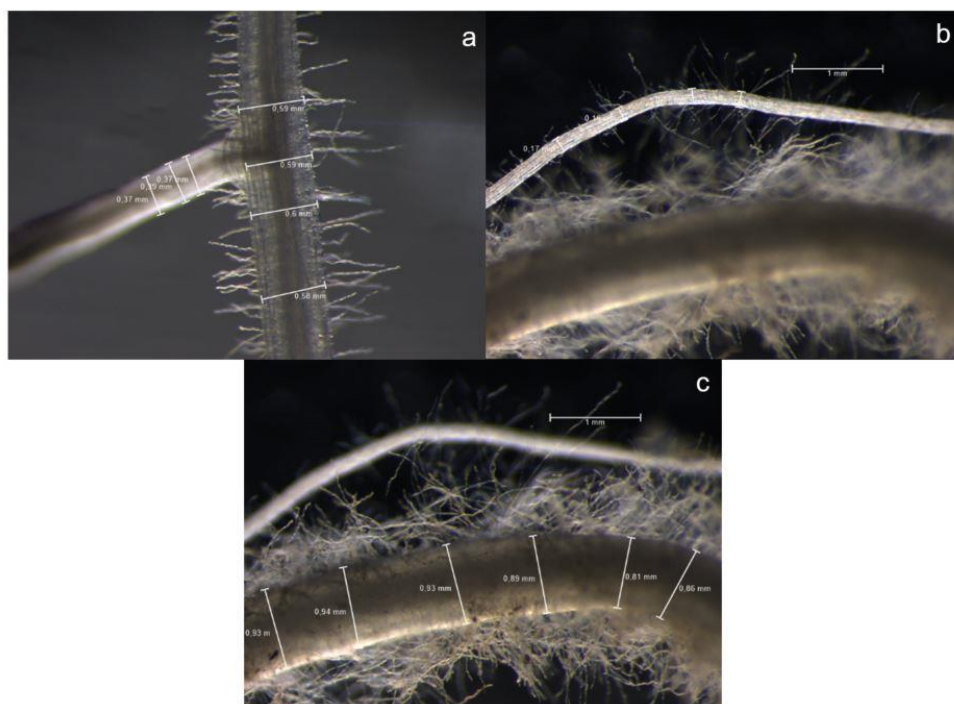


Fig. B.1. Stereomicroscopic pictures of winter wheat roots harvested after a growth period of 28 days. a) Primary root (width: 0.58-0.6 cm) and lateral root (width: 0.37-0.39 cm), b) Lateral root (width: 0.16-0.17 cm), c) Primary root (width: 0.81-0.94 cm).

Calculation of chlorophyll-a, -b, and carotenoid

The following equations as proposed by Lichtenthaler & Wellburn (1983) were used to determine the chlorophyll-a (chl_a), chlorophyll-b (chl_b) and the carotenoid (car) content of the extracts (Eqns. B.2-B.3). The values were further converted to μg per g leaf dry weight.

$$chl_a = abs_{663} \cdot 12.21 - abs_{646} \cdot 2.81 \quad (\text{B.2})$$

$$chl_b = abs_{646} \cdot 20.13 - abs_{663} \cdot 5.03 \quad (\text{B.3})$$

$$car = \frac{abs_{470} \cdot 1000 - chl_a \cdot 3.27 - chl_b \cdot 104}{229} \quad (\text{B.4})$$

With chl_a = chlorophyll-a in $\mu\text{g mL}^{-1}$, chl_b = chlorophyll-b in $\mu\text{g mL}^{-1}$, car = carotenoid in $\mu\text{g mL}^{-1}$, abs_x = absorption at wavelength x nm.

Shoot analysis

After the growth period of 28 days, the shoots had a mean dry mass in the range of 390 to 570 mg (Fig. B.2a), while their length ranged from 32 to 35 cm (Fig. B.2b). Similarly, no significant difference in shoot length development over time was observed with the addition of MP (Fig. B.4). The nitrogen (N) concentration ranged from 49 to 53 mg g^{-1} leaf mass (Fig. B.2c), while the average total leaf N content of each plant (Fig. B.2d) remained below 30 mg, and was not significantly different between the treatments.

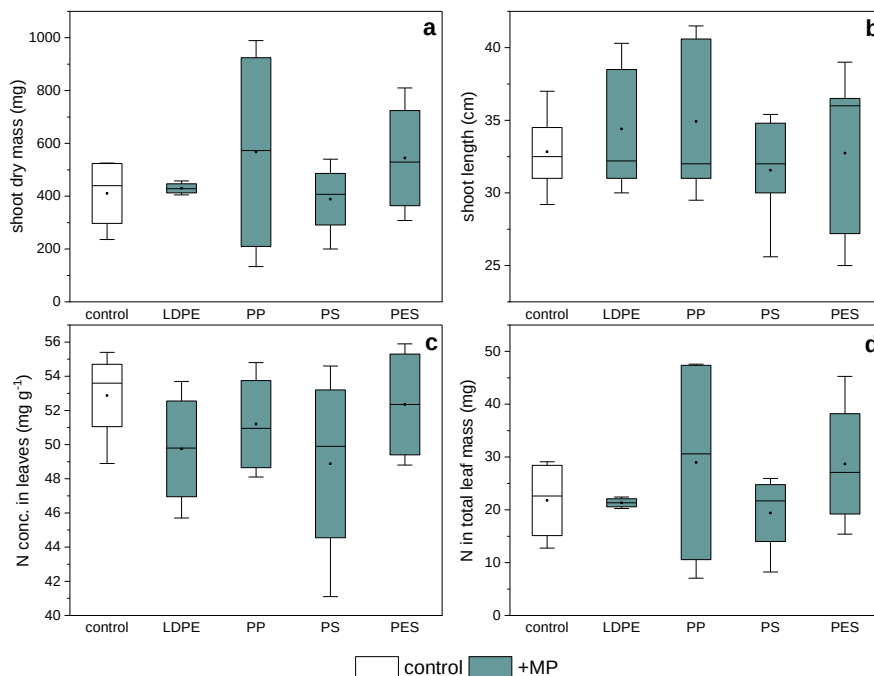


Fig. B.2. Shoot a) dry mass (in mg), b) length (in cm), c) nitrogen (N) concentration in leaves (in mg g⁻¹) and d) total leaf N content (mg) of winter wheat plants after the growth period of 28 days in rhizotrons. “+MP” represents microplastic (MP) addition to soil, while no MP was added to the soil of the control. The following MP types were added: LDPE = low-density polyethylene, PP = polypropylene, PS = polystyrene, PES = polyester fibers. Boxes are the interquartile range (IQR) from the 25th to 75th percentile, lines inside boxes indicate the median, white squares indicate the mean values and upper and lower whiskers are 1.5 times the IQR (n = 5).

The leaf chlorophyll content, *chl_a* (Fig. B.3a) and *chl_b* (Fig. B.3b), as well as the *car* content (Fig. B.3c) of plants grown in MP-amended soil were not significantly different from the control. The *chl_a* values ranged between 3 to 3.6 µg mg⁻¹, for *chl_b* between 1 to 1.15 µg mg⁻¹ and for *car* 0.40 to 0.51 µg mg⁻¹.

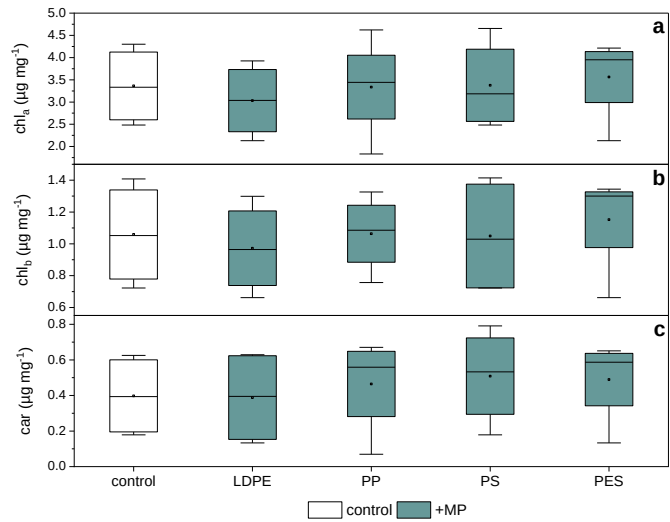


Fig. B.3. Chlorophyll, a) chlorophyll-a (chl_a) and b) chlorophyll-b (chl_b), and c) carotenoid (car) content (in $\mu\text{g mg}^{-1}$) of winter wheat plants grown for 28 days in rhizotrons. “+MP” represents microplastic (MP) addition to soil, while no MP was added to the soil of the control. The following MP types were added: LDPE = low-density polyethylene, PP = polypropylene, PS = polystyrene, PES = polyester fiber. Boxes are the interquartile range (IQR) from the 25th to 75th percentile, lines inside boxes indicate the median, white squares indicate the mean values and upper and lower whiskers are 1.5 times the IQR (n = 5).

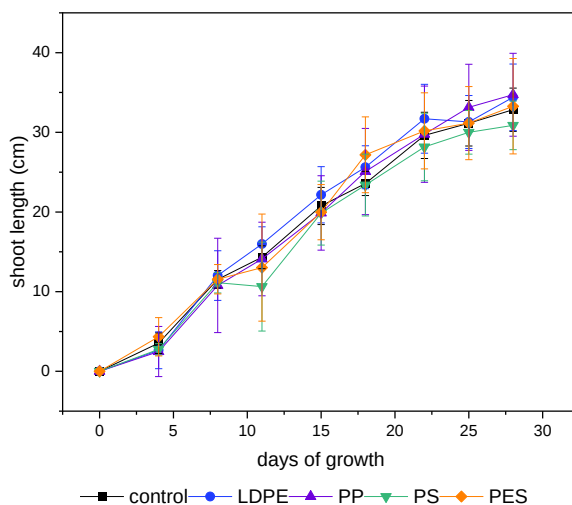


Fig. B.4. Shoot length (in cm) development over the growth period of winter wheat plants in rhizotrons for 28 days. Winter wheat plants grew without (control, in black squares) and with the addition of different microplastic (MP) types in soil (0.4% w/w). The following MP types were added: LDPE = low-density polyethylene particles in blue circles, PP = polypropylene particles in purple triangles, PS = polystyrene particles in green triangles, PES = polyester fibers in orange diamonds.

Soil nutrient analysis in $\mu\text{g g}^{-1}$

In general the soil nutrient analysis of plant-available mineral nitrogen species, e.g. NO_3^- , NH_4^+ and NO_2^- and K_{av} and P_{av} did not show large differences with MP load compared to the control (Fig. B.5).

In the bulk soil of LDPE- and PP-amended soil it was noticeable that the NO_3^- content was slightly reduced with LDPE by $50\% \pm 16\%$ ($p = 0.1$) and with PP by $59\% \pm 22\%$ ($p = 0.08$). The NO_2^- concentration in rhizo soil was slightly enhanced with presence of PP particles ($+33\% \pm 34\%$, $p = 0.08$). With PS particles the soils' NH_4^+ content was enhanced in the rhizo soil by $62\% \pm 68\%$ ($p = 0.07$) with a weak significance and significantly enhanced in bulk soil by $48\% \pm 26\%$ ($p = 0.03$). Moreover, the NH_4^+ content in the bulk soil of PES treatment was slightly enhanced ($+71\% \pm 90\%$, $p = 0.1$).

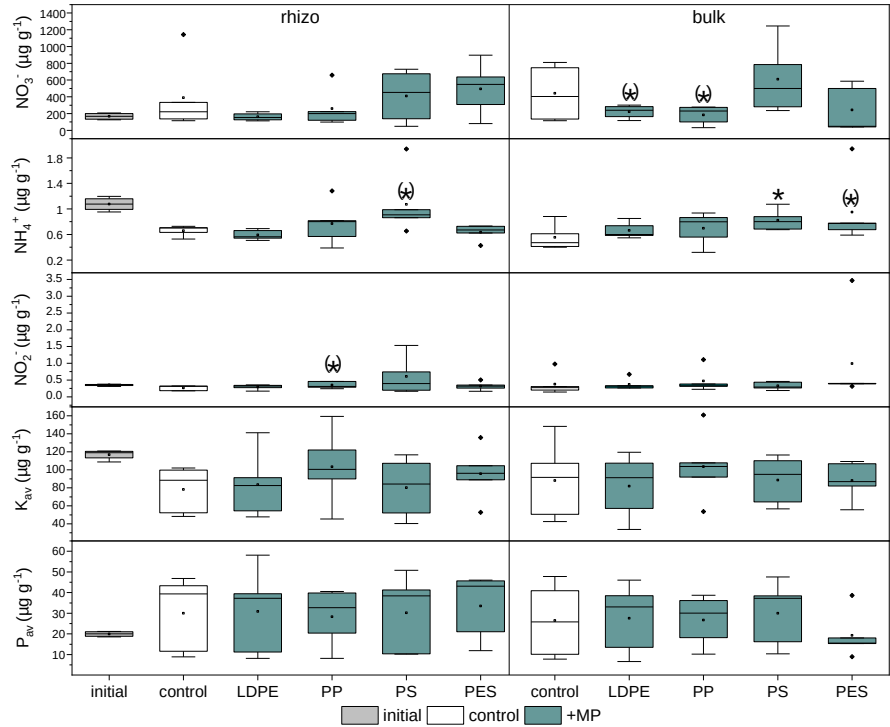


Fig. B.5. Soil nutrient analysis of plant-available nitrate (NO_3^-), ammonia (NH_4^+), nitrite (NO_2^-), potassium (K_{av}) and phosphate (P_{av}) (in $\mu\text{g g}^{-1}$) in rhizosphere (rhizo) and bulk soil before (“initial” in grey) and after the winter wheat growth period of 28 days in rhizotrons. The following microplastic (MP) types were added in soil (in dark cyan): LDPE = low-density polyethylene (in cyan), PP = polypropylene, PS = polystyrene, PES = polyester fiber. Control (in white) resembles no MP addition in soil. Boxes are the interquartile range (IQR) from the 25th to 75th percentile, lines inside boxes indicate the median, white squares indicate the mean values, upper and lower whiskers are 1.5 times the IQR and diamond-shaped markers are outliers (n = 5). The asterisk show significant difference ($p \leq 0.05$) towards the control and asterisks in parentheses show weak significant differences ($0.05 \leq p \leq 0.1$) to the control, tested with one-sided, heteroscedastic t-test.

Table B.2. Two-way ANOVA testing the effect of root zone and microplastic (MP) type on the amount of soil nutrients (NO_3^- = nitrate, NH_4^+ = ammonium, NO_2^- = nitrite, K_{av} = potassium, P_{av} = phosphate) per root surface area ($\mu\text{g cm}^{-2}$), described by the test statistic (F) and critical test statistic ($F_{crit.}$) values, the p-value lower than 0.05, and the effect size eta-squared (η^2).

Source of variation	NO_3^-			NH_4^+			NO_2^-			K_{av}			P_{av}		
	F and $F_{crit.}$	p-value	η^2	F and $F_{crit.}$	p-value	η^2	F and $F_{crit.}$	p-value	η^2	F and $F_{crit.}$	p-value	η^2	F and $F_{crit.}$	p-value	η^2
Effect of root zone	0.12 and 4.08	0.72	0.00	0.20 and 4.08	0.65	0	0.98 and 4.08	0.33	0.02	0.02 and 4.08	0.90	0.00	0.69 and 4.08	0.41	0.02
Effect of MP type	2.76 and 2.61	0.04*	0.20	3.65 and 2.61	0.01*	0.25	1.34 and 2.61	0.27	0.10	1.70 and 2.61	0.17	0.14	0.79 and 2.61	0.54	0.07
Interaction effect	1.09 and 2.61	0.38	0.08	0.63 and 2.61	0.64	0.04	1.26 and 2.61	0.30	0.10	0.17 and 2.61	0.95	0.01	0.26 and 2.61	0.90	0.02

*significantly different from control ($p < 0.05$).

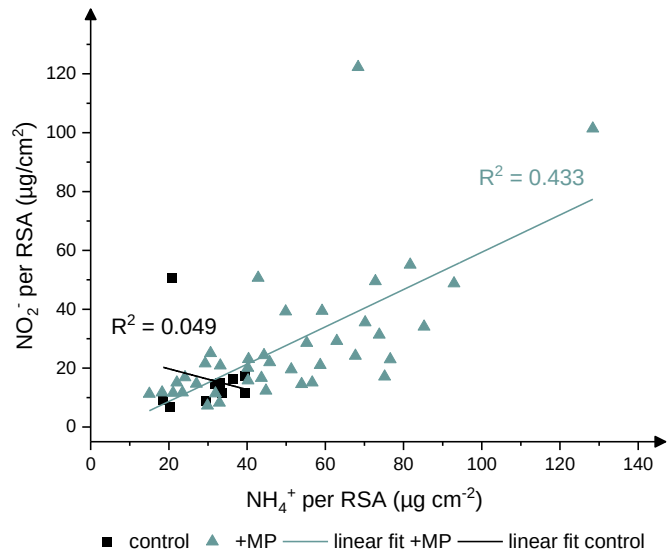


Fig. B.6. Plot of plant-available nitrite (NO_2^-) versus ammonium (NH_4^+) content of the soil per root surface area (RSA).

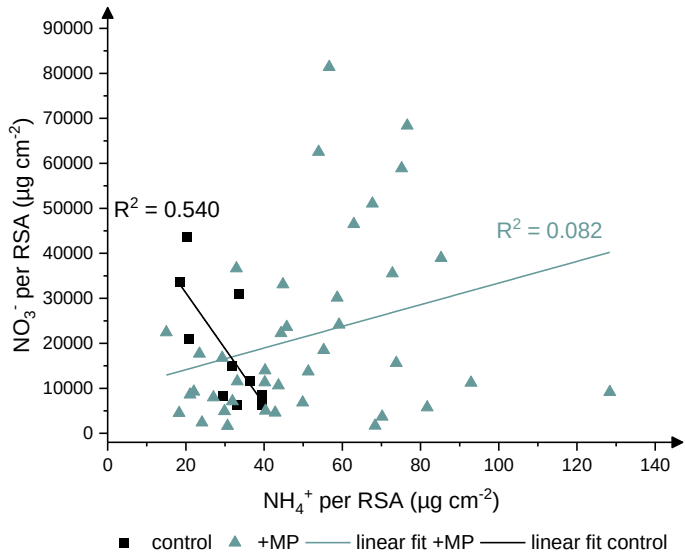


Fig. B.7. Plot of plant-available nitrate (NO_3^-) versus ammonium (NH_4^+) content of the soil per root surface area (RSA).

Microbiome analysis

Table B.3. Statistical analysis of different alpha diversity measures. Statistical analysis was performed using Kruskal-Wallis and the Wilcoxon Rank-Sum test with Benjamini-Hochberg correction for multiple comparisons.

<i>p</i> - values	Richness (Rich)	Shannon	Simpson	Evenness (Even)
Soil	0.652	0.598	0.309	0.832
MP	0.721	0.897	0.705	0.769
Soil* MP	0.755	0.608	0.409	0.851

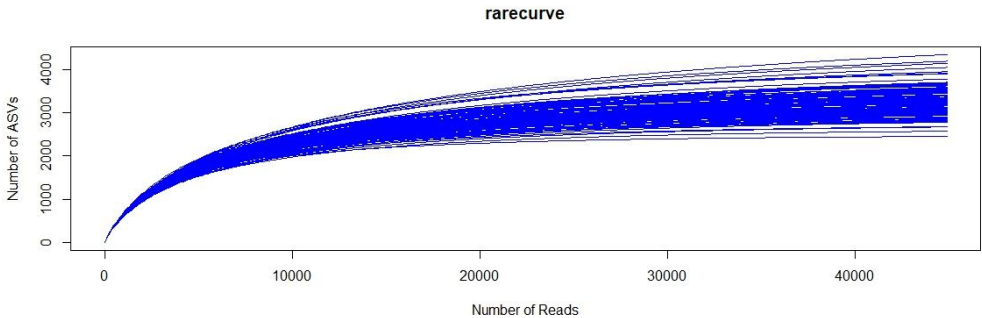


Fig. B.8. Rarefaction curve of amplicon sequence variants (ASVs) identified via 16S rRNA gene amplicon sequencing after subsampling to 44,905 reads per sample. The number of sequences ASVs reached a plateau for all samples, indicating a sufficient sequencing depth.

Table B.4. Global network properties of the microbial co-occurrence network based on Pearson correlation for untreated (control) and low-density polyethylene (LDPE) microplastics treated rhizosphere samples. Both treatments formed single-component networks with high clustering coefficients and edge densities. LDPE reduced vertex and edge connectivity by 50% and slightly lowered modularity. Average dissimilarity and path length remained comparable.

	Control	LDPE
Number of components	1.00000	1.00000
Clustering coefficient	0.94176	0.95691
Modularity	-0.00158	-0.00507
Positive edge percentage	49.5654	49.32838
Edge density	0.90178	0.91405
Natural connectivity	0.33447	0.34642
Vertex connectivity	28.00000	14.00000
Edge connectivity	28.00000	14.00000
Average dissimilarity*	0.65894	0.64355
Average path length**	0.97402	0.97445

*: Dissimilarity = 1- edge weight

** : Path length = Units with average dissimilarity

Table B.5. Global network properties of the microbial co-occurrence network based on Pearson correlation for untreated (control) and polyester (PES) fiber microplastics treated rhizosphere samples. Both treatments formed single-component networks with high clustering coefficients and edge densities. LDPE reduced vertex and edge connectivity by 50% and slightly lowered modularity. Average dissimilarity and path length remained comparable.

	Control	PES
Number of components	1.00000	1.00000
Clustering coefficient	0.94870	0.88754
Modularity	-0.00482	-0.00589
Positive edge percentage	49.46174	49.03877
Edge density	0.91874	0.82037
Natural connectivity	0.32672	0.28916
Vertex connectivity	28.00000	24.00000
Edge connectivity	28.00000	24.00000
Average dissimilarity*	0.65727	0.70149
Average path length**	0.97834	0.94937

*: Dissimilarity = 1- edge weight

**: Path length = Units with average dissimilarity

Table B.6. Global network properties of the microbial co-occurrence network based on Pearson correlation for untreated (Control) and polypropylene (PP) microplastics treated rhizosphere samples. Both treatments formed single-component networks. The PES network showed reduced clustering coefficient, edge density, connectivity, and natural connectivity, indicating decreased network complexity and cohesion.

	Control	PP
Number of components	1.00000	1.00000
Clustering coefficient	0.91942	0.88415
Modularity	-0.00161	-0.00505
Positive edge percentage	49.77390	49.42149
Edge density	0.84706	0.82763
Natural connectivity	0.33241	0.28624
Vertex connectivity	19.00000	28.00000
Edge connectivity	19.00000	28.0000
Average dissimilarity *	0.67588	0.70113
Average path length **	0.96045	0.95186

*: Dissimilarity = 1- edge weight

**: Path length = Units with average dissimilarity

Table B.7. Global network properties of the microbial co-occurrence network based on Pearson correlation for untreated (control) and polystyrene (PS) microplastics treated rhizosphere samples. Both treatments formed single-component networks. The PP network showed a lower clustering coefficient, lower edge density, and natural connectivity compared to control. Vertex and edge connectivity increased under PP.

	Control	PS
Number of components	1.00000	1.00000
Clustering coefficient	0.92931	0.92093
Modularity	-0.00372	-0.00591
Positive edge percentage	49.44391	49.71591
Edge density	0.86101	0.86676
Natural connectivity	0.33352	0.32253
Vertex connectivity	21.00000	23.00000
Edge connectivity	21.00000	23.00000
Average dissimilarity *	0.67057	0.67430
Average path length **	0.96231	0.95954

*: Dissimilarity = 1- edge weight

**: Path length = Units with average dissimilarity

Appendix C: Supplementary material of Chapter 4

Microplastic characterization

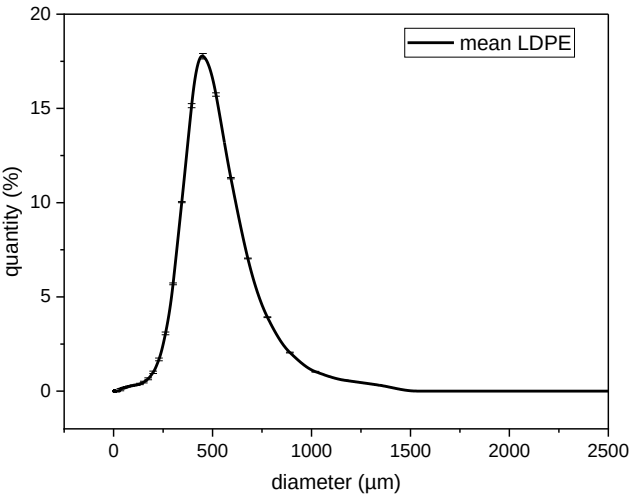


Fig. C.1. Particle size distribution of used low-density polyethylene (LDPE) microplastic particles, measured with a laser diffraction particle size analyzer (HORIBA LA-950). Particles were dispersed in ethanol and no ultrasound was used, n = 3. The mean particle diameter size was $444.6 \pm 1.1 \mu\text{m}$.

Experimental set-up

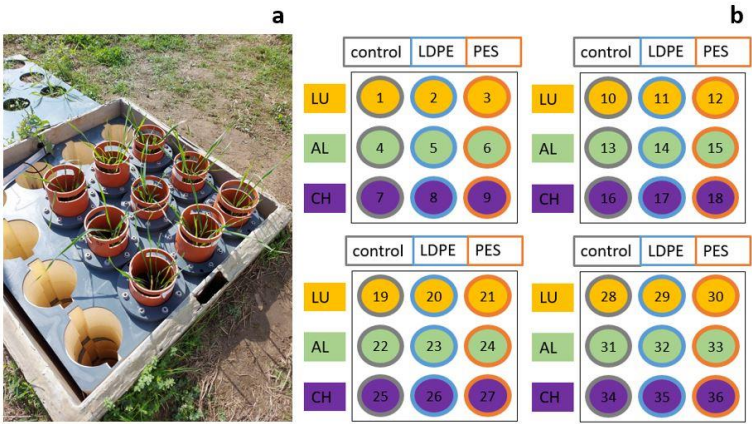


Fig. C.2. Lysimeter space filled with sewage pipes used as plant pots (a). Each pipe contains one treatment. Full layout of treatments and arrangement in all of the four lysimeter spaces (b); the controls of the respective soil type without microplastic (MP) are bordered in grey, LDPE is low-density polyethylene (blue bordered), PES is polyester microfiber (orange bordered); LU is silty loamy Luvisol soil (yellow filled), AL is loamy sandy Albic Luvisol soil (green filled), CH is silty clayey loamy Chernozem soil (purple filled).

Weather data during the experiment

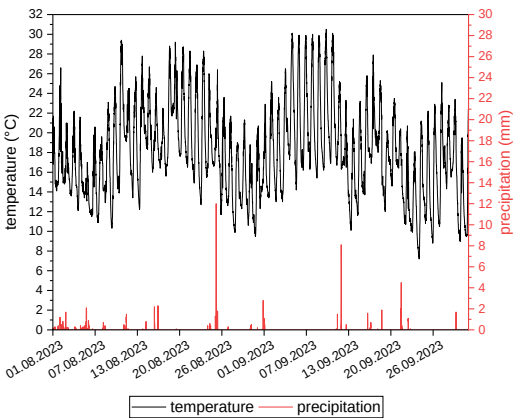


Fig. C.3. Weather data (temperature, in °C, black and precipitation, in mm, red) from 1st August 2023 to 30th September 2023. Data are taken from the weather station on the campus of Forschungszentrum Jülich.

Root temperature conditions

By embedding the pipes in empty lysimeter spaces below ground and isolating them with a custom-made Styrofoam structure, stable temperature conditions for roots could be created. Temperature probes were placed at half the depth of lysimeter spaces 1 to 4 and proved stable temperature (T) conditions throughout the growth period of 42 days from 18th August to 29th September. Mean temperatures inside lysimeter spaces 1 to 4 were 18.41 ± 1.71 °C, 18.62 ± 1.97 °C, 18.80 ± 1.83 °C and 18.86 ± 1.95 °C, respectively.

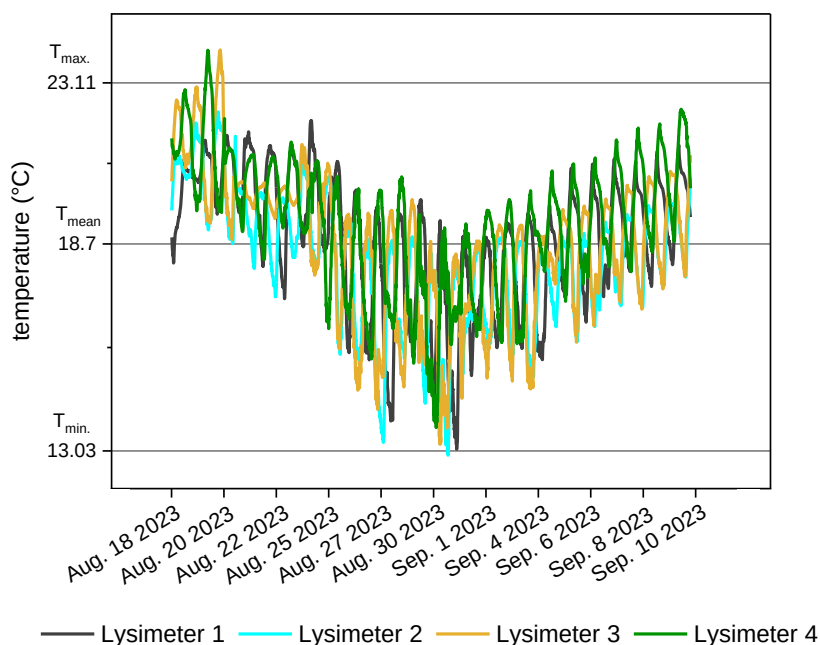


Fig. C.4. Temperature conditions for roots in the lysimeter spaces. The mean temperature (T_{mean}) for all four lysimeter spaces was 18.7 ± 0.18 °C. Maximum (T_{max}) and minimum (T_{min}) temperatures were: $T_{\text{max}} = 23.11 \pm 0.91$ °C and $T_{\text{min}} = 13.03 \pm 0.13$ °C.

Preparation of extraction media for soil analysis

Extraction medium for mineral plant-available nitrogen (N_{min}) analysis:

A 0.01 M calcium chloride (CaCl_2) solution was prepared by dissolving 7.35 g of $\text{CaCl}_2 \times 2 \text{H}_2\text{O}$ salt (Merck KGaA, Darmstadt, Germany) in 5 L deionized water.

Extraction medium for plant-available potassium (K_{av}) and phosphate (P_{av}) analysis:

The calcium acetate lactate (CAL) solution was prepared according to VDLUFA (2016) as 1 L stock solution by slowly dissolving 77.00 g of calcium lactate pentahydrate ($\text{C}_6\text{H}_{10}\text{CaO}_6 \times 5 \text{H}_2\text{O}$) and 39.50 g

of calcium acetate ($\text{Ca}(\text{CH}_3\text{COO})_2 \times \text{H}_2\text{O}$) respectively in 300 mL of hot deionized water while both solutions were stirred at 200 rpm. Both solutions were filled into a 1 L volumetric flask and were adjusted to 1 L after 90 mL of conc. acetic acid were added. The final stock solution was diluted to 5 liters.

Parameters for WinRHIZO analysis

Table C.1. Parameter set for WinRHIZO analysis of root characteristics.

Criteria	Set parameter
Root and Background Detection	220
Debris Rough Edges	Area smaller than: 0.0025 cm^2
Rough Edges and Noise Removal	high

Sample digestion for root and leaf nutrient analysis by ICP-OES

One to two parallel digestions per sample, each using approximately 25-50 mg, were carried out in closed vessels in a microwave system with 1 mL HNO_3 and 0.5 mL H_2O_2 . The vessels were heated to 160°C within 4 minutes and held at this temperature for 3 minutes. Each digestion solution was transferred and diluted to a final volume of 14 mL. Two parallel dilutions of each digestion solution (50-fold) were prepared and analyzed.

Appendix D: Supplementary material of Chapter 5

The obtained MP powder was sieved through analysis sieves ranking from 100 μm to 355 μm . The particle size distribution of LDPE, PP, and PS was measured in duplicate via a laser scattering particle size distribution analyzer (Partica LA-960, Horiba, Japan) by suspending the respective powder in ethanol.

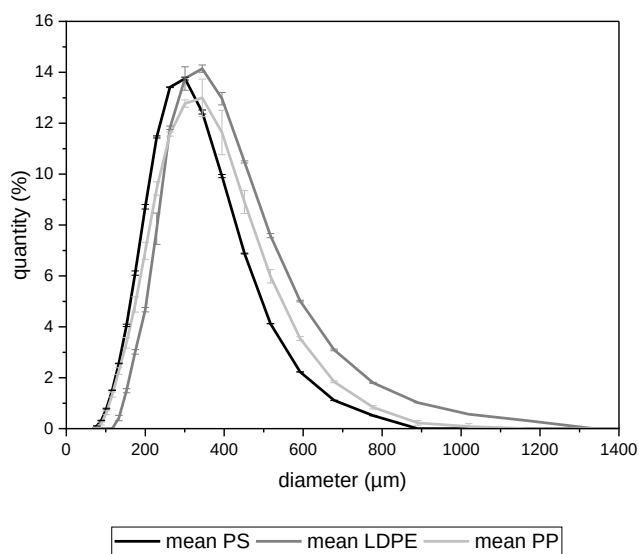


Fig. D.1. Particle size distribution of polystyrene (PS) in black, low-density polyethylene (LDPE) in grey and polypropylene (PP) in light grey for gas emission measurement. Mean particle diameters were $283 \pm 0.3 \mu\text{m}$, $353 \pm 0.8 \mu\text{m}$, and $305 \pm 1.6 \mu\text{m}$ for PS, LDPE, and PP, respectively.

Experimental setting

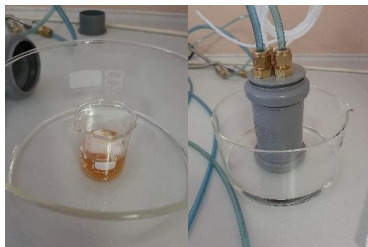


Fig. D.2. Experimental set-up of gas emission measurements of different MP types treated with Fenton's reagent.

Curve fitting and calculations

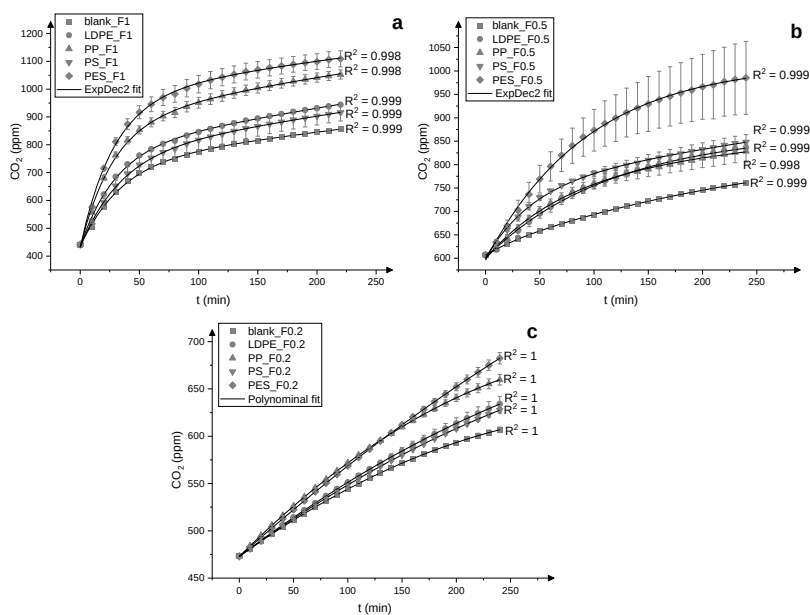


Fig. D.3. Curve fitting for experimental data of CO₂ emissions, in ppm, with standard deviation (whiskers) for the degradation of different microplastic types (LDPE = low-density polyethylene, PP = polypropylene, PS = polystyrene, PES = polyester microfiber) with Fenton's reagent of different hydrogen peroxide (H₂O₂) concentrations: F1 = 1 M H₂O₂ + Fe²⁺, F0.5 = 0.5 M H₂O₂ + Fe²⁺, F0.2 = 0.2 M H₂O₂ + Fe²⁺. F1 and F0.5 experimental data (dots) were fitted with an exponential fit (ExpDec2 in Origin, solid lines) and the F0.2 experimental data were fitted with a polynomial fit (solid lines).

Table D.1. Variable definition for graph fitting.

Treatment	Variable	blank	LDPE	PP	PS	PES
F0.2	y0	472.96	472.96	472.96	472.96	472.96
	B1	0.8218	0.8629	1.1381	0.8358	1.0168
	B2	-0.0011	-0.0008	-0.0015	-0.0008	-0.0006
F0.5	y0	882.95237	1051965.96	2471155.36	4114926.17	1014.3252
	A1	-17.65399	-213.76737	-179.65231	-162.04318	-213.39844
	t1	27.78501	91.248	66.20297	42.09751	91.8271
	A2	-259.71334	-1051156.2	-2470381.5	-4114168.3	-210.04057
	t2	314.70936	-1051156.2	1.01011E7	1.08241E7	91.83446
F1	y0	1675152.63	1250.23034	1232.66232	1704.72079	1242.88615
	A1	-313.67363	-342.57882	-436.06386	-337.24589	-520.66279
	t1	31.79092	25.52703	23.84535	30.83256	24.67715
	A2	-1674418.9	-494.01777	-391.53699	-952.1344	-319.23428
	t2	2975767.20	454.77248	278.17319	1157.32195	246.90237

Table D.2. Orders of reaction kinetics (n) of the degradation of different microplastic types (PES = polyester microfibers, PP = polypropylene particles, LDPE = low-density polyethylene particles) using Fenton's reagent. n was determined by using x, the factor by which the reaction rate was enhanced from 0.5 M to 1 M of H₂O₂.

Plastic type	x	n
PES	5.4	2.43
PP	6.2	2.63
LDPE	1.6	0.64

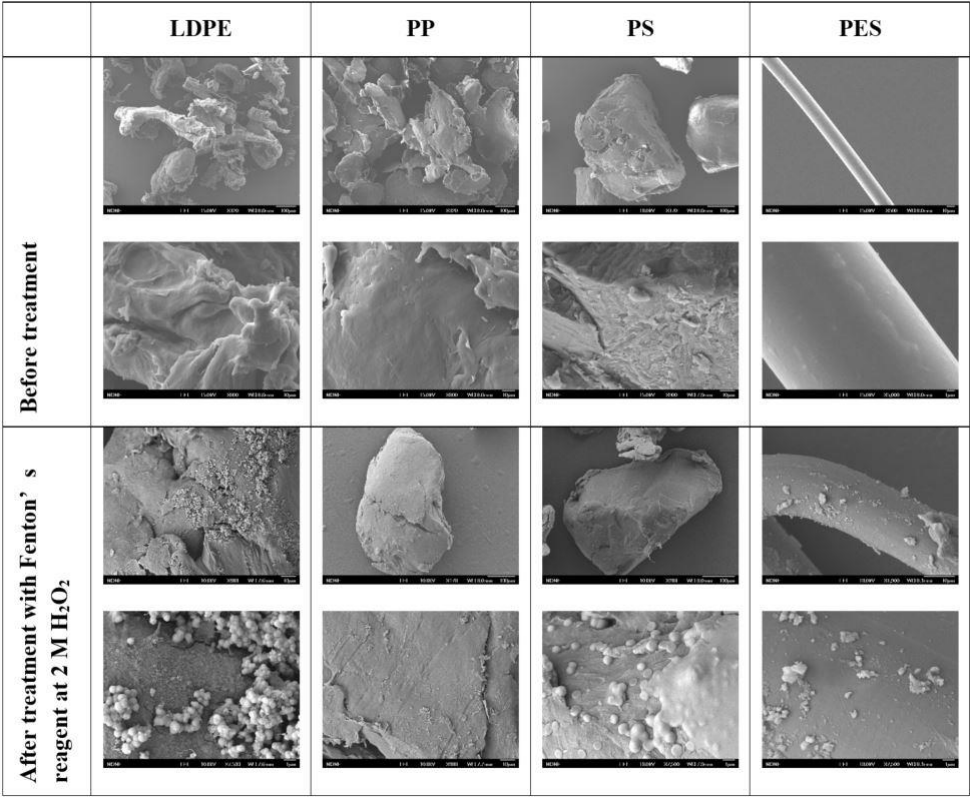


Fig. D.4. Scanning electron microscopy (SEM) images of microplastic particles (LDPE = low-density polyethylene, PP = polypropylene, PS = polystyrene) and polyester (PES) microfibers before and after the treatment with Fenton's reagent of 2 M hydrogen peroxide (H₂O₂).

Literature

- Abedsoltan, H. (2023). A focused review on recycling and hydrolysis techniques of polyethylene terephthalate. *Polymer Engineering & Science*, 63(9), 2651-2674. <https://doi.org/10.1002/pen.26406>
- AdAlsteinsson, S., Schjørring, J. K., & Jensen, P. (1994). Regulation of Phosphate Influx in Winter Wheat: Root-Shoot Phosphorus Interactions. *Journal of Plant Physiology*, 143(6), 681-686. [https://doi.org/10.1016/S0176-1617\(11\)81157-0](https://doi.org/10.1016/S0176-1617(11)81157-0)
- Al-Malaika, S., Axtell, F., Rothon, R., & Gilbert, M. (2017). Additives for Plastics. In *Brydson's Plastics Materials* (pp. 127-168). Elsevier. <https://doi.org/10.1016/B978-0-323-35824-8.00007-4>
- Aldas, M., Paladines, A., Valle, V., Pazmino, M., & Quiroz, F. (2018). Effect of the Prodegradant-Additive Plastics Incorporated on the Polyethylene Recycling. *International Journal of Polymer Science*, 2018, 1-10. <https://doi.org/10.1155/2018/2474176>
- Alexiou, V. F., Mathioudakis, G. N., Andrikopoulos, K. S., Soto Beobide, A., & Voyiatzis, G. A. (2020). Poly(ethylene Terephthalate) Carbon-Based Nanocomposites: A Crystallization and Molecular Orientation Study. *Polymers*, 12(11). <https://doi.org/10.3390/polym12112626>
- Ali, S. S., Elsamahy, T., Koutra, E., Kornaros, M., El-Sheekh, M., Abdelkarim, E. A., Zhu, D., & Sun, J. (2021). Degradation of conventional plastic wastes in the environment: A review on current status of knowledge and future perspectives of disposal. *Science of The Total Environment*, 771, 144719. <https://doi.org/10.1016/j.scitotenv.2020.144719>
- Allemann, M. N., Tessman, M., Reindel, J., Scofield, G. B., Evans, P., Pomeroy, R. S., Burkart, M. D., Mayfield, S. P., & Simkovsky, R. (2024). Rapid biodegradation of microplastics generated from bio-based thermoplastic polyurethane. *Scientific Reports*, 14(1), 6036. <https://doi.org/10.1038/s41598-024-56492-6>
- Allen, S., Allen, D., Phoenix, V. R., Le Roux, G., Durántez Jiménez, P., Simonneau, A., Binet, S., & Galop, D. (2019). Atmospheric transport and deposition of microplastics in a remote mountain catchment. *Nature Geoscience*, 12(5), 339-344. <https://doi.org/10.1038/s41561-019-0335-5>
- Álvarez-Barragán, J., Domínguez-Malfavón, L., Vargas-Suárez, M., González-Hernández, R., Aguilar-Osorio, G., & Loza-Tavera, H. (2016). Biodegradative activities of selected environmental fungi on a polyester polyurethane varnish and polyether polyurethane foams. *Appl Environ Microbiol*, 82(17), 5225-5235. <https://doi.org/10.1128/AEM.01344-16>
- Andrady, A. L. (2011). Microplastics in the marine environment. *Marine pollution bulletin*, 62(8), 1596-1605. <https://doi.org/10.1016/j.marpolbul.2011.05.030>
- Aralappanavar, V. K., Mukhopadhyay, R., Yu, Y., Liu, J., Bhatnagar, A., Praveena, S. M., Li, Y., Paller, M., Adyel, T. M., Rinklebe, J., Bolan, N. S., & Sarkar, B. (2024). Effects of microplastics on soil microorganisms and microbial functions in nutrients and carbon cycling – A review. *Science of The Total Environment*, 924, 171435. <https://doi.org/10.1016/j.scitotenv.2024.171435>
- Arunrat, N., Uttaratotai, T., Kongsurakan, P., Sereenonchai, S., & Hatano, R. (2025). Bacterial Community Structure in Soils With Fire-Deposited Charcoal Under Rotational Shifting Cultivation of Upland Rice in Northern Thailand. *Ecology and Evolution*, 15(2), e70851. <https://doi.org/10.1002/ece3.70851>
- Auta, H. S., Emenike, C. U., & Fauziah, S. H. (2017). Screening of Bacillus strains isolated from mangrove ecosystems in Peninsular Malaysia for microplastic degradation. *Environmental Pollution*, 231, 1552-1559. <https://doi.org/10.1016/j.envpol.2017.09.043>

- Ayiti, O. E., & Babalola, O. O. (2022). Factors Influencing Soil Nitrification Process and the Effect on Environment and Health [Review]. *Frontiers in Sustainable Food Systems*, 6. <https://doi.org/10.3389/fsufs.2022.821994>
- Bagnall, D. K., Morgan, C. L. S., Bean, G. M., Liptzin, D., Cappellazzi, S. B., Cope, M., Greub, K. L. H., Rieke, E. L., Norris, C. E., Tracy, P. W., Aberle, E., Ashworth, A., Tavarez, O. B., Bary, A. I., Baumhardt, R. L., Gracia, A. B., Brainard, D. C., Brennan, J. R., Reyes, D. B., ... Honeycutt, C. W. (2022). Selecting soil hydraulic properties as indicators of soil health: Measurement response to management and site characteristics. *Soil Science Society of America Journal*, 86(5), 1206-1226. <https://doi.org/10.1002/saj2.20428>
- Baldrian, P., & Brabcová, V. (2008). Baldrian P, Valaskova V.. Degradation of cellulose by basidiomycete fungi. *FEMS Microbiol Rev* 32: 501-521. *FEMS microbiology reviews*, 32, 501-521. <https://doi.org/10.1111/j.1574-6976.2008.00106.x>
- Barnes, D. K., Galgani, F., Thompson, R. C., & Barlaz, M. (2009). Accumulation and fragmentation of plastic debris in global environments. *Philos Trans R Soc Lond B Biol Sci*, 364(1526), 1985-1998. <https://doi.org/10.1098/rstb.2008.0205>
- Bashyal, J. (2024). Nylon: History, Types, Properties, Uses, Importance. Accessed 16.09.2024 from <https://scienceinfo.com/nylon-history-types-properties-uses-importance>
- Beach, E. S., Weeks, B., Stern, R., & Anastas, P. (2010). Plastics additives and green chemistry. *Pure and Applied Chemistry*, 85, 1611-1624. <https://doi.org/10.1351/PAC-CON-12-08-08>
- Behnami, A., Aghayani, E., Benis, K. Z., Sattari, M., & Pourakbar, M. (2022). Comparing the efficacy of various methods for sulfate radical generation for antibiotics degradation in synthetic wastewater: degradation mechanism, kinetics study, and toxicity assessment. *RSC Advances*, 12(23), 14945-14956. <https://doi.org/10.1039/D2RA01618D>
- Bell, M. J., Ransom, M. D., Thompson, M. L., Hinsinger, P., Florence, A. M., Moody, P. W., & Guppy, C. N. (2021). Considering Soil Potassium Pools with Dissimilar Plant Availability. *Improving Potassium Recommendations for Agricultural Crops*, Cham
- Beman, J. M., Vargas, S. M., Wilson, J. M., Perez-Coronel, E., Karolewski, J. S., Vazquez, S., Yu, A., Cairo, A. E., White, M. E., Koester, I., Aluwihare, L. I., & Wankel, S. D. (2021). Substantial oxygen consumption by aerobic nitrite oxidation in oceanic oxygen minimum zones. *Nature Communications*, 12(1), 7043. <https://doi.org/10.1038/s41467-021-27381-7>
- Bengough, A. G. (2012). Water Dynamics of the Root Zone: Rhizosphere Biophysics and Its Control on Soil Hydrology. *Vadose Zone Journal*, 11(2), vzj2011.0111. <https://doi.org/10.2136/vzj2011.0111>
- Berry, D., & Widder, S. (2014). Deciphering microbial interactions and detecting keystone species with co-occurrence networks [Original Research]. *Frontiers in Microbiology*, Volume 5 - 2014. <https://doi.org/10.3389/fmicb.2014.00219>
- Bian, W., An, L., Zhang, S., Feng, J., Sun, D., Yao, Y., Shen, T., Yang, Y., & Zhang, M. (2022). The long-term effects of microplastics on soil organomineral complexes and bacterial communities from controlled-release fertilizer residual coating. *Journal of Environmental Management*, 304, 114193. <https://doi.org/10.1016/j.jenvman.2021.114193>
- Biba, R., Cvjetko, P., Jakopčić, M., Komazec, B., Tkalec, M., Dimitrov, N., Begović, T., & Balen, B. (2023). Phytotoxic effects of polystyrene and polymethyl methacrylate microplastics on *Allium cepa* roots. *Plants*, 12(4), 747. <https://doi.org/10.3390/plants12040747>
- Björkner, B. (2000). Plasticizers and Other Additives in Synthetic Polymers. In Kanerva, L., Wahlberg, J. E., Elsner, P., & Maibach, H. I. (Eds.), *Handbook of Occupational Dermatology* (pp. 688-690). Springer Berlin Heidelberg. https://doi.org/10.1007/978-3-662-07677-4_85

- Boots, B., Russell, C. W., & Green, D. S. (2019). Effects of Microplastics in Soil Ecosystems: Above and Below Ground. *Environ Sci Technol*, 53(19), 11496-11506. <https://doi.org/10.1021/acs.est.9b03304>
- Bosker, T., Bouwman, L. J., Brun, N. R., Behrens, P., & Vijver, M. G. (2019). Microplastics accumulate on pores in seed capsule and delay germination and root growth of the terrestrial vascular plant *Lepidium sativum*. *Chemosphere*, 226, 774-781. <https://doi.org/10.1016/j.chemosphere.2019.03.163>
- Boucher, J., & Friot, D. (2017). Primary Microplastics in the Oceans: A Global Evaluation of Sources. <https://doi.org/10.2305/IUCN.CH.2017.01.en>
- Brandes, E., Henseler, M., & Kreins, P. (2021). Identifying hot-spots for microplastic contamination in agricultural soils—a spatial modelling approach for Germany. *Environmental Research Letters*, 16(10), 104041. <https://doi.org/10.1088/1748-9326/ac21e6>
- Braun, M., Mail, M., Krupp, A. E., & Amelung, W. (2023). Microplastic contamination of soil: Are input pathways by compost overridden by littering? *Science of The Total Environment*, 855, 158889. <https://doi.org/10.1016/j.scitotenv.2022.158889>
- Briassoulis, D. (2023). Agricultural plastics as a potential threat to food security, health, and environment through soil pollution by microplastics: Problem definition. *Science of The Total Environment*, 892, 164533. <https://doi.org/10.1016/j.scitotenv.2023.164533>
- Brown, R. W., Chadwick, D. R., Thornton, H., Marshall, M. R., Bei, S., Distaso, M. A., Bargiela, R., Marsden, K. A., Clode, P. L., Murphy, D. V., Pagella, S., & Jones, D. L. (2022). Field application of pure polyethylene microplastic has no significant short-term effect on soil biological quality and function. *Soil Biology and Biochemistry*, 165, 108496. <https://doi.org/10.1016/j.soilbio.2021.108496>
- Browne, M. A., Crump, P., Niven, S. J., Teuten, E., Tonkin, A., Galloway, T., & Thompson, R. (2011). Accumulation of microplastic on shorelines worldwide: sources and sinks. *Environ Sci Technol*, 45(21), 9175-9179. <https://doi.org/10.1021/es201811s>
- Büks, F., & Kaupenjohann, M. (2020). Global concentrations of microplastics in soils – a review. *SOIL*, 6(2), 649-662. <https://doi.org/10.5194/soil-6-649-2020>
- Bullard, J. E., Zhou, Z., Davis, S., & Fowler, S. (2023). Breakdown and Modification of Microplastic Beads by Aeolian Abrasion. *Environ Sci Technol*, 57(1), 76-84. <https://doi.org/10.1021/acs.est.2c05396>
- Burrows, S. D., Frustaci, S., Thomas, K. V., & Galloway, T. (2020). Expanding exploration of dynamic microplastic surface characteristics and interactions. *TrAC Trends in Analytical Chemistry*, 130, 115993. <https://doi.org/10.1016/j.trac.2020.115993>
- Cai, Z., Li, M., Zhu, Z., Wang, X., Huang, Y., Li, T., Gong, H., & Yan, M. (2023). Biological Degradation of Plastics and Microplastics: A Recent Perspective on Associated Mechanisms and Influencing Factors. *Microorganisms*, 11(7). <https://doi.org/10.3390/microorganisms11071661>
- Calvert, P. D. (1986). Polymer degradation and stabilisation. N. Grassie and G. Scott, Cambridge University Press, Cambridge, 1985. pp. 222, price £27.50, \$54.50. ISBN 0-521-24961-9. *British Polymer Journal*, 18(4), 278-278. <https://doi.org/10.1002/pi.4980180422>
- Carney Almroth, B. M., Åström, L., Roslund, S., Petersson, H., Johansson, M., & Persson, N.-K. (2018). Quantifying shedding of synthetic fibers from textiles; a source of microplastics released into the environment. *Environmental Science and Pollution Research*, 25(2), 1191-1199. <https://doi.org/10.1007/s11356-017-0528-7>
- Carson, J. K., Gonzalez-Quiñones, V., Murphy, D. V., Hinz, C., Shaw, J. A., & Gleeson, D. B. (2010). Low pore connectivity increases bacterial diversity in soil. *Appl Environ Microbiol*, 76(12), 3936-3942. <https://doi.org/10.1128/aem.03085-09>

- Chamas, A., Moon, H., Zheng, J., Qiu, Y., Tabassum, T., Jang, J. H., Abu-Omar, M., Scott, S. L., & Suh, S. (2020). Degradation Rates of Plastics in the Environment. *ACS Sustainable Chemistry & Engineering*, 8(9), 3494-3511. <https://doi.org/10.1021/acssuschemeng.9b06635>
- Chan, C. K. M., Park, C., Chan, K. M., Mak, D. C. W., Fang, J. K. H., & Mitrano, D. M. (2021). Microplastic fibre releases from industrial wastewater effluent: a textile wet-processing mill in China. *Environmental Chemistry*, 18(3), 93-100. <https://doi.org/10.1071/EN20143>
- Chen, F., Aqeel, M., Khalid, N., Nazir, A., Irshad, M. K., Akbar, M. U., Alzuaibr, F. M., Ma, J., & Noman, A. (2023). Interactive effects of polystyrene microplastics and Pb on growth and phytochemicals in mung bean (*Vigna radiata* L.). *Journal of Hazardous Materials*, 449, 130966. <https://doi.org/10.1016/j.jhazmat.2023.130966>
- Chen, H., Ingraffia, R., Schlöter, M., Brüggemann, N., & Rillig, M. C. (2025a). Effects of multiple microplastic types on growth of winter wheat and soil properties vary in different agricultural soils. *PLANTS, PEOPLE, PLANET*, 7(1), 194-203. <https://doi.org/10.1002/ppp3.10573>
- Chen, S., Feng, Y., Han, L., Li, D., Feng, Y., Jeyakumar, P., Sun, H., Shi, W., & Wang, H. (2022a). Responses of rice (*Oryza sativa* L.) plant growth, grain yield and quality, and soil properties to the microplastic occurrence in paddy soil. *Journal of Soils and Sediments*, 22(8), 2174-2183. <https://doi.org/10.1007/s11368-022-03232-w>
- Chen, Y., Stemple, B., Kumar, M., & Wei, N. (2016). Cell Surface Display Fungal Laccase as a Renewable Biocatalyst for Degradation of Persistent Micropollutants Bisphenol A and Sulfamethoxazole. *Environmental Science & Technology*, 50(16), 8799-8808. <https://doi.org/10.1021/acs.est.6b01641>
- Chen, Y., Wang, X., Wang, X., Cheng, T., Fu, K., Qin, Z., & Feng, K. (2022b). Biofilm Structural and Functional Features on Microplastic Surfaces in Greenhouse Agricultural Soil. *Sustainability*, 14(12).
- Chen, Y., Li, Y., Liang, X., Lu, S., Ren, J., Zhang, Y., Han, Z., Gao, B., & Sun, K. (2024). Effects of microplastics on soil carbon pool and terrestrial plant performance. *Carbon Research*, 3(1), 37. <https://doi.org/10.1007/s44246-024-00124-1>
- Chen, Z., Carter, L. J., Banwart, S. A., & Kay, P. (2025b). Microplastics in Soil–Plant Systems: Current Knowledge, Research Gaps, and Future Directions for Agricultural Sustainability. *Agronomy*, 15(7).
- Cho, Y., Shim, W. J., Ha, S. Y., Han, G. M., Jang, M., & Hong, S. H. (2023). Microplastic emission characteristics of stormwater runoff in an urban area: Intra-event variability and influencing factors. *Science of The Total Environment*, 866, 161318. <https://doi.org/10.1016/j.scitotenv.2022.161318>
- Clavel, J., Julliard, R., & Devictor, V. (2011). Worldwide decline of specialist species: toward a global functional homogenization? *Frontiers in Ecology and the Environment*, 9(4), 222-228. <https://doi.org/https://doi.org/10.1890/080216>
- Cole, M., Lindeque, P., Halsband, C., & Galloway, T. S. (2011). Microplastics as contaminants in the marine environment: a review. *Mar Pollut Bull*, 62(12), 2588-2597. <https://doi.org/10.1016/j.marpolbul.2011.09.025>
- Coleman, E. A. (2017). 21 - Plastics Additives. In Kutz, M. (Ed.), *Applied Plastics Engineering Handbook* (Second Edition) (pp. 489-500). William Andrew Publishing. <https://doi.org/10.1016/B978-0-323-39040-8.00021-3>
- Corradini, F., Meza, P., Eguiluz, R., Casado, F., Huerta-Lwanga, E., & Geissen, V. (2019). Evidence of microplastic accumulation in agricultural soils from sewage sludge disposal. *Science of the total environment*, 671, 411-420. <https://doi.org/10.1016/j.scitotenv.2019.03.368>

- Crincoli, K. R., Green, C., & Huling, S. G. (2020). Sulfate Radical Scavenging by Mineral Surfaces in Persulfate-Driven Oxidation Systems: Reaction Rate Constants and Implications. *Environ Sci Technol*, 54(3), 1955-1962. <https://doi.org/10.1021/acs.est.9b06442>
- Cusworth, S. J., Davies, W. J., McAinsh, M. R., Gregory, A. S., Storkey, J., & Stevens, C. J. (2024). Agricultural fertilisers contribute substantially to microplastic concentrations in UK soils. *Communications Earth & Environment*, 5(1), 7. <https://doi.org/10.1038/s43247-023-01172-y>
- da Luz, J. M. R., Paes, S. A., Nunes, M. D., da Silva, M. d. C. S., & Kasuya, M. C. M. (2013). Degradation of Oxo-Biodegradable Plastic by *Pleurotus ostreatus*. *PLOS ONE*, 8(8), e69386. <https://doi.org/10.1371/journal.pone.0069386>
- da Luz, J. M. R., Paes, S. A., Bazzolli, D. M. S., Tótola, M. R., Demuner, A. J., & Kasuya, M. C. M. (2014). Abiotic and Biotic Degradation of Oxo-Biodegradable Plastic Bags by *Pleurotus ostreatus*. *PLOS ONE*, 9(11), e107438. <https://doi.org/10.1371/journal.pone.0107438>
- da Luz, J. M. R., Paes, S. A., Ribeiro, K. V. G., Mendes, I. R., & Kasuya, M. C. M. (2015). Degradation of Green Polyethylene by *Pleurotus ostreatus*. *PLOS ONE*, 10(6), e0126047. <https://doi.org/10.1371/journal.pone.0126047>
- Daebeler, A., Kitzinger, K., Koch, H., Herbold, C. W., Steinfeder, M., Schwarz, J., Zechmeister, T., Karst, S. M., Albertsen, M., Nielsen, P. H., Wagner, M., & Daims, H. (2020). Exploring the upper pH limits of nitrite oxidation: diversity, ecophysiology, and adaptive traits of haloalkalitolerant *Nitrospira*. *Isme j*, 14(12), 2967-2979. <https://doi.org/10.1038/s41396-020-0724-1>
- Daims, H., Lückner, S., & Wagner, M. (2016). A New Perspective on Microbes Formerly Known as Nitrite-Oxidizing Bacteria. *Trends Microbiol*, 24(9), 699-712. <https://doi.org/10.1016/j.tim.2016.05.004>
- Daneshian, B., Habibagahi, G., & Nikooee, E. (2021). Determination of unsaturated hydraulic conductivity of sandy soils: a new pore network approach. *Acta Geotechnica*, 16(2), 449-466. <https://doi.org/10.1007/s11440-020-01088-3>
- Daou, M., & Faulds, C. B. (2017). Glyoxal oxidases: their nature and properties. *World Journal of Microbiology and Biotechnology*, 33(5), 87. <https://doi.org/10.1007/s11274-017-2254-1>
- Darby Richard, T., & Kaplan Arthur, M. (1968). Fungal Susceptibility of Polyurethanes. *Applied Microbiology*, 16(6), 900-905. <https://doi.org/10.1128/am.16.6.900-905.1968>
- de Souza Machado, A. A., Lau, C. W., Till, J., Kloas, W., Lehmann, A., Becker, R., & Rillig, M. C. (2018). Impacts of Microplastics on the Soil Biophysical Environment. *Environ Sci Technol*, 52(17), 9656-9665. <https://doi.org/10.1021/acs.est.8b02212>
- de Souza Machado, A. A., Lau, C. W., Kloas, W., Bergmann, J., Bachelier, J. B., Faltin, E., Becker, R., Görlich, A. S., & Rillig, M. C. (2019). Microplastics Can Change Soil Properties and Affect Plant Performance. *Environmental Science & Technology*, 53(10), 6044-6052. <https://doi.org/10.1021/acs.est.9b01339>
- Ding, L., Wang, X., Ouyang, Z., Chen, Y., Wang, X., Liu, D., Liu, S., Yang, X., Jia, H., & Guo, X. (2021). The occurrence of microplastic in Mu Us Sand Land soils in northwest China: Different soil types, vegetation cover and restoration years. *Journal of Hazardous Materials*, 403, 123982. <https://doi.org/10.1016/j.jhazmat.2020.123982>
- Draschoff, A. (2024). Die sieben Türen. Ullstein Buchverlag GmbH.
- Dris, R., Gasperi, J., Saad, M., Mirande, C., & Tassin, B. (2016). Synthetic fibers in atmospheric fallout: A source of microplastics in the environment? *Marine Pollution Bulletin*, 104(1), 290-293. <https://doi.org/10.1016/j.marpolbul.2016.01.006>
- Du, C., Liang, H., Li, Z., & Gong, J. (2020). Pollution Characteristics of Microplastics in Soils in Southeastern Suburbs of Baoding City, China. *Int J Environ Res Public Health*, 17(3). <https://doi.org/10.3390/ijerph17030845>

- Duan, Z., Zhao, S., Zhao, L., Duan, X., Xie, S., Zhang, H., Liu, Y., Peng, Y., Liu, C., & Wang, L. (2020). Microplastics in Yellow River Delta wetland: Occurrence, characteristics, human influences, and marker. *Environmental Pollution*, 258, 113232. <https://doi.org/10.1016/j.envpol.2019.113232>
- Dube, E., & Okuthe, G. E. (2023). Plastics and Micro/Nano-Plastics (MNPs) in the Environment: Occurrence, Impact, and Toxicity. *Int J Environ Res Public Health*, 20(17). <https://doi.org/10.3390/ijerph20176667>
- Duesterberg, C. K., & Waite, T. D. (2006). Process Optimization of Fenton Oxidation Using Kinetic Modeling. *Environmental Science & Technology*, 40(13), 4189-4195. <https://doi.org/10.1021/es060311v>
- Edo, C., Fernández-Piñas, F., & Rosal, R. (2022). Microplastics identification and quantification in the composted Organic Fraction of Municipal Solid Waste. *Science of The Total Environment*, 813, 151902. <https://doi.org/10.1016/j.scitotenv.2021.151902>
- Erdei, L., Oláh, Z., & Bérczi, A. (2006). Nutrition of winter wheat during the life cycle. II. Influx and translocation of potassium and phosphorus. *Physiologia Plantarum*, 58, 131-135. <https://doi.org/10.1111/j.1399-3054.1983.tb04155.x>
- Eriksen, M., Lebreton, L. C., Carson, H. S., Thiel, M., Moore, C. J., Borerro, J. C., Galgani, F., Ryan, P. G., & Reisser, J. (2014). Plastic Pollution in the World's Oceans: More than 5 Trillion Plastic Pieces Weighing over 250,000 Tons Afloat at Sea. *PLoS One*, 9(12), e111913. <https://doi.org/10.1371/journal.pone.0111913>
- Eshahbanati, S., & Fahrenfeld, N. L. (2016). Influence of wastewater treatment plant discharges on microplastic concentrations in surface water. *Chemosphere*, 162, 277-284. <https://doi.org/10.1016/j.chemosphere.2016.07.083>
- Esterhuizen, M., & Kim, Y. J. (2022). Effects of polypropylene, polyvinyl chloride, polyethylene terephthalate, polyurethane, high-density polyethylene, and polystyrene microplastic on *Nelumbo nucifera* (Lotus) in water and sediment. *Environmental Science and Pollution Research*, 29(12), 17580-17590. <https://doi.org/10.1007/s11356-021-17033-0>
- Esterhuizen, M., Vikfors, S., Penttinen, O.-P., Kim, Y. J., & Pflugmacher, S. (2022). *Lolium multiflorum* germination and growth affected by virgin, naturally, and artificially aged high-density polyethylene microplastic and leachates. *Frontiers in Environmental Science*, 10, 964230. <https://doi.org/10.3389/fenvs.2022.964230>
- Facciola, A., Visalli, G., Pruiti Ciarello, M., & Di Pietro, A. (2021). Newly Emerging Airborne Pollutants: Current Knowledge of Health Impact of Micro and Nanoplastics. *Int J Environ Res Public Health*, 18(6). <https://doi.org/10.3390/ijerph18062997>
- Fajardo, C., Martín, C., Costa, G., Sánchez-Fortún, S., Rodríguez, C., de Lucas Burneo, J. J., Nande, M., Mengs, G., & Martín, M. (2022). Assessing the role of polyethylene microplastics as a vector for organic pollutants in soil: Ecotoxicological and molecular approaches. *Chemosphere*, 288, 132460. <https://doi.org/10.1016/j.chemosphere.2021.132460>
- Farow, D., Lebel, R., Crossman, J., & Proctor, C. (2024). Root traits of soybeans exposed to polyethylene films, polypropylene fragments, and biosolids. *Environmental Pollution*, 363, 125141. <https://doi.org/10.1016/j.envpol.2024.125141>
- Fei, Y., Huang, S., Zhang, H., Tong, Y., Wen, D., Xia, X., Wang, H., Luo, Y., & Barceló, D. (2020). Response of soil enzyme activities and bacterial communities to the accumulation of microplastics in an acid cropped soil. *Science of The Total Environment*, 707, 135634. <https://doi.org/10.1016/j.scitotenv.2019.135634>

- Feng, S., Lu, H., & Liu, Y. (2021). The occurrence of microplastics in farmland and grassland soils in the Qinghai-Tibet plateau: Different land use and mulching time in facility agriculture. *Environmental Pollution*, 279, 116939. <https://doi.org/10.1016/j.envpol.2021.116939>
- Fierer, N., Bradford, M. A., & Jackson, R. B. (2007). TOWARD AN ECOLOGICAL CLASSIFICATION OF SOIL BACTERIA. *Ecology*, 88(6), 1354-1364. <https://doi.org/10.1890/05-1839>
- Fujisawa, M., Hirai, H., & Nishida, T. (2001). Degradation of Polyethylene and Nylon-66 by the Laccase-Mediator System. *Journal of Polymers and the Environment*, 9(3), 103-108. <https://doi.org/10.1023/A:1020472426516>
- Fuller, S., & Gautam, A. (2016). A procedure for measuring microplastics using pressurized fluid extraction. *Environmental science & technology*, 50(11), 5774-5780.
- Galafassi, S., Nizzetto, L., & Volta, P. (2019). Plastic sources: A survey across scientific and grey literature for their inventory and relative contribution to microplastics pollution in natural environments, with an emphasis on surface water. *Sci Total Environ*, 693, 133499. <https://doi.org/10.1016/j.scitotenv.2019.07.305>
- Gallo, F., Fossi, C., Weber, R., Santillo, D., Sousa, J., Ingram, I., Nadal, A., & Romano, D. (2020). Marine litter plastics and microplastics and their toxic chemicals components: the need for urgent preventive measures. In *Analysis of Nanoplastics and Microplastics in Food* (pp. 159-179). CRC Press. <https://doi.org/10.1186/s12302-018-0139-z>
- Gao, J., Pan, S., Li, P., Wang, L., Hou, R., Wu, W. M., Luo, J., & Hou, D. (2021). Vertical migration of microplastics in porous media: Multiple controlling factors under wet-dry cycling. *J Hazard Mater*, 419, 126413. <https://doi.org/10.1016/j.jhazmat.2021.126413>
- Garside, M. (2024). Global plastic production 1950-2022 <https://www.statista.com/statistics/282732/global-production-of-plastics-since-1950/>
- George, T. S., Bulgarelli, D., Carminati, A., Chen, Y., Jones, D., Kuzyakov, Y., Schnepf, A., Wissuwa, M., & Roose, T. (2024). Bottom-up perspective – The role of roots and rhizosphere in climate change adaptation and mitigation in agroecosystems. *Plant and Soil*, 500(1), 297-323. <https://doi.org/10.1007/s11104-024-06626-6>
- Geyer, R., Jambeck, J. R., & Law, K. L. (2017). Production, use, and fate of all plastics ever made. *Science Advances*, 3(7), e1700782. <https://doi.org/10.1126/sciadv.1700782>
- Gijsman, P., Meijers, G., & Vitarelli, G. (1999). Comparison of the UV-degradation chemistry of polypropylene, polyethylene, polyamide 6 and polybutylene terephthalate. *Polymer Degradation and Stability*, 65(3), 433-441. [https://doi.org/10.1016/S0141-3910\(99\)00033-6](https://doi.org/10.1016/S0141-3910(99)00033-6)
- Giles, M., Morley, N., Baggs, E. M., & Daniell, T. J. (2012). Soil nitrate reducing processes - drivers, mechanisms for spatial variation, and significance for nitrous oxide production. *Front Microbiol*, 3, 407. <https://doi.org/10.3389/fmicb.2012.00407>
- Gkoutselis, G., Rohrbach, S., Harjes, J., Obst, M., Brachmann, A., Horn, M. A., & Rambold, G. (2021). Microplastics accumulate fungal pathogens in terrestrial ecosystems. *Scientific Reports*, 11(1), 13214. <https://doi.org/10.1038/s41598-021-92405-7>
- Gómez-Méndez, L. D., Moreno-Bayona, D. A., Poutou-Pinales, R. A., Salcedo-Reyes, J. C., Pedroza-Rodríguez, A. M., Vargas, A., & Bogoya, J. M. (2018). Biodeterioration of plasma pretreated LDPE sheets by *Pleurotus ostreatus*. *PloS one*, 13(9), e0203786. <https://doi.org/10.1371/journal.pone.0203786>
- Gonçalves, O. S., Fernandes, A. S., Tupy, S. M., Ferreira, T. G., Almeida, L. N., Creevey, C. J., & Santana, M. F. (2024). Insights into plant interactions and the biogeochemical role of the globally widespread Acidobacteriota phylum. *Soil Biology and Biochemistry*, 192, 109369. <https://doi.org/10.1016/j.soilbio.2024.109369>

- Gonda, K. E., Jendrossek, D., & Molitoris, H. P. (2000). Fungal degradation of the thermoplastic polymer poly- β -hydroxybutyric acid (PHB) under simulated deep sea pressure. *Hydrobiologia*, 426(1), 173-183. <https://doi.org/10.1023/A:1003971925285>
- Gong, M., Yang, G., Zhuang, L., & Zeng, E. Y. (2019). Microbial biofilm formation and community structure on low-density polyethylene microparticles in lake water microcosms. *Environmental Pollution*, 252, 94-102. <https://doi.org/10.1016/j.envpol.2019.05.090>
- Gong, Z., Jin, L., Yu, X., Wang, B., Hu, S., Ruan, H., Sung, Y.-J., Lee, H.-G., & Jin, F. (2023). Biodegradation of Low Density Polyethylene by the Fungus *Cladosporium* sp. Recovered from a Landfill Site. *Journal of Fungi*, 9(6).
- Griffiths, B. S., & Philippot, L. (2013). Insights into the resistance and resilience of the soil microbial community. *FEMS Microbiology Reviews*, 37(2), 112-129. <https://doi.org/10.1111/j.1574-6976.2012.00343.x>
- Gu, J.-D. (2003). Microbiological deterioration and degradation of synthetic polymeric materials: recent research advances. *International Biodeterioration & Biodegradation*, 52(2), 69-91. [https://doi.org/10.1016/S0964-8305\(02\)00177-4](https://doi.org/10.1016/S0964-8305(02)00177-4)
- Guedes, A. M. F. M., Madeira, L. M. P., Boaventura, R. A. R., & Costa, C. A. V. (2003). Fenton oxidation of cork cooking wastewater—overall kinetic analysis. *Water Research*, 37(13), 3061-3069. [https://doi.org/10.1016/S0043-1354\(03\)00178-7](https://doi.org/10.1016/S0043-1354(03)00178-7)
- Gunaalan, K., Fabbri, E., & Capolupo, M. (2020). The hidden threat of plastic leachates: A critical review on their impacts on aquatic organisms [Article]. *Water Research*, 184, Article 116170. <https://doi.org/10.1016/j.watres.2020.116170>
- Guo, J.-J., Huang, X.-P., Xiang, L., Wang, Y.-Z., Li, Y.-W., Li, H., Cai, Q.-Y., Mo, C.-H., & Wong, M.-H. (2020). Source, migration and toxicology of microplastics in soil. *Environment International*, 137, 105263. <https://doi.org/10.1016/j.envint.2019.105263>
- Guo, W., Ye, Z., Zhao, Y., Lu, Q., Shen, B., Zhang, X., Zhang, W., Chen, S.-C., & Li, Y. (2024). Effects of different microplastic types on soil physicochemical properties, enzyme activities, and bacterial communities. *Ecotoxicology and Environmental Safety*, 286, 117219. <https://doi.org/10.1016/j.ecoenv.2024.117219>
- Guo, Z., Li, P., Yang, X., Wang, Z., Lu, B., Chen, W., Wu, Y., Li, G., Zhao, Z., Liu, G., Ritsema, C., Geissen, V., & Xue, S. (2022). Soil texture is an important factor determining how microplastics affect soil hydraulic characteristics. *Environment International*, 165, 107293. <https://doi.org/10.1016/j.envint.2022.107293>
- Hahladakis, J. N., Velis, C. A., Weber, R., Iacovidou, E., & Purnell, P. (2018). An overview of chemical additives present in plastics: Migration, release, fate and environmental impact during their use, disposal and recycling. *J Hazard Mater*, 344, 179-199. <https://doi.org/10.1016/j.jhazmat.2017.10.014>
- Hangele, P., Luise Müller, K., Laermanns, H., & Bogner, C. (2020, May 01, 2020). The influence of microplastic on soil hydraulic properties EGU General Assembly Conference Abstracts, <https://ui.adsabs.harvard.edu/abs/2020EGUGA..22.9337>
- Harms, I. K., Diekötter, T., Troegel, S., & Lenz, M. (2021). Amount, distribution and composition of large microplastics in typical agricultural soils in Northern Germany. *Science of The Total Environment*, 758, 143615. <https://doi.org/10.1016/j.scitotenv.2020.143615>
- Hatakka, A. (Ed.). (2001). *Lignin, Humic Substances and Coal* (Vol. 1). Wiley-VHC.
- Hegan, D., Tong, L., Zhiquan, H., Qinming, S., & Ru, L. (2015). Determining time limits of continuous film mulching and examining residual effects on cotton yield and soil properties. *Journal of Environmental Biology*, 36(3), 677.

- Heinrich-Böll-Stiftung, B. (2019). Plastikatlas 2019: Daten und Fakten über eine Welt voller Kunststoff (Fuhr, L., Buschmann, R., Freund, J., Schächtele, K., & Schmidt-Landenberger, E., Eds. 6 ed.). Heinrich-Böll-Stiftung. https://www.boell.de/sites/default/files/2022-01/Boell_Plastikatlas%202019%206.Auflage_V01_kommentierbar.pdf
- Heinze, W. M., Steinmetz, Z., Klemmensen, N. D. R., Vollertsen, J., & Cornelis, G. (2024). Vertical distribution of microplastics in an agricultural soil after long-term treatment with sewage sludge and mineral fertiliser. *Environmental Pollution*, 356, 124343. <https://doi.org/10.1016/j.envpol.2024.124343>
- Helliwell, J. R., Sturrock, C. J., Miller, A. J., Whalley, W. R., & Mooney, S. J. (2019). The role of plant species and soil condition in the structural development of the rhizosphere. *Plant, Cell & Environment*, 42(6), 1974-1986. <https://doi.org/10.1111/pce.13529>
- Helmberger, M. S., Miesel, J. R., Tiemann, L. K., & Grieshop, M. J. (2022). Soil Invertebrates Generate Microplastics From Polystyrene Foam Debris. *Journal of Insect Science*, 22(1), 21. <https://doi.org/10.1093/jisesa/ieac005>
- Hernandez, D. J., David, A. S., Menges, E. S., Searcy, C. A., & Afkhami, M. E. (2021). Environmental stress destabilizes microbial networks. *The ISME Journal*, 15(6), 1722-1734. <https://doi.org/10.1038/s41396-020-00882-x>
- Hoang, V.-H., Nguyen, M.-K., Hoang, T.-D., Ha, M. C., Huyen, N. T. T., Bui, V. K. H., Pham, M.-T., Nguyen, C.-M., Chang, S. W., & Nguyen, D. D. (2024). Sources, environmental fate, and impacts of microplastic contamination in agricultural soils: A comprehensive review. *Science of The Total Environment*, 950, 175276. <https://doi.org/10.1016/j.scitotenv.2024.175276>
- Hu, K., Zhou, P., Yang, Y., Hall, T., Nie, G., Yao, Y., Duan, X., & Wang, S. (2022). Degradation of Microplastics by a Thermal Fenton Reaction. *ACS ES&T Engineering*, 2(1), 110-120. <https://doi.org/10.1021/acsestengg.1c00323>
- Huang, D., Wang, X., Yin, L., Chen, S., Tao, J., Zhou, W., Chen, H., Zhang, G., & Xiao, R. (2022). Research progress of microplastics in soil-plant system: Ecological effects and potential risks. *Sci Total Environ*, 812, 151487. <https://doi.org/10.1016/j.scitotenv.2021.151487>
- Huang, R., McGrath, S. P., Hirsch, P. R., Clark, I. M., Storkey, J., Wu, L., Zhou, J., & Liang, Y. (2019a). Plant-microbe networks in soil are weakened by century-long use of inorganic fertilizers. *Microbial Biotechnology*, 12(6), 1464-1475. <https://doi.org/10.1111/1751-7915.13487>
- Huang, Y., Zhao, Y., Wang, J., Zhang, M., Jia, W., & Qin, X. (2019b). LDPE microplastic films alter microbial community composition and enzymatic activities in soil. *Environmental Pollution*, 254, 112983. <https://doi.org/10.1016/j.envpol.2019.112983>
- Huang, Y., Liu, Q., Jia, W., Yan, C., & Wang, J. (2020). Agricultural plastic mulching as a source of microplastics in the terrestrial environment. *Environmental Pollution*, 260, 114096. <https://doi.org/10.1016/j.envpol.2020.114096>
- Igiehon, N. O., & Babalola, O. O. (2018). Rhizosphere Microbiome Modulators: Contributions of Nitrogen Fixing Bacteria towards Sustainable Agriculture. *Int J Environ Res Public Health*, 15(4). <https://doi.org/10.3390/ijerph15040574>
- Ikhajagbe, B., Omoregie, G. O., Adama, S. O., & Eshey, K. U. (2023). Growth responses of *Celosia argentea* L. in soils polluted with microplastics. *bioRxiv*, 2023-2001. <https://doi.org/10.1101/2023.01.07.523084>
- Ingraffia, R., Amato, G., Bagarello, V., Carollo, F. G., Giambalvo, D., Iovino, M., Lehmann, A., Rillig, M. C., & Frenda, A. S. (2021). Polyester Microplastic Fibers affect Soil Physical Properties and Erosion as a Function of Soil Type. *SOIL Discuss.*, 2021, 1-22. <https://doi.org/10.5194/soil-2021-136>

- Ingraffia, R., Amato, G., Bagarello, V., Carollo, F. G., Giambalvo, D., Iovino, M., Lehmann, A., Rillig, M. C., & Frenda, A. S. (2022a). Polyester microplastic fibers affect soil physical properties and erosion as a function of soil type. *SOIL*, 8(1), 421-435. <https://doi.org/10.5194/soil-8-421-2022>
- Ingraffia, R., Amato, G., Iovino, M., Rillig, M. C., Giambalvo, D., & Frenda, A. S. (2022b). Polyester microplastic fibers in soil increase nitrogen loss via leaching and decrease plant biomass production and N uptake. *Environmental Research Letters*, 17(5), 054012. <https://doi.org/10.1088/1748-9326/ac652d>
- Iqbal, R., Khan, M. T., Bilal, H., Aslam, M. M., Khan, I. A., Raja, S., Arslan, M., & Nguyen, P. M. (2022). Microplastics as vectors of environmental contaminants: Interactions in the natural ecosystems. *Human and Ecological Risk Assessment: An International Journal*, 28(9), 1022-1042. <https://doi.org/10.1080/10807039.2022.2120846>
- Ivanic, F. M., Guggenberger, G., Woche, S. K., Bachmann, J., Hoppe, M., & Carstens, J. F. (2023). Soil organic matter facilitates the transport of microplastic by reducing surface hydrophobicity. *Colloids and Surfaces A: Physicochemical and Engineering Aspects*, 676, 132255. <https://doi.org/10.1016/j.colsurfa.2023.132255>
- Jambeck, J. R., Geyer, R., Wilcox, C., Siegler, T. R., Perryman, M., Andrady, A., Narayan, R., & Law, K. L. (2015). Plastic waste inputs from land into the ocean. *Science*, 347(6223), 768-771.
- Jansen, M. A. K., Andrady, A. L., Bornman, J. F., Aucamp, P. J., Bais, A. F., Banaszak, A. T., Barnes, P. W., Bernhard, G. H., Bruckman, L. S., Busquets, R., Häder, D.-P., Hanson, M. L., Heikkilä, A. M., Hylander, S., Lucas, R. M., Mackenzie, R., Madronich, S., Neale, P. J., Neale, R. E., Olsen, C. M., Ossola, R., Pandey, K. K., Petropavlovskikh, I., Revell, L. E., Robinson, S. A., Robson, T. M., Rose, K. C., Solomon, K. R., Andersen, M. P. S., Sulzberger, B., Wallington, T. J., Wang, Q.-W., Wängberg, S.-Å., White, C. C., Young, A. R., Zepp, R. G., & Zhu, L. (2024). Plastics in the environment in the context of UV radiation, climate change and the Montreal Protocol: UNEP Environmental Effects Assessment Panel, Update 2023. *Photochemical & Photobiological Sciences*, 23(4), 629-650. <https://doi.org/10.1007/s43630-024-00552-3>
- Jeon, H. J., & Kim, M. N. (2013). Isolation of a thermophilic bacterium capable of low-molecular-weight polyethylene degradation. *Biodegradation*, 24, 89-98.
- Jia, H., Wu, D., Yu, Y., Han, S., Sun, L., & Li, M. (2022). Impact of microplastics on bioaccumulation of heavy metals in rape (*Brassica napus* L.). *Chemosphere*, 288, 132576.
- Jia, L., Liu, L., Zhang, Y., Fu, W., Liu, X., Wang, Q., Tanveer, M., & Huang, L. (2023). Microplastic stress in plants: effects on plant growth and their remediations. *Front Plant Sci*, 14, 1226484. <https://doi.org/10.3389/fpls.2023.1226484>
- Jiang, X., Chen, H., Liao, Y., Ye, Z., Li, M., & Klobučar, G. (2019). Ecotoxicity and genotoxicity of polystyrene microplastics on higher plant *Vicia faba*. *Environmental Pollution*, 250, 831-838.
- Jin, T., Liu, X., Cui, J., Peng, J., & Liu, Q. (2024). Macroplastics Quantity and Its Influence on Soil Nutrients in Typical Plastic Film Mulching Farmland in Northern Xinjiang. *Agriculture*, 14(5).
- Joos, L., & De Tender, C. (2022). Soil under stress: The importance of soil life and how it is influenced by (micro)plastic pollution. *Comput Struct Biotechnol J*, 20, 1554-1566. <https://doi.org/10.1016/j.csbj.2022.03.041>
- Kabir, M. S., Wang, H., Luster-Teasley, S., Zhang, L., & Zhao, R. (2023). Microplastics in landfill leachate: Sources, detection, occurrence, and removal. *Environ Sci Ecotechnol*, 16, 100256. <https://doi.org/10.1016/j.ese.2023.100256>
- Karlsson, T. M., Arneborg, L., Broström, G., Almroth, B. C., Gipperth, L., & Hassellöv, M. (2018). The unaccountability case of plastic pellet pollution. *Mar Pollut Bull*, 129(1), 52-60. <https://doi.org/10.1016/j.marpolbul.2018.01.041>

- Kausch, H. H. (2005). The Effect of Degradation and Stabilization on the Mechanical Properties of Polymers Using Polypropylene Blends as the Main Example. *Macromolecular Symposia*, 225(1), 165-178. <https://doi.org/10.1002/masy.200550713>
- Kedzierski, M., Ciredorf-Boulant, D., Palazot, M., Yvin, M., & Bruzard, S. (2023). Continents of plastics: An estimate of the stock of microplastics in agricultural soils. *Science of The Total Environment*, 880, 163294. <https://doi.org/10.1016/j.scitotenv.2023.163294>
- Keefer, R. F. (1999). Parts of a Soil (Soil Constituents) Air, Water, Minerals, and Organic Matter. In Keefer, R. F. (Ed.), *Handbook of Soils for Landscape Architects* (pp. 0). Oxford University Press. <https://doi.org/10.1093/oso/9780195121025.003.0005>
- Kernchen, S., Schmalz, H., Löder, M. G. J., Georgi, C., Einhorn, A., Greiner, A., Nölscher, A. C., Laforsch, C., & Held, A. (2024). Atmospheric deposition studies of microplastics in Central Germany. *Air Quality, Atmosphere & Health*. <https://doi.org/10.1007/s11869-024-01571-w>
- Khan, M. A., Kumar, S., Wang, Q., Wang, M., Fahad, S., Nizamani, M. M., Chang, K., Khan, S., Huang, Q., & Zhu, G. (2023). Influence of polyvinyl chloride microplastic on chromium uptake and toxicity in sweet potato. *Ecotoxicology and Environmental Safety*, 251, 114526. <https://doi.org/10.1016/j.ecoenv.2023.114526>
- Khan, S., Nadir, S., Shah, Z. U., Shah, A. A., Karunarathna, S. C., Xu, J., Khan, A., Munir, S., & Hasan, F. (2017). Biodegradation of polyester polyurethane by *Aspergillus tubingensis*. *Environmental Pollution*, 225, 469-480. <https://doi.org/10.1016/j.envpol.2017.03.012>
- Klein, M., & Fischer, E. K. (2019). Microplastic abundance in atmospheric deposition within the Metropolitan area of Hamburg, Germany. *Science of the Total Environment*, 685, 96-103.
- Krehl, A., Schöllkopf, U., Májeková, M., Tielbörger, K., & Tomiolo, S. (2022). Effects of plastic fragments on plant performance are mediated by soil properties and drought. *Scientific Reports*, 12(1), 17771. <https://doi.org/10.1038/s41598-022-22270-5>
- Krenos, J. R. (2001). *Chemical Kinetics and Reaction Dynamics* (Houston, Paul L.). Journal of Chemical Education, 78, 1466.
- Krueger, M. C., Hofmann, U., Moeder, M., & Schlosser, D. (2015). Potential of Wood-Rotting Fungi to Attack Polystyrene Sulfonate and Its Depolymerisation by *Gloeophyllum trabeum* via Hydroquinone-Driven Fenton Chemistry. *PLoS One*, 10(7), e0131773. <https://doi.org/10.1371/journal.pone.0131773>
- Kublik, S., Gschwendtner, S., Magritsch, T., Radl, V., Rillig, M. C., & Schlöter, M. (2022). Microplastics in soil induce a new microbial habitat, with consequences for bulk soil microbiomes. *Frontiers in Environmental Science*, 10. <https://doi.org/10.3389/fenvs.2022.989267>
- Kukharchyk, T. I., & Chernyk, V. D. (2022). Soil Pollution with Microplastic in the Impact Area of a Plant Producing Expanded Polystyrene. *Eurasian Soil Science*, 55(3), 377-386. <https://doi.org/10.1134/S1064229322030085>
- Kwan, C. S., & Takada, H. (2019). Release of Additives and Monomers from Plastic Wastes. In Takada, H. & Karapanagioti, H. K. (Eds.), *Hazardous Chemicals Associated with Plastics in the Marine Environment* (pp. 51-70). Springer International Publishing. <https://doi.org/10.1007/978-2016-122>
- Laermanns, H., Lehmann, M., Klee, M., J. Löder, M. G., Gekle, S., & Bogner, C. (2021). Tracing the horizontal transport of microplastics on rough surfaces. *Microplastics and Nanoplastics*, 1(1), 11. <https://doi.org/10.1186/s43591-021-00010-2>
- Lambert, S., Scherer, C., & Wagner, M. (2017). Ecotoxicity testing of microplastics: Considering the heterogeneity of physicochemical properties. *Integr Environ Assess Manag*, 13(3), 470-475. <https://doi.org/10.1002/ieam.1901>

- Lang, M., Yu, X., Liu, J., Xia, T., Wang, T., Jia, H., & Guo, X. (2020). Fenton aging significantly affects the heavy metal adsorption capacity of polystyrene microplastics. *Science of the total environment*, 722, 137762. <https://doi.org/10.1016/j.scitotenv.2020.137762>
- Lareen, A., Burton, F., & Schäfer, P. (2016). Plant root-microbe communication in shaping root microbiomes. *Plant Mol Biol*, 90(6), 575-587. <https://doi.org/10.1007/s11103-015-0417-8>
- Lärm, L., Bauer, F. M., Hermes, N., van der Kruk, J., Vereecken, H., Vanderborght, J., Nguyen, T. H., Lopez, G., Seidel, S. J., Ewert, F., Schnepf, A., & Klotzsche, A. (2023). Multi-year belowground data of minirhizotron facilities in Selhausen. *Scientific Data*, 10(1), 672. <https://doi.org/10.1038/s41597-023-02570-9>
- Laurent, F., Cébron, A., Schwartz, C., & Leyval, C. (2012). Oxidation of a PAH polluted soil using modified Fenton reaction in unsaturated condition affects biological and physico-chemical properties. *Chemosphere*, 86(6), 659-664. <https://doi.org/10.1016/j.chemosphere.2011.11.018>
- Lechner, A., & Ramler, D. (2015). The discharge of certain amounts of industrial microplastic from a production plant into the River Danube is permitted by the Austrian legislation. *Environmental Pollution*, 200, 159-160. <https://doi.org/10.1016/j.envpol.2015.02.019>
- Lee, J., Jeong, S., Long, C., & Chandran, K. (2022a). Size dependent impacts of a model microplastic on nitrification induced by interaction with nitrifying bacteria. *Journal of Hazardous Materials*, 424, 127363. <https://doi.org/10.1016/j.jhazmat.2021.127363>
- Lee, T.-Y., Kim, L., Kim, D., An, S., & An, Y.-J. (2022b). Microplastics from shoe sole fragments cause oxidative stress in a plant (*Vigna radiata*) and impair soil environment. *Journal of Hazardous Materials*, 429, 128306. <https://doi.org/10.1016/j.jhazmat.2022.128306>
- Lehmann, A., Fitschen, K., & Rillig, M. C. (2019). Abiotic and Biotic Factors Influencing the Effect of Microplastic on Soil Aggregation. *Soil Systems*, 3(1).
- Lehmann, A., Leifheit, E. F., Gerdawischke, M., & Rillig, M. C. (2021). Microplastics have shape- and polymer-dependent effects on soil aggregation and organic matter loss – an experimental and meta-analytical approach. *Microplastics and Nanoplastics*, 1(1), 7. <https://doi.org/10.1186/s43591-021-00007-x>
- Lei, S., Xu, X., Cheng, Z., Xiong, J., Ma, R., Zhang, L., Yang, X., Zhu, Y., Zhang, B., & Tian, B. (2019). Analysis of the community composition and bacterial diversity of the rhizosphere microbiome across different plant taxa. *MicrobiologyOpen*, 8(6), e00762. <https://doi.org/10.1002/mbo3.762>
- Li, C., Ma, B., & Zhang, T. (2002). Soil bulk density effects on soil microbial populations and enzyme activities during the growth of maize (*Zea mays* L.) planted in large pots under field exposure. *Canadian Journal of Soil Science*, 82, 147-154. <https://doi.org/10.4141/S01-026>
- Li, C., Cui, Q., Li, Y., Zhang, K., Lu, X., & Zhang, Y. (2022a). Effect of LDPE and biodegradable PBAT primary microplastics on bacterial community after four months of soil incubation. *Journal of Hazardous Materials*, 429, 128353. <https://doi.org/10.1016/j.jhazmat.2022.128353>
- Li, H., & Liu, L. (2022). Short-term effects of polyethylene and polypropylene microplastics on soil phosphorus and nitrogen availability. *Chemosphere*, 291(Pt 2), 132984. <https://doi.org/10.1016/j.chemosphere.2021.132984>
- Li, H., Chang, X., Zhang, J., Wang, Y., Zhong, R., Wang, L., Wei, J., & Wang, Y. (2023a). Uptake and distribution of microplastics of different particle sizes in maize (*Zea mays*) seedling roots. *Chemosphere*, 313, 137491. <https://doi.org/10.1016/j.chemosphere.2021.132984>
- Li, J., Yu, S., Yu, Y., & Xu, M. (2022b). Effects of Microplastics on Higher Plants: A Review. *Bull Environ Contam Toxicol*, 109(2), 241-265. <https://doi.org/10.1007/s00128-022-03566-8>

- Li, J., Yu, Y., Zhang, Z., & Cui, M. (2023b). The positive effects of polypropylene and polyvinyl chloride microplastics on agricultural soil quality. *Journal of Soils and Sediments*, 23(3), 1304-1314. <https://doi.org/10.1007/s11368-022-03387-6>
- Li, L., Zhou, Q., Yin, N., Tu, C., & Luo, Y. (2019). Uptake and accumulation of microplastics in an edible plant. *Chinese Science Bulletin*, 64(9), 928-934. <https://doi.org/10.1016/j.jhazmat.2021.126700>
- Li, Z., Yang, Y., Chen, X., He, Y., Bolan, N., Rinklebe, J., Lam, S. S., Peng, W., & Sonne, C. (2023c). A discussion of microplastics in soil and risks for ecosystems and food chains. *Chemosphere*, 313, 137637. <https://doi.org/10.1016/j.chemosphere.2022.137637>
- Lian, Y., Shi, R., Liu, J., Zeb, A., Wang, Q., Wang, J., Yu, M., Li, J., Zheng, Z., Ali, N., Bao, Y., & Liu, W. (2024). Effects of polystyrene, polyethylene, and polypropylene microplastics on the soil-rhizosphere-plant system: Phytotoxicity, enzyme activity, and microbial community. *Journal of Hazardous Materials*, 465, 133417. <https://doi.org/10.1016/j.jhazmat.2023.133417>
- Liang, Y., Lehmann, A., Yang, G., Leifheit, E. F., & Rillig, M. C. (2021). Effects of Microplastic Fibers on Soil Aggregation and Enzyme Activities Are Organic Matter Dependent [Original Research]. *Frontiers in Environmental Science*, Volume 9 - 2021. <https://doi.org/10.3389/fenvs.2021.650155>
- Lichtenthaler, H. K., & Wellburn, A. R. (1983). Determinations of total carotenoids and chlorophylls a and b of leaf extracts in different solvents. *Biochemical Society Transactions*, 11(5), 591-592. <https://doi.org/10.1042/bst0110591>
- Lin, P.-J., Yeh, C.-H., & Jiang, J.-C. (2021). Theoretical insight into hydroxyl production via H₂O₂ decomposition over the Fe₃O₄(311) surface [10.1039/D1RA06943H]. *RSC Advances*, 11(57), 36257-36264. <https://doi.org/10.1039/D1RA06943H>
- Linderman, R. G. (1991). Mycorrhizal interactions in the rhizosphere. In Keister, D. L. & Cregan, P. B. (Eds.), *The Rhizosphere and Plant Growth: Papers presented at a Symposium held May 8–11, 1989, at the Beltsville Agricultural Research Center (BARC), Beltsville, Maryland* (pp. 343-348). Springer Netherlands. https://doi.org/10.1007/978-94-011-3336-4_73
- Ling, N., Wang, T., & Kuzyakov, Y. (2022). Rhizosphere bacteriome structure and functions. *Nature Communications*, 13(1), 836. <https://doi.org/10.1038/s41467-022-28448-9>
- Liu, B., Fan, H., Han, W., Zhu, L., Zhao, X., Zhang, Y., & Ma, R. (2021). Linking soil water retention capacity to pore structure characteristics based on X-ray computed tomography: Chinese Mollisol under freeze-thaw effect. *Geoderma*, 401, 115170. <https://doi.org/10.1016/j.geoderma.2021.115170>
- Liu, B., Jiang, Q., Qiu, Z., Liu, L., Wei, R., Zhang, X., & Xu, H. (2022a). Process analysis of microplastic degradation using activated PMS and Fenton reagents. *Chemosphere*, 298, 134220. <https://doi.org/10.1016/j.chemosphere.2022.134220>
- Liu, H., Li, X. Z., Leng, Y. J., & Wang, C. (2007). Kinetic modeling of electro-Fenton reaction in aqueous solution. *Water Research*, 41(5), 1161-1167. <https://doi.org/10.1016/j.watres.2006.12.006>
- Liu, H., Yang, X., Liu, G., Liang, C., Xue, S., Chen, H., Ritsema, C. J., & Geissen, V. (2017). Response of soil dissolved organic matter to microplastic addition in Chinese loess soil. *Chemosphere*, 185, 907-917. <https://doi.org/10.1016/j.chemosphere.2017.07.064>
- Liu, M., Lu, S., Song, Y., Lei, L., Hu, J., Lv, W., Zhou, W., Cao, C., Shi, H., Yang, X., & He, D. (2018). Microplastic and mesoplastic pollution in farmland soils in suburbs of Shanghai, China. *Environmental Pollution*, 242, 855-862. <https://doi.org/10.1016/j.envpol.2018.07.051>
- Liu, S., Deng, W.-K., Niu, S.-H., Mo, C.-H., Liao, X.-D., & Xing, S.-C. (2023a). Doxycycline combined manure microbes to enhances biofilm formation of the soil plastisphere and increases the surface bio-risk of microplastics vehicle. *Chemical Engineering Journal*, 454, 140530. <https://doi.org/10.1016/j.cej.2022.140530>

- Liu, Y., Xiao, M., Shahbaz, M., Hu, Z. e., Zhu, Z., Lu, S., Yu, Y., Yao, H., Chen, J., & Ge, T. (2022b). Microplastics in soil can increase nutrient uptake by wheat. *Journal of Hazardous Materials*, 438, 129547. <https://doi.org/10.1016/j.jhazmat.2022.129547>
- Liu, Y., Jin, T., Wang, L., & Tang, J. (2023b). Polystyrene micro and nanoplastics attenuated the bioavailability and toxic effects of Perfluorooctane sulfonate (PFOS) on soybean (*Glycine max*) sprouts. *Journal of Hazardous Materials*, 448, 130911. <https://doi.org/10.1016/j.jhazmat.2023.130911>
- Liu, Y., Xu, F., Ding, L., Zhang, G., Bai, B., Han, Y., Xiao, L., Song, Y., Li, Y., Wan, S., & Li, G. (2023c). Microplastics reduce nitrogen uptake in peanut plants by damaging root cells and impairing soil nitrogen cycling. *Journal of Hazardous Materials*, 443, 130384. <https://doi.org/10.1016/j.jhazmat.2022.130384>
- Lozano, Y. M., & Rillig, M. C. (2020). Effects of Microplastic Fibers and Drought on Plant Communities. *Environmental Science & Technology*, 54(10), 6166-6173. <https://doi.org/10.1021/acs.est.0c01051>
- Lozano, Y. M., Aguilar-Trigueros, C. A., Onandia, G., Maaß, S., Zhao, T., & Rillig, M. C. (2021a). Effects of microplastics and drought on soil ecosystem functions and multifunctionality. *Journal of Applied Ecology*, 58(5), 988-996. <https://doi.org/10.1111/1365-2664.13839>
- Lozano, Y. M., Lehnert, T., Linck, L. T., Lehmann, A., & Rillig, M. C. (2021b). Microplastic Shape, Polymer Type, and Concentration Affect Soil Properties and Plant Biomass [Original Research]. *Frontiers in Plant Science*, 12. <https://doi.org/10.3389/fpls.2021.616645>
- Lozano, Y. M., & Rillig, M. C. (2022). Legacy effect of microplastics on plant–soil feedbacks [Original Research]. *Frontiers in Plant Science*, 13. <https://doi.org/10.3389/fpls.2022.965576>
- Lozano, Y. M., Dueñas, J. F., Zordick, C., & Rillig, M. C. (2024). Microplastic fibres affect soil fungal communities depending on drought conditions with consequences for ecosystem functions. *Environ Microbiol*, 26(2), e16549. <https://doi.org/10.1111/1462-2920.16549>
- Luo, H., Liu, C., He, D., Sun, J., Li, J., & Pan, X. (2022). Effects of aging on environmental behavior of plastic additives: Migration, leaching, and ecotoxicity. *Science of The Total Environment*, 849, 157951. <https://doi.org/10.1016/j.scitotenv.2022.157951>
- Lyu, X., Liu, Y., Li, N., Ku, L., Hou, Y., & Wen, X. (2022). Foliar applications of various nitrogen (N) forms to winter wheat affect grain protein accumulation and quality via N metabolism and remobilization. *The Crop Journal*, 10(4), 1165-1177. <https://doi.org/10.1016/j.cj.2021.10.009>
- Ma, L., Hoogenboom, G., Saseendran, S. A., Bartling, P. N. S., Ahuja, L. R., & Green, T. R. (2009). Effects of Estimating Soil Hydraulic Properties and Root Growth Factor on Soil Water Balance and Crop Production. *Agronomy Journal*, 101(3), 572-583. <https://doi.org/10.2134/agronj2008.0206x>
- Maddela, N. R., Ramakrishnan, B., Kadiyala, T., Venkateswarlu, K., & Megharaj, M. (2023). Do Microplastics and Nanoplastics Pose Risks to Biota in Agricultural Ecosystems? *Soil Systems*, 7(1), 19. <https://www.mdpi.com/2571-8789/7/1/19>
- Magalhães, S., Alves, L., Romano, A., Medronho, B., & Rasteiro, M. D. (2022). Extraction and Characterization of Microplastics from Portuguese Industrial Effluents. *Polymers*, 14(14).
- Mahon, A. M., O'Connell, B., Healy, M. G., O'Connor, I., Officer, R., Nash, R., & Morrison, L. (2017). Microplastics in Sewage Sludge: Effects of Treatment. *Environmental Science & Technology*, 51(2), 810-818. <https://doi.org/10.1021/acs.est.6b04048>
- Malpass, D. B. (2010). *Introduction to Industrial Polyethylene: Properties, Catalysts, and Processes*. Wiley. <https://books.google.de/books?id=-7cWD7MrtfEC>

- Manikanda Bharath, K., Muthulakshmi, A. L., & Natesan, U. (2023). Microplastic contamination around the landfills: Distribution, characterization and threats: A review. *Current Opinion in Environmental Science & Health*, 31, 100422. <https://doi.org/10.1016/j.coesh.2022.100422>
- Mantoglou, A., & Gelhar, L. W. (1987). Effective hydraulic conductivities of transient unsaturated flow in stratified soils. *Water Resources Research*, 23(1), 57-67. <https://doi.org/10.1029/WR023i001p00057>
- Matúš, P., Littera, P., Farkas, B., & Urík, M. (2023). Review on Performance of *Aspergillus* and *Penicillium* Species in Biodegradation of Organochlorine and Organophosphorus Pesticides. *Microorganisms*, 11(6). <https://doi.org/10.3390/microorganisms11061485>
- McNeill, I. C. (1997). Thermal degradation of polystyrene in different environments. *Die Angewandte Makromolekulare Chemie*, 247(1), 179-195. <https://doi.org/10.1002/apmc.1997.052470112>
- Meixner, K., Kubiczek, M., & Fritz, I. (2020). Microplastic in soil—current status in Europe with special focus on method tests with Austrian samples. *AIMS Environmental Science*, 7(2).
- Meizoso-Regueira, T., Fuentes, J., Cusworth, S. J., & Rillig, M. C. (2024). Prediction of future microplastic accumulation in agricultural soils. *Environmental Pollution*, 359, 124587. <https://doi.org/10.1016/j.envpol.2024.124587>
- Mendes, R., Garbeva, P., & Raaijmakers, J. M. (2013). The rhizosphere microbiome: significance of plant beneficial, plant pathogenic, and human pathogenic microorganisms. *FEMS Microbiology Reviews*, 37(5), 634-663. <https://doi.org/10.1111/1574-6976.12028>
- Meng, J., Xu, B., Liu, F., Li, W., Sy, N., Zhou, X., & Yan, B. (2021). Effects of chemical and natural ageing on the release of potentially toxic metal additives in commercial PVC microplastics. *Chemosphere*, 283, 131274. <https://doi.org/10.1016/j.chemosphere.2021.131274>
- Merino, C., Kuzyakov, Y., Godoy, K., Cornejo, P., & Matus, F. (2020). Synergy effect of peroxidase enzymes and Fenton reactions greatly increase the anaerobic oxidation of soil organic matter. *Sci Rep*, 10(1), 11289. <https://doi.org/10.1038/s41598-020-67953-z>
- Merino, C., Matus, F., Kuzyakov, Y., Dyckmans, J., Stock, S., & Dippold, M. A. (2021). Contribution of the Fenton reaction and ligninolytic enzymes to soil organic matter mineralisation under anoxic conditions. *Sci Total Environ*, 760, 143397. <https://doi.org/10.1016/j.scitotenv.2020.143397>
- Mészáros, E., Bodor, A., Kovács, E., Papp, S., Kovács, K., Perei, K., & Feigl, G. (2023). Impacts of Plastics on Plant Development: Recent Advances and Future Research Directions. *Plants (Basel)*, 12(18). <https://doi.org/10.3390/plants12183282>
- Meyerstein, D. (2022). What Are the Oxidizing Intermediates in the Fenton and Fenton-like Reactions? A Perspective. *Antioxidants*, 11(7).
- Miao, J., Chen, Y., Zhang, E., Yang, Y., Sun, K., & Gao, B. (2023). Effects of microplastics and biochar on soil cadmium availability and wheat plant performance. *GCB Bioenergy*, 15(8), 1046-1057. <https://doi.org/10.1111/gcbb.13083>
- Miao, L., Wang, P., Hou, J., Yao, Y., Liu, Z., Liu, S., & Li, T. (2019). Distinct community structure and microbial functions of biofilms colonizing microplastics. *Science of the Total Environment*, 650, 2395-2402. <https://doi.org/10.1016/j.scitotenv.2018.09.378>
- Mincheva, R., & Raquez, J.-M. (2019). The Surface of Polymers. In *Surface Modification of Polymers* (pp. 1-30). <https://doi.org/10.1002/9783527819249.ch1>
- Mintenig, S. M., Int-Veen, I., Löder, M. G. J., Primpke, S., & Gerdt, G. (2017). Identification of microplastic in effluents of waste water treatment plants using focal plane array-based micro-Fourier-transform infrared imaging. *Water Research*, 108, 365-372. <https://doi.org/10.1016/j.watres.2016.11.015>

- Mishra, S., Swain, S., Sahoo, M., Mishra, S., & Das, A. P. (2022). Microbial Colonization and Degradation of Microplastics in Aquatic Ecosystem: A Review. *Geomicrobiology Journal*, 39(3-5), 259-269. <https://doi.org/10.1080/01490451.2021.1983670>
- Mohan, A. J., Sekhar, V. C., Bhaskar, T., & Nampoothiri, K. M. (2016). Microbial assisted High Impact Polystyrene (HIPS) degradation. *Bioresource Technology*, 213, 204-207. <https://doi.org/10.1016/j.biortech.2016.03.021>
- Molazadeh, M., Liu, F., Simon-Sánchez, L., & Vollersten, J. (2023). Buoyant microplastics in freshwater sediments – How do they get there? *Science of The Total Environment*, 860, 160489. <https://doi.org/10.1016/j.scitotenv.2022.160489>
- Morrison, O. (2024). Microplastics in soils...how big is the problem? https://www.agtechnavigator.com/Article/2024/01/25/Microplastics-in-soils-how-big-is-the-problem?utm_source=copyright&utm_medium=OnSite&utm_campaign=copyright
- Naderi Beni, N., Karimifard, S., Gilley, J., Messer, T., Schmidt, A., & Bartelt-Hunt, S. (2023). Higher concentrations of microplastics in runoff from biosolid-amended croplands than manure-amended croplands. *Communications Earth & Environment*, 4(1), 42. <https://doi.org/10.1038/s43247-023-00691-y>
- Nakachew, K., Yigermal, H., Assefa, F., Gelaye, Y., & Ali, S. (2024). Review on enhancing the efficiency of fertilizer utilization: Strategies for optimal nutrient management. 9(1). <https://doi.org/10.1515/opag-2022-0356>
- Narula, N., Kothe, E., & Behl, R. (2009). Role of root exudates in plant-microbe interactions. *Journal of Applied Botany and Food Quality*, 82, 122-130.
- Nasrabadi, A. E., Ramavandi, B., & Bonyadi, Z. (2023). Recent progress in biodegradation of microplastics by *Aspergillus* sp. in aquatic environments. *Colloid and Interface Science Communications*, 57, 100754. <https://doi.org/10.1016/j.colcom.2023.100754>
- NatureCommunications. (2018). The future of plastic (2041-1723). (Nature Communications, Issue. <https://doi.org/10.1038/s41467-018-04565-2>
- Neubert, K. J., & Brüggemann, N. (2025). Soil texture modifies the impact of microplastics on winter wheat growth. *Journal of Soils and Sediments*, 25(4), 1340-1357. <https://doi.org/10.1007/s11368-025-04016-8>
- Neubert, K. J., Weihermüller, L., Vereecken, H., & Brüggemann, N. (2025). Soil texture governs the influence of different microplastics on soil hydraulic properties. *Vadose Zone Journal*, 24(5), e70044. <https://doi.org/10.1002/vzj2.70044>
- Nimmo, J. R. (1997). Modeling Structural Influences on Soil Water Retention. *Soil Science Society of America Journal*, 61(3), 712-719. <https://doi.org/10.2136/sssaj1997.03615995006100030002x>
- Nowak, B. D., Pająk, J., & Karcz, J. (2012). Biodegradation of Pre-Aged Modified Polyethylene Films. In Kazmiruk, V. (Ed.), *Scanning Electron Microscopy*. IntechOpen. <https://doi.org/10.5772/35128>
- Op De Beeck, M., Troein, C., Peterson, C., Persson, P., & Tunlid, A. (2018). Fenton reaction facilitates organic nitrogen acquisition by an ectomycorrhizal fungus. *New Phytol*, 218(1), 335-343. <https://doi.org/10.1111/nph.14971>
- Ortiz, D., Munoz, M., Nieto-Sandoval, J., Romera-Castillo, C., de Pedro, Z. M., & Casas, J. A. (2022). Insights into the degradation of microplastics by Fenton oxidation: From surface modification to mineralization. *Chemosphere*, 309, 136809. <https://doi.org/10.1016/j.chemosphere.2022.136809>
- Pais, I. P., Moreira, R., Semedo, J. N., Ramalho, J. C., Lidon, F. C., Coutinho, J., Maças, B., & Scotti-Campos, P. (2023). Wheat Crop under Waterlogging: Potential Soil and Plant Effects. *Plants*, 12(1), 149. <https://doi.org/10.3390/plants12010149>

- Pan, Y., Kang, P., Tan, M., Hu, J., Zhang, Y., Zhang, J., Song, N., & Li, X. (2022). Root exudates and rhizosphere soil bacterial relationships of *Nitraria tangutorum* are linked to k-strategists bacterial community under salt stress [Original Research]. *Frontiers in Plant Science*, Volume 13 - 2022. <https://doi.org/10.3389/fpls.2022.997292>
- Peters, B., Blume-Werry, G., Gillert, A., Schwieger, S., von Lukas, U. F., & Kreyling, J. (2023). As good as human experts in detecting plant roots in minirhizotron images but efficient and reproducible: the convolutional neural network "RootDetector". *Scientific Reports*, 13(1), 1399. <https://doi.org/10.1038/s41598-023-28400-x>
- Pfaendner, R. (2006). How will additives shape the future of plastics. *Polymer Degradation and Stability*, 91, 2249-2256. <https://doi.org/10.1016/J.POLYMDEGRADSTAB.2005.10.017>
- Pflugmacher, S., Sulek, A., Mader, H., Heo, J., Noh, J. H., Penttinen, O.-P., Kim, Y., Kim, S., & Esterhuizen, M. (2020). The influence of new and artificial aged microplastic and leachates on the germination of *Lepidium sativum* L. *Plants*, 9(3), 339. <https://doi.org/10.3390/plants9030339>
- Piccardi, P., Ulrich, E., Garcia-Garcerà, M., Martino, R. D., Testa, S. E. A., & Mitri, S. (2024). The evolution of reduced facilitation in a four-species bacterial community. *Evolution Letters*, 8(6), 828-840. <https://doi.org/10.1093/evlett/qrae036>
- Piehl, S., Leibner, A., Löder, M. G. J., Dris, R., Bogner, C., & Laforsch, C. (2018). Identification and quantification of macro- and microplastics on an agricultural farmland. *Scientific Reports*, 8(1), 17950. <https://doi.org/10.1038/s41598-018-36172-y>
- Pignatello, J. J., Oliveros, E., & MacKay, A. (2006). Advanced oxidation processes for organic contaminant destruction based on the Fenton reaction and related chemistry. *Critical reviews in environmental science and technology*, 36(1), 1-84. <https://doi.org/10.1080/10643380500326564>
- Pii, Y., Mimmo, T., Tomasi, N., Terzano, R., Cesco, S., & Crecchio, C. (2015). Microbial interactions in the rhizosphere: beneficial influences of plant growth-promoting rhizobacteria on nutrient acquisition process. A review. *Biology and fertility of soils*, 51, 403-415. <https://doi.org/10.1007/s00374-015-0996-1>
- Piñon-Colin, T. d. J., Rodriguez-Jimenez, R., Rogel-Hernandez, E., Alvarez-Andrade, A., & Wakida, F. T. (2020). Microplastics in stormwater runoff in a semiarid region, Tijuana, Mexico. *Science of The Total Environment*, 704, 135411. <https://doi.org/10.1016/j.scitotenv.2019.135411>
- Ponti, M. G., Allen, D., White, C. J., Bertram, D., & Switzer, C. (2022). A framework to assess the impact of flooding on the release of microplastics from waste management facilities. *Journal of Hazardous Materials Advances*, 7, 100105. <https://doi.org/10.1016/j.hazadv.2022.100105>
- Prajapati, A., Narayan Vaidya, A., & Kumar, A. R. (2022). Microplastic properties and their interaction with hydrophobic organic contaminants: a review. *Environmental Science and Pollution Research*, 29(33), 49490-49512. <https://doi.org/10.1007/s11356-022-20723-y>
- Pratiwi, O. A., Achmadi, U. F., & Kurniawan, R. (2024). Microplastic pollution in landfill soil: Emerging threats the environmental and public health. *Environ Anal Health Toxicol*, 39(1), e2024009-2024000. <https://doi.org/10.5620/eaht.2024009>
- Priya, K. L., Renjith, K. R., Joseph, C. J., Indu, M. S., Srinivas, R., & Haddout, S. (2022). Fate, transport and degradation pathway of microplastics in aquatic environment — A critical review. *Regional Studies in Marine Science*, 56, 102647. <https://doi.org/10.1016/j.rsma.2022.102647>
- Qi, R., Jones, D. L., Li, Z., Liu, Q., & Yan, C. (2020a). Behavior of microplastics and plastic film residues in the soil environment: A critical review. *Science of the Total Environment*, 703, 134722.

- Qi, X., Yan, W., Cao, Z., Ding, M., & Yuan, Y. (2021). Current Advances in the Biodegradation and Bioconversion of Polyethylene Terephthalate. *Microorganisms*, 10(1). <https://doi.org/10.3390/microorganisms10010039>
- Qi, Y., Yang, X., Pelaez, A. M., Lwanga, E. H., Beriot, N., Gertsen, H., Garbeva, P., & Geissen, V. (2018). Macro-and micro-plastics in soil-plant system: effects of plastic mulch film residues on wheat (*Triticum aestivum*) growth. *Science of the Total Environment*, 645, 1048-1056. <https://doi.org/10.1016/j.scitotenv.2018.07.229>
- Qi, Y., Beriot, N., Gort, G., Huerta Lwanga, E., Gooren, H., Yang, X., & Geissen, V. (2020b). Impact of plastic mulch film debris on soil physicochemical and hydrological properties. *Environmental Pollution*, 266, 115097. <https://doi.org/10.1016/j.envpol.2020.115097>
- Qi, Y., Ossowicki, A., Yang, X., Huerta Lwanga, E., Dini-Andreote, F., Geissen, V., & Garbeva, P. (2020c). Effects of plastic mulch film residues on wheat rhizosphere and soil properties. *Journal of Hazardous Materials*, 387, 121711. <https://doi.org/10.1016/j.jhazmat.2019.121711>
- Qiu, G., Wang, Q., Wang, T., Zhang, S., Song, N., Yang, X., Zeng, Y., Sun, Z., Wu, G., & Yu, H. (2024). Microplastic risk assessment and toxicity in plants: a review. *Environmental Chemistry Letters*, 22(1), 209-226. <https://doi.org/10.1007/s10311-023-01665-4>
- Qiu, R., Song, Y., Zhang, X., Xie, B., & He, D. (2020). Microplastics in Urban Environments: Sources, Pathways, and Distribution. In He, D. & Luo, Y. (Eds.), *Microplastics in Terrestrial Environments: Emerging Contaminants and Major Challenges* (pp. 41-61). Springer International Publishing. <https://doi.org/10.1007/978-2020-447>
- R Core Team (2023). R: A language and environment for statistical computing. R Foundation for Statistical Computing. Accessed 26.02.2026 from <https://www.R-project.org/>
- Rajandas, H., Parimannan, S., Sathasivam, K., Ravichandran, M., & Yin, L. S. (2012). A novel FTIR-ATR spectroscopy based technique for the estimation of low-density polyethylene biodegradation. *Polymer Testing*, 31(8), 1094-1099.
- Rana, M. M., Haque, M. R., Tasnim, S. S., & Rahman, M. M. (2023). The potential contribution of microplastic pollution by organic fertilizers in agricultural soils of Bangladesh: quantification, characterization, and risk appraisals [Original Research]. *Frontiers in Environmental Science*, 11. <https://doi.org/10.3389/fenvs.2023.1205603>
- Rebollar, E., Pérez, S., Hernandez, M., Domingo, C., Martín, M., Ezquerro, T., García-Ruiz, J., & Castillejo, M. (2014). Physicochemical modifications accompanying UV laser induced surface structures on poly(ethylene terephthalate) and their effect on adhesion of mesenchymal cells. *Physical chemistry chemical physics : PCCP*, 16. <https://doi.org/10.1039/c4cp02434f>
- Reynolds, W. D., Drury, C. F., Tan, C. S., Fox, C. A., & Yang, X. M. (2009). Use of indicators and pore volume-function characteristics to quantify soil physical quality. *Geoderma*, 152(3), 252-263. <https://doi.org/10.1016/j.geoderma.2009.06.009>
- Richardson, A. E., Barea, J.-M., McNeill, A. M., & Prigent-Combaret, C. (2009). Acquisition of phosphorus and nitrogen in the rhizosphere and plant growth promotion by microorganisms. *Plant and Soil*, 321, 305-339. <https://doi.org/10.1007/s11104-009-9895-2>
- Richardson, J. T. E. (2011). Eta squared and partial eta squared as measures of effect size in educational research. *Educational Research Review*, 6(2), 135-147. <https://doi.org/10.1016/j.edurev.2010.12.001>
- Rillig, M. C. (2012). Microplastic in Terrestrial Ecosystems and the Soil? *Environmental Science & Technology*, 46(12), 6453-6454. <https://doi.org/10.1021/es302011r>
- Rillig, M. C., Ingraffia, R., & de Souza Machado, A. A. (2017a). Microplastic Incorporation into Soil in Agroecosystems [Opinion]. *Frontiers in Plant Science*, 8. <https://doi.org/10.3389/fpls.2017.01805>

- Rillig, M. C., Ziersch, L., & Hempel, S. (2017b). Microplastic transport in soil by earthworms. *Scientific Reports*, 7(1), 1362. <https://doi.org/10.1038/s41598-017-01594-7>
- Rillig, M. C. (2018). Microplastic Disguising As Soil Carbon Storage. *Environmental Science & Technology*, 52(11), 6079-6080. <https://doi.org/10.1021/acs.est.8b02338>
- Rillig, M. C., Lehmann, A., de Souza Machado, A. A., & Yang, G. (2019). Microplastic effects on plants. *New Phytologist*, 223(3), 1066-1070. <https://doi.org/10.1111/nph.15794>
- Rillig, M. C., & Lehmann, A. (2020). Microplastic in terrestrial ecosystems. *Ecology*, 368(6498), 2. <https://doi.org/10.1126/science.abb5979>
- Rillig, M. C., Kim, S. W., & Zhu, Y.-G. (2024). The soil plastisphere. *Nature Reviews Microbiology*, 22(2), 64-74. <https://doi.org/10.1038/s41579-023-00967-2>
- Rimmer, D. L. (2006). Free radicals, antioxidants, and soil organic matter recalcitrance. *European Journal of Soil Science*, 57(2), 91-94. <https://doi.org/10.1111/j.1365-2389.2005.00735.x>
- Rocha-Santos, T., & Duarte, A. C. (2015). A critical overview of the analytical approaches to the occurrence, the fate and the behavior of microplastics in the environment. *TrAC Trends in Analytical Chemistry*, 65, 47-53. <https://doi.org/10.1016/j.trac.2014.10.011>
- Rochman, C. M., Kross, S. M., Armstrong, J. B., Bogan, M. T., Darling, E. S., Green, S. J., Smyth, A. R., & Veríssimo, D. (2015). Scientific Evidence Supports a Ban on Microbeads. *Environ Sci Technol*, 49(18), 10759-10761. <https://doi.org/10.1021/acs.est.5b03909>
- Rødland, E. S., Lind, O. C., Reid, M., Heier, L. S., Skogsberg, E., Snilsberg, B., Gryteselv, D., & Meland, S. (2022). Characterization of tire and road wear microplastic particle contamination in a road tunnel: From surface to release. *Journal of Hazardous Materials*, 435, 129032. <https://doi.org/10.1016/j.jhazmat.2022.129032>
- Rojas, J. P., Ruge, J. C., & Carrillo, G. A. (2022). Unsaturated Hydraulic Conductivity in Composite Porous Media. *Applied Sciences*, 12(18).
- Römer, W., Augustin, J., & Schilling, G. (1988). The relationship between phosphate absorption and root length in nine wheat cultivars. *Plant and Soil*, 111, 199-201. <https://doi.org/10.1007/BF02139938>
- Rong, L., Zhao, L., Zhao, L., Cheng, Z., Yao, Y., Yuan, C., Wang, L., & Sun, H. (2021). LDPE microplastics affect soil microbial communities and nitrogen cycling. *Science of The Total Environment*, 773, 145640. <https://doi.org/10.1016/j.scitotenv.2021.145640>
- Roy, T., Dey, T. K., & Jamal, M. (2022). Microplastic/nanoplastic toxicity in plants: an imminent concern. *Environmental Monitoring and Assessment*, 195(1), 27. <https://doi.org/10.1007/s10661-022-10654-z>
- Roychand, R., Patel, S., Halder, P., Kundu, S., Hampton, J., Bergmann, D., Surapaneni, A., Shah, K., & Pramanik, B. K. (2021). Recycling biosolids as cement composites in raw, pyrolyzed and ashed forms: A waste utilisation approach to support circular economy. *Journal of Building Engineering*, 38, 102199. <https://doi.org/10.1016/j.jobe.2021.102199>
- Rozman, U., Turk, T., Skalar, T., Zupančič, M., Čelan Korošin, N., Marinšek, M., Olivero-Verbel, J., & Kalčíková, G. (2021). An extensive characterization of various environmentally relevant microplastics – Material properties, leaching and ecotoxicity testing. *Science of The Total Environment*, 773, 145576. <https://doi.org/10.1016/j.scitotenv.2021.145576>
- Russell Jonathan, R., Huang, J., Anand, P., Kucera, K., Sandoval Amanda, G., Dantzer Kathleen, W., Hickman, D., Jee, J., Kimovec Farrah, M., Koppstein, D., Marks Daniel, H., Mittermiller Paul, A., Núñez Salvador, J., Santiago, M., Townes Maria, A., Vishnevetsky, M., Williams Neely, E., Vargas Mario Percy, N., Boulanger, L.-A., Bascom-Slack, C., & Strobel Scott, A. (2011). Biodegradation of

- Polyester Polyurethane by Endophytic Fungi. *Appl Environ Microbiol*, 77(17), 6076-6084. <https://doi.org/10.1128/AEM.00521-11>
- Saljnikov, E., Grujić, T., Jovković, M., Perović, V., Čakmak, D., Zhapparova, A., Radović, V., Stefanović, S., Miladinović, V., Stanković, S., Marjanović, Ž., Kenzhegulova, S., Tleppayeva, A., Kunypiyaeva, G., & Krnjajić, S. (2025). Changes in Soil Properties Under the Influence of Microplastics in Plastic and Open Field Production in Three Serbian Valleys. *Horticulturae*, 11(3).
- Sánchez, C. (2020). Fungal potential for the degradation of petroleum-based polymers: An overview of macro- and microplastics biodegradation. *Biotechnology Advances*, 40, 107501. <https://doi.org/10.1016/j.biotechadv.2019.107501>
- Sangale, M. K., Shah Nawaz, M., & Ade, A. B. (2019). Potential of fungi isolated from the dumping sites mangrove rhizosphere soil to degrade polythene. *Scientific Reports*, 9(1), 5390. <https://doi.org/10.1038/s41598-019-41448-y>
- Schindler, U., Durner, W., von Unold, G., & Müller, L. (2010). Evaporation Method for Measuring Unsaturated Hydraulic Properties of Soils: Extending the Measurement Range. *Soil Science Society of America Journal*, 74(4), 1071-1083. <https://doi.org/10.2136/sssaj2008.0358>
- Schlosser, D. (2021). Chapter Four - Cultivation of filamentous fungi for attack on synthetic polymers via biological Fenton chemistry. In Weber, G., Bornscheuer, U. T., & Wei, R. (Eds.), *Methods in Enzymology* (Vol. 648, pp. 71-94). Academic Press. <https://doi.org/10.1016/bs.mie.2020.12.006>
- Sforzini, S., Oliveri, L., Chinaglia, S., & Viarengo, A. (2016). Application of Biotests for the Determination of Soil Ecotoxicity after Exposure to Biodegradable Plastics [Technology Report]. *Frontiers in Environmental Science*, 4. <https://doi.org/10.3389/fenvs.2016.00068>
- Shafea, L., Felde, V. J. M. N. L., Woche, S. K., Bachmann, J., & Peth, S. (2023). Microplastics effects on wettability, pore sizes and saturated hydraulic conductivity of a loess topsoil. *Geoderma*, 437, 116566. <https://doi.org/10.1016/j.geoderma.2023.116566>
- Shah, F., Nicolás, C., Bentzer, J., Ellström, M., Smits, M., Rineau, F., Canbäck, B., Floudas, D., Carleer, R., Lackner, G., Braesel, J., Hoffmeister, D., Henrissat, B., Ahrén, D., Johansson, T., Hibbett, D. S., Martin, F., Persson, P., & Tunlid, A. (2016). Ectomycorrhizal fungi decompose soil organic matter using oxidative mechanisms adapted from saprotrophic ancestors. *New Phytologist*, 209(4), 1705-1719. <https://doi.org/10.1111/nph.13722>
- Sharma, H. B., Vanapalli, K. R., Cheela, V. S., Ranjan, V. P., Jaglan, A. K., Dubey, B., Goel, S., & Bhattacharya, J. (2020). Challenges, opportunities, and innovations for effective solid waste management during and post COVID-19 pandemic. *Resour Conserv Recycl*, 162, 105052. <https://doi.org/10.1016/j.resconrec.2020.105052>
- Shi, W., Wu, N., Zhang, Z., Liu, Y., Chen, J., & Li, J. (2024). A global review on the abundance and threats of microplastics in soils to terrestrial ecosystem and human health. *Science of The Total Environment*, 912, 169469. <https://doi.org/10.1016/j.scitotenv.2023.169469>
- Shirkhorshidi, B., Ghanatghehstani, M. D., Moeinpour, F., & Parvaresh, H. (2023). Exploring the interaction between microplastics and heavy metals: unveiling the impact of microplastics on lead sorption and desorption in soil. *Environmental Monitoring and Assessment*, 195(9), 1017. <https://doi.org/10.1007/s10661-023-11640-9>
- Sirtori, C., Agüera, A., Carra, I., & Sánchez Pérez, J. A. (2014). Identification and monitoring of thiabendazole transformation products in water during Fenton degradation by LC-QTOF-MS. *Analytical and Bioanalytical Chemistry*, 406(22), 5323-5337. <https://doi.org/10.1007/s00216-014-7942-1>
- Skariyachan, S., Patil, A. A., Shankar, A., Manjunath, M., Bachappanavar, N., & Kiran, S. (2018). Enhanced polymer degradation of polyethylene and polypropylene by novel thermophilic

- consortia of *Brevibacillus* sps. and *Aneurinibacillus* sp. screened from waste management landfills and sewage treatment plants. *Polymer Degradation and Stability*, 149, 52-68. <https://doi.org/10.1016/j.polyimdegradstab.2018.01.018>
- Sørensen, L., Groven, A. S., Hovsbakken, I. A., Del Puerto, O., Krause, D. F., Sarno, A., & Booth, A. M. (2021). UV degradation of natural and synthetic microfibers causes fragmentation and release of polymer degradation products and chemical additives. *Science of The Total Environment*, 755, 143170. <https://doi.org/10.1016/j.scitotenv.2020.143170>
- Speight, J. G. (2020). Chapter 14 - Monomers, polymers, and plastics. In Speight, J. G. (Ed.), *Handbook of Industrial Hydrocarbon Processes* (Second Edition) (pp. 597-649). Gulf Professional Publishing. <https://doi.org/10.1016/B978-0-12-809923-0.00014-X>
- Sun, H., Shi, Y., Zhao, P., Long, G., Li, C., Wang, J., Qiu, D., Lu, C., Ding, Y., & Liu, L. (2023). Effects of polyethylene and biodegradable microplastics on photosynthesis, antioxidant defense systems, and arsenic accumulation in maize (*Zea mays* L.) seedlings grown in arsenic-contaminated soils. *Science of The Total Environment*, 868, 161557. <https://doi.org/10.1016/j.scitotenv.2023.161557>
- Sustr, M., Soukup, A., & Tylova, E. (2019). Potassium in Root Growth and Development. *Plants* (Basel), 8(10). <https://doi.org/10.3390/plants8100435>
- Tadsuwan, K., & Babel, S. (2021). Microplastic contamination in a conventional wastewater treatment plant in Thailand. *Waste Management & Research*, 39(5), 754-761. <https://doi.org/10.1177/0734242X20982055>
- Tagg, A., Harrison, J., Ju-Nam, Y., Sapp, M., Bradley, E., Sinclair, C., & Ojeda, J. (2017). Fenton's reagent for the rapid and efficient isolation of microplastics from wastewater. *Chemical Communications*, 53, 372-375. <https://doi.org/10.1039/C6CC08798A>
- Ter Halle, A., Ladirat, L., Gendre, X., Goudouneche, D., Pusineri, C., Routaboul, C., Tenailleau, C., Duployer, B., & Perez, E. (2016). Understanding the Fragmentation Pattern of Marine Plastic Debris. *Environ Sci Technol*, 50(11), 5668-5675. <https://doi.org/10.1021/acs.est.6b00594>
- Teuten, E. L., Saquing, J. M., Knappe, D. R. U., Barlaz, M. A., Jonsson, S., Björn, A., Rowland, S. J., Thompson, R. C., Galloway, T. S., Yamashita, R., Ochi, D., Watanuki, Y., Moore, C., Viet, P. H., Tana, T. S., Prudente, M., Boonyatumanond, R., Zakaria, M. P., Akkhavong, K., Ogata, Y., Hirai, H., Iwasa, S., Mizukawa, K., Hagino, Y., Imamura, A., Saha, M., & Takada, H. (2009). Transport and release of chemicals from plastics to the environment and to wildlife. *Philosophical Transactions of the Royal Society B: Biological Sciences*, 364(1526), 2027-2045. <https://doi.org/10.1098/rstb.2008.0284>
- TextileExchange. (2024). *Materials Market Report 2024*. Accessed 07.11.2024 from <https://textileexchange.org/knowledge-center/reports/materials-market-report-2024/>
- Tian, L., Jinjin, C., Ji, R., Ma, Y., & Yu, X. (2022). Microplastics in agricultural soils: sources, effects, and their fate. *Current Opinion in Environmental Science & Health*, 25, 100311. <https://doi.org/10.1016/j.coesh.2021.100311>
- Tiso, T., Narancic, T., Wei, R., Pollet, E., Beagan, N., Schröder, K., Honak, A., Jiang, M., Kenny, S. T., Wierckx, N., Perrin, R., Avérous, L., Zimmermann, W., O'Connor, K., & Blank, L. M. (2021). Towards bio-upcycling of polyethylene terephthalate. *Metabolic Engineering*, 66, 167-178. <https://doi.org/10.1016/j.ymben.2021.03.011>
- Topp, G. C., Reynolds, W. D., Cook, F. J., Kirby, J. M., & Carter, M. R. (1997). Chapter 2 Physical attributes of soil quality. In Gregorich, E. G. & Carter, M. R. (Eds.), *Developments in Soil Science* (Vol. 25, pp. 21-58). Elsevier. [https://doi.org/10.1016/S0166-2481\(97\)80029-3](https://doi.org/10.1016/S0166-2481(97)80029-3)
- Trainic, M., Flores, J. M., Pinkas, I., Pedrotti, M. L., Lombard, F., Bourdin, G., Gorsky, G., Boss, E., Rudich, Y., Vardi, A., & Koren, I. (2020). Airborne microplastic particles detected in the remote marine atmosphere. *Communications Earth & Environment*, 1(1), 64. <https://doi.org/10.1038/s43247-020-00061-y>

- Tribedi, P., & Sil, A. K. (2013). Low-density polyethylene degradation by *Pseudomonas* sp. AKS2 biofilm. *Environmental Science and Pollution Research*, 20(6), 4146-4153. <https://doi.org/10.1007/s11356-012-1378-y>
- Tumwet, F. C., Richter, A., Kleint, T., & Scheytt, T. (2024). Vertical movement of microplastics by roots of wheat plant (*Triticum aestivum*) and the plant response in sandy soil. *Microplastics and Nanoplastics*, 4(1), 15. <https://doi.org/10.1186/s43591-024-00092-8>
- Turner, B. L. (2021). Soil as an Archetype of Complexity: A Systems Approach to Improve Insights, Learning, and Management of Coupled Biogeochemical Processes and Environmental Externalities. *Soil Systems*, 5(3).
- UBA. (2022). Kompost und Klärschlamm. Accessed 16.10.2024 from <https://www.umweltbundesamt.de/themen/landwirtschaft/umweltbelastungen-der-landwirtschaft/kompost-klaerschlamm#einfuehrung>
- Ullah, R., Tsui, M. T.-K., Chen, H., Chow, A., Williams, C., & Ligaba-Osena, A. (2021). Microplastics interaction with terrestrial plants and their impacts on agriculture. *Journal of Environmental Quality*, 50(5), 1024-1041. <https://doi.org/10.1002/jeq2.20264>
- Union, P. O. o. t. E. (2020). Verwendung von Klärschlamm in der Landwirtschaft. Accessed 09.05.2025 from <https://eur-lex.europa.eu/legal-content/DE/ALL/?uri=legissum:l28088>
- Urbina, M. A., Correa, F., Aburto, F., & Ferrio, J. P. (2020). Adsorption of polyethylene microbeads and physiological effects on hydroponic maize. *Science of The Total Environment*, 741, 140216. <https://doi.org/10.1016/j.scitotenv.2020.140216>
- van den Berg, P., Huerta-Lwanga, E., Corradini, F., & Geissen, V. (2020). Sewage sludge application as a vehicle for microplastics in eastern Spanish agricultural soils. *Environmental Pollution*, 261, 114198. <https://doi.org/10.1016/j.envpol.2020.114198>
- van Genuchten, M. T. (1980). A Closed-form Equation for Predicting the Hydraulic Conductivity of Unsaturated Soils. *Soil Science Society of America Journal*, 44(5), 892-898. <https://doi.org/10.2136/sssaj1980.03615995004400050002x>
- van Wezel, A., Caris, I., & Kools, S. A. E. (2016). Release of primary microplastics from consumer products to wastewater in the Netherlands. *Environmental Toxicology and Chemistry*, 35(7), 1627-1631. <https://doi.org/10.1002/etc.3316>
- VDLUFA. (2016). Methodenbuch Band I, Die Untersuchung von Böden. VDLUFA-Verlag.
- Wan, J., & Wilson, J. L. (1994). Colloid transport in unsaturated porous media. *Water Resources Research*, 30(4), 857-864. <https://doi.org/10.1029/93WR03017>
- Wan, Y., Wu, C., Xue, Q., & Hui, X. (2019). Effects of plastic contamination on water evaporation and desiccation cracking in soil. *Science of The Total Environment*, 654, 576-582. <https://doi.org/10.1016/j.scitotenv.2018.11.123>
- Wang, C. (2018). Fractional kinetics of photocatalytic degradation. *Journal of Advanced Dielectrics*, 08. <https://doi.org/10.1142/S2010135X18500340>
- Wang, F., Zhang, X., Zhang, S., Zhang, S., Adams, C. A., & Sun, Y. (2020a). Effects of Co-Contamination of Microplastics and Cd on Plant Growth and Cd Accumulation. *Toxics*, 8(2), 36. <https://doi.org/10.3390/toxics8020036>
- Wang, F., Zhang, X., Zhang, S., Zhang, S., & Sun, Y. (2020b). Interactions of microplastics and cadmium on plant growth and arbuscular mycorrhizal fungal communities in an agricultural soil. *Chemosphere*, 254, 126791. <https://doi.org/10.1016/j.chemosphere.2020.126791>

- Wang, F., Wang, Q., Adams, C. A., Sun, Y., & Zhang, S. (2022a). Effects of microplastics on soil properties: Current knowledge and future perspectives. *Journal of Hazardous Materials*, 424, 127531. <https://doi.org/10.1016/j.jhazmat.2021.127531>
- Wang, J., Tan, Z., Peng, J., Qiu, Q., & Li, M. (2016). The behaviors of microplastics in the marine environment. *Mar Environ Res*, 113, 7-17. <https://doi.org/10.1016/j.marenvres.2015.10.014>
- Wang, J., Liu, X., Li, Y., Powell, T., Wang, X., Wang, G., & Zhang, P. (2019). Microplastics as contaminants in the soil environment: A mini-review. *Sci Total Environ*, 691, 848-857. <https://doi.org/10.1016/j.scitotenv.2019.07.209>
- Wang, J., Huang, M., Wang, Q., Sun, Y., Zhao, Y., & Huang, Y. (2020c). LDPE microplastics significantly alter the temporal turnover of soil microbial communities. *Science of The Total Environment*, 726, 138682. <https://doi.org/10.1016/j.scitotenv.2020.138682>
- Wang, J., Wang, C., Zhang, J., Wu, X., Hou, Y., Zhao, G., & Sun, H. (2022b). Decreased precipitation reduced the complexity and stability of bacterial co-occurrence patterns in a semiarid grassland. *Frontiers in Microbiology*, Volume 13 - 2022. <https://doi.org/10.3389/fmicb.2022.1031496>
- Wang, Q., Adams, C. A., Wang, F., Sun, Y., & Zhang, S. (2022c). Interactions between microplastics and soil fauna: a critical review. *Critical Reviews in Environmental Science and Technology*, 52(18), 3211-3243. <https://doi.org/10.1080/10643389.2021.1915035>
- Wang, S. (2008). A Comparative study of Fenton and Fenton-like reaction kinetics in decolourisation of wastewater. *Dyes and Pigments*, 76(3), 714-720. <https://doi.org/10.1016/j.dyepig.2007.01.012>
- Wang, Z., Li, W., Li, W., Yang, W., & Jing, S. (2023). Effects of microplastics on the water characteristic curve of soils with different textures. *Chemosphere*, 317, 137762. <https://doi.org/10.1016/j.chemosphere.2023.137762>
- Wang, Z., Li, J., Qu, Z., Ayurzana, B., Zhao, G., & Li, W. (2024). Effects of microplastics on the pore structure and connectivity with different soil textures: Based on CT scanning. *Environmental Technology & Innovation*, 36, 103791. <https://doi.org/10.1016/j.eti.2024.103791>
- Webb, H. K., Arnott, J., Crawford, R. J., & Ivanova, E. P. (2012). Plastic degradation and its environmental implications with special reference to poly (ethylene terephthalate). *Polymers*, 5(1), 1-18. <https://doi.org/10.3390/polym5010001>
- Weber, C. J., Santowski, A., & Chiffard, P. (2022). Investigating the dispersal of macro- and microplastics on agricultural fields 30 years after sewage sludge application. *Scientific Reports*, 12(1), 6401. <https://doi.org/10.1038/s41598-022-10294-w>
- Witt, T., Robinson, N., Palma, A. C., Cernusak, L. A., Pratt, S., Redding, M., Batstone, D. J., Schmidt, S., & Laycock, B. (2024). Evaluating novel biodegradable polymer matrix fertilizers for nitrogen-efficient agriculture. *Journal of Environmental Quality*, 53(3), 287-299. <https://doi.org/10.1002/jeq2.20552>
- Xiang, P., Zhang, T., Wu, Q., & Li, Q. (2023). Systematic Review of Degradation Processes for Microplastics: Progress and Prospects. *Sustainability*, 15(17).
- Xiang, Y., Rillig, M. C., Peñuelas, J., Sardans, J., Liu, Y., Yao, B., & Li, Y. (2024). Global Responses of Soil Carbon Dynamics to Microplastic Exposure: A Data Synthesis of Laboratory Studies. *Environ Sci Technol*, 58(13), 5821-5831. <https://doi.org/10.1021/acs.est.3c06177>
- Xiao, M., Shahbaz, M., Liang, Y., Yang, J., Wang, S., Chadwick, D. R., Jones, D., Chen, J., & Ge, T. (2021). Effect of microplastics on organic matter decomposition in paddy soil amended with crop residues and labile C: A three-source-partitioning study. *Journal of Hazardous Materials*, 416, 126221. <https://doi.org/10.1016/j.jhazmat.2021.126221>

- Xie, Y., Wang, H., Chen, Y., Guo, Y., Wang, C., Cui, H., & Xue, J. (2023). Water retention and hydraulic properties of a natural soil subjected to microplastic contaminations and leachate exposures. *Science of The Total Environment*, 901, 166502. <https://doi.org/10.1016/j.scitotenv.2023.166502>
- Xing, R., Sun, H., Du, X., Lin, H., Qin, S., Chen, Z., & Zhou, S. (2023). Enhanced degradation of microplastics during sludge composting via microbially-driven Fenton reaction. *Journal of Hazardous Materials*, 449, 131031. <https://doi.org/10.1016/j.jhazmat.2023.131031>
- Xu, B., Liu, F., Cryder, Z., Huang, D., Lu, Z., He, Y., Wang, H., Lu, Z., Brookes, P. C., Tang, C., Gan, J., & Xu, J. (2020). Microplastics in the soil environment: Occurrence, risks, interactions and fate – A review. *Critical Reviews in Environmental Science and Technology*, 50(21), 2175-2222. <https://doi.org/10.1080/10643389.2019.1694822>
- Xu, G., Liu, Y., & Yu, Y. (2021a). Effects of polystyrene microplastics on uptake and toxicity of phenanthrene in soybean. *Science of the Total Environment*, 783, 147016. <https://doi.org/10.1016/j.scitotenv.2021.147016>
- Xu, Q., Fu, H., Zhu, B., Hussain, H. A., Zhang, K., Tian, X., Duan, M., Xie, X., & Wang, L. (2021b). Potassium Improves Drought Stress Tolerance in Plants by Affecting Root Morphology, Root Exudates, and Microbial Diversity. *Metabolites*, 11(3).
- Xu, Y., Ou, Q., van der Hoek, J. P., Liu, G., & Lompe, K. M. (2024). Photo-oxidation of Micro- and Nanoplastics: Physical, Chemical, and Biological Effects in Environments. *Environ Sci Technol*, 58(2), 991-1009. <https://doi.org/10.1021/acs.est.3c07035>
- Xu, Z., Zhang, J., Qi, R., Liu, Q., Cao, H., Wen, F., Liao, Y., Shih, K., & Tang, Y. (2025). Complex release dynamics of microplastic additives: An interplay of additive degradation and microplastic aging. *Journal of Hazardous Materials*, 490, 137711. <https://doi.org/10.1016/j.jhazmat.2025.137711>
- Ya, H., Jiang, B., Xing, Y., Zhang, T., Lv, M., & Wang, X. (2021). Recent advances on ecological effects of microplastics on soil environment. *Science of The Total Environment*, 798, 149338. <https://doi.org/10.1016/j.scitotenv.2021.149338>
- Ya, H., Xing, Y., Zhang, T., Lv, M., & Jiang, B. (2022). LDPE microplastics affect soil microbial community and form a unique plastisphere on microplastics. *Applied Soil Ecology*, 180, 104623. <https://doi.org/10.1016/j.apsoil.2022.104623>
- Yamada-Onodera, K., Mukumoto, H., Katsuyaya, Y., Saiganji, A., & Tani, Y. (2001). Degradation of polyethylene by a fungus, *Penicillium simplicissimum* YK. *Polymer Degradation and Stability*, 72(2), 323-327. [https://doi.org/10.1016/S0141-3910\(01\)00027-1](https://doi.org/10.1016/S0141-3910(01)00027-1)
- Yan, S., Zhang, S., Xu, B., Yan, P., Wang, J., Wang, H., & Aurangzeib, M. (2023). Microplastics change the leaching of nitrogen and potassium in Mollisols. *Science of The Total Environment*, 878, 163121. <https://doi.org/10.1016/j.scitotenv.2023.163121>
- Yang, Y., Chen, J., Chen, Z., Yu, Z., Xue, J., Luan, T., Chen, S., & Zhou, S. (2022). Mechanisms of polystyrene microplastic degradation by the microbially driven Fenton reaction. *Water Res*, 223, 118979. <https://doi.org/10.1016/j.watres.2022.118979>
- Yousif, E., & Haddad, R. (2013). Photodegradation and photostabilization of polymers, especially polystyrene: review. *SpringerPlus*, 2(1), 398. <https://doi.org/10.1186/2193-1801-2-398>
- Yu, G.-H., & Kuzyakov, Y. (2021). Fenton chemistry and reactive oxygen species in soil: Abiotic mechanisms of biotic processes, controls and consequences for carbon and nutrient cycling. *Earth-Science Reviews*, 214, 103525. <https://doi.org/10.1016/j.earscirev.2021.103525>
- Yu, J., Adingo, S., Liu, X., Li, X., Sun, J., & Zhang, X. (2022). Micro plastics in soil ecosystem - A review of sources, fate, and ecological impact. *Plant, Soil and Environment*, 68(1), 1-17. <https://doi.org/10.17221/242/2021-PSE>

- Yu, L., Zhang, J., Liu, Y., Chen, L., Tao, S., & Liu, W. (2021). Distribution characteristics of microplastics in agricultural soils from the largest vegetable production base in China. *Science of The Total Environment*, 756, 143860. <https://doi.org/10.1016/j.scitotenv.2020.143860>
- Yu, M., Ploeg, M., Huerta, E., Yang, X., Zhang, S., Ma, X., Ritsema, C., & Geissen, V. (2019). Leaching of microplastics by preferential flow in earthworm (*Lumbricus terrestris*) burrows. *Environmental Chemistry*, 16. <https://doi.org/10.1071/EN18161>
- Yu, Y., & Flury, M. (2022, May 01, 2022). Effects of Microplastics on Soil Hydraulic Properties EGU General Assembly Conference Abstracts, Vienna, Austria. <https://ui.adsabs.harvard.edu/abs/2022EGUGA..24..819Y>
- Zahra, S., Abbas, S. S., Mahsa, M.-T., & Mohsen, N. (2010). Biodegradation of low-density polyethylene (LDPE) by isolated fungi in solid waste medium. *Waste Management*, 30(3), 396-401. <https://doi.org/10.1016/j.wasman.2009.09.027>
- Zandieh, M., Griffiths, E., Waldie, A., Li, S., Honek, J., Rezanezhad, F., Van Cappellen, P., & Liu, J. (2024). Catalytic and biocatalytic degradation of microplastics. *Exploration*, 4(3), 20230018. <https://doi.org/10.1002/EXP.20230018>
- Zang, H., Zhou, J., Marshall, M. R., Chadwick, D. R., Wen, Y., & Jones, D. L. (2020). Microplastics in the agroecosystem: Are they an emerging threat to the plant-soil system? *Soil Biology and Biochemistry*, 148, 107926. <https://doi.org/10.1016/j.soilbio.2020.107926>
- Zeb, A., Liu, W., Meng, L., Lian, J., Wang, Q., Lian, Y., Chen, C., & Wu, J. (2022). Effects of polyester microfibers (PMFs) and cadmium on lettuce (*Lactuca sativa*) and the rhizospheric microbial communities: a study involving physio-biochemical properties and metabolomic profiles. *Journal of Hazardous Materials*, 424, 127405. <https://doi.org/10.1016/j.jhazmat.2021.127405>
- Zettler, E. R., Mincer, T. J., & Amaral-Zettler, L. A. (2013). Life in the "plastisphere": microbial communities on plastic marine debris. *Environmental science & technology*, 47(13), 7137-7146. <https://doi.org/10.1021/es401288x>
- Zhang, G. S., & Liu, Y. F. (2018). The distribution of microplastics in soil aggregate fractions in southwestern China. *Sci Total Environ*, 642, 12-20. <https://doi.org/10.1016/j.scitotenv.2018.06.004>
- Zhang, G. S., Zhang, F. X., & Li, X. T. (2019). Effects of polyester microfibers on soil physical properties: Perception from a field and a pot experiment. *Science of The Total Environment*, 670, 1-7. <https://doi.org/10.1016/j.scitotenv.2019.03.149>
- Zhang, K., Gao, N., Li, Y., Dou, S., Liu, Z., Chen, Y., Ma, C., & Zhang, H. (2023). Responses of maize (*Zea mays* L.) seedlings growth and physiological traits triggered by polyvinyl chloride microplastics is dominated by soil available nitrogen. *Ecotoxicology and Environmental Safety*, 252, 114618. <https://doi.org/10.1016/j.ecoenv.2023.114618>
- Zhang, K., Wang, M., Li, Y., Zhang, X., Xiao, K., Ma, C., Zhang, X., Zhang, H., & Chen, Y. (2024). Wheat (*Triticum aestivum* L.) seedlings performance mainly affected by soil nitrate nitrogen under the stress of polyvinyl chloride microplastics. *Scientific Reports*, 14(1), 4962. <https://doi.org/10.1038/s41598-024-54838-8>
- Zhang, Y., Cai, C., Gu, Y., Shi, Y., & Gao, X. (2022a). Microplastics in plant-soil ecosystems: A meta-analysis. *Environ Pollut*, 308, 119718. <https://doi.org/10.1016/j.envpol.2022.119718>
- Zhang, Y., Zhang, C., Jiang, M., & Zhou, G. (2022b). Bio-effects of bio-based and fossil-based microplastics: Case study with lettuce-soil system. *Environmental Pollution*, 306, 119395. <https://doi.org/10.1016/j.envpol.2022.119395>

- Zhao, T., Lozano, Y. M., & Rillig, M. C. (2021). Microplastics Increase Soil pH and Decrease Microbial Activities as a Function of Microplastic Shape, Polymer Type, and Exposure Time [Original Research]. *Frontiers in Environmental Science*, 9. <https://doi.org/10.3389/fenvs.2021.675803>
- Zheng, W., Morris, E. K., Lehmann, A., & Rillig, M. C. (2016). Interplay of soil water repellency, soil aggregation and organic carbon. A meta-analysis. *Geoderma*, 283, 39-47. <https://doi.org/10.1016/j.geoderma.2016.07.025>
- Zhou, H., Zhou, L., & Ma, K. (2020a). Microfiber from textile dyeing and printing wastewater of a typical industrial park in China: Occurrence, removal and release. *Science of The Total Environment*, 739, 140329. <https://doi.org/10.1016/j.scitotenv.2020.140329>
- Zhou, J., Gui, H., Banfield, C. C., Wen, Y., Zang, H., Dippold, M. A., Charlton, A., & Jones, D. L. (2021). The microplastisphere: Biodegradable microplastics addition alters soil microbial community structure and function. *Soil Biology and Biochemistry*, 156, 108211. <https://doi.org/10.1016/j.soilbio.2021.108211>
- Zhou, Y., Wang, J., Zou, M., Jia, Z., Zhou, S., & Li, Y. (2020b). Microplastics in soils: A review of methods, occurrence, fate, transport, ecological and environmental risks. *Sci Total Environ*, 748, 141368. <https://doi.org/10.1016/j.scitotenv.2020.141368>
- Zhou, Y., Yao, Q., & Zhu, H. (2020c). Soil Organic Carbon Attenuates the Influence of Plants on Root-Associated Bacterial Community. *Front Microbiol*, 11, 594890. <https://doi.org/10.3389/fmicb.2020.594890>
- Zhu, F., Zhu, C., Wang, C., & Gu, C. (2019a). Occurrence and Ecological Impacts of Microplastics in Soil Systems: A Review. *Bull Environ Contam Toxicol*, 102(6), 741-749. <https://doi.org/10.1007/s00128-019-02623-z>
- Zhu, F., Yan, Y., Doyle, E., Zhu, C., Jin, X., Chen, Z., Wang, C., He, H., Zhou, D., & Gu, C. (2022). Microplastics altered soil microbiome and nitrogen cycling: The role of phthalate plasticizer. *Journal of Hazardous Materials*, 427, 127944. <https://doi.org/10.1016/j.jhazmat.2021.127944>
- Zhu, Y. H., Weiner, J., Yu, M. X., & Li, F. M. (2019b). Evolutionary agroecology: Trends in root architecture during wheat breeding. *Evol Appl*, 12(4), 733-743. <https://doi.org/10.1111/eva.12749>
- Zubris, K. A. V., & Richards, B. K. (2005). Synthetic fibers as an indicator of land application of sludge. *Environmental Pollution*, 138(2), 201-211. <https://doi.org/10.1016/j.envpol.2005.04.013>

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