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Monitoring of microplastics and tyre wear particles in the Norwegian environment

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Abstract
The 2024 MIKRONOR campaign, coordinated by NIVA and NILU on behalf of the Norwegian Environment Agency, significantly expanded the national monitoring framework for microplastics (MPs) to encompass diverse environmental compartments, including surface waters (Oslofjord and Lake Mjøsa), urban runoff, marine sediments, atmospheric deposition, and coastal beach sediments. Urban stormwater runoff was identified as a predominant source of MPs, particularly tyre wear particles (TWP). Sediment samples from stormwater traps in Oslo exhibited high TWP concentrations up to 240 mg/g, constituting approximately 25% of the total sediment mass. Corresponding runoff water samples revealed MP concentrations as high as 733 ± 142 particles/L, indicating substantial episodic fluxes of MPs into receiving aquatic or marine systems. Inner Oslofjord sediments contained 0.6–3.5 % TWP by mass, confirming the high levels found in 2023. Microplastic concentrations in surface waters were generally low, ranging from 0 to 0.6 MP/m³. However, two hydrodynamic accumulation zones within the Oslofjord exhibited anomalously high concentrations, with levels approximately two orders of magnitude greater than outside the accumulation zones. One net tow recovered >7,000 fragments of expanded polystyrene, highlighting localized retention. Atmospheric deposition peaked in urban Sofienbergparken (1514 µg/m²/d; 68 % TWP) and showed a clear urban-to-remote gradient. Beach sediments at Akerøya remained low in MPs, with most samples below detection limits. The findings highlight urban runoff, especially TWP, as a dominant source to the Oslofjord, and reveal critical hotspots in both water and air pathways.

Keywords: Microplastic pollution, Environmental contaminants, monitoring, tyre wear particles
Emneord: Mikroplast, forurensning, overvåking, dekkpartikler

Table of contents

Preface

The Norwegian Institute for Water Research (NIVA), acting on behalf of the Norwegian Environment Agency (NEA, Miljødirektoratet), organised the sampling and carried out subsequent quantitative and qualitative determination of microplastics in samples from the Norwegian environments for the fourth round/year of the national microplastic monitoring program "Microplastic monitoring 2024-2025" (MIKRONOR). Eivind Farmen coordinated the project at the NEA. The project was initiated in 2021, and Vanja Alling has been project manager at NIVA since May 2023. Before that, Bert van Bavel was project manager and he has also been responsible for the overall method development and quality assurance of NIVAs analytical work. The sampling was a collaborative effort, incorporating and coordinating collection of highly different sample materials from seven ongoing national monitoring programs run by the NEA. Sverre Hjelset and Cecilie Singdahl-Larsen managed the coordination of sampling equipment and logistics. Manta sampling was planned by Amy Lusher and conducted by Svetlana Pakhomova, Bjørnar Beylich and Asle Økelsrud. Urban sampling was planned and conducted by Morten Jartun. Sample preparation and microplastic analyses was conducted by Svetlana Pakhomova, Sverre Hjelset, Chiara Consolaro, France Collard, Mari Gjeitnes, Kristin Galtung and Vilde Kloster Snekkevik. Elisabeth Rødland analysed samples for the content of tyre wear particles with Pyrolysis-GCMS, while air sampling was coordinated and samples were analysed and reported by Dorte Herzke and Natascha Schmidt from the Climate and Environmental Research Institute NILU. Data analyses were performed by Espen Lund, Vanja Alling, Elisabeth Rødland, Natascha Schmidt, Amy Lusher and Jemmima Knight.

The scientific quality assurance was provided by Bert van Bavel, Amy Lusher and Morten Jartun. This report has been collaboratively written by Vanja Alling, Espen Lund, Amy Lusher, Elisabeth Rødland, Jemmima Knight, Svetlana Pakhomova, Natascha Schmidt and Dorte Herzke.

Artificial intelligence (AI) has been used as a tool in the preparation of this report in the following way:

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Summary

MIKRONOR, Norway's national microplastics (MP) monitoring program, is dedicated to building a robust knowledge base to support policy development and increase public awareness. Key objectives include identifying major sources and hotspots of MP pollution and establishing baselines for long-term monitoring and trend assessment. Since 2021, the Norwegian Institute for Water Research (NIVA), in collaboration with the climate and environmental research institute NILU, has managed MIKRONOR on behalf of the Norwegian Environment Agency. This report summarizes the findings from the 2024 sampling campaign.

2024 Sampling Campaign Overview

In 2024, MIKRONOR expanded its scope to include, for the first time, surface water samples collected using manta trawls from the Oslofjord and Lake Mjøsa. The campaign also included urban runoff samples—both water and sediments—collected from stormwater traps in Oslo. Additionally, an extended marine sediment sampling campaign was conducted in the inner Oslofjord to follow up on the exceptionally high concentrations of tyre wear particles (TWP) and other microplastics (MP) observed in 2022. Atmospheric deposition samples were collected from three distinct locations: the urban Sofienbergparken (Oslo), the semi-remote Birkenes (Agder), and the remote Zeppelin Station (Svalbard). The 2024 air sampling had an increased focus on TWP, following findings from the 2023 MIKRONOR report.

Urban Runoff: Water and Sediments in Stormwater Sediment Traps

Water and sediment samples from urban stormwater traps were analyzed to assess the contribution of microplastics (MP) from urban runoff to the Oslofjord. High concentrations of both tyre wear particles (TWP) and other MPs were detected, particularly in sediments.

In the water phase, MP levels (number of MP/L) reached a maximum of 733 ± 142 MP/L, with a mean level of 261 ± 121 MP/L. The highest levels were observed in traps located near the harbor. Tyre wear particle (TWP) mass concentrations were detected at mean levels up to 0.5 mg/L, with peak values recorded at Filipstadskaia and Akershuskaia by the harbor.

Sediment samples collected from stormwater sediment traps in Oslo exhibited the highest concentrations of tyre wear particles (TWP) recorded during the 2024 MIKRONOR campaign, with peak levels reaching 240 **mg/g** and a mean concentration of 81 ± 59 mg/g. Additional MP polymers, were detected at concentrations up to 22 mg/g (mean concentration of 8.7 ± 7.8 mg/g) and number of MP particles (besides TWP), reaching 3000 MP/g dw (mean level of 1600 ± 970 MP/g). Also the levels of MP are the highest recorded in any solid sample in the MiKRONOR programme. These values suggest that microplastics—predominantly TWPs—constitute approximately 25% of the total sediment mass in these traps. The data indicates that while sediment traps serve as effective sinks for MPs, particularly TWPs, substantial quantities remain suspended in runoff *water* and are subsequently transported downstream. Consistent with the water phase results, the traps closest to the shoreline showed the highest levels of MPs and TWP. Enhancing sedimentation efficiency, along with proper maintenance and emptying of stormwater traps, within urban stormwater infrastructure may represent a viable strategy for mitigating MP fluxes into ecologically sensitive aquatic or marine recipients such as the Oslofjord.

Marine Sediments

High concentrations of both tire wear particles (TWP) and microplastics (MPs) were observed in the inner Oslofjord, confirming the findings from the 2023 annual report. Particularly concerning are the TWP levels, where sediments were estimated to consist of 0.6–3.5 % TWP by weight. This is considered alarming due to the potential ecological risks associated with TWP, including the leaching of potentially toxic additives

and physical impacts on benthic organisms. Combined measurements of other MPs, besides TWP, from two years of marine sediment sampling within the MIKRONOR project consistently demonstrated elevated MP levels in urbanized and near-shore areas, particularly within the Inner Oslofjord. The highest values reported by Alling et al. (2023) were approximately 5 MP/g dw, while the 2024 samples showed similar or higher levels—up to 15.6 ±2.8 MP/g dw at stations in the inner Oslofjord —confirming and expanding upon previous findings; however, comparisons outside the Oslofjord are less informative due to most measurements falling between LOD and LOQ, indicating presence but uncertain quantification.

Microplastic particles, including TWP, were also found in marine sediment cores from inner Oslofjord (Steilene), Bergen (Raunefjord), and Tromsø (Nordbotn), with relatively high concentrations detected down to 14 cm depth. Dating analyses indicated that these layers correspond to sediments deposited from the late 1970s to early 1980s. The relatively even distribution of microplastics throughout the upper layers—particularly in the Oslo and Tromsø cores—suggests that bioturbation has likely contributed to vertical mixing, redistributing particles over time and obscuring a clear chronological deposition pattern.

Lower MP levels and TWP concentrations were observed in surface sediments along the outer Oslofjord coastline and Norwegian coast, consistent with reduced urban and industrial influence, and in line with the results from the 2023 annual MIKRONOR report. This is consistent with the Mareano 2023 study reported microplastic levels in surface sediments from the North Sea and Skagerrak ranging from 0.096 to 0.8 MP/g dw, with a mean of 0.4±0.1 MP/g dw.

Manta Trawls: Surface Waters of the Oslofjord and Lake Mjøsa

Surface water sampling was conducted in accordance with internationally recognized protocols, aligning with methodological standards anticipated for future reporting under OSPAR and the Marine Strategy Framework Directive (MSFD). The majority of the samples contained low concentrations of microplastics (MPs, 300 µm–5 mm), typically 0–0.6 MP/m^3. Results also showed that reporting as particle counts versus mass revealed different patterns: while the number of MPs per m^3 was similar in Lake Mjøsa and Oslofjord, the mass concentrations were consistently lower in the lake, reflecting the dominance of small particles (<1 mm) compared to larger particles in Oslofjord. Particles from the Oslofjord accumulation zones were largely in the 1–5 mm fraction. Mesoplastics (5–25 mm) were also common in Oslofjord samples, especially in the accumulation zones, but almost absent in Mjøsa.

Two hydrodynamic accumulation zones in the Oslofjord stood out with markedly elevated MP levels up to 65 MP/m^3—two orders of magnitude higher than surrounding waters. One tow alone (120 m^3) recovered over 7,000 expanded polystyrene fragments and more than 200 meso- and macroplastic particles. These accumulation zones are key to understanding spatial dynamics, transport, and sources of plastics in semi-urban marine systems, and their systematic mapping will be critical for effective monitoring and mitigation strategies.

Atmospheric deposition

Atmospheric deposition samples were collected year-long in 2024 at three stations: remote Zeppelin (Svalbard), semi-remote Birkenes (southern Norway), and urban Sofienbergparken (Oslo). Samples were analyzed for microplastics (MP), UV stabilizers, and tire-related chemicals.

Zeppelin generally showed the lowest MP deposition fluxes, with a maximum value of 165 µg/m^2/d. This was comparable to the highest flux observed at the semi-remote Birkenes station (163 µg/ m^2/d). In contrast, the urban site Sofienbergparken exhibited a maximum MP flux of 1514 µg/ m^2/d, approximately ten times higher. Clear differences were also observed concerning the presence of tire wear particles (TWP): they were detected in only 9% of samples from Zeppelin, compared to 68% from Birkenes and 95% from Sofienbergparken. A gradient in the amount of tire-related chemicals was also observable,

reflecting the degree of remoteness (remote < semi-remote < urban). TWP further dominated the polymeric composition in Sofienbergparken, with an average relative contribution of 68%.

MP deposition fluxes generally increased during the sampling period at Zeppelin and Sofienbergparken, while no clear temporal trend was distinguished in Birkenes. At Sofienbergparken, low temperatures followed by snow cover on the road surface resulted in a reduction of TWP fluxes. For the other two stations, no clear links between MP deposition fluxes and environmental parameters were observed. UV stabilizers were detected in MP extracts from all stations, suggesting that MP particles can transport these and other organic chemicals, even to remote areas.

Beach samples

In 2024, microplastic (MP) sampling was repeated at Akerøya, incorporating both the previously monitored OSPAR-designated beach and an additional reference beach to assess spatial variability in MP contamination. The results corroborate findings from the 2023 campaign, indicating consistently low levels of MPs in beach sediments across both sites.

The majority of samples from both years were below the limits of detection (LOD) or quantification (LOQ), suggesting very low levels of MP present at Akerøya. Where MPs were detected, the size distribution was dominated by small particles in the 50–300 µm range, with similar profiles observed across both beaches and sampling years.

The consistency in MP concentrations and particle size distributions between the OSPAR and reference beaches suggests that low MP contamination is representative of Akerøya Island as a whole, rather than being confined to the previously studied OSPAR site.

Sammendrag

MIKRONOR, Norges nasjonale overvåkingsprogram for mikroplast (MP), har som mål å bygge et solid kunnskapsgrunnlag for å støtte forvaltningen, bidra til politikkutvikling og øke offentlig bevissthet. Hovedmålene inkluderer å identifisere viktige kilder og hotspots for mikroplastforurensning, samt å etablere referanseverdier for langsiktig overvåking og trendanalyse. Siden 2021 har Norsk institutt for vannforskning (NIVA), i samarbeid med NILU, ledet MIKRONOR på oppdrag fra Miljødirektoratet. Denne rapporten oppsummerer funnene fra prøvetakingskampanjen i 2024.

Oversikt over prøvetakingskampanje 2024

I 2024 utvidet MIKRONOR sitt omfang til å inkludere, for første gang, overflatevannsprøver tatt med mantatrål fra Oslofjorden og Mjøsa. Kampanjen inkluderte også prøver fra urban avrenning – både vann og sedimenter – samlet fra sandfang i Oslo. I tillegg ble det gjennomført en utvidet prøvetaking av marine sedimenter i indre Oslofjord for å følge opp de svært høye konsentrasjonene av bildekkpartikler (TWP) og annen mikroplast (MP) som ble observert i 2022 års prøver. Atmosfæriske nedbørsprøver ble samlet inn fra tre ulike lokaliteter: urbane Sofienbergparken (Oslo), semi-avsidesliggende Birkenes (Agder), og avsidesliggende Zeppelin-stasjonen (Svalbard). Luftprøvetakingen i 2024 hadde økt fokus på TWP, basert på funn fra MIKRONOR-rapporten 2023.

Urban avrenning: Vann og sedimenter fra sandfang

Vann- og sedimentprøver fra urbane sandfang ble analysert for å vurdere mikroplastbidraget fra urban avrenning til Oslofjorden. Høye konsentrasjoner av både bildekkpartikler (TWP) og annen mikroplast ble påvist, særlig i bunnsedimentene i sandfangene.

I vannfasen nådde MP-nivåene (antall MP/L) opptil 733 ± 142 MP/L, med et gjennomsnitt på 261 ± 121 MP/L. De høyeste nivåene ble observert i sandfang nær havnen. TWP-konsentrasjoner ble målt opp til 0,5 mg/L, med høyeste verdier ved Filipstadskaia og Akershuskaia ved havnen.

Sedimentprøvene fra sandfangene viste de høyeste TWP-konsentrasjonene i hele 2024-kampanjen, med høyeste verdier på 240 mg/g og et gjennomsnitt på 81 ± 59 mg/g. Andre MP-polymerer ble påvist med konsentrasjoner opp til 22 mg/g (gjennomsnitt 8,7 ± 7,8 mg/g), og antall MP-partikler (utenom TWP) nådde 3000 MP/g tørrvekt (gjennomsnitt 1600 ± 970 MP/g). Disse verdiene antyder at mikroplast – hovedsakelig TWP – utgjør omtrent 25 % av den totale sedimentmassen i disse sandfangene. I tråd med resultatene fra vannfasen viste sedimentprøver fra sandfangene nærmest strandlinjen de høyeste nivåene av MP og TWP. Dataene indikerer at sandfang fungerer som effektive fangere for mikroplast, spesielt TWP, men at betydelige mengder fortsatt er suspendert i vannet og transporteres videre. Økt sedimenteringseffektivitet i urbane avrenningssystemer, samt gode rutiner for tømming av sandfang kan være en strategi for å redusere MP-tilførsel til økologisk sårbare vannmiljøer som Oslofjorden.

Marine sedimenter

Høye konsentrasjoner av både TWP og andre MP ble observert i indre Oslofjord, noe som bekrefter funnene fra årsrapporten 2023. Spesielt bekymringsverdig er nivåene av TWP, hvor sedimentene ble estimert til å bestå av 0,6–3,5 % TWP etter vekt. Dette må anses som alarmerende på grunn av de potensielle økologiske risikoene knyttet til TWP, inkludert utlekking av potensielt giftige tilsetningsstoffer og fysiske effekter på bunnlevende organismer.

Målinger av andre MP-typer (utenom TWP) fra to års prøvetaking av marine sedimenter i MIKRONOR-prosjektet viser konsekvent forhøyede nivåer i urbaniserte og kystnære områder, særlig i indre Oslofjord. De høyeste verdiene rapportert av Alling et al. (2023) var omtrent 5 MP/g tørrvekt, mens prøvene fra 2024 viste lignende eller høyere nivåer – opptil 16±2,8 MP/g tørrvekt ved stasjoner i indre Oslofjord – noe som

bekrefter og utvider tidligere funn. Sammenligninger utenfor Oslofjorden er mindre informative, ettersom de fleste målinger ligger mellom LOD og LOQ, og dermed indikerer tilstedeværelse av mikroplast, men med usikker kvantifisering.

Mikroplastpartikler, inkludert TWP, ble også funnet i sedimentkjerner fra indre Oslofjord (Steilene), Bergen (Raunefjorden) og Tromsø (Nordbotn), med relativt høye konsentrasjoner ned til 14 cm dyp. Dateringsanalyser indikerer at disse lagene tilsvarer sedimenter avsatt fra slutten av 1970-tallet til begynnelsen av 1980-tallet. Den relativt jevne fordelingen av mikroplast i de øvre lagene – spesielt i Oslo- og Tromsø-kjernene – tyder på at bioturbasjon sannsynligvis har bidratt til vertikal blanding, og dermed omfordelt partiklene over tid og gjort det vanskeligere å tolke den kronologiske avsetningen.

Lavere nivåer av MP og TWP ble observert i overflatesedimenter langs ytre Oslofjord-kysten og den norske kysten, i tråd med redusert urban og industriell påvirkning, og samsvarer med resultatene fra MIKRONORs årsrapport 2023. Dette er også i samsvar med Mareano-studien fra 2023, som rapporterte mikroplastnivåer i overflatesedimenter fra Nordsjøen og Skagerrak i området 0,096 til 0,8 MP/g tørrvekt, med et gjennomsnitt på 0,4 ± 0,1 MP/g tørrvekt.

Mantatrål: Overflatevann i Oslofjorden og Mjøsa

Prøvetaking av overflatevann ble gjennomført i henhold til internasjonalt anerkjente protokoller, i tråd med metodiske standarder som forventes bli obligatorisk i fremtidig rapportering under OSPAR og EUs rammedirektiv for havstrategi (MSFD). De fleste prøvene inneholdt lave konsentrasjoner av mikroplast (MP) i størrelsesområdet 300 µm til 5 mm, typisk mellom 0 og 0.6 MP/m³. Resultatene viste også at rapportering som partikkeltall versus masse avdekket ulike mønstre: mens antall MP per m³ var relativt likt i Mjøsa og Oslofjorden, var massen konsistent lavere i innsjøen, noe som reflekterer dominansen av små partikler (<1 mm) sammenlignet med større partikler i Oslofjorden. Partikler fra akkumulasjonssonene i Oslofjorden befant seg i stor grad i størrelsesfraksjonen 1–5 mm. Mesoplast (5–25 mm) var også vanlige i Oslofjord-prøvene, særlig i akkumulasjonssonene, men nesten fraværende i Mjøsa.

To hydrodynamiske akkumuleringssoner i Oslofjorden viste imidlertid betydelig forhøyede nivåer av mikroplast, med opptil 65 MP/m³ – opptil to størrelsesordener høyere enn nivåene utenfor akkumuleringssonene. I én enkelt mantatrål ble det samlet inn over 7000 fragmenter av ekspandert polystyren (isopor). Disse sonene inneholdt også betydelige mengder mesoplast og makroplast (5–25 mm og >25 mm), med opptil over 200 partikler i én tråling av et vannvolum på 120 m³.

Identifikasjon og karakterisering av slike akkumuleringssoner er avgjørende for å forstå romlig dynamikk, transportmekanismer og skjebnen til plastavfall i semiurbane marine systemer. Videre kan karakterisering av partiklene i disse sonene gi viktig informasjon om kildene til marin mikroplast og mesoplast. Systematisk kartlegging av disse sonenes variasjon i tid og rom vil være essensielt for å etablere robuste overvåkingsrammer og utvikle tiltak for å redusere plastforurensning i overflatevann.

Atmosfæriske prøver

Atmosfæriske nedbørsprøver ble samlet gjennom hele 2024 ved tre stasjoner: avsidesliggende Zeppelin (Svalbard), semi-avsidesliggende Birkenes (Sør-Norge), og urbane Sofienbergparken (Oslo). Prøvene ble analysert for mikroplast, UV-stabilisatorer og bildekkrelaterte kjemikalier.

Zeppelin viste generelt de laveste MP nivåene i nedbørsprøvene, med en maksimal verdi på 165 µg/m²/dag. Dette var sammenlignbart med den høyeste verdien observert ved Birkenes (163 µg/m²/dag). Til sammenligning hadde den urbane stasjonen Sofienbergparken et maksimalt MP-nedfall på 1514 µg/m²/dag, omtrent ti ganger høyere.

Det ble også observert tydelige forskjeller i forekomsten av bildekkpartikler (TWP): de ble påvist i kun 9 % av prøvene fra Zeppelin, sammenlignet med 68 % fra Birkenes og 95 % fra Sofienbergparken. Det var også en tydelig gradient i mengden bildekkrelaterte kjemikalier, som reflekterer graden av urbanisering (avsidesliggende < semi-avsides < urban). TWP dominerte polymersammensetningen i Sofienbergparken, med en gjennomsnittlig relativ andel på 68 %.

MP i nedbøren økte generelt gjennom prøvetakingsperioden ved Zeppelin og Sofienbergparken, mens det ikke ble observert noen tydelig tidsmessig trend ved Birkenes. I Sofienbergparken så det ut til at lave temperaturer og snødekke på veiene resulterte i lavere TWP-nivåer. For de to andre stasjonene ble det ikke funnet klare sammenhenger mellom MP-nedfall og miljøparametere. UV-stabilisatorer ble påvist i MP-ekstrakter fra alle stasjoner, noe som tyder på at MP-partikler kan transportere disse og andre organiske kjemikalier – selv til avsidesliggende områder.

Strandprøver

I 2024 ble prøvetaking av mikroplast gjentatt på Akerøya, og inkluderte både den tidligere overvåkede OSPAR-stranden og en ny referansestrand for å vurdere variasjonen i MP-forurensning på øya. Resultatene bekrefter funnene fra prøvetakingen i 2023, og viser konsekvent lave nivåer av mikroplast i strandsedimenter på begge lokaliteter.

Majoriteten av prøvene fra begge år lå under deteksjonsgrensen (LOD) eller kvantifiseringsgrensen (LOQ), noe som tyder på svært lave nivåer av mikroplast på Akerøya. Der MP ble påvist, var størrelsesfordelingen dominert av små partikler i størrelsesområdet 50–300 µm, med lignende profiler observert på begge strender og i begge år.

Den konsistente MP-konsentrasjonen og partikkelstørrelsesfordelingen mellom OSPAR-stranden og referansestranden tyder på at lav mikroplastforurensning er representativt for hele Akerøya, og ikke begrenset til den tidligere studerte OSPAR-lokaliteten.

1 Introduction to the microplastic monitoring program MIKRONOR

The Norwegian Institute for Water Research (NIVA), on behalf of the Norwegian Environment Agency is responsible for the Norwegian national microplastic monitoring program. The climate and environmental research institute NILU serves as a subcontractor, handling the analyses of air samples and screening for organic compounds. The program, titled "Microplastics in Norwegian Coastal Areas, Rivers, Lakes, and Air" (MIKRONOR), began in 2021 (van Bavel et al., 2022, Alling et al., 2023; Alling et al., 2024). It organizes the sampling of various environmental matrices from other ongoing national monitoring programs and is scheduled to run until 2026. This is the fourth report in this program.

MIKRONOR aims to establish a baseline for future microplastics monitoring programs and to investigate potential high-impact areas and sources to microplastic contamination in the environment. The ultimate, long-term goal is to create a robust knowledge base on microplastics pollution for policymaking and to ensure the public is well-informed about the state of the environment.

Given the pervasive presence of microplastics (MPs) across environmental compartments, the MIKRONOR campaign has employed a multi-matrix sampling strategy to establish baseline concentrations and support future assessments of temporal trends in MP pollution within the Norwegian environment. Analytical work was primarily conducted at NIVA, encompassing matrices such as surface water, sediments, and urban runoff. Atmospheric deposition samples, including eluates containing additives from car tyres, were analyzed by NILU, reflecting their specialized expertise in air quality and particulate matter characterization.

Terminology: Microplastics (MP) vs. Tyre Wear Particles (TWP)

Within the MIKRONOR program, we distinguish between microplastics (MP) and tyre wear particles (TWP) for methodological reasons. TWP, originating from tyre abrasion on roads or the use of tyre granulates (e.g., in artificial turf), are analyzed by Py-GC/MS and reported as mass. Other MPs are analyzed by µFTIR imaging, where particles are counted and identified by polymer type; their mass can be estimated from particle volume and density. This distinction is purely methodological and does not imply that TWP are excluded from the definition of microplastics. In line with EU Directive 2019/904, TWP are regarded as a subset of microplastics but are reported separately in this program. The separation ensures clarity and transparency, as the two analytical techniques yield results that are not directly comparable. Presenting MPs and TWP separately also improves readability and helps highlight the methodological differences in their analysis.

Size Fractions Analysed in the 2024 Sampling Campaign

In the 2024 monitoring campaign, microplastic (MP) and tyre wear particles (TWP) analyses were conducted across four distinct environmental compartments: coastal marine systems, freshwater lakes, urban runoff, and atmospheric deposition. The size fractions analyzed varied according to sample type and analytical methodology, reflecting both the specific objectives of each sampling and laboratory process effort and the technical constraints of the instrumentation employed.

For most environmental samples (e.g., marine sediments, water, mussels), MP analyses were carried out for particles larger than 50 µm, in line with instrumental capabilities for Fourier Transform Infrared Spectroscopy (FTIR). For marine surface water and lake water sampled with a manta net, the lower size

limit was 300 µm. In addition, mesoplastics (5-25 mm) were analyzed for the first time in the MIKRONOR program due to their presence in manta samples from marine waters.

Pyrolysis-GC/MS was used to quantify tyre wear particle (TWP) mass in different matrices. For sediments (coastal, beach sand, stormwater sediments), untreated freeze-dried material was analysed for the <500 µm fraction. For blue mussels and urban runoff water, TWP were collected on silver filters and analysed in the 50–300 µm size range. These upper cutoffs (500 µm for sediments; 300 µm for mussels and water) may lead to underestimation of total TWP mass. However, an even greater limitation is the lower cutoff in mussel and water samples: particles <50 µm, which are thought to dominate TWP occurrence, were not captured due to methodological constraints.

The atmospheric deposition samples were analyzed for smaller particles, with a lower size limit of 5 µm, reflecting the sampling method using high-volume air filters and precipitation collectors. These samples were analyzed for TWP and MPs by Pyrolysis-GCMS, with mass-based data generated down to the 5 µm level.

The MP particles analyzed by FTIR were identified according to polymer types defined by AMAP (2021), and their mass was estimated based on the calculated volume and known polymer density (Alling et al., 2023). TWP masses were measured using Pyrolysis-GCMS across selected environments, including air and sediment. Atmospheric deposition samples Pyr-GCMS analyses were conducted for nine polymer types. In addition, deposition samples were analyzed for UV compounds and tyre related additives.

A full overview of the size fractions, sample types, and methods used is provided in **Table 1**. All methods are described in detail in Alling et al., 2023, Alling et al., 2024 or in the appendices of this report.

2 Overview of sample types and analyses

Table 1. The 2024 sample types in four environments with sampling methods, number of samples and field blank samples (net blanks + atmospheric blanks), size fractions and methods for lab analyses, including Fourier-Transform Infrared Spectroscopy (FTIR) and pyrolysis GCMS (Pyr-GCMS) for microplastics (MP) and tyre wear particles (TWP).

Environment	Sample type	Sampling method	Number of samples	Number of field blanks	Size fractions analysed, lower limit for FTIR	FTIR for MP analysis	Pyr-GCMS for TWP
Coastal	Water, FerryBox	FerryBox system collecting seawater	9	1	100 µm	✓	
	Marine sediment	Sediment grab	57	57	50 µm	✓	*✓
					300 µm	✓	*✓
	Marine sediment	Sediment core	24	9	50 µm	✓	*✓
					300 µm	✓	*✓
	Beach sediment	Collection of sand from transects at two beaches	30	10	50 µm	✓	
					300 µm	✓	
	Blue mussel	Collection of live mussels	18	0	50 µm	✓	**✓
					300 µm	✓	
	Water, manta net	Trawl of marine surface water using manta net	15	5+5	300 µm	✓	
					***5-25 mm	✓	
Lake	Water, manta net	Trawl of lake surface water using manta net	12	4+4	300 µm	✓	
Urban	Water, urban run-off	Water sampling in urban sediment traps	39		50 µm	✓	**✓
					300 µm	✓	
	Sediment, urban run-off	Sediment sampling in urban sediment traps	21		50 µm	✓	*✓
					300 µm	✓	*✓
Air	Deposition	Full-metal bulk precipitation sampler	66	12	5 µm		****✓
Total number of samples			**291**				

* TWP measured in untreated, freeze-dried solid sample, size fraction <500 µm
** TWP measured on silver filters that have been processed for FTIR. This includes a sieving step and the size fraction analysed is therefore 50-300 µm (same as small fraction FTIR analyses).
*** The analyses of the manta samples from the Oslo fjord included this year the bigger fraction 5-25 mm, called the mesoplastic fraction.
**** Pyr-GCMS for air samples conducted for 9 polymers.

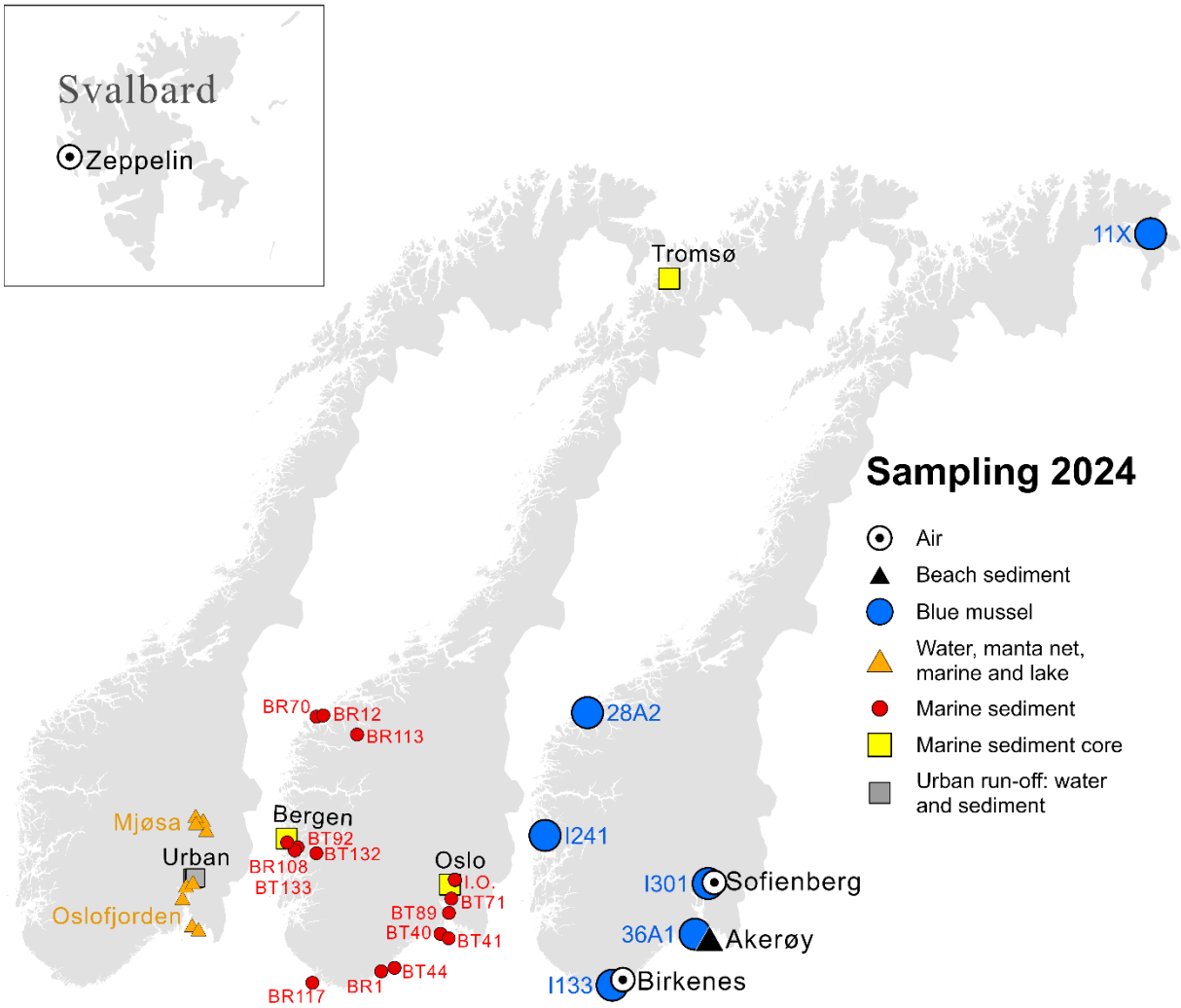


Figure 1. Stations for the analysis of microplastics in 2024. Stations names in Table 2. See Figure 2 and Table 3 for the FerryBox sampling.

Table 2. Codes and names of sampling stations for analysis of microplastics in 2024.

Code	Name	Code	Name	Code	Name
Mjøsa	Mjøsbrua	Tromsø	Sandnessundet	BT80	Bastøy
Mjøsa	Furnesfjorden	BR1	Grimstad	BT92	Bjørnafjorden
Mjøsa	HIAS_sør	BR108	Klokkarvika	I.O. (5x)	Sediment, Oslo
Mjøsa	Gjøvik	BR113	Korsen	I241	Bergen havn
Oslofjorden	Bunnefjorden	BR117	Lista	36A1	Tjøme
Oslofjorden	Slemmestad	BR12	Skinnbrokleia	I133	Kristiansand
Oslofjorden	Svelvik	BR70	Herøyfjorden	I301	Akershuskaia
Oslofjorden	Fredrikstad	BT132	Maurangsfjorden	28A2	Ålesund havn
Oslofjorden	Hvaler	BT133	Fusafjorden	11X	Brashavn
Urban (13x)	Runoff, Oslo	BT40	Færder, Ytre	Sofienberg	Sofienbergparken
Urban (7x)	Sediment,	BT41	Ytre Oslofjord	Birkenes	Birkenes
Oslo	Steilene	BT44	Arendal, Tromøy	Zeppelin	Zeppelin
Bergen	Raunesfjorden	BT71	Hvitsten	Akerøy	Akerøy

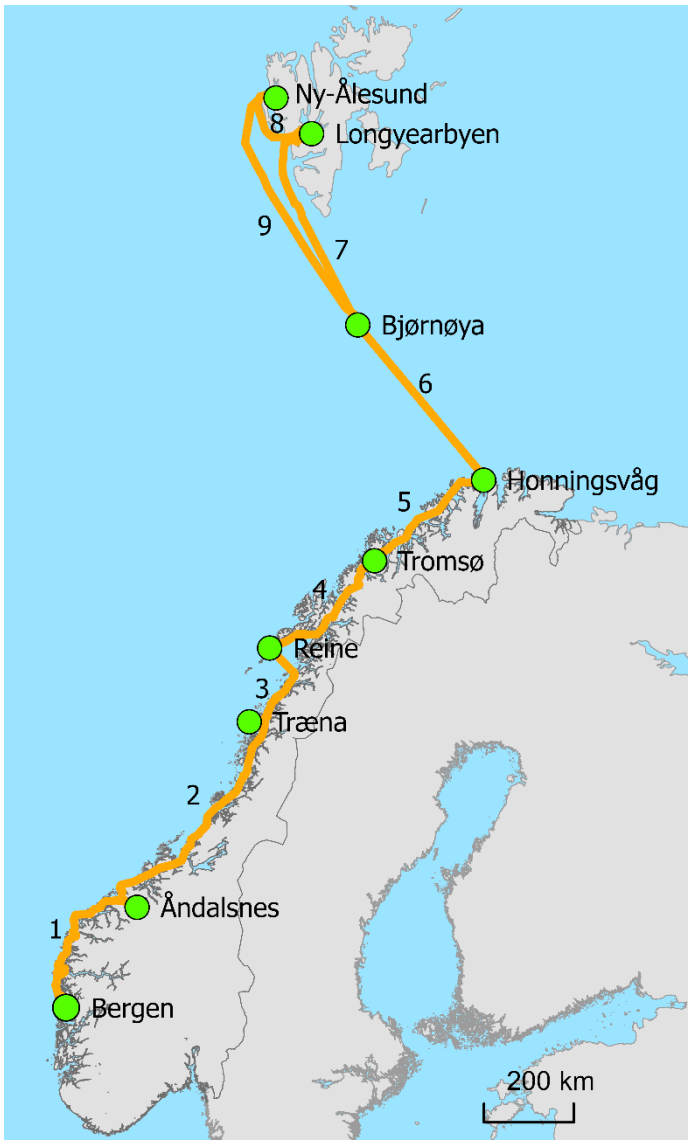


Figure 2. Transects for the analysis of microplastics in FerryBox samples in 2023. Transect names in Table 3 below.

Table 3. FerryBox sampling for analysis of microplastics in 2024.

Nr.	Transect	Nr.	Transect
1	Bergen – Åndalsnes	6	Honningsvåg – Bjørnøya
2	Åndalsnes – Træna	7	Bjørnøya – Longyearbyen
3	Træna – Reine	8	Longyearbyen – Ny-
4	Reine – Tromsø	9	Ny-Ålesund – Bjørnøya
5	Tromsø – Honningsvåg		

3 Key findings

3.1 Urban runoff – water and sediments from sediment traps in the Oslo urban area.

3.1.1. Urban run-off water

Urban runoff, also known as stormwater, is considered one of the largest transport routes of microplastic particles to the environment, containing high levels of particles including TWP from roads, paint from buildings and road markings, and breakdowns of plastic litter (Wang et al 2022, Alling et al 2023).

The traditional management practice for stormwater in urban areas has been to convey it in underground pipelines to the nearest creek or other available aquatic recipients. To prevent silting and subsequent clogging of the pipelines, stormwater traps have been installed all around the cities' streets and roads to collect rapidly settling particulate matter. For example, Oslo city has approximately 30000 stormwater traps (Ræstad 2014).

Urban runoff samples were collected in stormwater traps across various locations in Oslo (**Figure 3**).

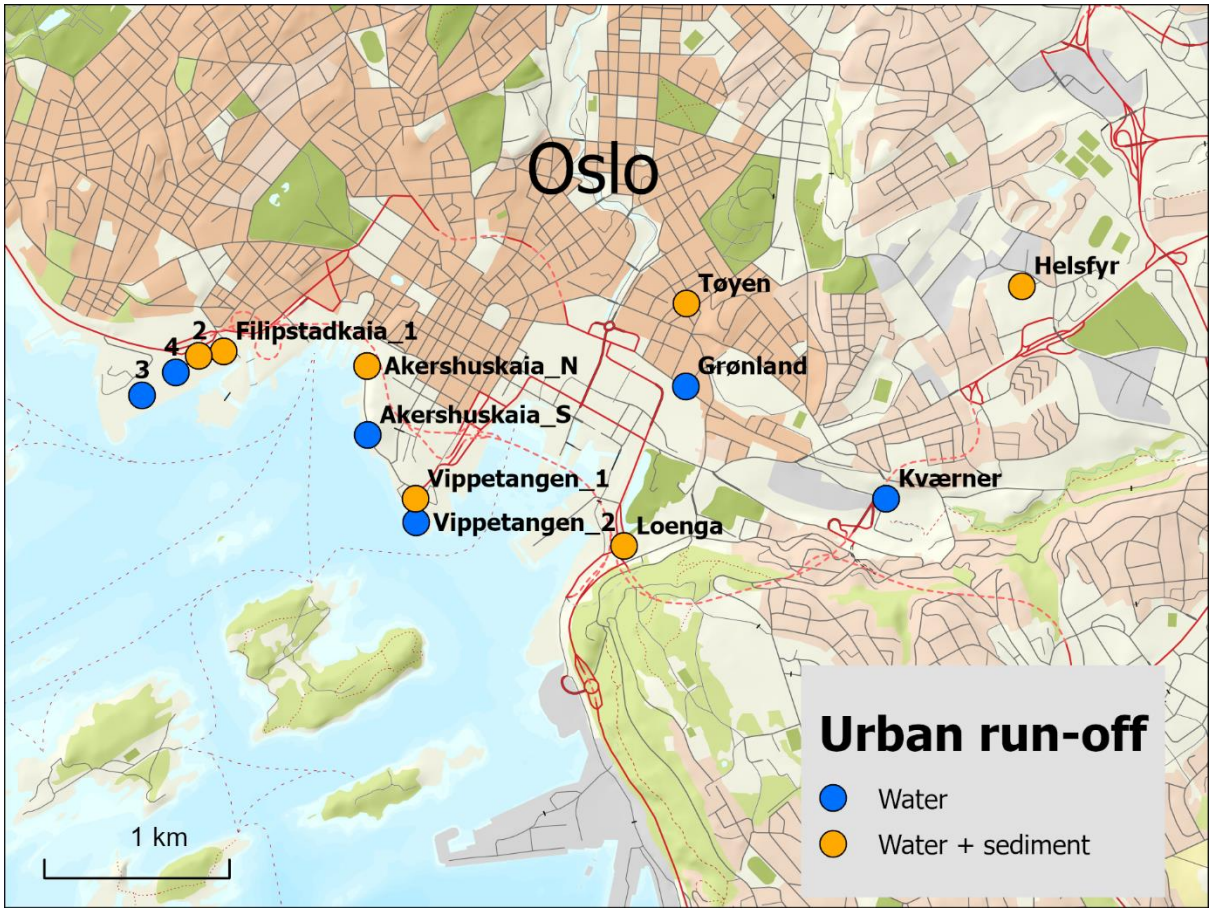


Figure 3. Map of sampled stormwater sediment traps in Oslo 2024. 13 stormwater traps were sampled for water, and of these 7 were also sampled for sediments (see map legend). All samples were taken in triplicates, resulting in 39 urban water samples and 21 urban sediment samples. © Kartverket

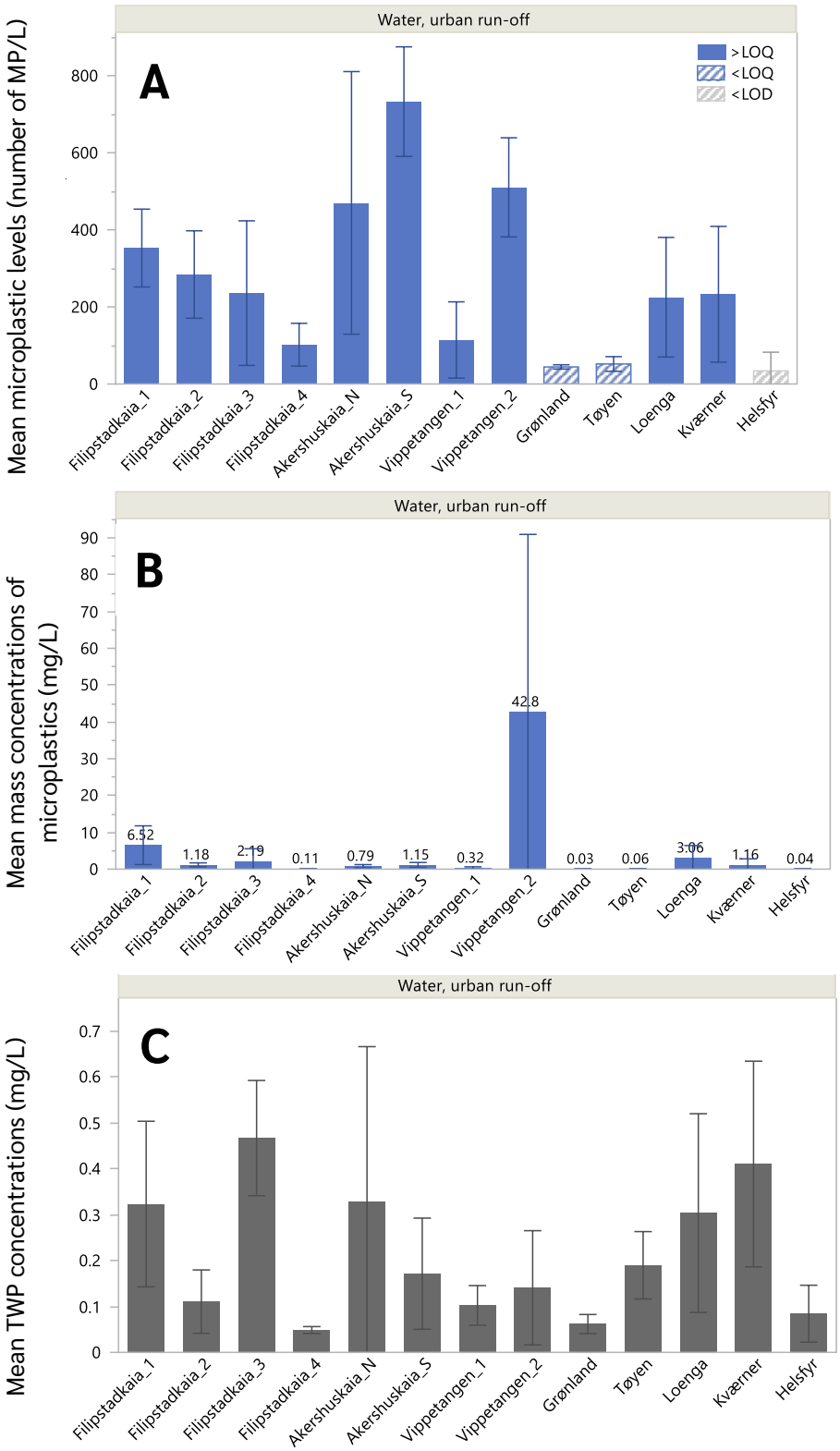


Figure 4. A: Microplastic levels (number of MP per liter) in urban runoff water collected from stormwater traps in 2024, corresponding to locations shown in Figure 3. Size fraction: 50µm – 5 mm **B**: Corresponding microplastic mass concentrations (mg/L) calculated for the same samples. Size fraction: 50µm – 5 mm **C**: Mean TWP concentration (mg/L) in the same urban stormwater runoff samples, size fraction 50-300 µm. Standard deviation is depicted by error bars.

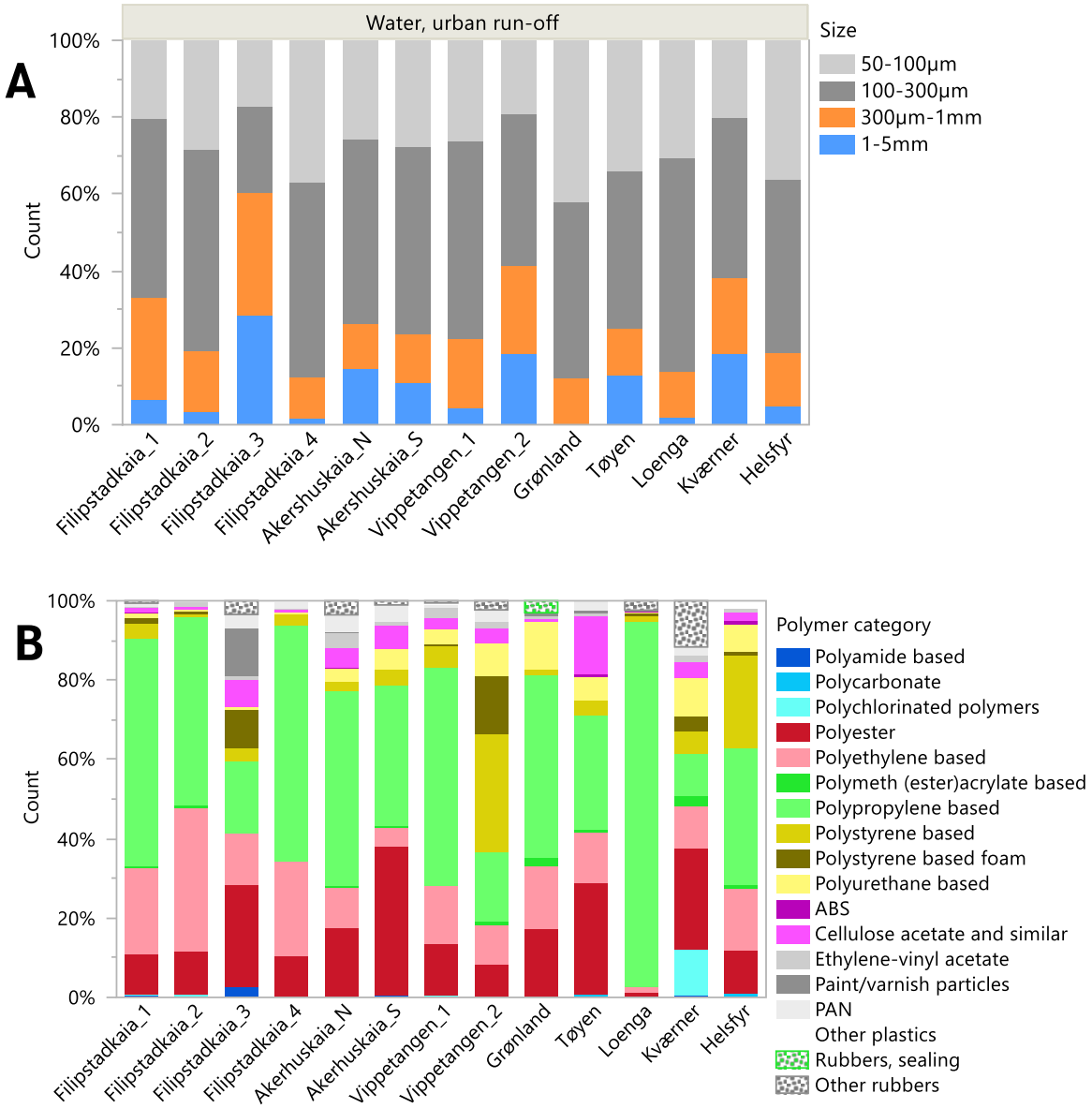


Figure 5. A: Size distribution of MP particles in urban runoff water samples per station. **B:** Polymer category distribution of MP particles in urban runoff water samples per station. TWP not included in the polymer categories (as it is measured as mass concentration not number of particles).

Microplastic measurements in the urban runoff water samples revealed high mean MP levels, ranging from 34±48 to 733±142 MP/L, with an average MP level across all samples of 261 ± 121 MP/L. Urban run-off measured during the MIKRONOR project in 2022 in Oslo and Hamar showed mean levels of 45 ± 18 MP/L, comparable to the lowest levels in this year's study, although collected from different stormwater traps (Alling et al., 2023). The majority of particles in most samples were smaller than 300 µm. However, a few samples contained a considerable fraction (up to almost 60% in one sample) of larger particles.

Wang et al., (2022) provided a broad overview of global MP levels found in literature in urban runoff (some studies including TWP), with a range of 0-8580 MPs/L that encompassed our reported values (30-730 MP/L). Highest values reported from road runoff in Sundsvall Sweden (Lange et al., 2021) It also highlighted the challenges of heterogeneity, and the need for better understanding of MP dynamics, regarding spatial and temporal variations.

Mass concentrations of MPs in urban runoff samples from the 2024 campaign exhibited substantial variability, primarily influenced by the presence or absence of larger particles within individual samples. Total concentrations of MP polymers—excluding TWP—reached up to 42.8 mg/L, underscoring the episodic nature of high-mass MP inputs. However, when particles exceeding 1 mm in size were excluded from the analysis, the measured concentrations ranged from <0.1 to 2 mg/L, indicating that larger particles disproportionately contribute to overall mass despite their lower numerical abundance. These findings highlight the importance of size fractionation in MP monitoring and the need for standardized approaches to ensure comparability across studies and temporal datasets. The TWP mass mean concentrations in the urban runoff water samples ranged from <0.1 mg/L to 0.5 mg /L, placing them in a similar range to the MP mass concentrations across different polymer types for particles <1 mm. The highest levels of TWP in the runoff water in the stormwater traps were reported from Akerhuskaia N (0.058 - 0.80 mg/L), Kværner (0.23-0.73 mg/L) and Filipstadkaia_3 (0.38 - 0.61 mg/L) (**Figure 5**). The average TWP concentration across all samples was 0.22 ± 0.19 mg/L. This is within the same range as reported in other comparable studies. In Oslo, at a residential urban area (Tåsen), the TWP values (>1.6 µm) in road runoff were monitored over one year during heavy rainfall. Samples were taken directly from the road (0.90-6.9 mg/L) and after treatment using a rain garden (0.26 - 0.76 mg/L) (Carvahlo et al., 2024). Similarly, a study from a highway outside of Oslo (E18 Tomter) measured the road runoff from the highway during heavy rainfall and reported TWP values from <detection limit to 25 mg/L during one year of monitoring (Rødland et al., 2025. in prep. B).

The TWP values are also comparable to recent findings in other countries. Two recent studies from Sweden have reported TWP values from roads; Gaggini et al., 2024 reported mean TWP values of 0.52-2.9 mg/L in water from stormwater traps from a highway road (Testsite E18 Sweden), and Lindfors et al. 2025 reported TWP values of 0.19-1.04 mg/L in runoff from the city Luleå in Sweden. In comparison, Rauert et al., 2022 have reported varying TWP values from urban sites in the Brisbane area (Australia), from below detection limit to approximately 1 mg/L TWP (originally reported as TRWP values, assuming 50% TWP) However, the values reported from Oslo stormwater traps is significantly lower compared to the stormwater values reported during storm peaks in another Brisbane study (Rauert et al., 2022), with levels ranging from below detection limit to 9.2 mg/L.

It is crucial to note, however, that for both MP and TWP, only the fraction greater than 50 µm was analysed in this study, which should be considered when comparing to other studies that includes the <50µm fraction. Studies have suggested that the main proportion of TWPs are in the size fraction <350µm (Chae et al., 2024), with the largest proportion of the particles being <20µm (Gaggini et al. 2024, Rødland et al., in prep). A recent study comparing MPs and TWPs in road runoff also reported that the main size fraction for all MPs combined were 11-75µm (>50% of the total particle mass) (Lindfors et al., 2025). TWPs also have a higher density than most other microplastic particles (1.2-1.7 g/cm3, Jung & Choi, 2022), causing TWP to settle more easily compared to other MPs. Both the size discrimination of the study and the density of TWPs could explain why TWPs are found in comparable and sometimes lower levels in the runoff water compared to other MPs, while other studies have reported that TWP contributes up to 96% of the total microplastic particle mass in road runoff (Lindfors et al., 2025). Future monitoring studies should consider the variations in size and density of MPs and TWPs to enhance the understanding of their transport and fate in the environment.

3.1.2. Sediments in stormwater traps

The main function of a stormwater trap is to retain and trap debris, sand, litter and larger particles from the water flowing into the stormwater drainage (sewage) system. Although the retention effect of these types of structures for tyre wear particles has been debated (Vogelsang et al., 2020), it has been demonstrated that TWP can be retained in the sediments in both highways (Gaggini et al., 2024) and in road tunnels (Rødland et al., 2022).

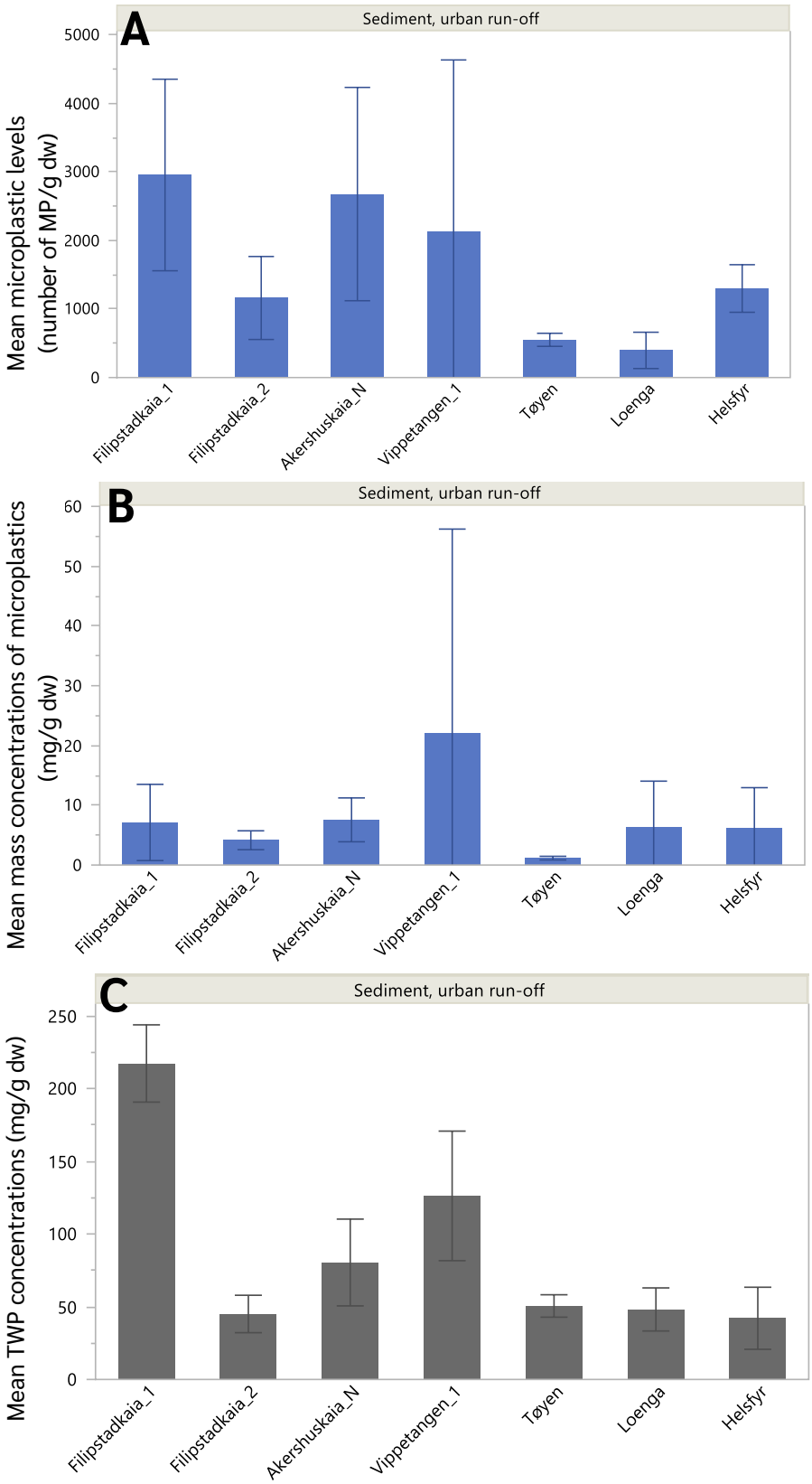


Figure 6. A: Mean microplastic levels (number of MP per g dw) in urban runoff sediment samples collected from stormwater traps in 2024, corresponding to locations shown in Figure 3. **B:** Corresponding mean microplastic mass concentrations (mg/g dw). **C:** Mean TWP concentration (mg/g) in the same urban runoff sediment samples. Standard deviation is depicted by error bars. (n=3 for each sampling site, except n=2 for Filipstadkaia_1 TWP).

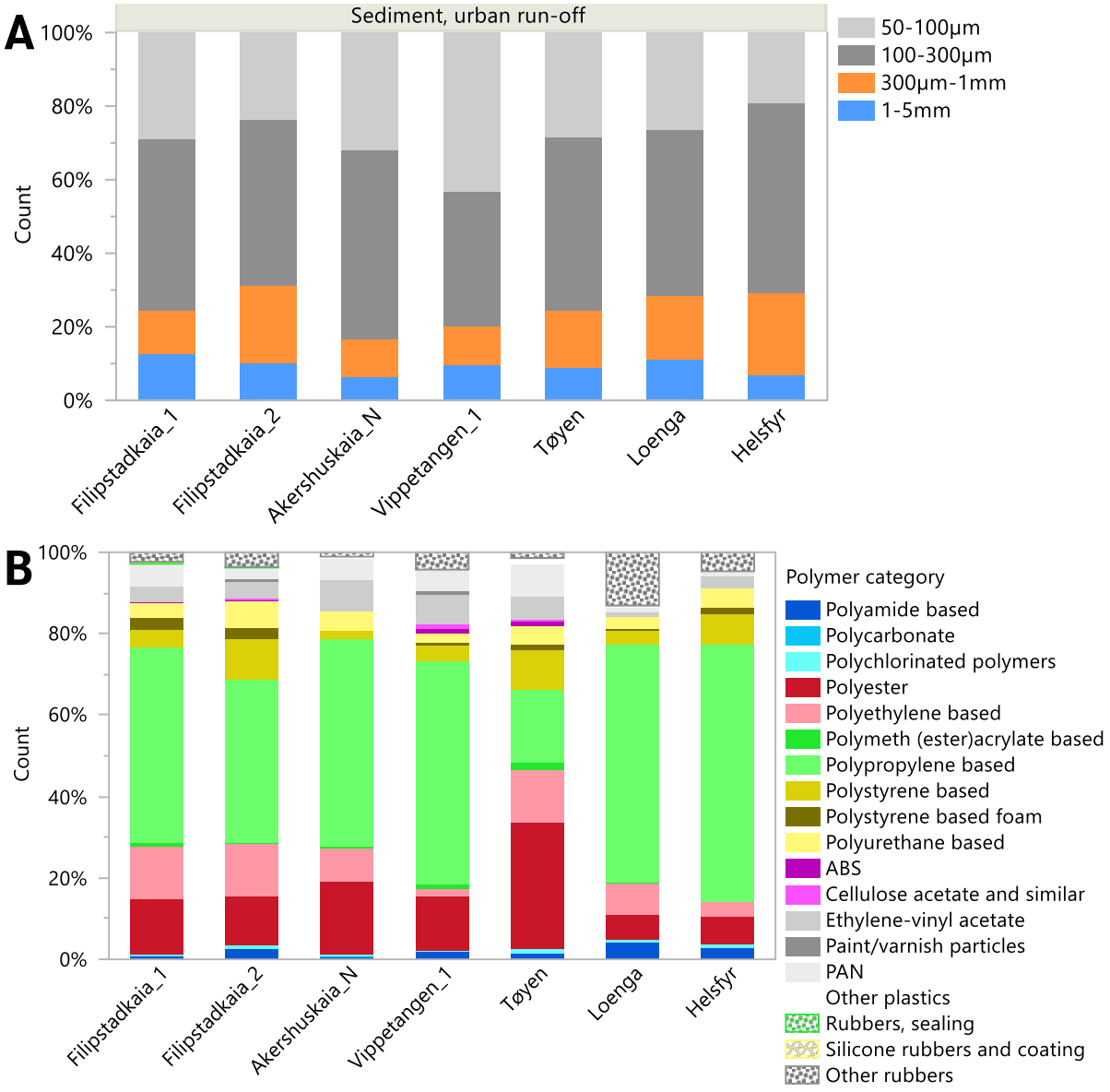


Figure 7. A: Size distribution in stormwater sediment samples per station. **B:** Polymer category distribution in stormwater sediment samples per station. TWP not included in the polymer categories (as it is measured as mass concentration not number of particles).

Microplastic measurements in the stormwater sediments revealed very high mean MP levels, ranging from 396±265 to 2957±1399 MP/g dw, with an average MP level across all samples of 1596 ± 969 MP/g dw. These levels are the highest MP levels measured in any solid matrix in MIKRONOR so far. The majority of MPs in most samples were smaller than 300 µm. Polypropylene, polyester and polyethylene dominated the polymer composition (TWP not included), and the MP particles identified comprised between 60 and 80 % of those three polymer types.

The levels of TWP in the stormwater sediments were substantially higher than what was measured for other MPs, with mass concentrations ranging from 22 - 236 mg/g (81 ± 59 mg/g) across all stormwater sediment traps. The highest levels of TWPs were detected in the sediments from Filipstadkaia 1 (236 mg/g), resulting in close to 24% TWP in the sediments (**Figure 6**). This is considerably higher than what

has previously been reported for stormwater sediments in a road tunnel in Oslo (53.1 ± 1.33 mg/g dw, Rødland et al., 2022). The stormwater traps with the highest levels of TWP were all close to the Oslo harbor (Filipstadkaia, Vippetangen, Akerhuskaia), up to ten times higher compared to the stormwater trap from Helsfyr, which is the furthest from the harbor in this sampling campaign. This could be related to the difference in how runoff is distributed in this area, especially considering if water from the urban areas will flow from the areas above and down towards the harbor. Although these are plausible explanations to why we see variations in the stormwater traps, this study did not include investigations of the sediment depth, nor do we have information about the age of the sediments, e.g. how long these have been accumulating for and when the stormwater sediment traps were last emptied. It has previously been reported that the frequency of emptying the sedimentation traps is important to remain sufficient volume for retention and for the functionality of the sedimentation traps (Vogelsang 2018).

3.2 Marine sediments

3.2.1. Sediment grab samples

The results for marine sediment samples taken with grab (**Figure 8**) reveals that samples from the Inner Oslofjord exhibited the highest MP levels, with Indre Oslofjord 01 and Indre Oslofjord 03 showing the highest mean MP/g dw among the marine sediment samples. Conversely, other stations displayed lower concentrations, with numerous measurements falling below the LOD or even the LOQ. This pattern indicates a heterogeneous distribution with distinct hotspots within the Inner Oslofjord, consistent with earlier findings in MIKRONOR (Alling et al. 2023), which also reported the highest MP levels in marine sediments in the Inner Oslofjord and lower levels along the Norwegian coast, with most values measured below LOQ (but above LOD).

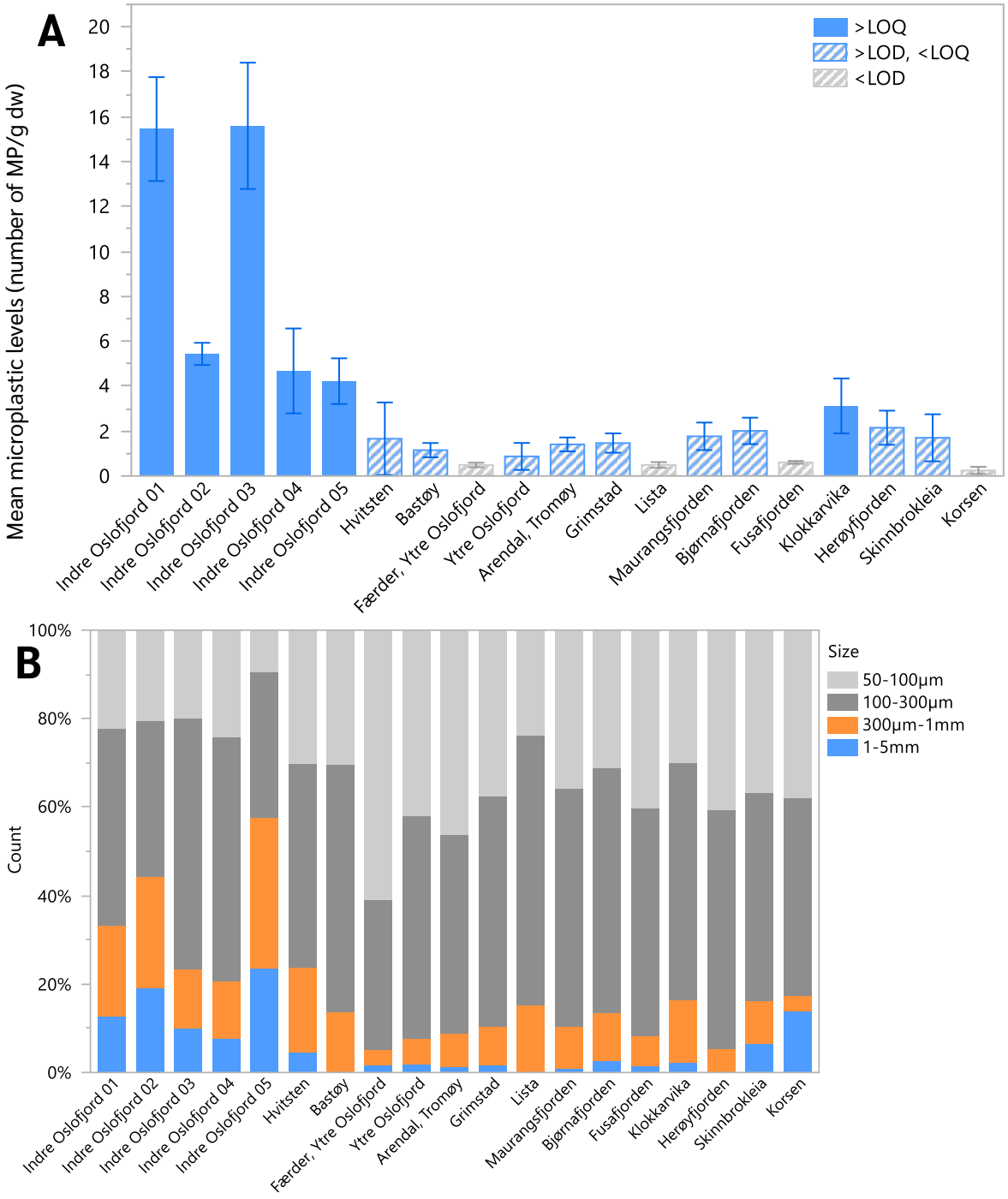


Figure 8. A: Average microplastic levels (number of MP/g dw) in marine sediment samples taken with grab sampler in 2024. Error bars = SD. Number of samples = 3 per station. **B:** Size distribution for the same samples/stations.

Combined measurements from two years of marine sediment sampling within the MIKRONOR project (2022 samples, Alling et al., 2023, and 2024 samples, as depicted in **Figure 12**) consistently demonstrate elevated microplastic abundance in urbanized and near-shore areas, particularly within the Inner Oslofjord. The highest values reported by Alling et al. (2023) were approximately 5 MP/g dw from two samples in the Inner Oslofjord. This is consistent with the levels observed in three of the 2024 samples from Inner Oslofjord (stations 02, 04, and 05). Furthermore, stations 01 and 03 in the 2024 dataset exhibited even higher levels, with 15.4 ±2.3 and 15.6 ±2.8 MP/g dw, respectively. While the current study

offers more granular data for specific stations within the Oslofjord, reinforcing previous findings, direct comparisons of MP levels at stations outside the Oslofjord are less informative. Most measurements from remote MIKRONOR stations fell between the LOD and LOQ, indicating the presence of microplastics but with uncertain quantitative reliability. To improve data quality in future surveys, larger sediment volumes should be analyzed to reduce the frequency of results below LOQ. In the present dataset, MP levels in remote samples ranged from <LOD to 3 MP/g dw, comparable to the Mareano 2023 study, which reported 0.096–0.80 MP/g dw (mean 0.38 ± 0.13) in North Sea and Skagerrak sediments.

Furthermore, the current study consistently shows a dominance of smaller microplastic particles (e.g., 50-300 µm) across the majority of stations, encompassing both Inner Oslofjord and the other coastal samples. (**Figure 8**). The percentage of the particles smaller than 300 µm (50- 300 µm) were calculated to approximately 45 % in the sample Indre Oslofjord 05 and increased in the samples further out in the Oslofjord to 60-80 % of the particles, and in the further remote stations along the Norwegian coast to comprising 85-95 % of the particles.

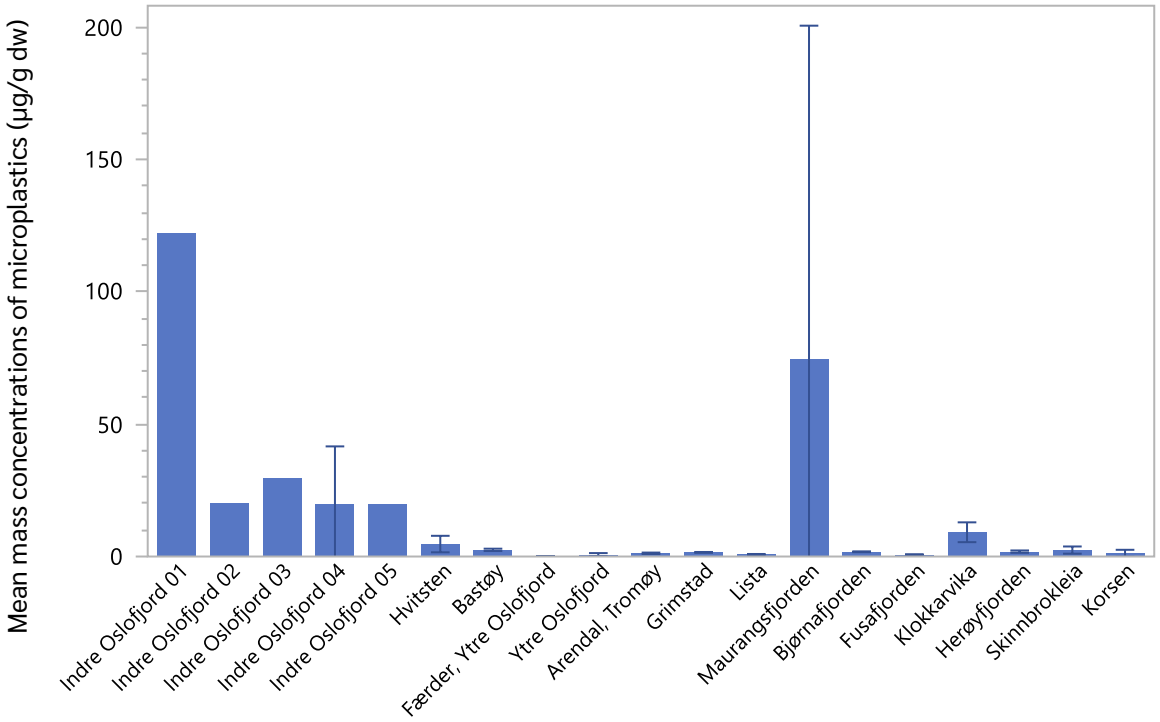


Figure 9. Marine sediment mean values (µg/g dw) per station. n=3 per station except for Indre Oslofjord 01, 02, 03 and 05, where n=1.

Our data show the disproportionate contribution of larger particles to the overall microplastic mass **(Figure 9)**, even if their numbers are lower. While smaller particles are numerically dominant, larger particles still contribute significantly to the total plastic load by mass. This is e.g. causing the high concentration at Maurangsfjorden, where one of three replicates had one big particle with high mass.

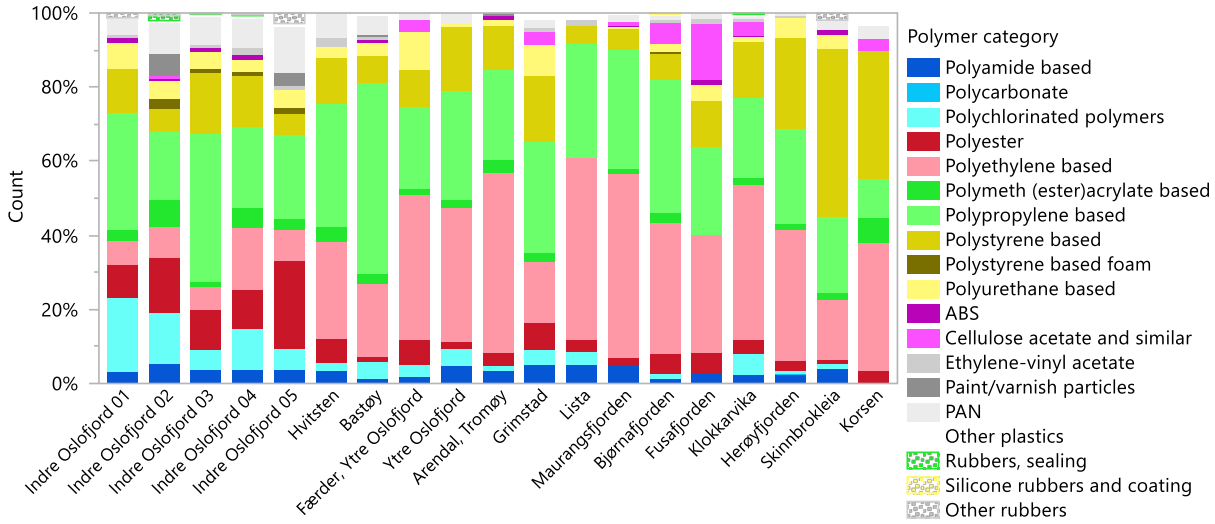


Figure 10. The polymer categories distribution in the sediment grab samples. Location of Inner Oslofjord samples are shown in Figure 12 map and other coastal samples are shown in Figure 1 map.

The polymers dominating the Inner Oslofjord grab samples differed slightly from the other coastal grab samples (**Figure 10**), mostly by having a more diverse composition, with the top three polymers accounting for just above 50 % of the total compared to almost 80 % in the other coastal samples. Notable is that the inner Oslofjord samples had a much lower percentage of Polyethylene, that was the dominating polymer in the other coastal samples. Polyethylene was also reported as the dominating polymer type in the MAREANO study (Bienfait et al., 2025).

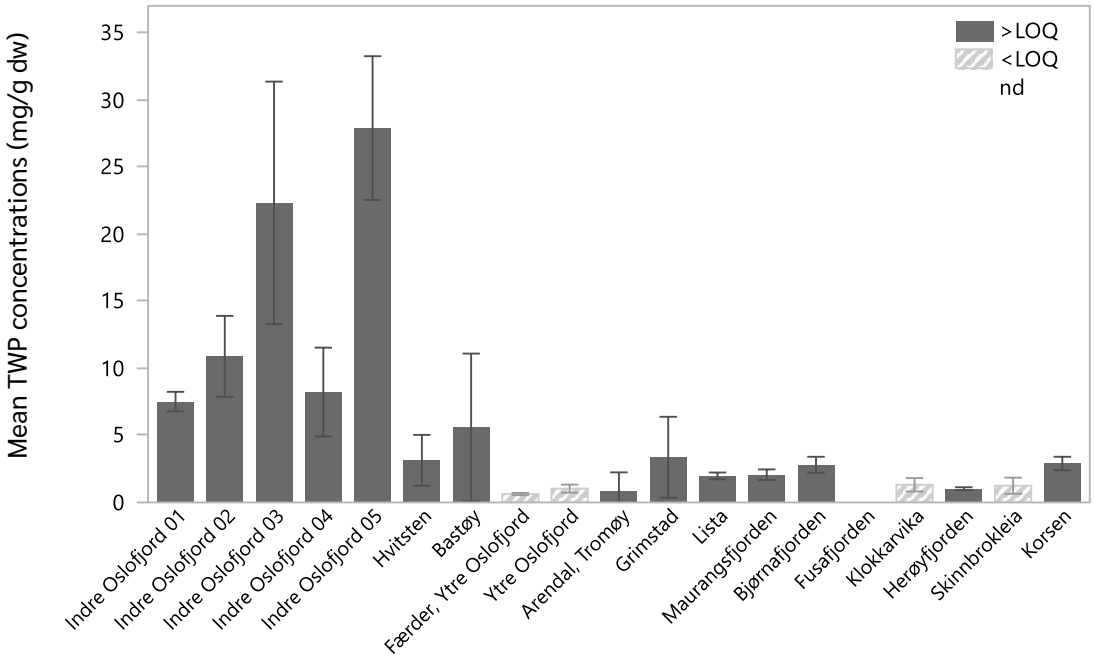


Figure 11. Mean TWP concentration (mg/g) in marine sediment samples. N=3 for each station. TWP was only detected in one sample from Arendal and in none of the samples from Fusafjorden. In both Klokkarvika and Skinnbrokleia, two samples were <LOQ and one samples in Herøyfjorden was <LOQ.

As for MPs, the TWP values show higher levels in the inner Oslofjord close to Akershus harbor, with TWP values of up to 34 mg/g (average 15.4 ± 4.3 mg/g dw, n=15), compared to the outer Oslofjord (0.567 mg/g

dw). The results from 2024 are comparable to the results of 2022, supporting the conclusions that there are higher levels of TWP close to land and close to urban areas compared to more remote marine sites. This is also supported by the results from the other MPs. The station Indre Oslofjord 05 exhibited the highest proportion of larger microplastic (MP) particles in the sediment, as well as the highest concentrations of TWP. This pattern may suggest that the site serves as a depositional area where both heavier TWP and larger microplastic particles settle rapidly. In contrast, station Indre Oslofjord 03 showed the highest total number of microplastic particles, despite the majority being in the smaller size fraction (<300 µm), indicating a different particle size distribution and potentially different transport or retention dynamics at this site. The values found in the other coastal areas varied but were generally lower than the inner Oslofjord (n.d. - 6.83 mg/g) (**Figure 11**) and showed similar variation as previous reports. In **Figure 12**, a detailed comparison of the results from 2021/2022 and 2024 for the inner Oslofjord is shown, confirming the high concentrations of TWP found in the Inner Oslofjord sediments.

Except for the results presented in Alling et al., 2022, only one study has been to report TWP in marine sediments using a comparable analytical technique (Barber et al., 2025). This study investigated the TWP levels in Osaka Bay in Japan and reported median concentrations of 0.16 mg/g for grab samples and 0.23 mg/g for sediment trap. Both values were converted from the reported unit TRWP to TWP (assuming 50% tyre tread). Their results suggested that sediment traps were better to evaluate the recent deposition of sediments compared to grab sampling. For future monitoring, and especially for coupling MP and TWP levels to recent weather events such as heavy storms and flooding, sediment traps should also be considered. The low number of studies reporting TWP values from marine sediments highlights the need to investigate this environmental matrix further, as the marine environment often acts as a sink of pollution coming directly from land sources such as roads, urban runoff and WWTP, and from rivers transporting these pollutants into estuaries (Kahn et al., 2024).

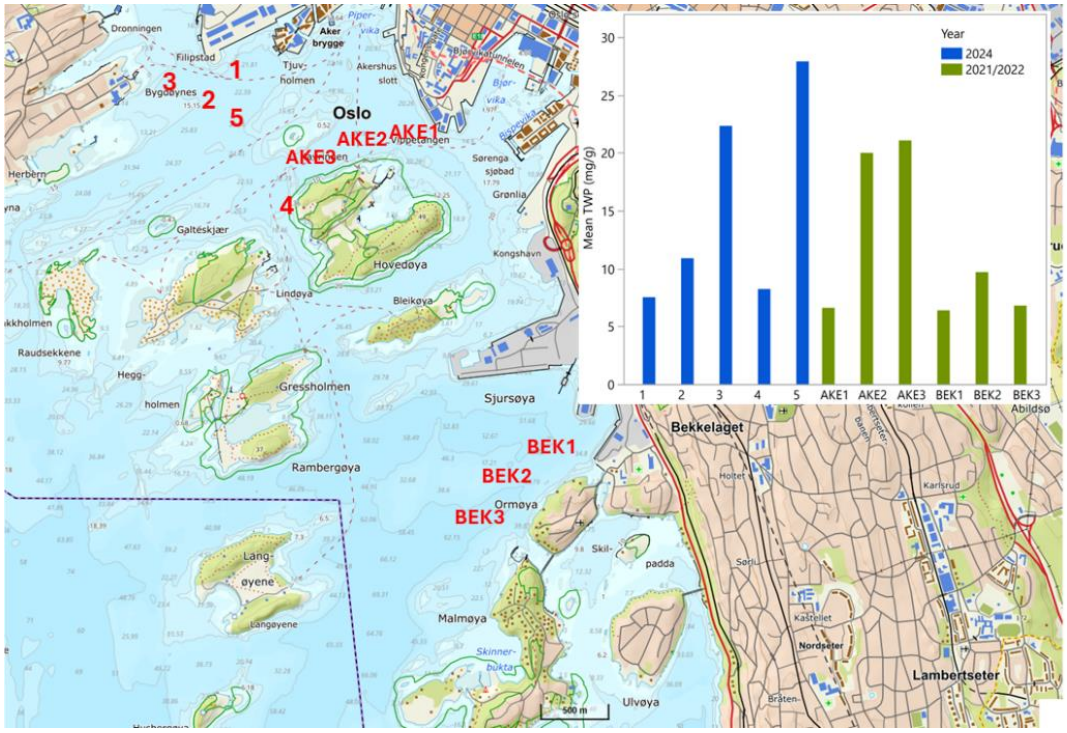


Figure 12. Map of stations (1-5 in 2024, and AKE1-3 and BEK1-3 from 2021/2022) and TWP concentrations in corresponding stations in inner Oslofjord. © Kartverket

In 2024, NIVA performed a power analyses of the TWP concentrations in sediments, showing that a sample size of 3-5 samples is enough to differentiate concentrations of TWP taken at different locations

in the Inner Oslofjord. Based on the limited data available, we assume that this sample size is adequate for other marine sediments. Confidence in this sample size will improve with further sampling. The full report of this study is presented in appendix 7.3.6.

3.2.2. Sediment cores

Sediment cores collected from Steilene in the Inner Oslofjord, from Raunefjord near Bergen, and from Nordbotn near Tromsø exhibited a weak correlation between depth and the concentrations of both microplastics (MP) and tyre wear particles (TWP) **(Figure 13)**. This weak correlation is likely attributable to bioturbation within the cores. Although the sediments were dated using Pb210 determination, the analyzing laboratory indicated that dating results are unreliable in the presence of bioturbation. This unreliability appears to apply to the Oslo and Tromsø cores, at minimum. While the Bergen core displayed a gradual decrease in microplastic concentrations with depth, this trend was not observed for tyre wear particles. The same pattern was observed in the MAREANO study, with weak correlations between depth of the core and number of MP particles. All cores were dominated by smaller MPs, typically below 300 µm.

Two of the cores contained low levels of MP and TWP in the reference slice at the bottom. This reference slice was intended to serve as a field blank, assuming the absence of MPs and TWPs below the historical introduction of plastics and the increase in vehicular traffic, which would have introduced TWPs as sediment contaminants. However, the Tromsø core unexpectedly showed detectable levels of both MP and TWP in its reference slice. This suggests that the cores, particularly the Tromsø core, may have been subjected to bioturbation throughout their entire length, an explanation more plausible than contamination during sampling, as contamination by TWP on the research vessel or within the laboratory is considered unlikely, a conclusion supported by both lab and atmospheric blank analyses.

In a recent study by Ni et al., 2024, the effect of bioturbation on the distribution of TWP in sediments was tested and found to lead to TWP being present in deeper layers of the sediments compared to controls without bioturbation. Burrowing activity has also been demonstrated for microplastics in previous studies (Gebhardt & Forster 2018), thus highlighting the challenge of using sediment cores to study the time-scale of microplastic presence. Lab studies have also demonstrated that the presence of TWPs and MPs in the sediment can lead to adverse effects on burrowing organisms, such as increased burrowing activity, oxidative stress, metabolic disorder and intestinal blockage (Song et al, 2024). However, results reported are both species- and polymer-dependent, e.g. the overall effect of MP and TWP presence in the sediments and its influence on sediment community is not yet fully understood. Thus, the results from monitoring programs such as MIKRONOR and other comparable studies are important to understand the pathways of MPs and TWPs to the marine environment, as a basis for more in-depth studies of the effect in the environment.

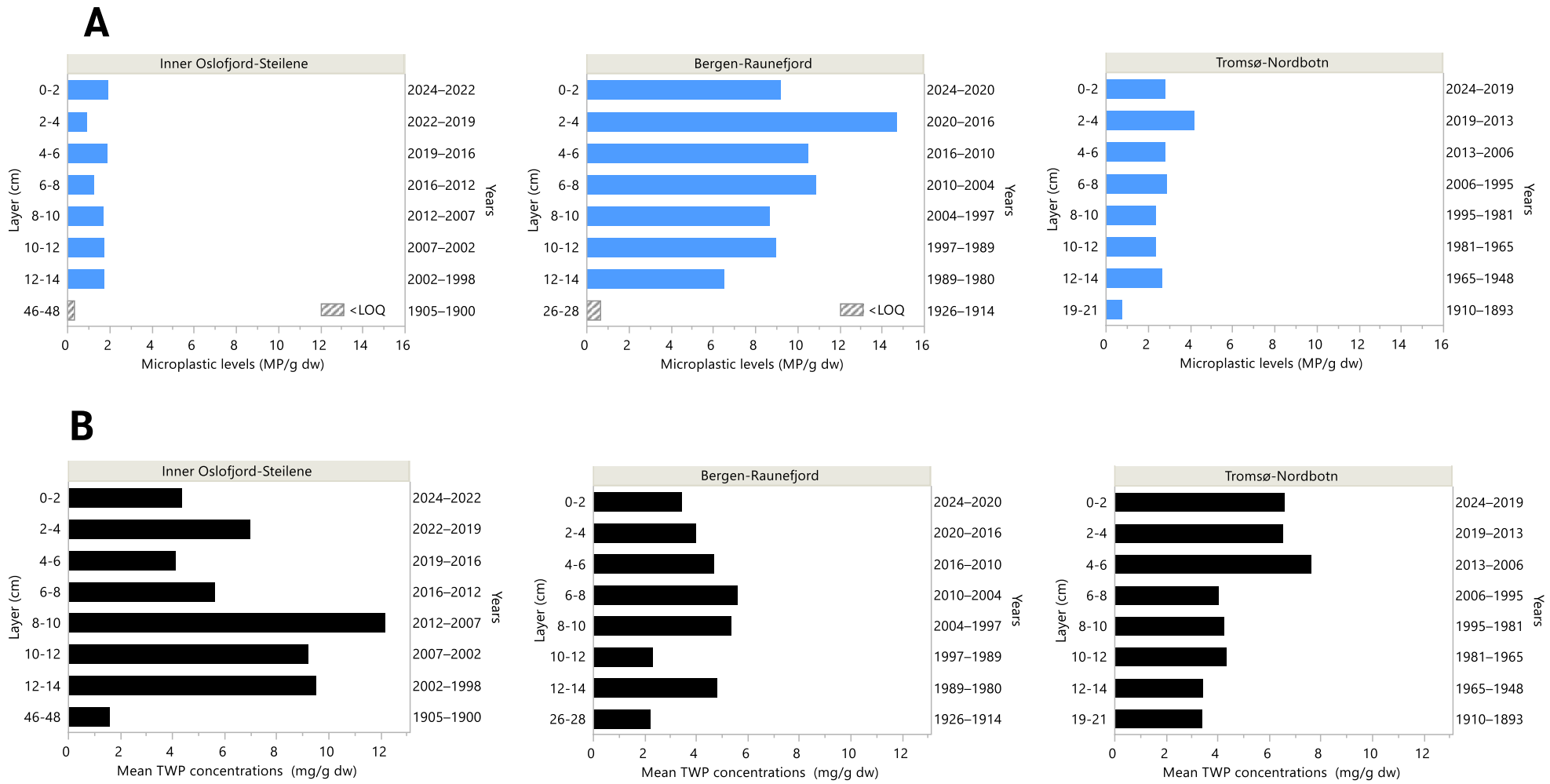


Figure 13. Results of sediment cores from Oslo, Bergen and Tromsø. **A:** Number of MP/g dw in the analyzed layers of the cores. **B**: TWP concentrations (mg/g dw

3.3 Manta samples

This is the first year that manta nets have been used to sample surface waters of the Oslofjord and Lake Mjøsa. Previously, we used high volume pump samples deployed just below the surface (van Bavel et al., 2022, Alling et al., 2023, Alling et al., 2024). For more detailed description of the sampling design and methods see chapter 7.2 in appendix

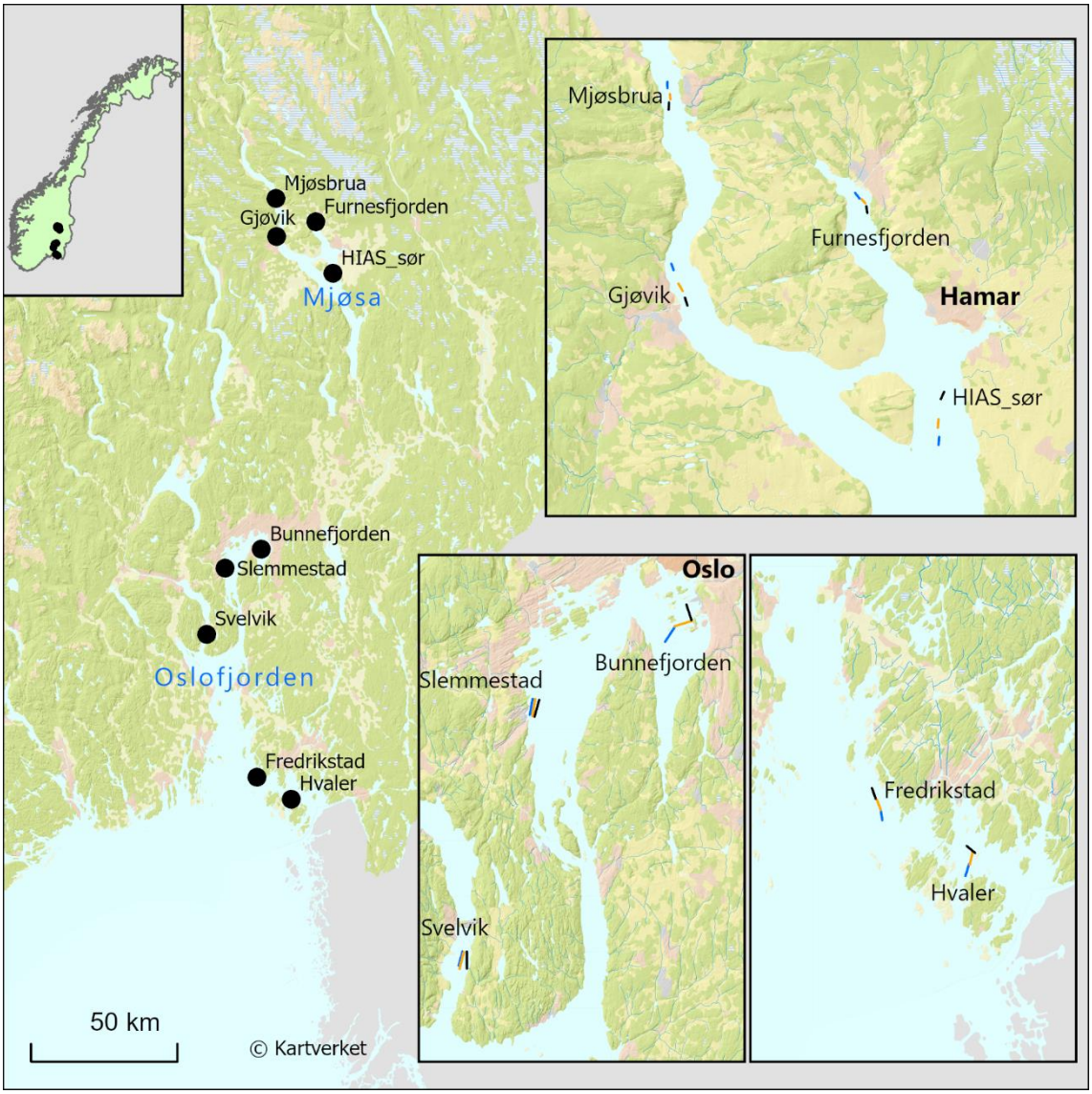


Figure 14. Map of sampling locations for manta net in Oslofjord (n=15) and Lake Mjøsa (n=12). Samples were taken in triplicate at each location (visualized as black, orange, blue transects). © Kartverket

Manta net sampling was designed to sample the Oslofjord in connection to point sources:

- Inner Oslofjord: Bekkelaget and VEAS
- The two water flows of the Glomma River
- Drammenselva River

These locations also allow comparisons with freshwater samples collected upstream. Furthermore, Manta trawl sampling was conducted in **Lake Mjøsa** to investigate four areas with known point sources (Clayer et al., 2018):

- Mjøsbrua (heavy traffic from the E6 highway, runoff from the bridge)
- HIAS (wastewater treatment plant near Hamar, with discharge into Lake Mjøsa)
- Hamar (urban area with significant industry, traffic, and development)
- Gjøvik (wastewater treatment plant and urban area)

Manta net samples were collected from Lake Mjøsa (n=12) and in Oslofjord (n=15) **(Figure 14)**. Plastic particles were identified in the microplastic range (300µm-5mm) as well as the mesoplastic range (5-25mm). Furthermore, during sampling in Oslofjord, two of the samples targeted visible accumulation zones at the stations Bunnefjorden and Fredrikstad. These accumulation zones are not known to be permanent, but rather moving zones where wind and water movements gather debris in large quanta. Due to the distinct difference in the sampling strategy (randomized locations vs. targeted areas with visible plastic debris), and the extreme differences in levels of microplastics and mesoplastics making it hard to visualize in the same figures, the results are shown and discussed partly separately below. Data for microplastic levels is reported as particles per m^3 and as mass per m^3 (MP/m^3 and $\mu g/m^3$, respectively). The data has also been generated in m^2, this has been presented in the Appendix 7.3.2.

3.3.1. Overall results

The levels of microplastics ranged from 0 to 0.6 MP/m^3 across all replicates, with the exception of the two samples in accumulation zones. Highest of those was Bunnefjorden_1 with 65 MP/m^3. Whilst there was variation between all stations, there was no *significant* difference outside the accumulation zones between samples in Oslofjord, compared to samples in Lake Mjøsa, nor between stations. The two samples from the accumulation zones in the Oslofjord two stations were significantly different from the other stations.

3.3.2. High levels of micro- and meso plastics in two accumulation zones in the Oslofjord

The number of microplastics observed in the samples from the accumulation zones was 1 to 2 orders of magnitude higher than reported in all other samples when reported as particles per m^3 (**Figure 15**). For example, Bunnefjorden_1 collected in the accumulation zone contained 65 MP/ m^3 compared to Bunnefjorden_2 and Bunnefjorden_3 which contained 0.41 and 0.37 MP/ m^3, respectively. Similar is true for Fredrikstad, 4.8 MP/ m^3 in the accumulation zone, compared to 0.49 MP/ $m^3$and 0.06 MP/ m^3 outside the accumulation zone.

The mass of microplastics reported in the Fredrikstad accumulation zone sample (11500 µg / m^3) is two orders of magnitude higher than in the samples from outside the accumulation zone (104 and 105 µg / m^3) compared. The mass of microplastics in the Bunnefjorden accumulation zone sample (21000 µg / m^3) is 1000 x larger than reported in the samples from outside the accumulation zone (50 and 15 µg/ m^3). The Fredrikstad accumulation zone formed at the boundary between fresh and salt water, creating a frontal zone. The plastics accumulated there could have originated from the Glomma River, from local coastal activities, or from more distant sources such as the Skagerrak and Kattegat regions. The Bunnefjorden accumulation zone, on the other hand, was more likely driven by wind drift and Langmuir circulation, which concentrated plastics from coastal and maritime activities in the inner Oslofjord. Surface accumulation zones are highly dynamic and short-lived, forming and dispersing within hours or days in response to changing wind, tides, and river flow. Identifying the precise origin of the observed distributions would require numerical backtracking models.

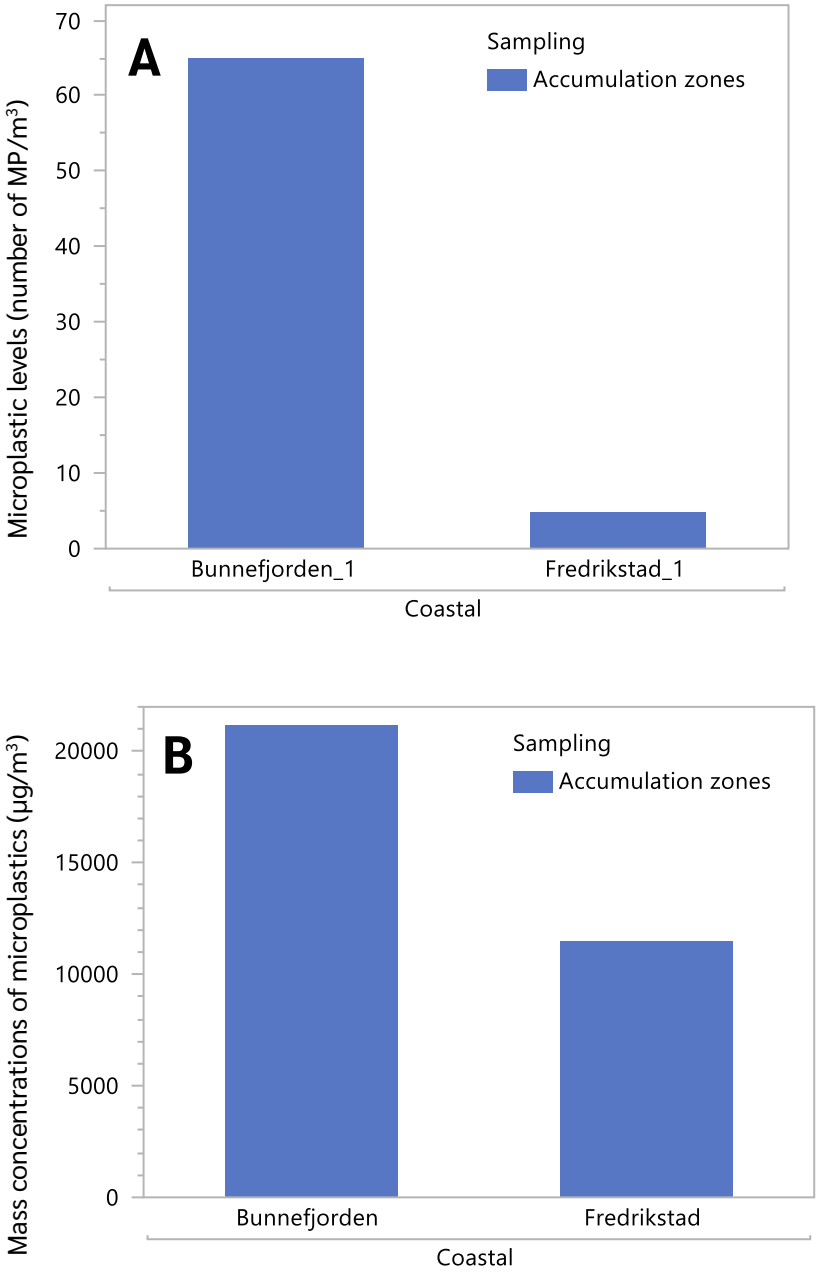


Figure 15. **A:** Microplastic levels (MP/m^3) in coastal manta net samples in accumulation zones, showing the results from Bunnefjorden_1 and Fredrikstad_1. Both stations n=1. **B:** Corresponding microplastic mass concentrations (µg/m^3).

The Bunnefjorden accumulation zone is particularly interesting, the sample had considerably more microplastic particles than any other station. There were more than 7000 expanded polystyrene foam particles (<5mm, **Figure 16**) in this single sample, resulting in a reported load of 65 MP/ m^3. (For details about the estimation and analyses of all these particles, see appendix 7.2.). The same trend was observed for the mesoplastic and microplastic fractions.

Expanded polystyrene has many uses and potential sources, and it is commonly observed in litter assessments. The material is light weight and buoyant, and it can travel far from sources of input. The particles can be seen as individual beads or as agglomerations/clumps of beads. Some of the beads look "fresh" whilst others show signs of environmental weathering: changes of color from white to grey/brown/yellow. The beads can become squashed (sometimes leading to misidentification as fragments), or also become fragmented themselves. Expanded polystyrene is used to make boxes and

packaging. About 4,800 tons of EPS are used in Norway each year. Most of this material is used to package household appliances, including white and brown goods, and sanitary ware. Fish boxes account for about 1,700 tons of annual consumption (35%). Due to the many possible sources in Oslofjord one cannot pinpoint the source of release, however, it may be interesting to establish a protocol that allows a detailed assessment of EPS.

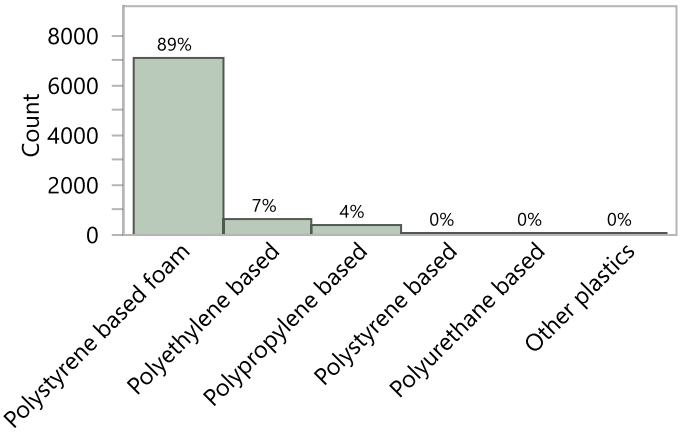


Figure 16. Distribution of polymer categories of microplastic particles in Bunnefjorden accumulation zone manta net water sample.

A challenge due to the highly uneven distribution of floating plastic particles in the Oslofjord will be how to find the zones, how to sample them in a representative way, how to report levels and how to estimate the total mass/number of particles in those zones best in future monitoring. Current monitoring practices (OSPAR, MOEJ) suggest neither targeting nor avoiding accumulation zones, but it is clear from our assessment that they should be handled differently due to the orders of magnitude difference in microplastic load. By targeting an accumulation zone, it increases the likelihood of encountering floating plastics, which can facilitate discussion on sources and polymers, although it may not represent the overall mean levels and concentrations of floating plastics. More information is presented in the section on mesoplastics in surface water of Oslofjord.

3.3.3. Low levels in areas without accumulation zones: Costal and Lake samples

While the accumulation zones encompassed large amounts of floating plastics, the general level outside the accumulation zones, measured as number of MP/m^3 in both Lake Mjøsa and the Oslofjord, were low.

The pattern in the levels of microplastics in manta nets were significantly different when reported as number of MP/m^3 compared to mass/m^3. Specifically, the mass of microplastics in the lake samples were consistently lower than those observed in Oslofjord. This is likely associated with the dominance of small particles in Lake Mjøsa compared to larger particles in Oslofjord. This is supported by the trend that mesoplastics (5 mm-2.5 cm) were observed in Oslofjord samples whereas only 1 mesoplastic was observed in Lake Mjøsa.

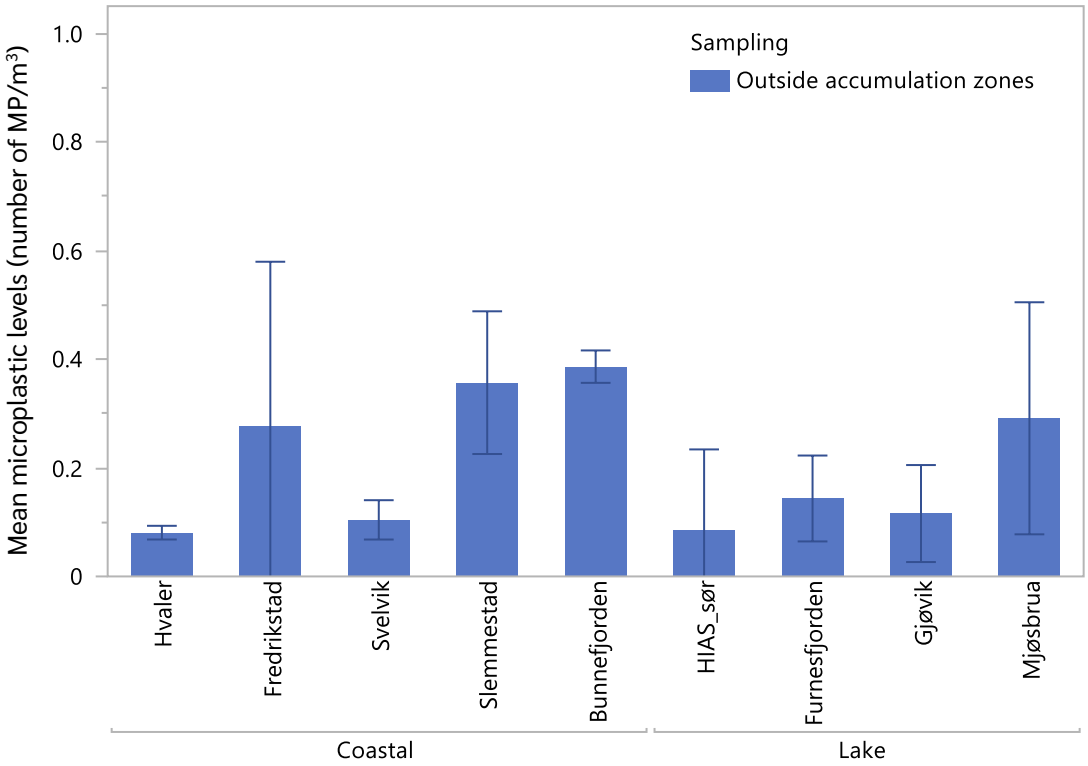


Figure 17. Mean microplastic levels (MP/m³) in coastal and lake (Mjøsa) manta samples outside accumulation zones. Number of samples per station n=3 except Fredrikstad and Bunnefjorden where n=2.

The number of microplastics (excluding textile fibres) in the net blanks was smaller than the total observed counts per sample in Oslofjord, however there was less of a difference for the Lake samples. see chapter 7.2.1.3 The polymers observed in the blanks were similar to those identified in the field samples. The values in the net blanks suggest that Lake Mjøsa samples could benefit from a longer deployment time in the water (sample a longer distance/bigger volume) to ensure that any potential contamination is surpassed by true environmental concentrations.

The sampling outside accumulation zones showed similar microplastic levels in both coastal and lake environments when expressed as number of MP/m³ (**Figure 17**). These concentrations are comparable to earlier Arctic studies using the same method (average 0.34 ± 0.31 SD; Lusher et al., 2015), indicating that the results are representing low levels of MP in surface waters of the Oslofjord and Lake Mjøsa, outside accumulation zones. Previous MIKRONOR results from the outer Oslofjord, based on FerryBox sampling of upper water layers, reported values of approximately 1 MP/m³ (van Bavel et al., 2022; Alling et al., 2023). The slightly lower levels found in this study can be explained by methodological differences—particularly the lower size detection limits: the Manta net samples down to 300 µm, whereas the FerryBox system detects particles as small as 100 µm. Within the Oslofjord, Slemmestad (mean: 0.36 MP/m³) and Bunnefjorden (mean: 0.39 MP/m³) had significantly higher concentrations than Hvaler (mean: 0.08 MP/m³) and Svelvik (mean: 0.10 MP/m³). No significant differences were observed between stations in Lake Mjøsa.

Before the 2024 survey, MIKRONOR used manta nets primarily to collect surface water samples from rivers. Among the five Norwegian rivers monitored, the Alnaelva showed the highest mean microplastic concentration (4.9 MP/m³; Alling et al., 2024), with substantial variability between samples (0.88–13.3 MP/m³). The Drammen River had concentrations between 1.0 and 1.5 MP/m³. Both rivers discharge into the Oslofjord, and their microplastic levels are intermediate between the fjord's background concentrations and the elevated levels observed in accumulation zones.

Although the number of microplastics per cubic meter was similar between the lake and coastal environments (outside the accumulation zones), the mass concentrations differed markedly (**Figure 18**). In the Oslofjord, Hvaler had the lowest mean mass concentration (5.3 µg/m³), while no significant differences were observed between the stations in Lake Mjøsa. The low mass concentration at Hvaler is likely due to both fewer particles and a dominance of smaller particles (<1 mm), including 13 expanded polystyrene foam beads, which have very low density. It should be noted that mass concentrations were calculated using assumed polymer densities; for expanded polystyrene, this introduces considerable uncertainty, meaning these values should not be regarded as absolute. The relatively low mass concentrations in Lake Mjøsa can likely be explained by the same factor—particle size distribution skewed towards smaller particles.

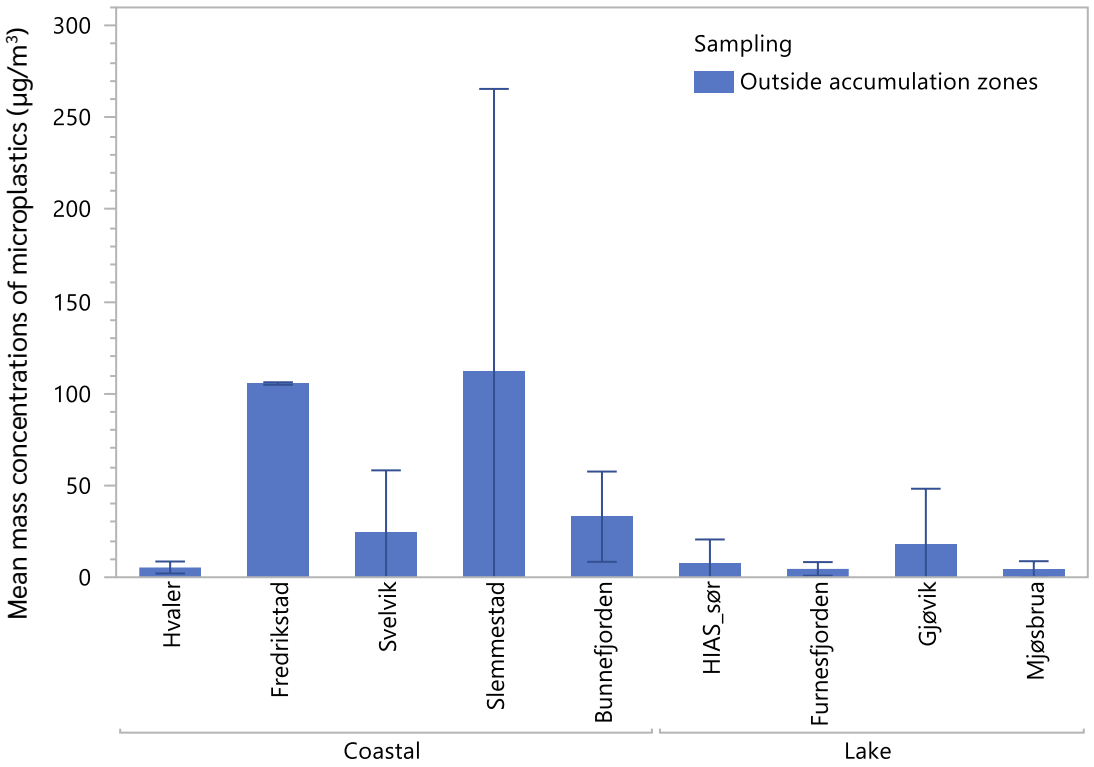


Figure 18. Mass concentrations of microplastics (µg/m³) in coastal and lake (Mjøsa) samples from outside accumulation zones. Number of samples per station n=3 except Fredrikstad and Bunnefjorden where n=2.

3.3.4. Comparisons between lake, coastal, and samples inside vs outside accumulation zones

Simply comparing microplastic concentrations inside and outside accumulation zones does not capture the full picture. The measured MP/m³ levels in these zones depend heavily on how precisely the zones were targeted during sampling and on the proportion of surrounding water that was sampled. Within the two samples from the accumulation zones in the Oslofjord, we recorded 8150 and in the 13 samples outside the accumulation zones only 438 particles. While the concentration data are important, the particles collected from the accumulation zones may provide deeper insights into the origin, characteristics, and potential fate of the most common microplastics in the fjord.

Microplastic size distributions differed markedly between the three sample groups: Lake Mjøsa, Oslofjord accumulation zones, and Oslofjord areas outside the accumulation zones. Lake Mjøsa samples were dominated by smaller particles (300 µm–1 mm), whereas particles from the Oslofjord accumulation zones were largely in the 1–5 mm fraction. This aligns with the high abundance of meso- and macroplastics also recorded in these zones (see below). Samples from outside the accumulation zones in the Oslofjord showed an intermediate pattern, with roughly half of the particles smaller than 1 mm (**Figure 19**). The mean particle size was 1 771 µm in the Oslofjord compared to 733 µm in Lake Mjøsa.

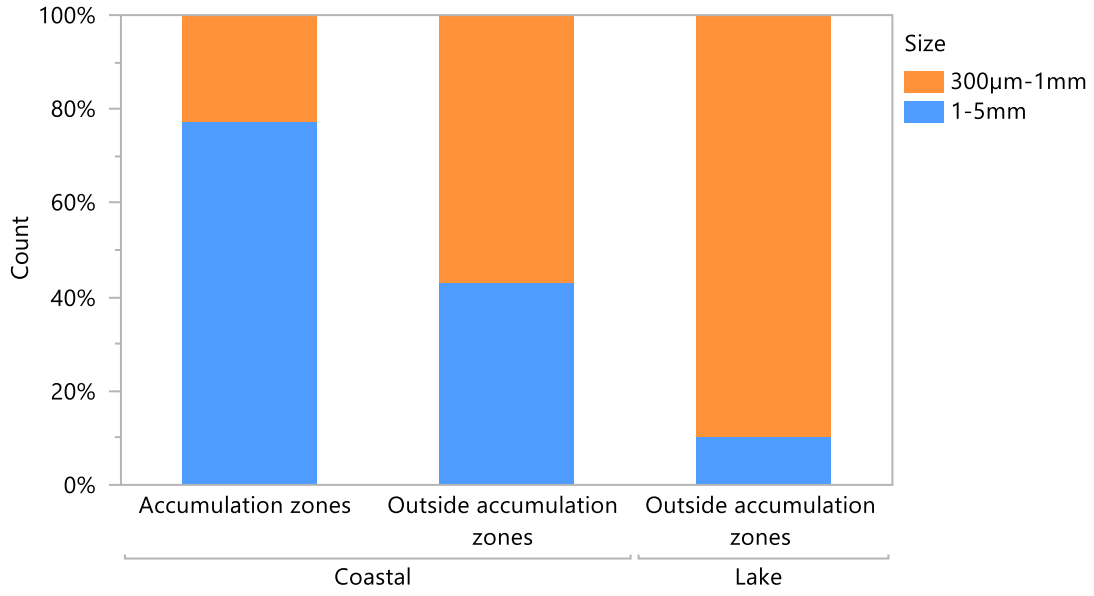


Figure 19. Size categories of microplastics observed in manta coastal and lake water samples.

Microplastics in the samples outside the accumulation zones (both coastal and lake samples) were dominated by fragments (45%) and foams (46%), followed by films (35%) and beads (1%). Coastal samples consisted of just over half the microplastics as foam (51%) followed by fragments (41%), whilst lake samples were dominated by fragments (80%), followed by films (18%) (**Figure 20**). The samples from the accumulation zones were dominated by foam (88%), this is the expanded polystyrene particles that were estimated to be at least 7000 in the Bunnefjorden accumulation zone sample.

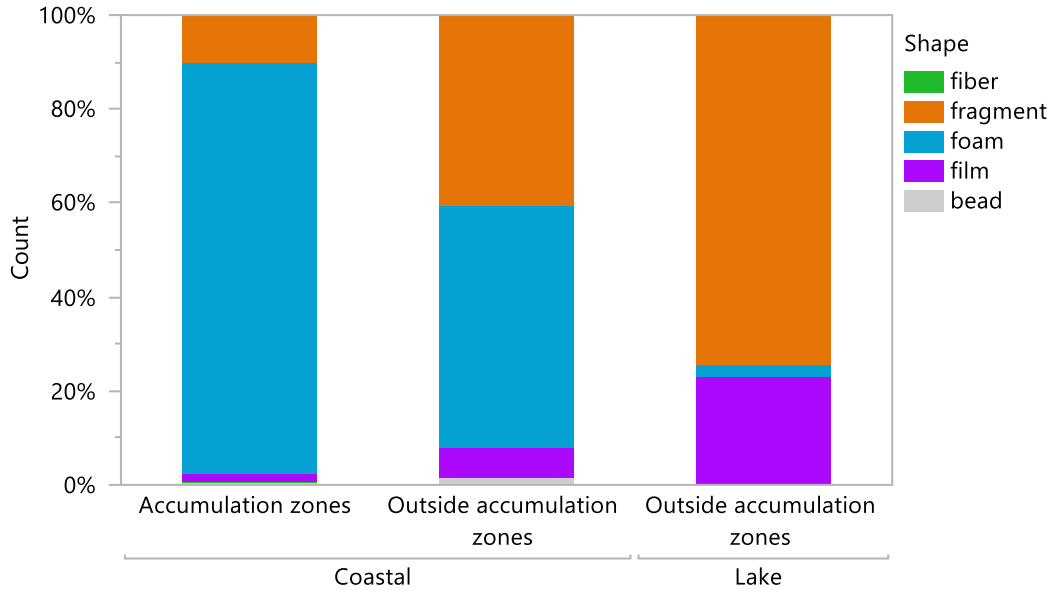


Figure 20. Distribution of shapes of microplastic particles in manta coastal and lake water samples. Fibres represent only technical fibers, for definition, see chapter 7.1.1.2.

There was a difference in polymers identified in samples from the accumulation zones compared to from outside accumulation zones (**Figure 21**). Although, there was a similar number of polymer types identified between the freshwater and coastal samples. The polyethylene and (expanded) polystyrene are not surprising since they are a lightweight polymer, although the presence of paint/varnish is surprising since

it is a denser polymer. Polystyrene foam was a polymer type found in much higher abundance in the coastal samples, compared to the lake samples.

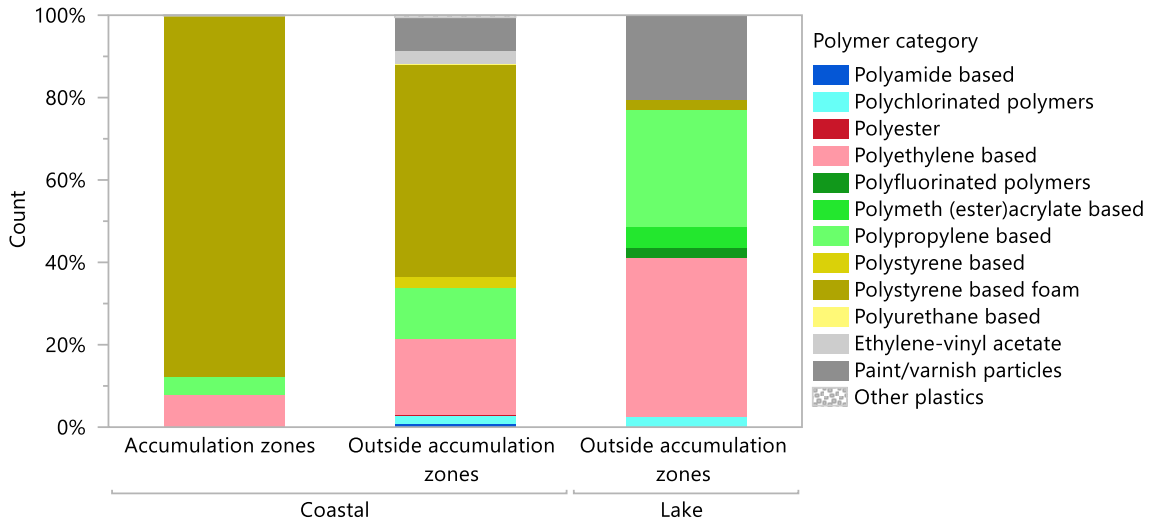


Figure 21. Distribution of polymer categories of microplastic particles in manta coastal and lake water samples.

3.3.5. Meso- and macroplastics in surface water samples

In total, 263 mesoplastic particles and 24 macroplastic particles were observed in the manta samples. All but one of the mesoplastics, and all macroplastics, were recorded in the Oslofjord; only a single mesoplastic was detected in Lake Mjøsa.

Nearly all sites in Oslofjord had at least 1 mesoplastic or macroplastic. Fragments accounted for 63%, followed by films (27%), foams (9%), and < 1% beads and fibre bundles. The samples from the accumulation zones had significantly higher numbers of mesoplastic and macroplastic compared to the samples outside the accumulation zones, and more polymer categories (**Figure 23**). As expected, the polymers found were types that have lower densities and float in surface waters.

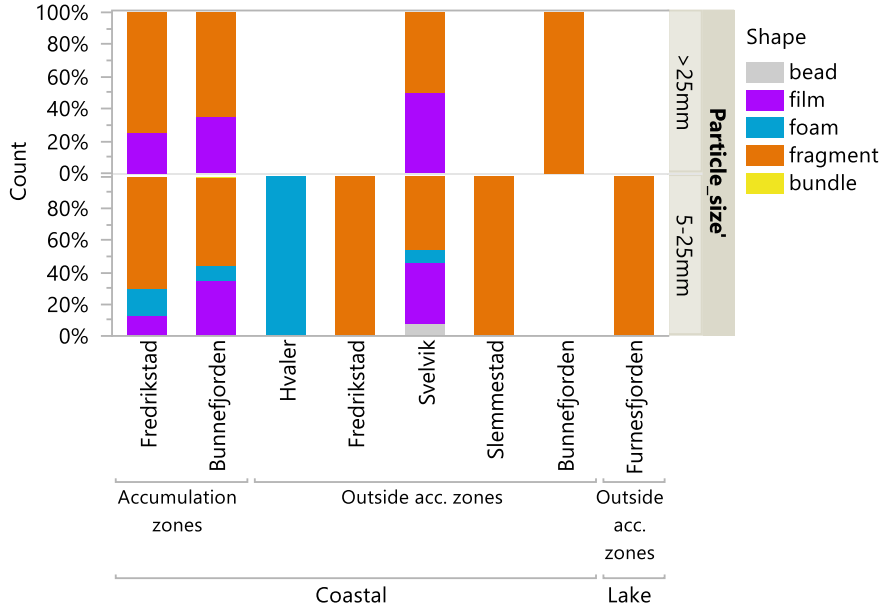


Figure 22. Percentage distribution of meso- and macroplastic shape categories observed in manta net water samples.

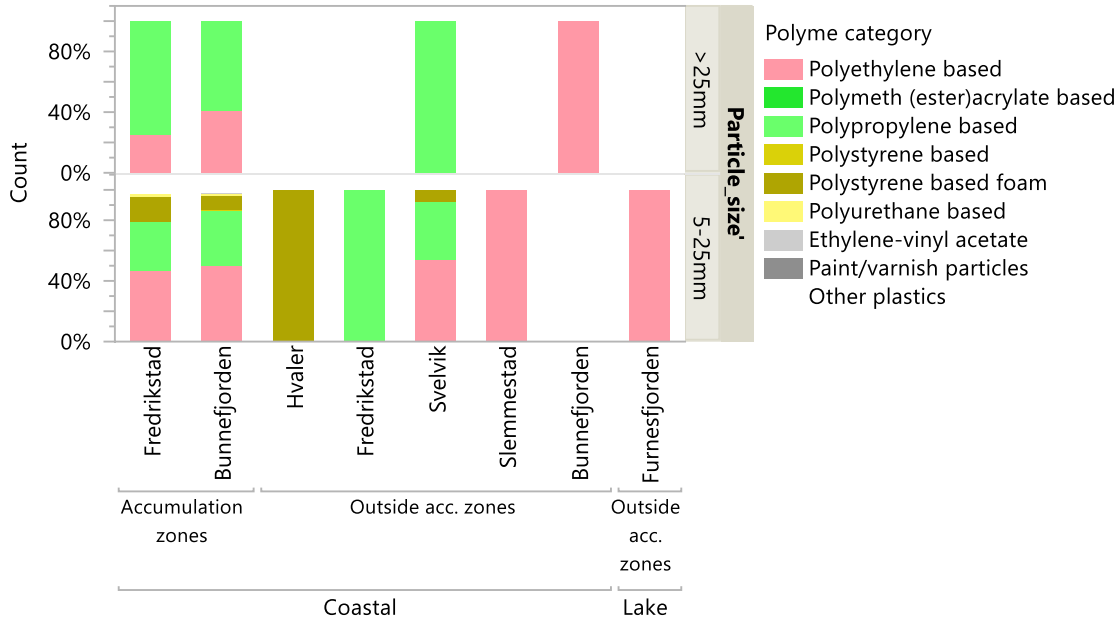


Figure 23. Percentage distribution of meso- and macroplastics polymer categories observed in manta net water samples.

Prior to this work, there is limited data on microplastics in surface water in Norway analyzed using nets (Lusher et al., 2021). Much of the work focuses on small research projects in Svalbard (Lusher et al., 2015, Pakhomova et al., 2024), historical data from Lake Mjøsa (NIVA), or river samples in MIKRONOR (2022 and 2023). The ongoing Marine Litter project (since 2010) in the Barents Sea (HI) through the BESS program introduced the Manta trawl as standard equipment from 2021, although data is not yet available for comparison. The data from MIKRONOR will be important for reporting surface water levels of MP internationally.

3.1 Atmospheric deposition

3.1.1. Summary of air samples

Atmospheric deposition samples were collected at three stations, Zeppelin (Svalbard), Birkenes (southern Norway) and Sofienbergparken (Oslo) year-long in 2024, with sampling periods covering 14 days each (n=22 per station), and analysed for MPs, UV stabilizers and tyre-related chemicals.

3.1.2. Zeppelin station, Svalbard

In 2024, microplastic deposition fluxes in the remote Zeppelin monitoring station averaged 34.5 ± 38.5 µg/m^2/d, with increasing fluxes towards the end of the sampling campaign (**Figure 24**). This was possibly influenced by changes in prevailing wind patterns and thus differing MP source regions (**Figure 65**). Generally, source regions can include continental as well as open ocean regions. The latter represents MP emitted from sea spray, while continental regions are a source of MP originating from road dust, agriculture or textiles, among others. While sea spray is presumed to be a major source for atmospheric MPs, only 1.8% of the MPs emitted by sea spray are estimated to be transferred to land (Evangeliou et al., 2022).

The mean MP deposition flux measured in 2024 is comparable to the average observed in 2023 in Zeppelin (21 ± 29 µg/m^2/d; Alling et al., 2023), though direct comparison is hampered by the limited temporal coverage of 2023 samples (end of August – mid-November in 2023 vs. beginning of February – beginning of December in 2024). The 2024 sampling campaign confirms the observation from the previous year that TWP can be detected in deposition samples from the Arctic Zeppelin monitoring station in Svalbard, likely originating from long-range atmospheric transport. The results further suggest that this represents sporadic events, with a frequency of occurrence (FO) of 9%.

The polymeric composition of samples from Zeppelin was dominated by PVC (63%), followed by PP (14%) and PMMA, PET and TWP (6% each). PE, PS, PA, PU, and PC contributed only marginally (≤ 3%) to the overall microplastic load.

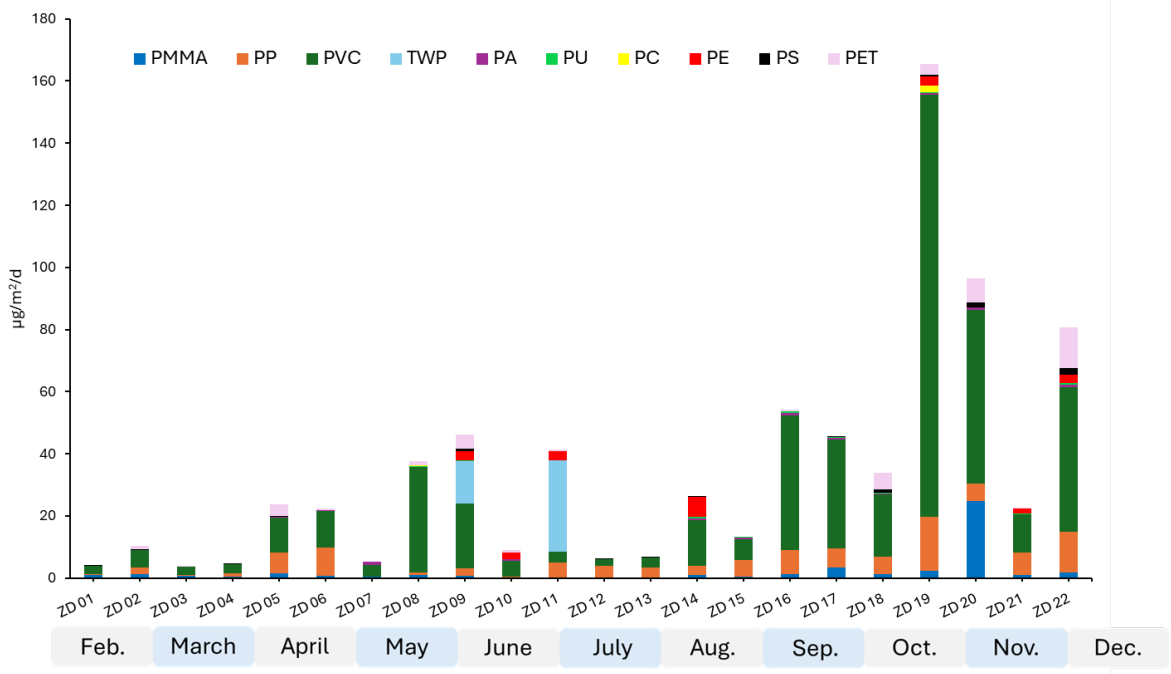


Figure 24. Microplastic fluxes (in µg/m^2/d) from deposition samples collected in Zeppelin station from 01.02.2024-04.12.2024. Sampling was conducted during 14-day periods each.

Concentrations of UV stabilizers in ethanol extracts from deposition samples from Zeppelin generally showed a similar temporal trend as MP deposition fluxes, with relatively low concentrations between ZD01-ZD15 (01.02.2024-28.08.2024) and generally higher concentrations in sampling periods ZD16-ZD22 (28.08.2024-04.12.2024). UV-328, a compound listed in the Stockholm Convention on Persistent Organic Pollutants (POPs), was detected in all samples from Zeppelin station, but this was at concentrations similar to those observed in the field blanks (0.04-0.13 ng/field blank from Zeppelin). A clear exception is sample ZD22, where a UV-328 concentration of 2.58 ng/sample was measured (**Figure 25**). A recent study suggested that modelled predictions of long-range atmospheric transport of particle-bound UV-328 are not supported by empirical evidence (Shunthirasingham et al., 2025). This conclusion might need to be revised if we consider direct transport via airborne MP particles.

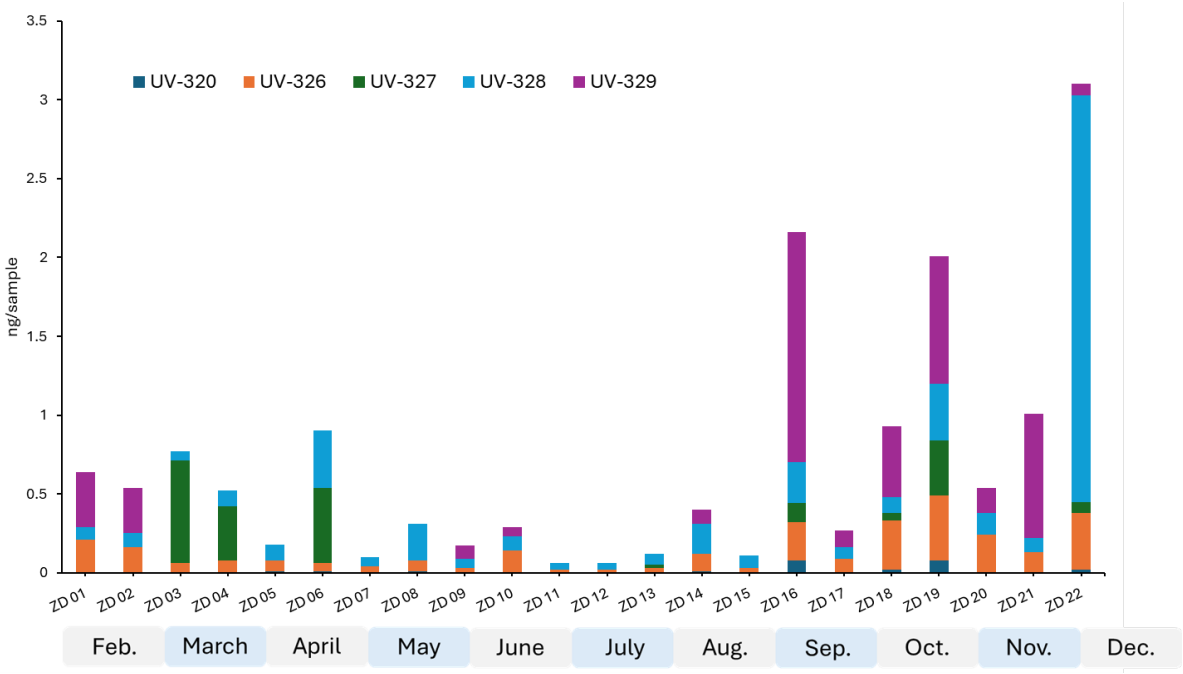


Figure 25. Concentrations (in ng/sample) of UV compounds in ethanol extracts of samples from Zeppelin station. Sampling was conducted during 14-day periods each.

3.1.3. Birkenes station, Southern Norway

In the semi-remote monitoring station Birkenes, MP deposition fluxes averaged 80.0 ± 41 µg/m^2/d in 2024 (**Figure 26**). A high variability between samples could be observed, without any clear temporal trends. The lowest flux was observed in April 2024 (BD07; 10.04.2024-24.04.2024) with only 4.34 µg/m^2/d (∑MPs) and no presence of TWP. This sampling period corresponded to sampled air masses mainly coming from the open ocean (**Figure 66**). In contrast, the sample exhibiting the highest MP deposition flux (BD10; 163 µg/m^2/d) with a 90% contribution of TWP corresponds to air masses originating predominantly from continental regions south of the monitoring station (**Figure 66**).

The polymer composition was dominated by PVC (44%), followed by TWP (40%). Minor contributors were PE (5%), PET (4%) and PC (4%). The pyrolyzate of PVC, naphthalene, is an unspecific marker as it can also origin from soot. Therefore, Py-GC/MS results attributed to PVC must be considered with care, especially in urban samples. As previously discussed, the presence of a PVC production plant located approximately 100 km from the monitoring station could also contribute as a local source to the observed PVC deposition fluxes in Birkenes (Alling et al., 2024).

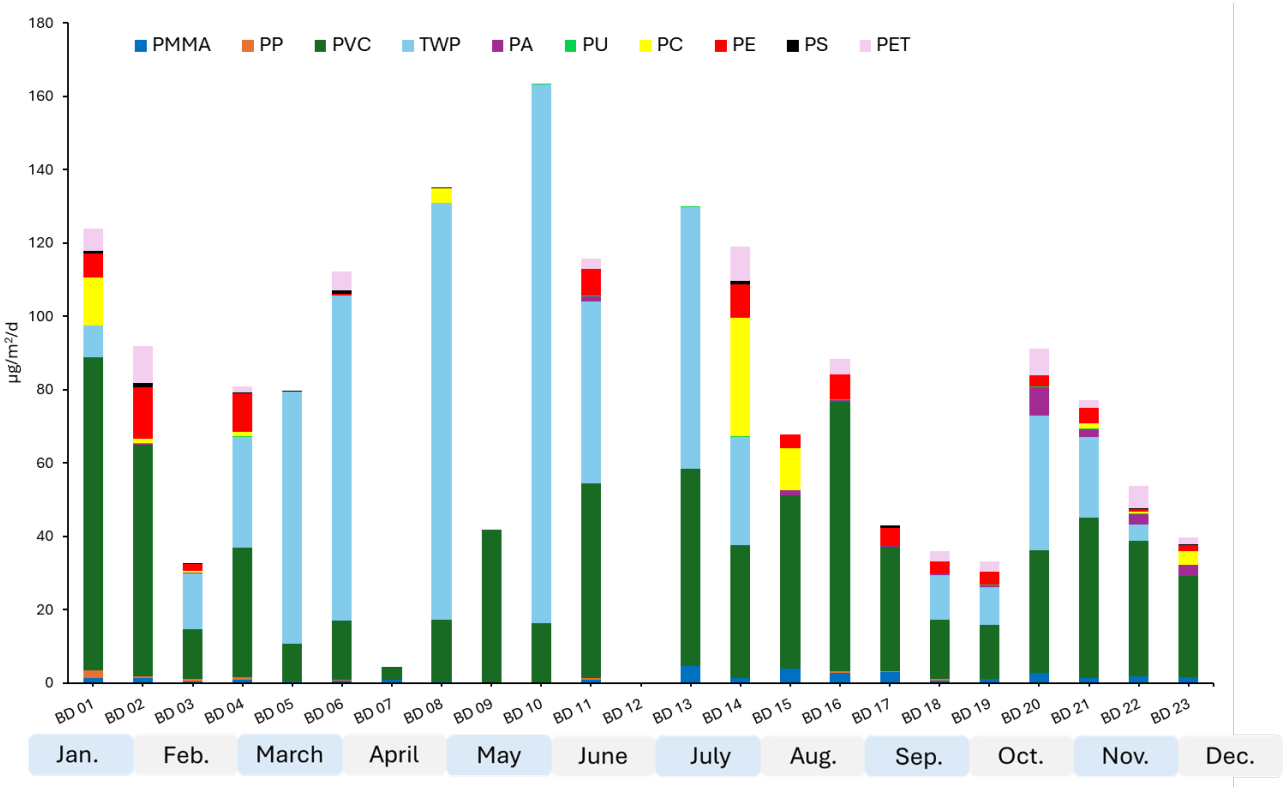


Figure 26. Microplastic fluxes (in µg/m²/d) from deposition samples collected in Birkenes station from 18.01.2024-04.12.2024. Sampling was conducted during 14-day periods each. Note that sample BD12 was lost due to instrumental issues.

The data on UV compounds present in the extracts from Birkenes station show some similar trends as the MP deposition fluxes (**Figure 27**). Indeed, when considering the MP and UV compound data from observations BD01-BD10 using a linear model, the coefficient of determination indicates a good fit (R^2-value = 0.73). Accordingly, the samples from this period showing the highest MP fluxes (BD10 = 163 µg/m²/d; BD08 = 135 µg/m²/d; BD01 = 124 µg/m²/d) also exhibited highest UV compound concentrations (0.68; 0.48 and 0.42 ng/sample, respectively). For BD11-BD23 however, no clear correlation between the MP and UV compound data can be observed. Benzotriazole UV stabilizers are used in a variety of polymers to prevent photodegradation and hence prolong the lifespan of plastic items. They are added during the production phase at a concentration of 0.05–3% by weight of the polymer (Hahladakis et al., 2018). Weathered plastics further contain significantly lower amounts of UV compounds than their new counterparts (Rani et al., 2017), due to leaching or degradation/transformation processes during use and disposal. The range of concentrations at which UV stabilizers are added to plastics as well as the unknown degree of weathering of airborne MPs suggest that direct correlation between MP fluxes and UV compound concentrations are not always evident.

UV-326 was present at a particularly high concentration (3.88 ng/sample) in sample BD13. Field blank values for this compound were generally low (0.05-0.07 ng/field blank from Birkenes), dismissing sample contamination as a likely reason.

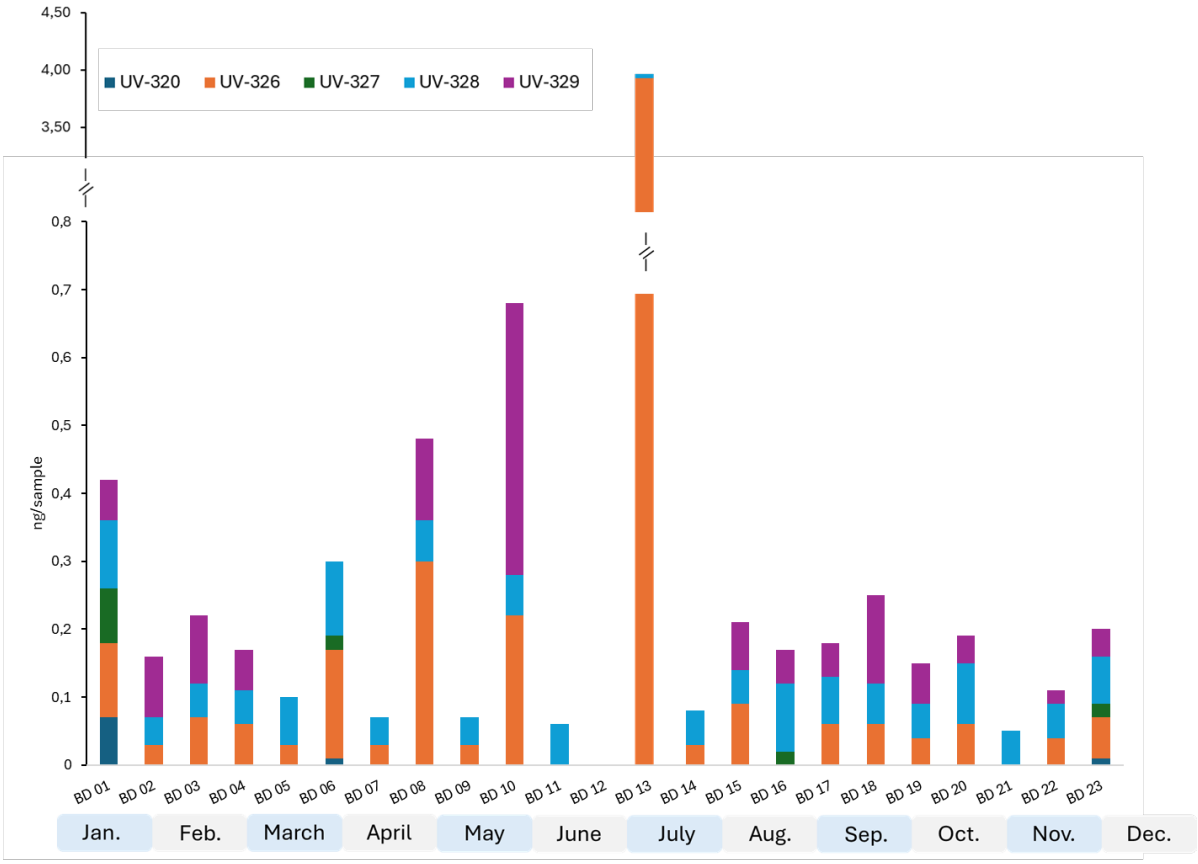


Figure 27. Concentrations (in ng/sample) of UV compounds in ethanol extracts of samples from Birkenes station. Sampling was conducted during 14-day periods each. Note that the y-axis is interrupted for better visibility of the results. Note that sample BD12 was lost due to instrumental issues.

3.1.4. Sofienbergparken station, Oslo

In the urban air monitoring station Sofienbergparken, located in Oslo, an average MP deposition flux of 599 (± 453) µg/m2/d was measured. TWP generally dominated the polymer composition, with 68% relative abundance on average (2-83% relative abundance per sample). Contribution of TWP to MP deposition fluxes was particularly low during the first three sampling periods (SD01-SD03, covering a total sampling period from 17.01.2024-28.02.2024), which correlates with average daily temperatures below or around 0 degrees, causing an ice-or snowcover of the road (**Figure 28**). It previously has been observed that when precipitation mainly occurs as snow, airborne TWP concentrations tend to be low (Herzke et al., submitted). A possible explanation could be that the snow cover temporarily 'binds' the TWP to the ground, avoiding their (re-) suspension into the air, before they get removed from the road via snow removal. Furthermore, snowy streets are likely to reduce the friction between tires and the road, thus limiting the production of TWP.

On the other hand, particularly high TWP deposition fluxes were observed during the last two sampling periods (SD21 and SD22, covering a total sampling period from 23.10.2024-20.11.2024), when temperatures were still above the freezing point and precipitation was low (**Figures 28-29**). This sampling period further coincides with the switch from summer tires to winter tires, which are known to generate a significantly higher amount of TWP than summer tires (Johannessen et al., 2022 and references therein).

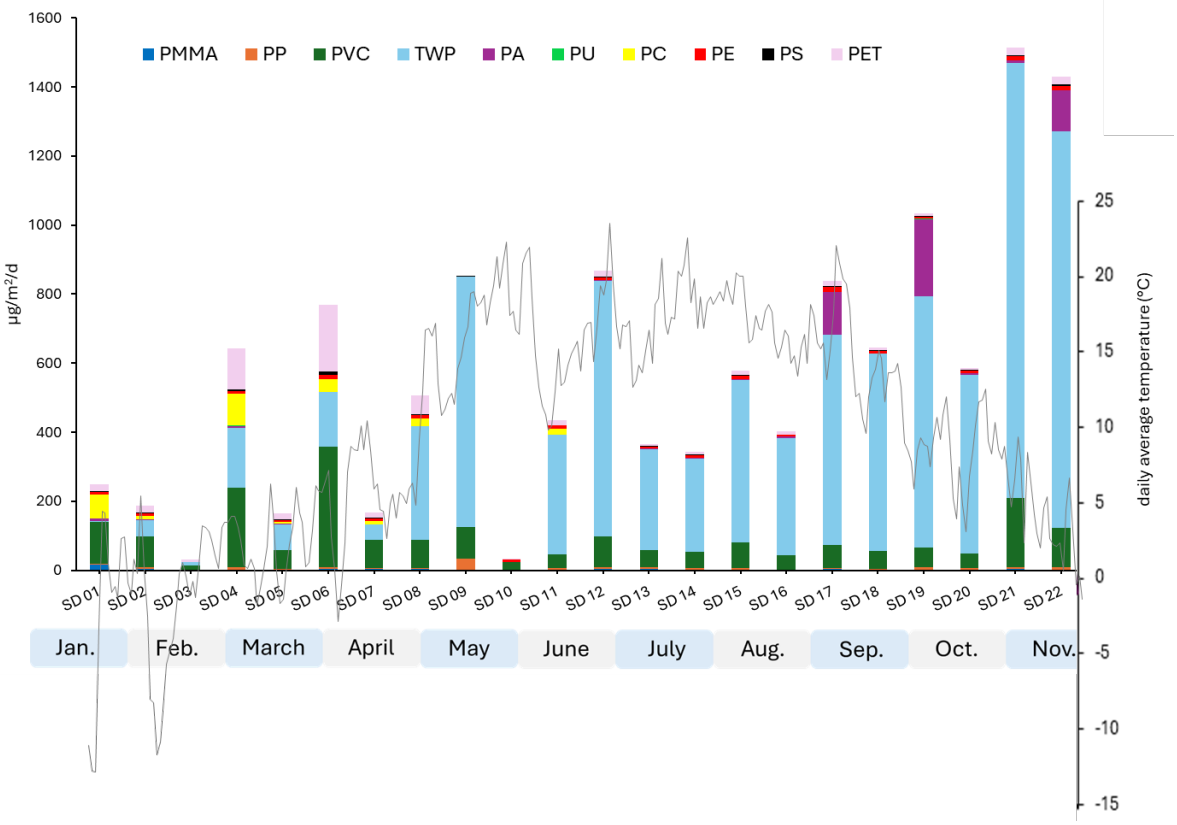


Figure 28. Microplastic deposition fluxes (in µg/m^2/d) and daily average temperature (in °C) at Sofienbergparken station during the sampling period 17.01.2024-20.11.2024. Sampling was conducted during 14-day periods each. Temperature data was sourced from the Meteorologisk institutt (MET) for the measurement station Oslo-Blindern (SN18700).

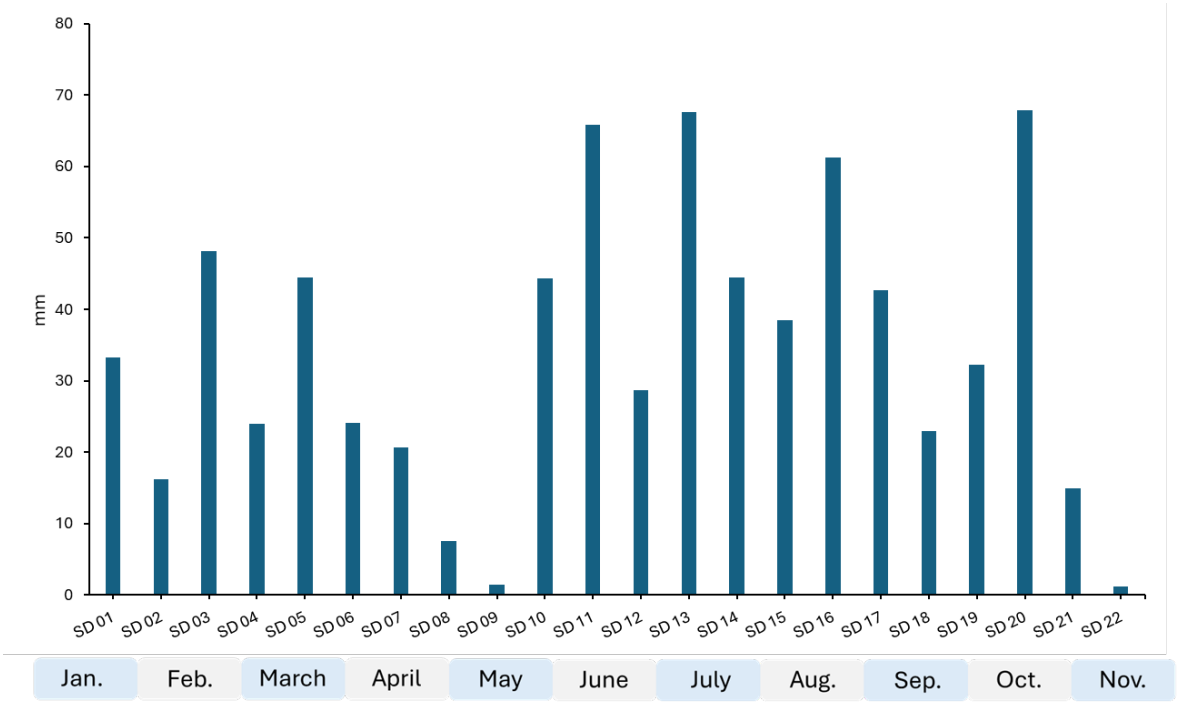


Figure 29. Accumulated precipitation (in mm) during each sampling period. Precipitation data was sourced from the Meteorologisk institutt (MET) for the measurement station Oslo-Elvebakken (SN18317).

Sample SD10 exhibited a remarkably low MP deposition flux of 32.8 µg/m^2/d and is the only sample collected in Sofienbergparken in which no TWP were detected. These observations coincided with a high load of organic matter on the filter (visual observation), which led to a first assumption that the organic matter could have resulted in matrix interferences, possibly suppressing signals during the Py-GC/MS analysis. However, this sample also exhibited a notably low concentration of UV compounds (0.19 ng/sample, **Figure 30**), suggesting that this hypothesis should be rejected. While it remains challenging to elucidate the reasons for this observation, there are some aspects that might be considered. For example, the sampling period (22.05.2024-05.06.2024) constitutes the sample taken directly after the Norwegian national day (17th of May), for which the streets of the cities are traditionally cleaned, which could have led to a removal of MPs from the streets surrounding the monitoring station. It is further noteworthy that the sampling period just before (i.e., SD09) was characterized by particularly low precipitation (**Figure 29**), though it is unclear how this might be linked to the MP fluxes observed in SD10.

In contrast to samples from Zeppelin and Birkenes, UV-329 was detected in every sample collected in Sofienbergparken and dominated the relative abundance (average 45%), followed by UV-326 (33%). Such differences in detection frequency and relative abundance of UV stabilizers between monitoring stations might be explained by specific uses (e.g. in tires, consumer products) or differing degradation times and thus a differing potential for long-range transport of individual UV compounds. Information on degradation times is scarce, however, and mostly limited to aqueous media, highlighting the need for further research specifically regarding the atmospheric compartment. As for MP deposition fluxes, UV stabilizer concentrations measured in samples from Sofienbergparken were generally higher (mean 1.8 ng/sample) than in Zeppelin (mean 0.69 ng/sample) and Birkenes (mean 0.37 ng/sample).

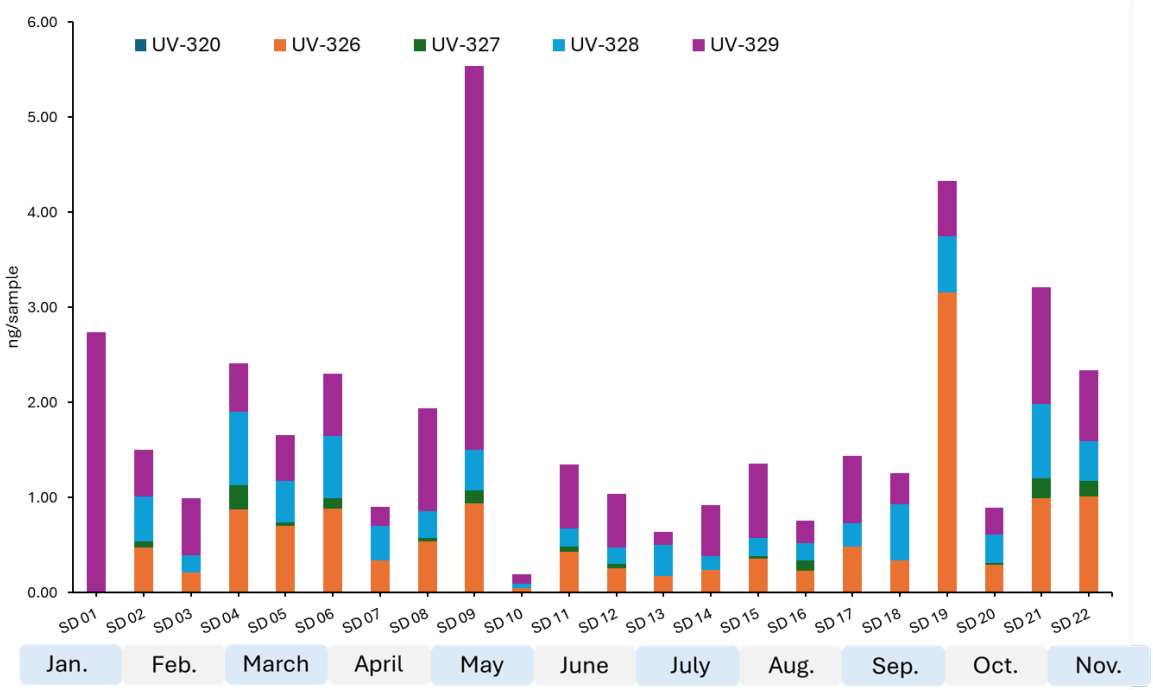


Figure 30. Concentrations (in ng/sample) of UV compounds in ethanol extracts of samples from Sofienbergparken. Sampling was conducted during 14-day periods each.

3.1.5. Reflections on remote vs. urban monitoring stations

The screening for and monitoring of contaminants in samples from remote sites has been proven to be of utmost importance to understand global distribution patterns, evaluate the potential for long-range transport and follow time-trends of regulated and unregulated substances. Urban monitoring stations on the other hand are highly influenced by local contaminant inputs and fluctuating wind patterns, and can thus only provide limited information regarding general patterns or transport pathways. However, they

are very important to evaluate local emission sources that contribute to long-range-transported depositions, while also serving as indicators for human exposure to contaminants and identify hotspot areas. This MIKRONOR report is the first to include an urban air monitoring station covering all seasons, namely Sofienbergparken in Oslo. As expected, the results from this station differ significantly from Zeppelin and Birkenes station with regards to detected MP mass fluxes and polymer composition. Our results suggest that in urban environments, MP deposition fluxes might be highly influenced by local sources (especially road traffic), environmental parameters (e.g. temperature, type and amount of precipitation) and human behavior/regulations (switch between summer and winter tires). They contribute to long range transported fluxes to the Arctic of not only the MPs but also plastic related chemicals. The inclusion of a remote, semi-remote and urban site in this year's MIKRONOR sampling campaign thus allowed to provide a comprehensive overview of the occurrence, composition, temporal trends and sources of atmospheric microplastics in Norway.

3.2 Beaches

Microplastics were assessed on the OSPAR beach on Akerøya (Akerøya-A) and a reference beach (Akerøya-B) in 2024. Samples were collected using a similar methodology to the 2023 study (Alling et al., 2023), employing a partially stratified and randomized sampling design to investigate the potential influence of position on the shoreline. Five transects were laid from the last high tide mark to the water line. Six samples were collected per transect, totaling 30 samples for each beach. The position of each sample was determined by random number generator, both for position on the transect and position within the quadrat. For full field protocol, refer to Appendix 5.2 of the 2024 report (Alling et al., 2024).

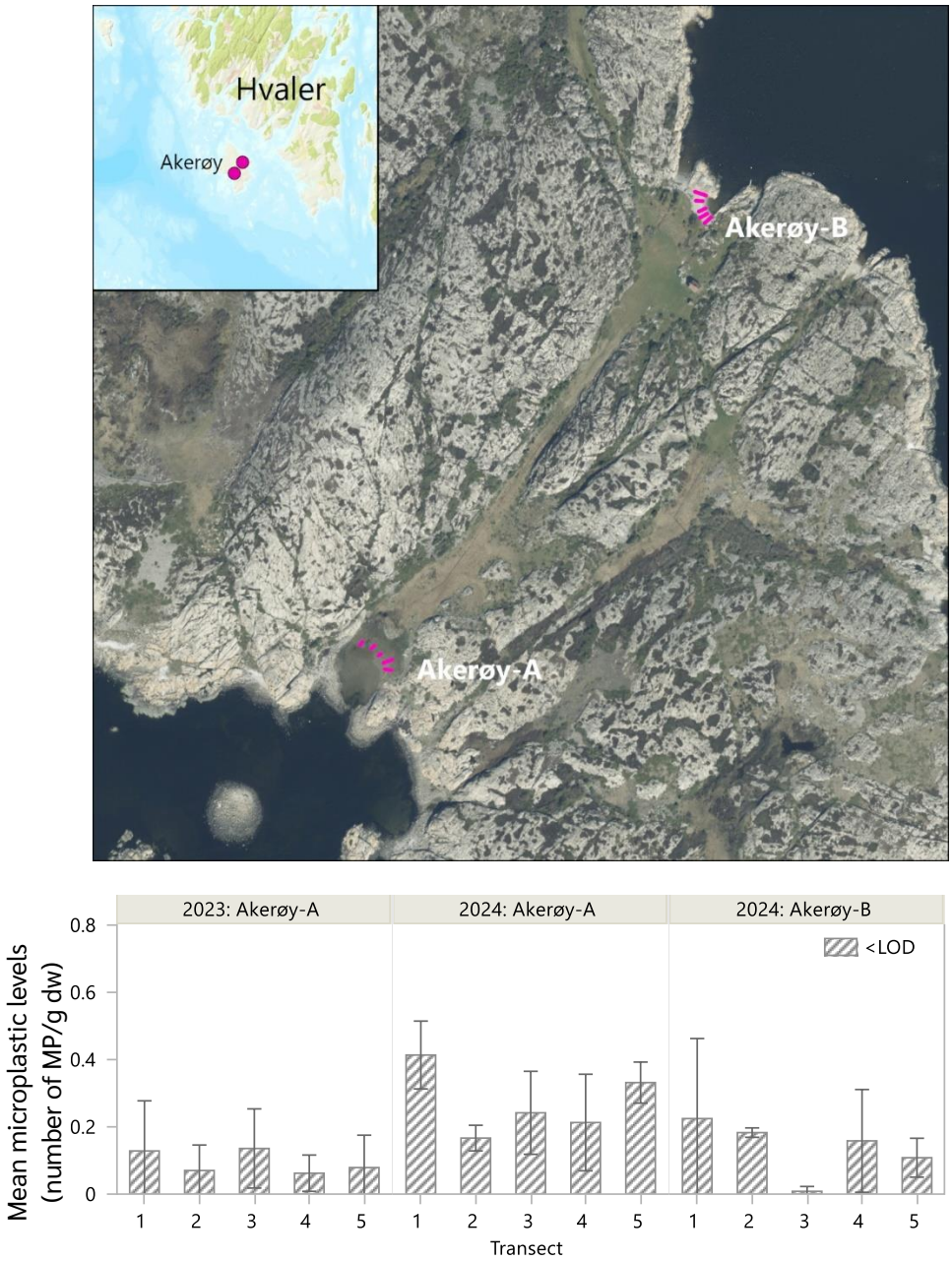


Figure 31. Upper: Map of the two beaches sampled at Akerøy. **Lower:** Microplastic levels (MP/g dw) in beach sediment samples per transect at Akerøya-A (2023+2024, left) and Akerøya-B (2024, right) .
© Kartverket

Microplastic levels were assessed at two beaches on Akerøya in 2024: the OSPAR beach (Akerøya-A) and a reference beach (Akerøya-B). Thirty sediment samples were collected from each beach using a stratified random sampling design to cover positions from the high tide mark to the waterline.

Results showed that all samples from both beaches were below the LOD (**Figure 31**). This was consistent with findings from 2023 where only one sample was above LOD but still below the LOQ. No significant differences were observed between the two beaches, between transects, or between positions along the transects. Both particle counts and mass concentrations were extremely low, and the polymer types detected—primarily polypropylene and polyethylene—were also present in the laboratory blanks, suggesting that measured values could reflect background contamination that cannot be separated from true environmental abundance.

The dominance of smaller particles (50–200 µm) in the samples matches the pattern seen in 2023, though concentrations were again below LOD in most samples (**Figure 32**). Even though below LOQ/LOD, our beach sediment samples in the MIKRONOR campaign indicate MP levels between 0.1 and 0.4 MP/g dw, comparable to the findings from Kiel Fjord, where Schröder et al. (2021) reported up to ~30 MP/kg dry sediment (≈ 0.03 MP/g dw) in beach drift line sediments. In future studiesof beach sediments a higher amount of sediments should be considered to be prosessed.

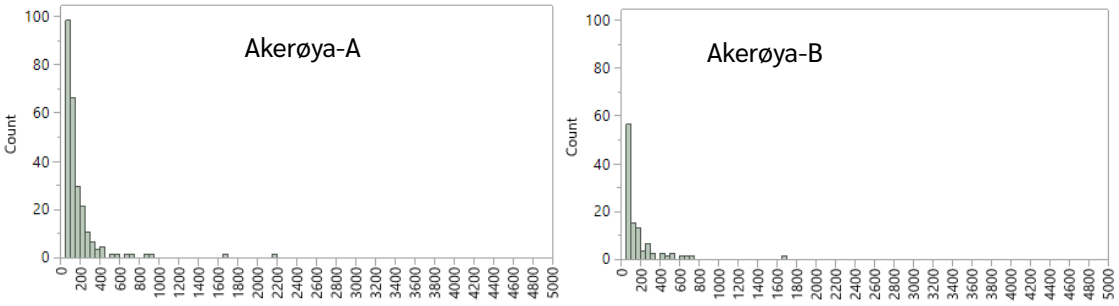


Figure 32. Count of microplastic particles per size class (µm) for Akerøya-A (left) and Akerøya-B (right) in 2024.

Both beaches consistently showed a dominance of smaller microplastic particles, primarily within the 50-200 µm range (**Figure 32**). This pattern is similar to the 2023 findings, where smaller microplastics (50-300 µm) also dominated. The method used has a lower size limit of 50 µm.

3.3 Coastal and open sea water samples taken with the FerryBox system

In 2024, microplastics were sampled during one Hurtigruten cruise between Bergen and Ny-Ålesund (Svalbard) June 2.-10. The FerryBox system enables large volume samples, and is consisting of a filter system with a 100 µm mesh size, deployed at a depth of approximately 4-5 m (Sensors/Samplers - Nautilos (nautilos-h2020.eu).

During the 2024 cruise, microplastic levels ranged from 0.01 MP/m^3 to 1.2 MP/m^3 (**Figure 33**), all values above LOD. The open waters between Svalbard and Bjørnøya -two transects, recorded the lowest levels. Only a few of the samples collected across three cruises showed elevated microplastic levels, indicating

that there is little microplastic over 100 µm in these marine areas. Highest levels (number of MP particles) were found in the two transects Tromsø- Honningsvåg – Bjørnøya.

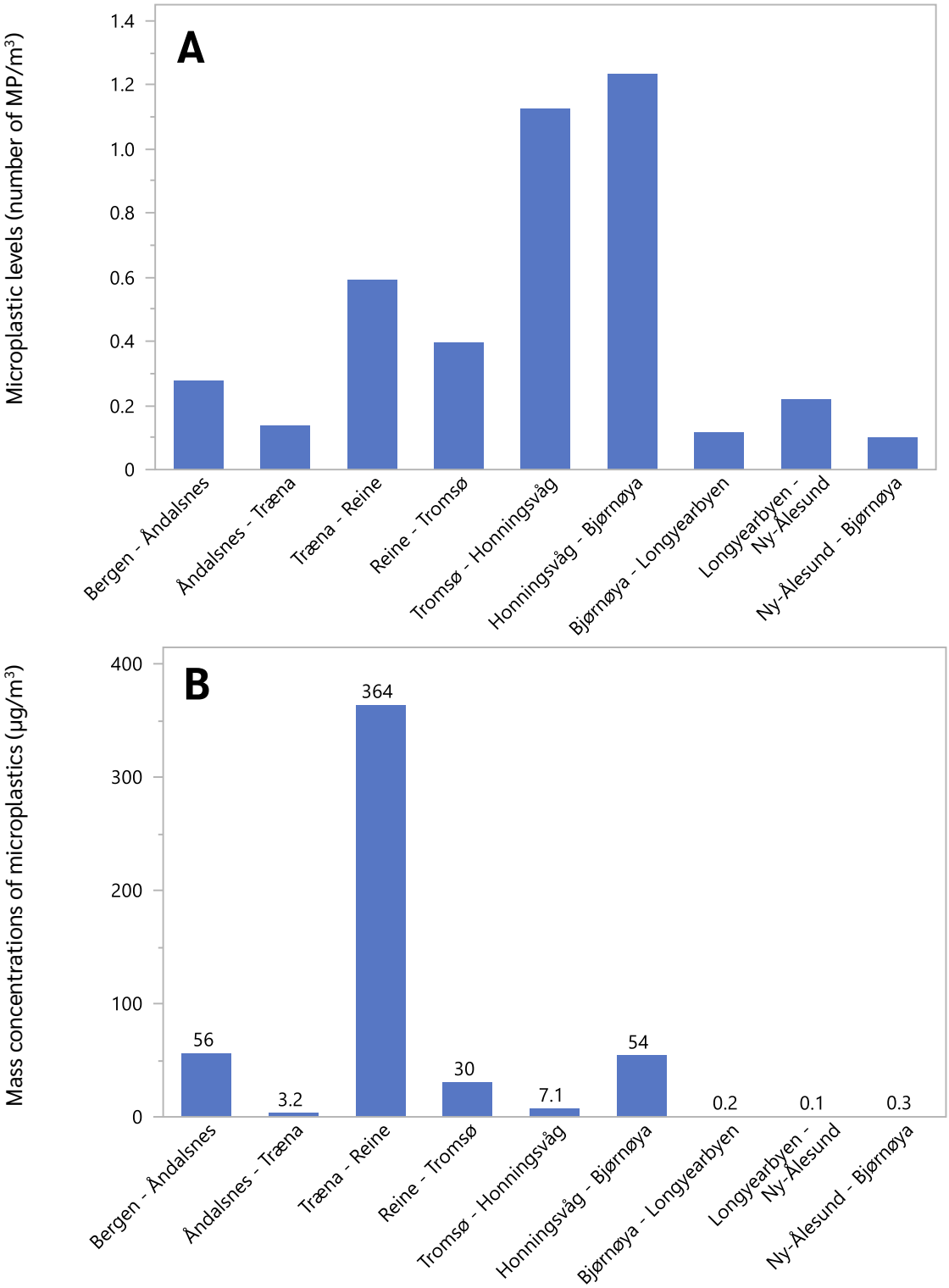


Figure 33. A: Microplastic levels (MP/m³) in FerryBox water samples per transect in 2024, particle size >100 µm and one sample per transect. All values above LOQ. **B:** Weight of microplastics (µg/m³) in FerryBox water samples per transect in 2024.

The mass concentrations reflected rather the sizes of the particles found in the samples, than the number of particles. Therefore, there were no correlations between the transects having the highest number of

plastics, and the transects exhibiting the highest mass concentrations. The mass concentrations range from <0.1 µg/m^3 to 56 µg/m^3, except one transect that had a very high mass concentration of 364 µg/m^3 due to a few large particles in that sample (**Figure** 34, particles above 4 mm). The levels of MP in the Ferrybox samples from the cruise 2024 correlated to the results from the three cruises in 2023, with levels generally well below 1 MP/m3 in most transects. This confirms the trends observed in previous MIKRONOR programs, and this year, all data from the FerryBox sampling have been above LOQ. A recent publication reported large amounts of nanoplastics in the North Atlantic claiming the combined mass of nano plastic exceeded the total estimates of both macro and micro plastics together (ten Hietbrink et al 2025). Although this publication received much attention these estimates were based on questionable analysis using a experimental technology without QA/QC.

The size distribution of microplastic particles (**Figure 34**) revealed that the majority were within the 300–1500 µm size range. A similar pattern was observed in the FerryBox dataset from 2023 (Alling et al., 2024). This contrasts with findings from most other sample matrices, where particle numbers typically increase with decreasing size. Whether this pattern reflects a true size distribution in surface waters of the open sea or is influenced by sampling biases remains an open question. Future sampling campaigns using the FerryBox system should aim to address this uncertainty.

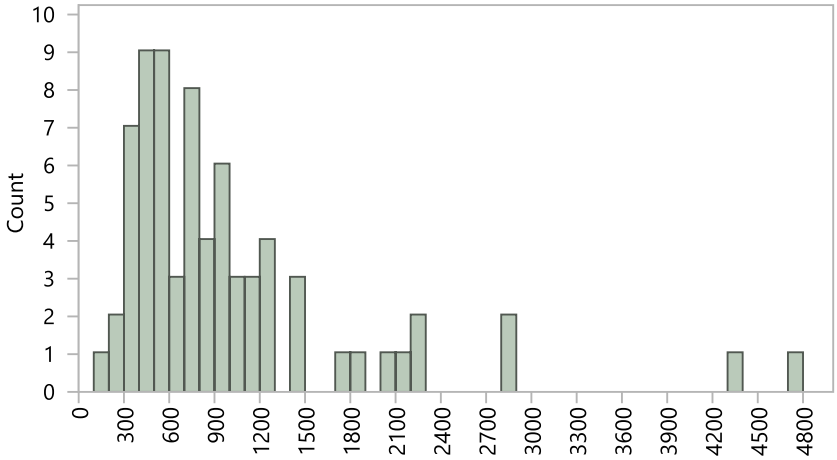


Figure 34. Distribution of microplastic particle sizes (µm) in water samples on FerryBox in 2024.

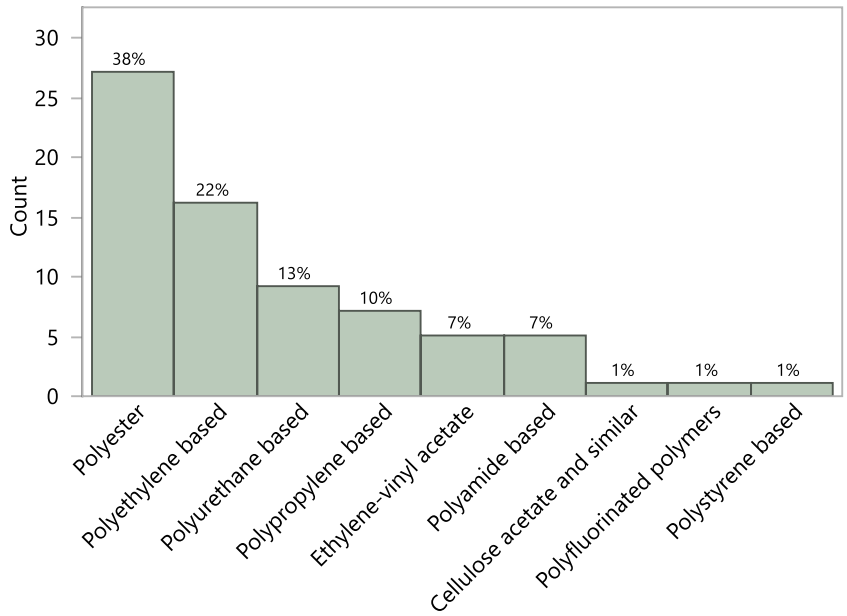
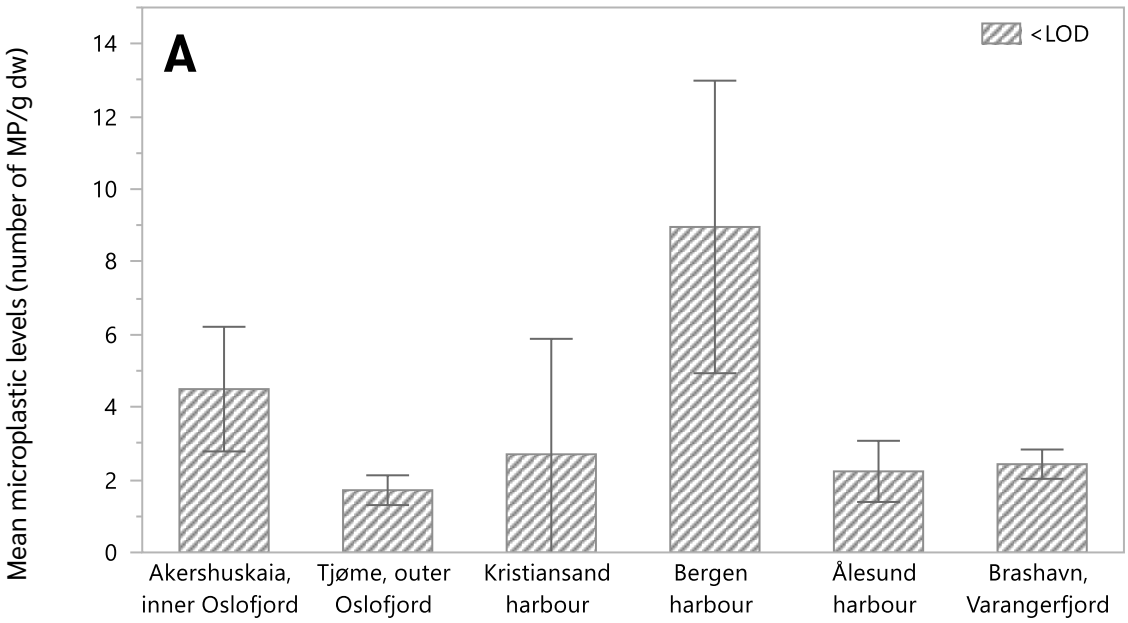


Figure 35. Distribution of polymer categories of microplastic particles in water samples on FerryBox in 2024.

The polymer composition of microplastic particles in the FerryBox samples was dominated by the common polymers polyester and polyethylene (**Figure 35**). In contrast, samples from 2023 showed greater variability in dominant polymer types across different cruises. Although the results from the current dataset were above LOQ, the number of particles on which the polymer distribution is based was relatively small. Therefore, firm conclusions regarding polymer composition should be drawn with caution.

3.4 Blue mussels

Blue mussels were collected from six stations as in 2022 and 2023 (Akerhuskaia Oslo, Tjøme, Kristiansand harbour, Bergen harbour, Ålesund harbour, Varangerfjorden -different station in Varangerfjorden compared to 2023). Three replicate samples, containing 6-10 individuals mussels were analyzed for both microplastics and TWP on the same sample.



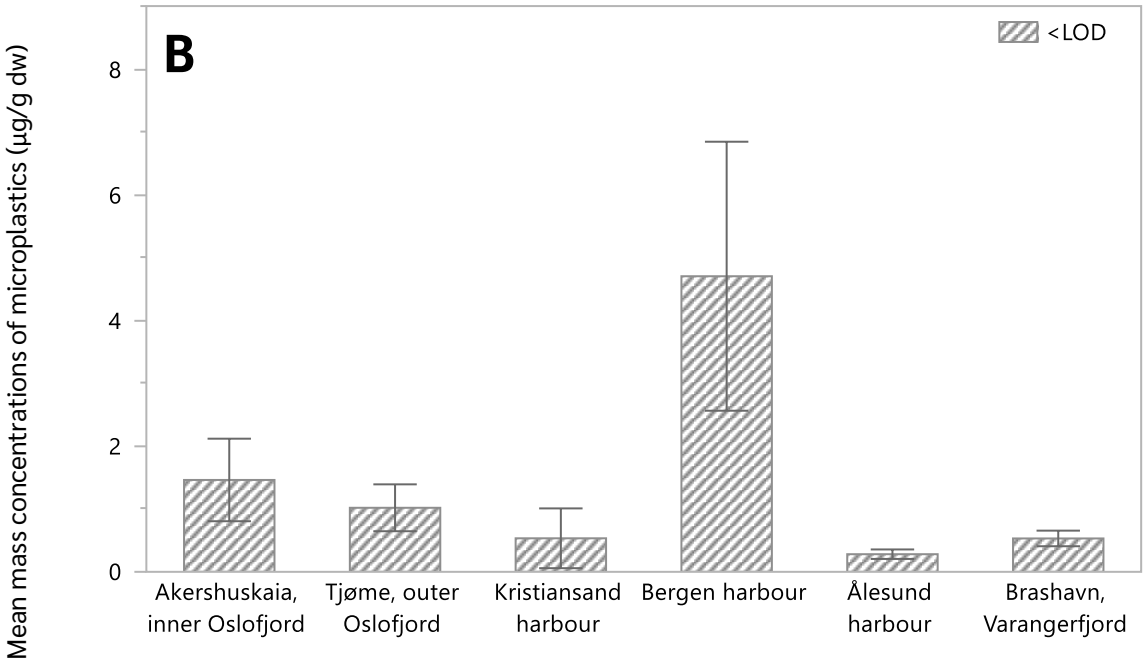


Figure 36. A: Mean microplastic levels (MP/g dw) in blue mussel stations in 2024. Error bars = SD. Number of samples = 3 per station. **B:** Mean weight of microplastics (µg/g dw) in blue mussel stations in 2024. Error bars = SD. Number of samples = 3 per station.

The levels of microplastics were below the Limit of Quantification (LOQ) for all samples, with only a few replicates having values above the Limit of Detection (LOD). This was also the case for the 2022 and 2023 samples, as reported in Alling et al., 2023 and 2024. LOD and LOQ were determined as MP/sample, as described in appendix 5.1. QA/QC.

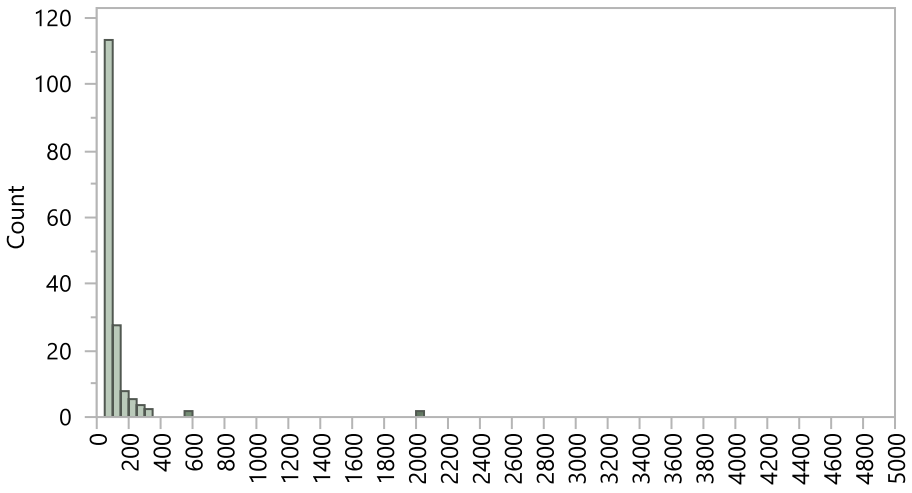


Figure 37. Distribution of microplastic particle sizes (µm) in all blue mussel samples in 2024.

The distribution of microplastic particle sizes is shown in **Figure 37**. Across all 18 samples, a total of approximately 150 particles were detected. More than 95% of these particles were within the smaller size fraction (<300 µm). The most frequently identified polymer types were polyamide and polypropylene. However, given that the microplastic concentrations in the samples were below the limit of detection

(LOD), and these same polymers were also commonly found in the laboratory blanks, the results should be interpreted with caution.

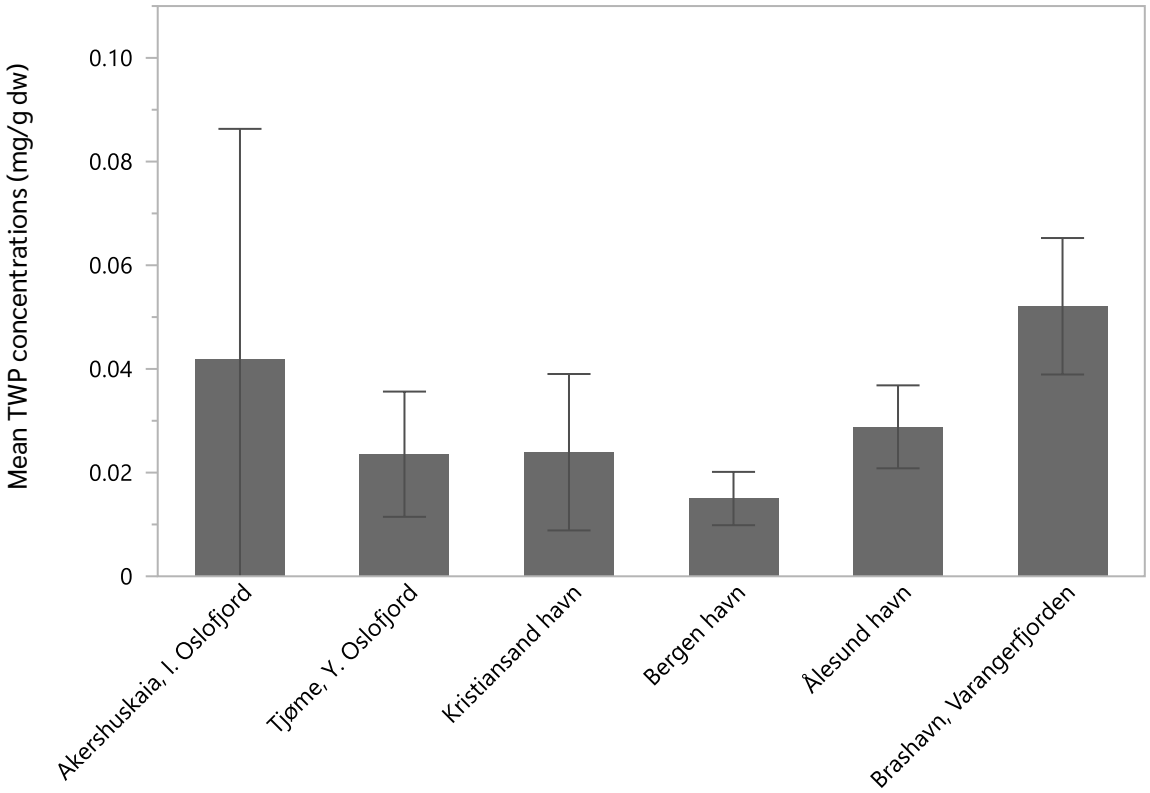


Figure 38. Concentrations (mg/g d.w.) of tyre wear particles (TWP) in the 50–300 µm size fraction in blue mussels. n = 3 replicates per station; each replicate consists of a pooled sample of 7–10 individual mussels.

Although the concentration levels of TWP observed in blue mussels are quite different between the sampling years, similar trends are observed for blue mussels collected in 2022 and 2024, with the highest levels close to the urban harbours in Oslo and Ålesund, followed by the Varangerfjord, Kristiansand and lower values in mussels from Tjøme and Bergen. The levels in 2024 ranged from 0.010 to 0.090 mg/g dw (**Figure 38**), whereas the TWP levels in 2022 were significantly higher (0.74- 58 mg/g dw, Alling et al., 2023). On the other hand, in 2023 TWP was only detected in two out of three replicate samples from Kristiansand harbour (Alling et al., 2024), and these samples had the highest TWP levels reported for blue mussel in Norway so far (180 mg/g dw). This shows that blue mussels have highly variable TWP levels between years, even from the same locations. This could be related to the size and age of the blue mussels, suggesting accumulation of TWP in blue mussels over time. The linear relationship between sample size and TWP was tested for the 2024 dataset, plotting the total dry weight of each sample (7-10 individuals) against the TWP concentration in each sample (**Figure 39**). Although the variations are small between samples, the results indicate that there might be linear relationship between the size of the blue mussels and the levels of TWP detected in the samples (R^2 adj 0.426, p = 0.0006).

Retention of TWP in mussels, including blue mussels, have been demonstrated in lab studies (Michelangeli et al., in prep and Weinstein et al., 2022). Exposure to tyre-derived compounds (TDCs) leaching to the water has also been demonstrated to reduce filtration rates of blue mussels (Thomsen et al., 2024), thus reducing the potential particle uptake of blue mussels in areas with higher levels of tyre-derived chemicals in the water. Although previous studies have reported the presence of tyre-derived compounds in discharge water from tunnels, wastewater treatment plants and rivers in Oslo (Gundersen et al., 2024), which is released into the Oslofjord, these compounds were not measured in this monitoring program and

therefore the direct correlation between TDCs and TWP uptake in blue mussels cannot be tested. The variations of TWP in blue mussels may also reflect variations of TWP levels in the water column, directly related to stormwater discharge from urban areas. Such comparisons were done in Alling et al. 2024, showing TWP levels measured in water in the Oslofjord did not follow the same pattern as MP levels, suggesting that MP are more closely linked to heavy rainfalls and river discharge during precipitation compared to TWP. It is also important to note that TWP has numerous potential discharge points in the urban areas to the fjord, including a large number of drainage pipes and stormwater traps, road tunnel discharges (Rosnes-Lundgaard) and diffuse drainage from the urban areas into the fjord, which may or may not be related to precipitation. It has also in recent years been increasingly challenging to find enough blue mussels for monitoring studies, across all of Norway, which both impacts the number of replicates, the age of the blue mussels and how representative these blue mussels are as indicator species. This further demonstrates the challenge of using blue mussels for microplastic monitoring, and environmental monitoring in general.

Figure 39. Correlation plot between the total concentration of TWP in each blue mussel sample and the total weight (dw) of blue mussels in a sample. Each blue mussel sample consists of 7-10 blue mussels, pooled and treated as one sample.

4 Microplastics and Tyre Wear Particles in the Norwegian Environment – Findings, Implications, and Global Context

The 2024 MIKRONOR campaign provides a comprehensive assessment of microplastic (MP) and tyre wear particle (TWP) contamination across multiple environmental compartments in Norway. The findings reaffirm that urbanized areas—particularly Oslo—serve as major sources of both MPs and TWPs, which are disseminated via stormwater runoff, atmospheric deposition, and subsequent accumulation in sediments and biota.

Stormwater traps in Oslo exhibited some of the highest TWP concentrations recorded during the campaign, with sediment levels reaching up to **236 mg/g**, indicating that these structures function as significant sinks for traffic-derived particulate pollution. Despite their retention capacity, runoff water continues to transport substantial MP loads downstream, posing a risk to ecologically sensitive recipients such as the Oslofjord. These results underscore the need for improved stormwater management and targeted mitigation strategies to reduce MP and TWP fluxes from urban environments.

Globally, the Norwegian results from MIKRONOR 2024 align with findings from other countries. For TWP in particular, there are currently limited number of studies using comparable analytical techniques, which limits the comparison between the Norwegian environment and other parts of the world. There are, however, comparable studies of sediments and runoff in Sweden, Japan and Australia. The TWP concentrations in Oslo runoff (0.1–0.8 mg/L) are found to be comparable to those in Luleå and Sundsvall, Sweden, and to peak values in Brisbane, Australia. However, marine sediment concentrations in the Oslofjord were notably higher than those reported in Osaka Bay, Japan, where median TWP levels were around 0.2 mg/g. This suggests that Norwegian urban and coastal environments may be more heavily impacted by TWPs than some international counterparts, possibly due to local traffic patterns, climate, or road maintenance practices. More comparable studies between different parts of the world are highly needed.

A global meta-analysis of microplastic pollution in reservoirs found that small-sized MPs (<1 mm) accounted for over 60% of total particles, with polypropylene and polyester being the most common polymers (Guo et al., 2021). This is consistent with MIKRONOR findings, which also reported a dominance of small particles and similar polymer types in both freshwater and marine samples (see this study and Alling et al 2023). However, MIKRONOR noted lower concentrations in freshwater systems like Mjøsa compared to urban runoff and fjord sediments, indicating stronger urban influence in Norway than in some global reservoir studies.

In European aquatic environments, MP concentrations were found to correlate more with stormwater and atmospheric deposition than with proximity to wastewater treatment plants or population density. This supports the MIKRONOR conclusion that urban runoff and atmospheric deposition are key pathways for MP and TWP transport in Norway. Marine ecosystems globally show MPs from surface waters to deep-sea sediments, influenced by hydrodynamics, land use, and seasonal factors (Rakib et al., 2023). MIKRONOR similarly found high MP and TWP levels in inner Oslofjord sediments, especially near urbanized zones, and even in remote areas, suggesting long-range transport and deposition.

MP ingestion by aquatic organisms is a well-documented global phenomenon, with impacts ranging from physical harm to trophic transfer (Rakib et al., 2023). MIKRONOR confirmed high TWP concentrations in

blue mussels from the inner Oslofjord, indicating bioaccumulation and potential ecological effects similar to those observed internationally.

Atmospheric deposition is increasingly recognized as a major MP source. Global models suggest oceans contribute only ~0.008% of airborne MPs, with land-based sources like roads and agriculture dominating (Yang et al., 2025). MIKRONOR's air sampling at Birkenes and Zeppelin stations supports this, showing higher MP deposition near urban areas and detectable levels even in remote Arctic-like environments.

It is also important to highloight that MPs and TWPs are not merely inert particles; they can serve as significant vectors for the transport of harmful chemical additives, such as UV stabilizers, which are incorporated into plastics during manufacturing. This direct transport of additives poses a critical concern, as these chemicals can be highly detrimental to ecosystems and human health. MIKRONOR has highlighted similar concerns:

Regarding specific chemical additives, UV-328, a compound listed in the Stockholm Convention on Persistent Organic Pollutants (POPs), was detected in a sample from the remote Zeppelin station (ZD22) at a concentration of 2.58 ng/sample. This challenges recent suggestions that modelled predictions of long-range atmospheric transport of particle-bound UV-328 are not supported by empirical evidence. Our results, particularly the elevated concentration in sample ZD22, suggest that direct transport via airborne MP particles, carrying their inherent additives, is indeed a relevant pathway for the distribution of such harmful substances, even to remote areas.

Finally, the ecological implications of microplastic and tyre wear particle contamination are still not fully understood. For TWP, a recent risk assessment study has summarized the current knowledge of ecological risk associated with TWP (Müller et al., 2025) and found that there are major knowledge gaps in the current literature on TWP and leachate concentrations and the toxicity of these in terrestrial and aquatic environment, and for those studies that are available, comparison is challenging due to lack of standardization. Least knowledge is found for the marine environment, both in terms of TWP concentrations in water and sediment, and toxicity studies on marine organisms. This highlights the need to agree on standard protocols for sampling, analysis and toxicity testing related to TWP and tire leachates, as well as the need for relevant environmental concentrations to set realistic exposure scenarios. The findings of Mikronor are important, as they are contributing to more data on TWP levels in the environment, especially for those matrices with very limited data today, such as marine sediments. The data provided by Mikronor already shows higher levels compared to those that are available through previous studies (Müller et al., 2025), supporting the need for sharing and utilizing data from Mikronor and other monitoring programs to set realistic test conditions for toxicity testing.

In conclusion, the MIKRONOR 2024 results provide critical insights into the distribution and impact of microplastics and tyre wear particles in Norway. They mirror global trends and reinforce the need for improved monitoring techniques, standardized methodologies, and targeted mitigation strategies. Norway's integrated approach serves as a model for other countries seeking to understand and manage microplastic contamination in diverse environmental contexts.

5 Reflections on monitoring program approach and data quality

5.1 Recommendations for future monitoring

- Lake and coastal surface waters should continue to be monitored. Special focus should be put into the accumulation zones. The load of MP in the surface waters of the Oslofjord seems to be driven by the plastic particles in these zones, and efforts to estimate their distribution, and by that total loads of MP in surface waters is recommended. Identifying similar accumulation zones in other semi-urban areas could be a valuable approach for MIKRONOR. Future campaigns could combine modeling of wind- and water-driven transport, combined with drone surveys and targeted sampling using more flexible collection devices.
- The effects of storms, floods and high precipitation on transport of MP particles should be prioritized in coming years for MIKRONOR.
- Studies aiming to target textile fibres should rely on high-volume pump methods, as nets are generally unsuited for fibre analysis. Textile fibres can easily pass through the net mesh—if oriented lengthwise—and are also prone to contamination. This was evident in the MIKRONOR study, where fibres had to be excluded due to high levels of contamination, likely introduced from storage conditions and sampler clothing.

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7 Appendix

7.1 QA/QC

Large parts of this chapter reuse text from our previous reports (Alling et al., 2023, Alling et al., 2024)

7.1.1. Field blanks

7.1.1.1. Atmospheric blanks

Atmospheric blanks were used during field sampling to account for any microplastic contamination that may have occurred during the sampling process due to deposits from the surrounding air. The air at the sampling site could have been contaminated with particles from the clothing and skin of the sampler, as well as from other sources (boats, equipment onboard etc.) at the sampling site.

Atmospheric blanks were taken together with the following sample types:

- Manta nets (three per three parallel transects)
- Beach samples (three per beach)
- Sediment grab (three per station)
- Sediment core (three per core)

All net blanks from 2024 year's sampling campaign showed very low levels of microplastics.

7.1.1.2. Net blanks

To mitigate contamination arising from the nets used for sampling (manta trawls), a net blank was conducted following net cleaning. This net blank was taken by attaching a freshly cleaned cod-end (the cup collecting the sample at the end of the net) and flushing the net multiple times (a minimum of four) from the outside with a seawater hose to transfer its content into the cod-end. Subsequently, the material from the cod-end was moved to a sample glass using RO-water.

During the laboratory analyses, the net blanks were analysed before the samples to get an indication of potential net-related contamination. Net related contamination is dominated by textile fibres that get attached to the nets by static electricity, and not easily could be removed. On the other side, fibres also can escape through the net if flushed towards the net with the shortest dimension first. This is a well-recognised challenge in microplastic research, and most studies exclude textile fibres from net samples such as samples taken with manta trawls. This is also the experience we have made in the MIKRONOR programme. We decided this year, especially as the manta trawl samples should be reported in the ICES database and into other international databases, that textile fibres wouldn't be included. However, we have included the "technical fibres" or "hard fibres" – i.e. fibres/filaments from shipping/fishing nets and ropes, which are made of mainly PP or PE, are wider than textile fibres and solid.

The next stage in quality control involved comparing the count in the net blank with the count in the environmental samples. In line with international recommendations, net blanks with equal or higher particle counts than environmental samples were regarded contaminated (Montoto-Martínez et al., 2022; Ryan et al., 2020).

7.1.2. Laboratory procedural blanks

Laboratory (lab) procedural blanks monitor potential contamination that may occur during processing and analysing the samples in the laboratory. Particles in the lab blanks might come from airborne contamination (such as ventilation and clothes), equipment and chemicals used to process the samples. In all our methods, each batch of samples was accompanied by ca. three lab blanks, consisting of 200 ml

RO water, that were treated in the same way as the environmental samples[1]. One batch is defined as samples that are processed on the same day(s). The number of environmental samples analysed within a single batch differ between sample types as the methods for sample processing differ in complexity and time used. Number of samples in one batch may also differ within each sample type.

Table 4. The mean MP per sample type and respective lab-blanks, LOD and LOQ for each sample type.

Matrix	Sample type	Mean MP/sample	Mean MP/lab- blank	LOD	LOQ
Sediment	Beach	8.2	9.0	33.1	89.2
	Marine grab	109.3	9.6	34.1	91.2
	Marine core	170.5	4.2	11.1	27.0
	Urban run-off sediment	393.0	19.2	34.7	71.
Biota	Blue mussel	8.8	3.7	12.2	32.0
Water	FerryBox	8.0	0.0	0.0	0.0
	Manta trawl	318.0	0.0	0.0	0.0
	Urban run-off water	261.0	10.2	37.0	99.6

The difference in the number of particles in lab blanks between sample types mainly reflects the time and complexity of processing different types of samples (**Table 4**). The blue mussel and sediment samples were exposed to possible contamination for a longer time in the lab, as they required more cleaning and processing steps compared to many of the water samples, such as Manta net samples. The FerryBox samples represent an intermediate situation in terms of time and steps involved in the preparation of the samples. In this year's samples, we encountered extremely high levels of MP in some, but not all of the Urban samples, forcing us to sub-sample our aliquots and causing rather high values in the blanks compared to other sample types. For most of the urban samples, this was not a problem as the levels in the samples were so high. Different methods in the lab are known to cause different contamination risks, as described by Noonan et al. (2023).

7.1.3. Calculation of LOD/LOQ from lab blanks

The lab blanks in MIKRONOR were used to calculate the LOD and LOQ for the different sample types, as follows:

- LOD: the mean number of microplastics in the lab blanks for that sample type + 3 x SD
- LOQ: the mean number of microplastics in the lab blanks for that sample type + 10 x SD

Blue mussels and beach samples were found to have number of particles below/in the same range as LOD and/or LOQ. The lab blanks for the water samples had very low numbers of microplastics (0 in almost all samples), resulting in low LOD/LOQ. The results for all environmental sample types are plotted and shown together with the calculated LOD and LOQ (as dotted lines) in **Figures 40-47**.

[1] *In a few batches, a lab blank has been lost in the procedure.*

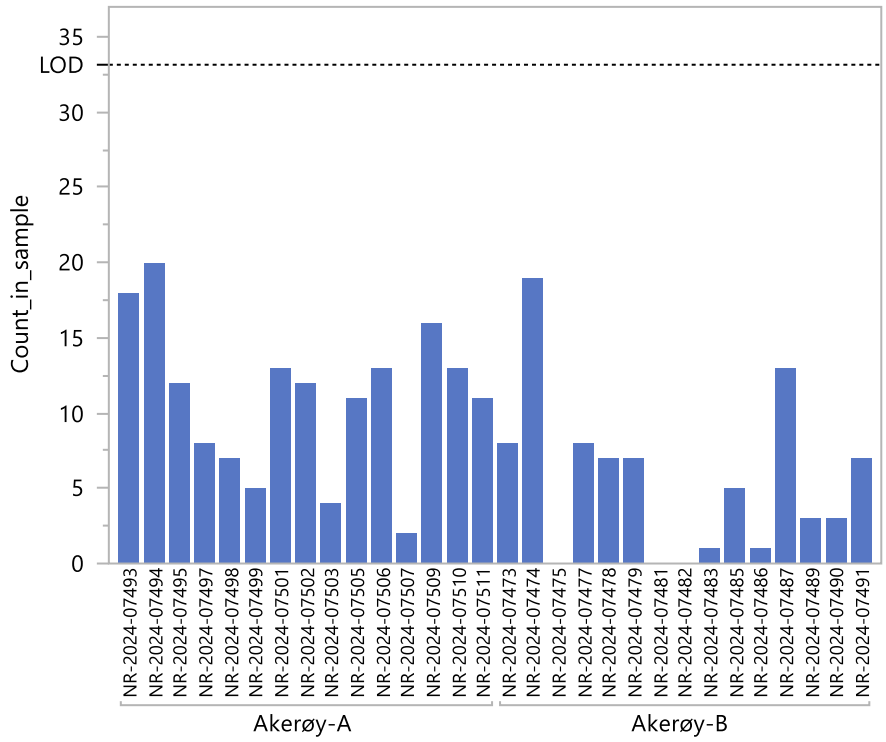


Figure 40. Beach sediment: MP count in samples vs. LOD

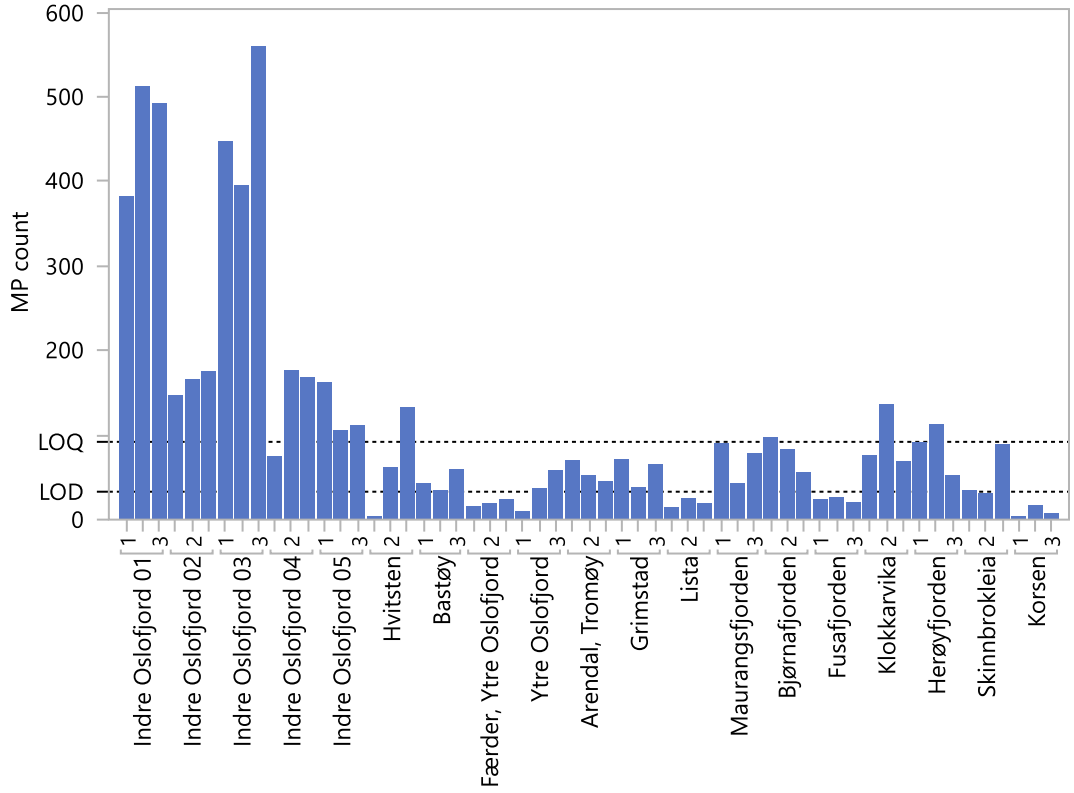


Figure 41. Marine sediment grabs: MP count in samples vs. LOD and LOQ

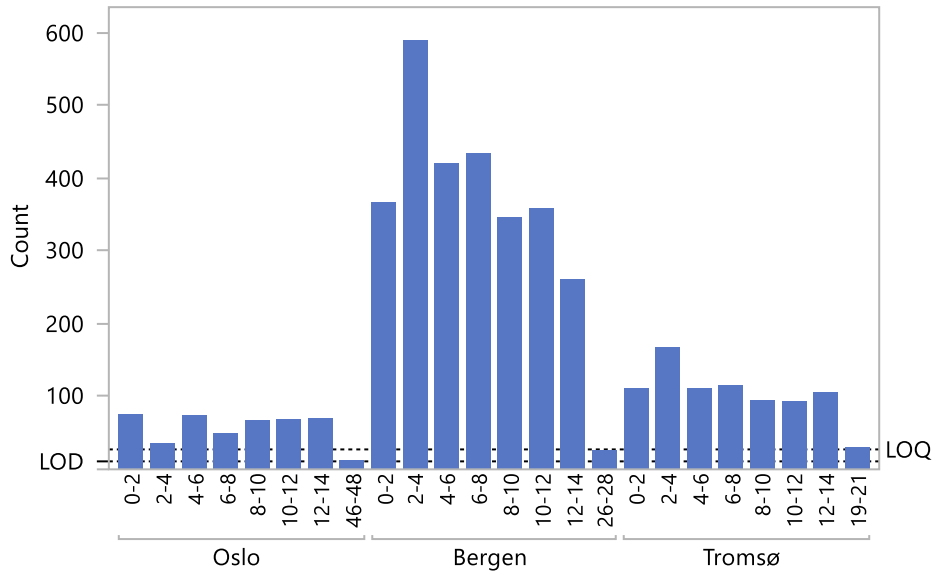


Figure 42. Marine sediment cores: MP count in samples vs. LOD and LOQ

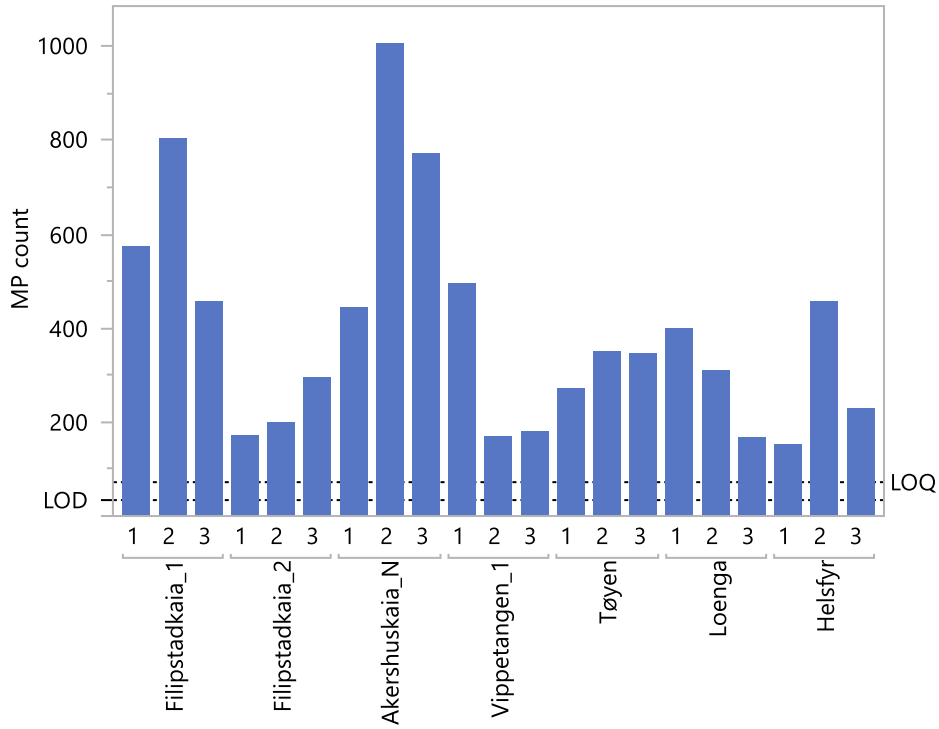


Figure 43. Urban run-off sediment. Number of microplastic per sample per station with LOD and LOQ.

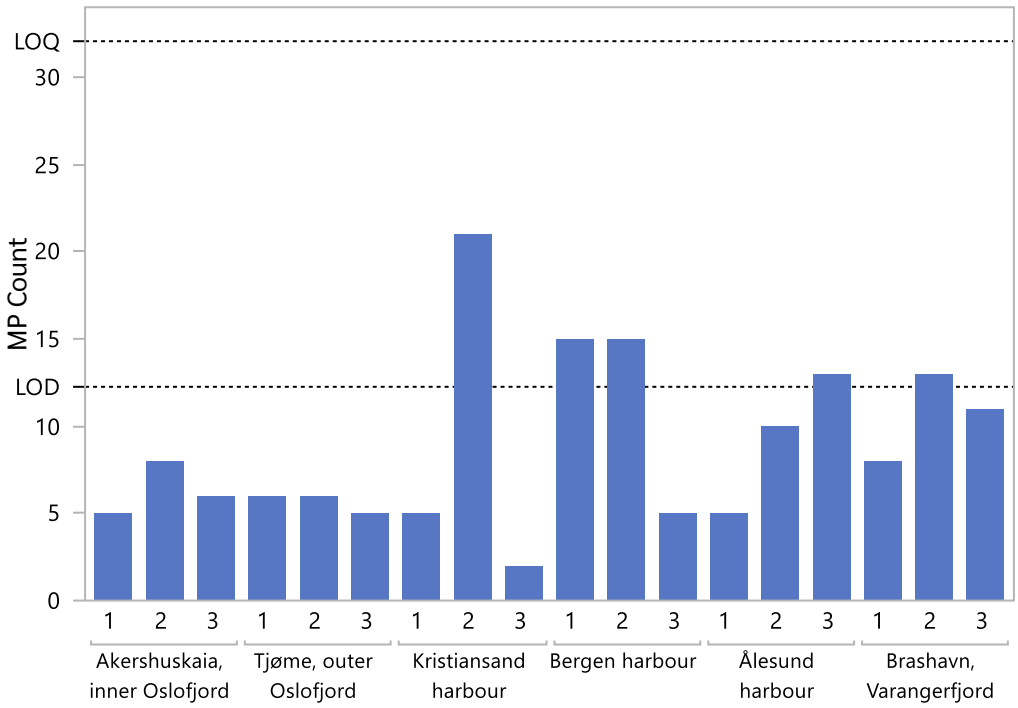


Figure 44. Blue mussel. Number of microplastic per sample per station with LOD and LOQ.

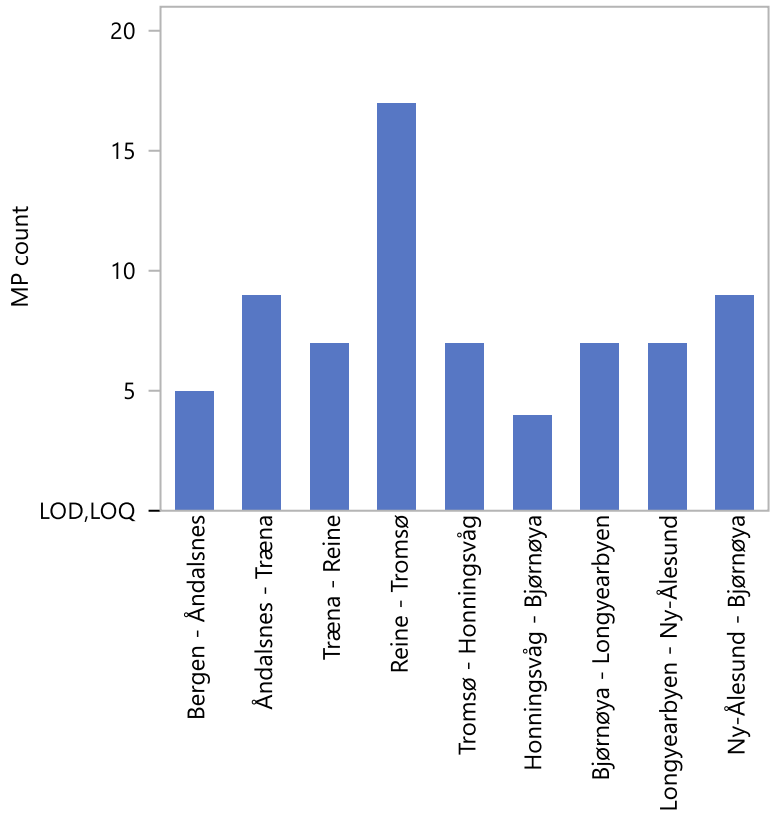


Figure 45. FerryBox. Number of microplastic per sample per station with LOD and LOQ.

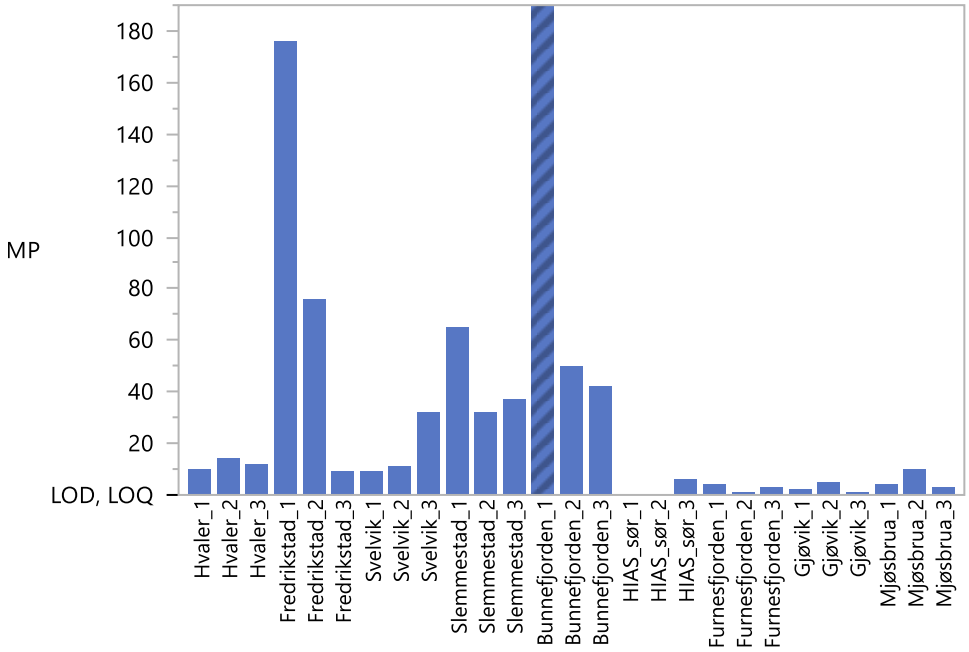


Figure 46. Manta. Number of microplastic per sample per station with LOD and LOQ. Count for sample Bunnefjorden_1 = 7971.

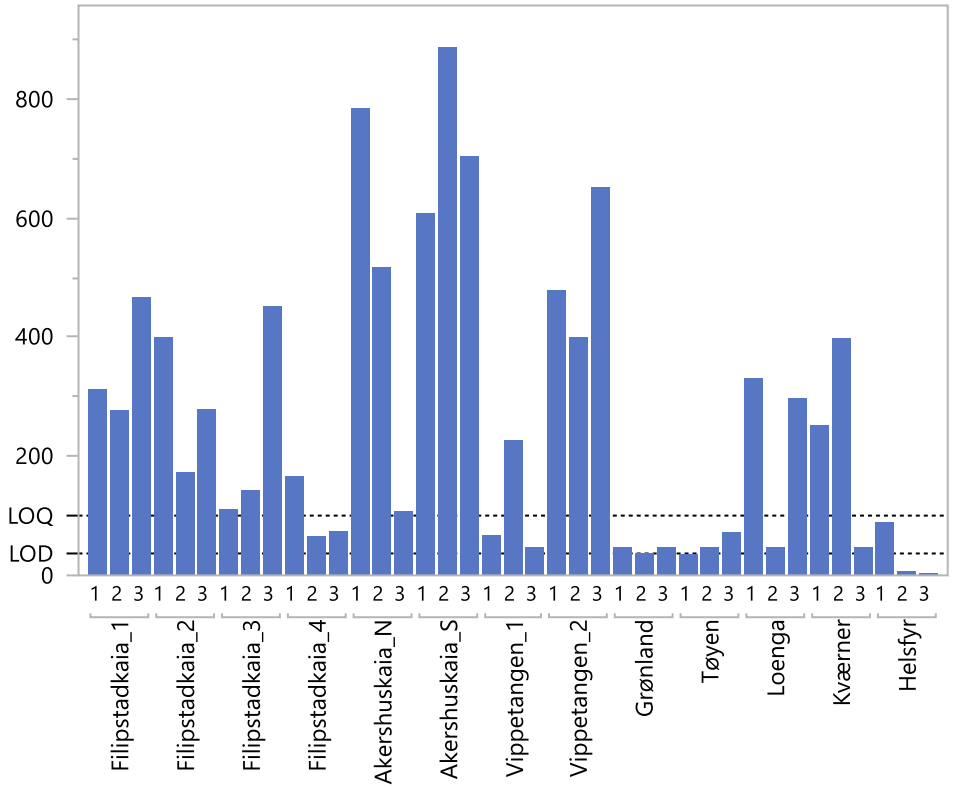


Figure 47. Urban run-off water. Number of microplastic per sample per station with LOD and LOQ.

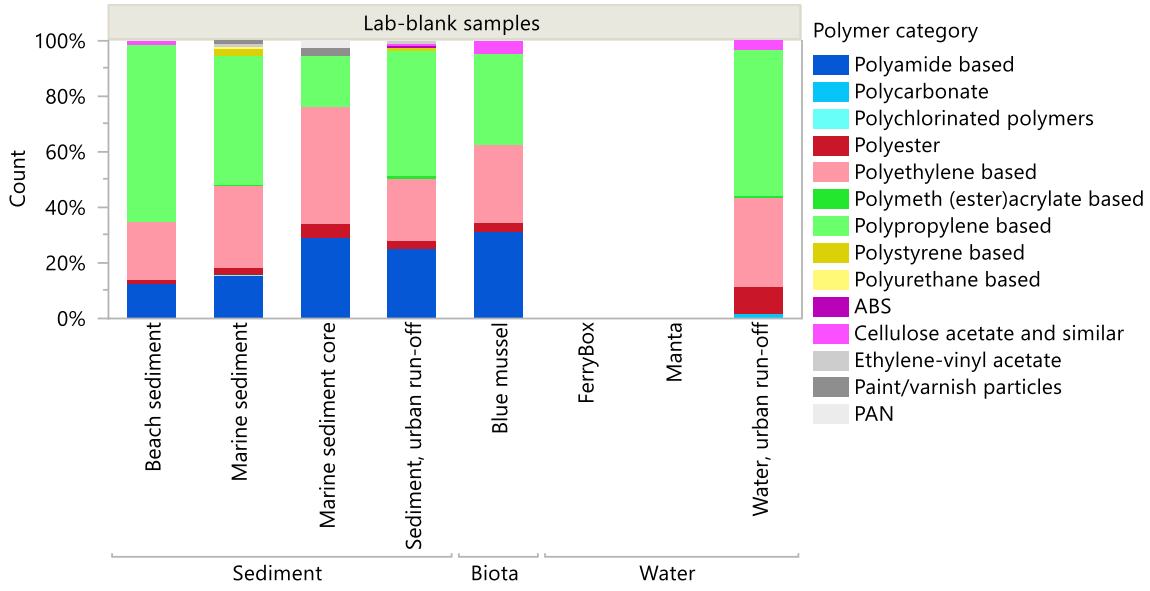


Figure 48. Distribution of polymer categories of microplastic particles found in laboratory blank samples corresponding to the 2024 environmental samples.

The **Figure 48** illustrates the distribution of polymer types found across the lab blanks divided by data set. Among the polymers detected in the lab blanks, polypropylene and polyethylene were notably present and are suspected to originate partly from laboratory equipment and consumables. However, those are also the most present MP polymers in the samples, and it can also reflect cross-over contamination from sample to blanks. Polyamide, also found in the blanks, was likely introduced through airborne fibers and clothing, reflecting general background contamination. Water samples (FerryBox and Manta) were processed quickly in the lab, reflecting in low contamination risk in the laboratory, and no plastic particles were detected in the lab blanks

7.1.4. About blank corrections for microplastic analyses (count and FTIR)

Within microplastics research, there has been an ongoing discussion whether to blank correct microplastic samples or not. As pointed out by Hermsen et al., 2018, who assessed potential airborne contamination in the laboratory; *blank correcting by particle count can often lead to an incorrect final sample number*.

Our approach has been to align microplastic monitoring with procedures that are common among other environmental monitoring analyses, to establish LOD and LOQ for methods and matrices, and to adjust number of replicates and amount of sample material to the established LOD/LOQ. Even though MPs are very different from measurements of other environmental contaminants this has been our starting point for establishing LOD/LOQs. This is also in line with the latest updates of international guidelines, e.g., EU Marine Strategy Framework Directive (MSFD)

In samples from areas without known nearby sources, the analyses generally revealed low levels of microplastics, often below the detection limits. However, in samples from highly impacted areas, the levels were higher and above the detection limits (see figures above). For some sample types, it was challenging to obtain values above the LOD and LOQ. When necessary, we have tried to modify our methodological approach. We have increased the number of individuals in the blue mussel samples to raise the number of microplastic particles in each processed sample. Last three years, the blue mussels provided to MIKRONOR were much smaller than in previous years. Several stations did not have enough blue mussels to provide 30 individuals to MIKRONOR, and in several stations, the blue mussels were smaller than MIKRONOR's specifications. The blue mussel samples did not increase in size, and we encountered the

same situation this year as in 2023 and 2024, resulting in blue mussel microplastic counts, below LOD in most samples.

Munno et al., (2023), recommended a complex correction by specific characteristics using combined methods. While this is more precise, and the total corrected value subtracts the lowest number of particles, it is time consuming and uncertain if it is applicable to a lab where lots of samples are handled in a large monitoring program.

In two cases, our standard procedures include subtractions of microplastic particles:

- If the contamination source was known, and particles with certainty came from that source, we have been subtracting those particles from the sample. A variation of this approach has been applied this year, see below.
- Fibres are a known problem in all net samples. If the number of fibres in the field blank were higher or equal to the numbers of fibres in the sample, fibres were excluded from further analyses. Our procedures are in line with international recommendations Guidance on the Monitoring of Marine Litter in European Seas, (Michida et al., 2019).

This year we have trialed a new approach for blank correction. There was a case where there was a significant occurrence of polypropylene contamination in the first batch of marine sediment grab samples. The source of this contamination was quickly identified and addressed, resulting in low polypropylene contamination in subsequent batches. There was a noticeable visual difference in µFTIR spectra for the polypropylene particles in the lab blanks compared to the majority of polypropylene found in the environmental samples. The hypothesis was developed that particles originating in lab contamination are from the same source, and will be more similar to each other than to polypropylene from the environment, which are from more diverse sources, and have likely been exposed to degradation for longer. To test if lab contamination sourced polypropylene could be differentiated from environmental polypropylene, a random forest machine learning classification model from the Python library scikit-learn (skikit-learn, 2025) was used. The classifier was trained on polypropylene particles from environmental samples with no known contamination, and from the laboratory procedural blanks where contamination was identified. The spectra were found to be significantly different, and the classifier was able to correctly identify contamination polypropylene with 80% accuracy when combining spectra and particle size. Using this method, 28 out of a total of 98 polypropylene particles were removed from the batch.

7.1.5. QA/QC for atmospheric deposition samples

At each monitoring station, four field blanks for atmospheric deposition samples were collected (**Table 5**). Field blanks were obtained by rinsing the bulk deposition sampler – after sample collection and two rinsing steps – once more with approx. 500 mL of water and transferring it to a separate glass bottle. The mean value and standard deviations of field blanks were used to calculate LODs and LOQs:

$LOD = mean_{field\ blank} + 3 \times SD$

$LOQ = mean_{field\ blank} + 5 \times SD$

Table 5. Field blanks, LODs and LOQs (all in µg/m^2/d) for each deposition sample monitoring station. LODs and LOQs were calculated from field blank data and set to 0.01 if not detected in the field blanks.

Station	PMMA	PP	PVC	TWP	PA	PU	PC	PE	PS	PET
Zeppelin										
Mean field blank	0.09	1.02	2.16	0.00	0.00	0.03	0.00	0.38	0.06	0.27
LOD	0.41	5.76	9.77	0.01	0.01	0.20	0.01	2.69	0.39	1.88
LOQ	0.63	8.92	14.8	0.01	0.01	0.32	0.01	4.23	0.61	2.95
Birkenes										
Mean field blank	0.17	0.17	9.71	0.00	0.22	0.04	0.00	1.20	0.16	0.50
LOD	0.67	0.80	20.4	0.01	1.39	0.21	0.01	6.26	0.25	1.56
LOQ	1.00	1.21	27.5	0.01	2.16	0.33	0.01	9.64	0.31	2.28
Sofienbergparken										
Mean field blank	0.04	0.06	1.94	0.01	0.02	0.01	0.00	0.00	0.13	0.41
LOD	0.12	0.39	5.99	0.09	0.11	0.04	0.01	0.01	0.33	1.74
LOQ	0.17	0.61	8.69	0.14	0.17	0.06	0.01	0.01	0.46	2.63

Table 6. Mean field blanks (in ng/sample), mean recovery rates (in %), LODs and LOQs of UV compounds in deposition samples. Recovery rates were assessed by spiking each sample with deuterium labelled standards before extraction. Recovery rates for UV-329 are not available due to an unavailability of the isotope-labelled standard. LODs and LOQs were calculated using laboratory blanks.

	UV-320	UV-326	UV-327	UV-328	UV-329
Zeppelin					
Mean field blank	0.01	0.07	0.03	0.08	0.15
Mean recovery (%)	102	108	86	97	n.a.
Birkenes					
Mean field blank	0.00	0.06	0.00	0.06	0.04
Mean recovery (%)	118	134	103	113	n.a.
Sofienbergparken					
Mean field blank	0.00	0.09	0.02	0.13	0.24
Mean recovery (%)	114	138	100	110	n.a.
LOD	0.01	0.02	0.02	0.02	0.05
LOQ	0.03	0.06	0.06	0.06	0.15

7.2 Methods for MIKRONOR 2024 sampling and analyses

This is the fourth annual report for MIKRONOR. Most sampling procedures and analytical methods are thoroughly described in van Bavel et al., 2022, and Alling et al., 2023. **Table 7** provides an overview of sample collection, laboratory treatments and analytical methods for each sample type. The methods and sampling procedures or field data that are new for this report are provided in chapter 7.2.1 and 7.2.2.

Table7. Overview of sampling and methods, and references to description of these for each sample type

Environment	Sample type	Methods
Air	Deposition	Sample collection, laboratory treatment and analytical methods, see Alling et al., 2023. Sampling periods for each station and sample type in Table 9.
Lake	Water, manta trawl	Sample collection see chapter 7.2.1., laboratory treatment and analytical methods, see Alling et al., 2023. For a description of some statistical estimation techniques used for samples with an unusually high number of particles, see chapter 7.2.2.
Urban	Water, urban run-off in stormwater traps	Sample collection see chapter 7.2.1 Laboratory treatment and analytical methods, see Alling et al., 2023. For a description of some statistical estimation techniques used for samples with an unusually high number of particles, see chapter 7.2.2.
	Sediment, stormwater traps	Sample collection see chapter7.2.1. Laboratory treatment, same as for sediments. For description of some statistical estimation techniques used for samples with an unusually high number of particles, see chapter 7.2.2
Coastal/Marine	Water, FerryBox	Sample collection, laboratory treatment and analytical methods, see Alling et al., 2023. For this year's sampling volumes per transect, see chapter 7.2.1.
	Water, manta trawl	Sample collection is described in chapter 3.3, laboratory treatment and analytical methods, see Alling et al., 2023. For a description of some statistical estimation techniques used for samples with an unusually high number of particles, see chapter 7.2.2.
	Blue Mussels	Sample collection under monitoring program MILKYS. Laboratory treatment and analytical methods, see Alling et al., 2023.
	Sediment, Beach samples	Sampling collection, see protocol in Alling et. al., 2024. Laboratory treatment and analytical methods, same as for sediments, see Alling et al., 2023
	Sediment, grab samples	Sampling and analyses of MPs and TWP see methods and results in Alling et al., 2023.
	Sediment, core samples	Sampling conducted under the monitoring program MILKYS. See chapter 7.2.1 for sampling strategy and 7.2.2 for dating of cores. Analytical methods same as for sediment grab samples and beach samples.

7.2.1. Sampling

7.2.1.1. Urban samples

In 2024, urban samples were included in the MIKRONOR programme. A total of 60 urban samples were collected in the inner city of Oslo, focusing on areas that have direct runoff to the Oslofjord.

The samples were collected after rainfall in Oslo on the 6th and the 8th of December 2024 (**Table 8**). Water and sediment were sampled from the same stormwater trap, totaling 39 water samples and 21 sediment samples, from a total of 13 different stormwater trap (only 7 of which were sampled for sediment). All stations were sampled in triplicates for respective matrix.

Table 8. Sample info for 2024 urban runoff samples.

Urban station	N samples, urban runoff water	N samples (stormwater sediment)	Date	North (degrees, decimal min.)	East (degrees, decimal min.)	UTM 33N	UTM 33Ø
1	3	3	06.12.2024	59°54.1990	10°44.4204	6648309	261822
2	3	3	06.12.2024	59°54.5763	10°44.0900	6649028	261559
3	3	3	06.12.2024	59°54.1010	10°45.6419	6648054	262947
4	3	3	08.12.2024	59°54.5910	10°43.2545	6649106	260783
5	3	3	08.12.2024	59°54.5726	10°43.1153	6649080	260652
6	3	3	08.12.2024	59°54.8158	10°45.9127	6649364	263284
7	3	3	08.12.2024	59°54.9290	10°47.8450	6649489	265030
8	3	-	08.12.2024	59°54.5762	10°45.9380	6648918	263280
9	3	-	08.12.2024	59°54.2870	10°47.1384	6648310	264363
10	3	-	08.12.2024	59°54.4481	10°42.8009	6648868	260344
11	3	-	08.12.2024	59°54.5213	10°42.9855	6648992	260525
12	3	-	06.12.2024	59°54.3750	10°44.1200	6648654	261563
13	3	-	06.12.2024	59°54.1316	10°44.4333	6648183	261826

Figure 49. Pictures from urban run-off sampling.

7.2.1.2. Air samples

Table 9. Sample ID, station, sample type and sampling period for 2024 for the air samples.

Sample ID	Station	Sample Type	Sampling start	Sampling end
ZD01	Zeppelin	Deposition	01.02.2024	15.02.2024
ZD02	Zeppelin	Deposition	15.02.2024	28.02.2024
ZD03	Zeppelin	Deposition	28.02.2024	13.03.2024
ZD04	Zeppelin	Deposition	13.03.2024	27.03.2024
ZD05	Zeppelin	Deposition	27.03.2024	10.04.2024
ZD06	Zeppelin	Deposition	10.04.2024	25.04.2024
ZD07	Zeppelin	Deposition	25.04.2024	08.05.2024
ZD08	Zeppelin	Deposition	08.05.2024	21.05.2024
ZD09	Zeppelin	Deposition	21.05.2024	05.06.2024
ZD10	Zeppelin	Deposition	05.06.2024	19.06.2024

Sample ID	Station	Sample Type	Sampling start	Sampling end
ZD11	Zeppelin	Deposition	19.06.2024	03.07.2024
ZD12	Zeppelin	Deposition	03.07.2024	17.07.2024
ZD13	Zeppelin	Deposition	17.07.2024	31.07.2024
ZD14	Zeppelin	Deposition	31.07.2024	14.08.2024
ZD15	Zeppelin	Deposition	14.08.2024	28.08.2024
ZD16	Zeppelin	Deposition	28.08.2024	11.09.2024
ZD17	Zeppelin	Deposition	11.09.2024	25.09.2024
ZD18	Zeppelin	Deposition	25.09.2024	09.10.2024
ZD19	Zeppelin	Deposition	09.10.2024	23.10.2024
ZD20	Zeppelin	Deposition	23.10.2024	07.11.2024
ZD21	Zeppelin	Deposition	07.11.2024	20.11.2024
ZD22	Zeppelin	Deposition	20.11.2024	04.12.2024
BD01	Birkenes	Deposition	18.01.2024	31.01.2024
BD02	Birkenes	Deposition	31.01.2024	14.02.2024
BD03	Birkenes	Deposition	14.02.2024	28.02.2024
BD04	Birkenes	Deposition	28.02.2024	13.03.2024
BD05	Birkenes	Deposition	13.03.2024	27.03.2024
BD06	Birkenes	Deposition	27.03.2024	10.04.2024
BD07	Birkenes	Deposition	10.04.2024	24.04.2024
BD08	Birkenes	Deposition	24.04.2024	08.05.2024
BD09	Birkenes	Deposition	08.05.2024	22.05.2024
BD10	Birkenes	Deposition	22.05.2024	05.06.2024
BD11	Birkenes	Deposition	05.06.2024	19.06.2024
BD12*	Birkenes	Deposition	19.06.2024	03.07.2024

Sample ID	Station	Sample Type	Sampling start	Sampling end
BD13	Birkenes	Deposition	03.07.2024	17.07.2024
BD14	Birkenes	Deposition	17.07.2024	31.07.2024
BD15	Birkenes	Deposition	31.07.2024	14.08.2024
BD16	Birkenes	Deposition	14.08.2024	28.08.2024
BD17	Birkenes	Deposition	28.08.2024	11.09.2024
BD18	Birkenes	Deposition	11.09.2024	25.09.2024
BD19	Birkenes	Deposition	25.09.2024	09.10.2024
BD20	Birkenes	Deposition	09.10.2024	23.10.2024
BD21	Birkenes	Deposition	23.10.2024	06.11.2024
BD22	Birkenes	Deposition	06.11.2024	20.11.2024
BD23	Birkenes	Deposition	20.11.2024	04.12.2024
SD01	Sofienbergparken	Deposition	17.01.2024	31.01.2024
SD02	Sofienbergparken	Deposition	31.01.2024	14.02.2024
SD03	Sofienbergparken	Deposition	14.02.2024	28.02.2024
SD04	Sofienbergparken	Deposition	28.02.2024	13.03.2024
SD05	Sofienbergparken	Deposition	13.03.2024	27.03.2024
SD06	Sofienbergparken	Deposition	27.03.2024	10.04.2024
SD07	Sofienbergparken	Deposition	10.04.2024	24.04.2024
SD08	Sofienbergparken	Deposition	24.04.2024	08.05.2024
SD09	Sofienbergparken	Deposition	08.05.2024	22.05.2024
SD10	Sofienbergparken	Deposition	22.05.2024	05.06.2024
SD11	Sofienbergparken	Deposition	05.06.2024	19.06.2024
SD12	Sofienbergparken	Deposition	19.06.2024	03.07.2024
SD13	Sofienbergparken	Deposition	03.07.2024	17.07.2024

Sample ID	Station	Sample Type	Sampling start	Sampling end
SD14	Sofienbergparken	Deposition	17.07.2024	31.07.2024
SD15	Sofienbergparken	Deposition	01.08.2024	15.08.2024
SD16	Sofienbergparken	Deposition	15.08.2024	28.08.2024
SD17	Sofienbergparken	Deposition	28.08.2024	11.09.2024
SD18	Sofienbergparken	Deposition	11.09.2024	25.09.2024
SD19	Sofienbergparken	Deposition	25.09.2024	09.10.2024
SD20	Sofienbergparken	Deposition	09.10.2024	23.10.2024
SD21	Sofienbergparken	Deposition	23.10.2024	06.11.2024
SD22	Sofienbergparken	Deposition	06.11.2024	20.11.2024

*Note that sample BD12 was lost due to instrumental issues.

In this report, monitoring stations for total deposition sampling were strategically placed to document the long-range transport of microplastics and minimize local influences. For this, two observatories were chosen to represent different regions of Norway and areas that receive air from diverse global source regions: Birkenes (58°23'17.9"N 8°15'08.1"E), situated in southern Norway, and Zeppelin (78°54'23.9"N 11°53'20.8"E), located on Svalbard in the Arctic Ocean (**Figure 1**). Furthermore, the monitoring station in Sofienbergparken (59°55'23.3"N, 10°45'56.0"E), Oslo, was chosen as an urban background station. For additional details about the sampling sites, visit NILU's observatories and monitoring stations - NILU:

Collection of total deposition samples for microplastic analysis was conducted over 14-day periods each, by applying the methods described in Alling et al., 2023.

Deposition Samples (Dry and Wet):

- Full metal bulk precipitation samplers (https://innovation.nilu.no/wp-content/uploads/sites/5/2024/10/innovation_nilu_Atmospheric_Microplastic_Collector_2020.pdf) were employed.
- After collection, samples were transferred to 2-L glass bottles and covered with aluminium foil to prevent contact with the cap.
- The metal sampler was rinsed twice, and the rinse water was added to the sample.

Field blanks were obtained by rinsing the sampler once more with approximately 500 mL of water and transferring it to a separate glass bottle. Four field blanks per monitoring station were made and data is presented in **Table 5**. MP deposition fluxes presented in this report are not blank-corrected.

7.2.1.3. Manta samples

Manta nets are one of the most commonly employed tools to sample water bodies for floating plastics. Significant effort has been made by the Ministry of Environment Japan to standardise monitoring approaches, and we used a combination of method recommendations from AMAP (2022) and updated guidelines from Japan's Ministry of the Environment (MOEJ) (Michida et al., 2023). The method also aligns with the ongoing Barents Sea survey with the ambition that the data will be available for comparative assessment between the Barents Sea, Oslofjord, and Lake Mjøsa.

Table 10 and **Figure 50** present the number of MPs and the polymer categories in the net and atmospheric blanks compared to those in the actual samples. As shown, some samples contained MP levels within the same range as the net blanks, and these results should therefore be interpreted with caution.

Table 10. Number of microplastics reported per collected sample (no volume correction) compared to particles in net blanks and atmospheric blanks.

		Net blank	Atm. blank	Sample Min	Sample Max
Coastal	Hvaler	4	-	10	14
	Fredrikstad	4	-	9	176
	Svelvik	1	-	9	32
	Slemmestad	0	-	32	65
	Bunnefjorden	4	-	42	7971
Lake	HIAS sør	3	1	0	6
	Furnesfjorden	1	-	1	4
	Gjøvik	0	1	1	5
	Mjøsabrua	4	1	3	10

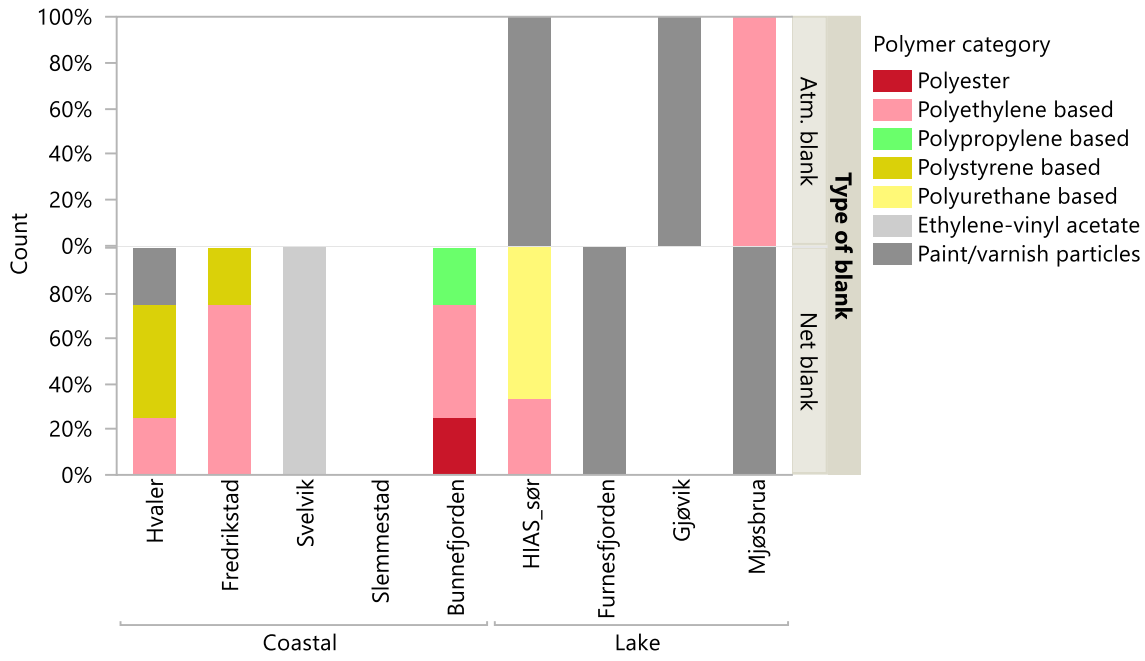


Figure 50. Polymer categories in manta field blank samples (atmospheric and net).

7.2.1.4. FerryBox samples

FerryBox transect samples were taken along the Norwegian coastline, in approximately the same starting positions as in 2023.

Table 11. Date of sampling, start and end location and volume per sample for FerryBox transects in 2024.

Date	Start	Stop	Volume (L)
02.06.2024	Bergen	Åndalsnes	17965
03.06.2024	Åndalsnes	Træna	66188
04.06.2024	Træna	Reine	11843
05.06.2024	Reine	Tromsø	42871
06.06.2024	Tromsø	Honningsvåg	6222
07.06.2024	Honningsvåg	Bjørnøya	3245
08.06.2024	Bjørnøya	Longyearbyen	60098
09.06.2024	Longyearbyen	Ny-Ålesund	32036
10.06.2024	Ny-Ålesund	Bjørnøya	91018

7.2.2. New or changed analytical methods

7.2.2.1. Imputation of polymer category in large fraction of some high concentration Oslofjord surface water and urban stormwater sediments

Elevated concentrations of microplastic particles were detected in samples collected from marine surface waters in Bunnefjorden, located within the inner Oslofjord, as well as from sediment accumulated in urban sediment traps in Oslo. Multiple analytical approaches were employed to estimate the concentration and polymer composition of microplastics with the highest possible accuracy.

Bunnefjorden accumulation zone:

In one sample obtained from a designated accumulation zone in Bunnefjorden, the particle count exceeded the capacity for manual enumeration. The majority of particles exhibited uniform visual characteristics—white coloration, foam-like texture, and size range between 400 and 3000 µm. A subset of these particles was subjected to Fourier-transform infrared spectroscopy (FTIR), confirming their composition as expanded polystyrene (EPS) foam. Based on visual assessment, the total number of EPS particles in the sample was estimated to range between 7000 and 14000. For conservative reporting purposes, a count of 7000 particles was adopted. These particles were classified as white polystyrene foam beads within the aforementioned size range.

For the remaining particles in the sample, approximately 70% underwent full FTIR analysis to determine polymer type. The remaining 30% were characterized based on visual parameters including size, shape, and color. Polymer classification for these particles was inferred probabilistically, using statistical correlations derived from the fully analyzed subset.

7.2.2.2. Urban stormwater samples:

Due to methodological constraints, it was not feasible to perform comprehensive polymer identification via micro-Fourier Transform Infrared Spectroscopy (µFTIR) on all particles within the 300 µm–5 mm size fraction extracted from urban stormwater sediment samples. Nevertheless, all particles were enumerated and characterized in terms of color, shape, and size, following the protocol described by Alling et al. (2023). For each stormwater trap, one representative sample underwent full polymer identification using µFTIR. In the remaining two samples, polymer types were inferred probabilistically based on morphological parameters (color, shape, and size), using statistical models calibrated against the fully analyzed sample. This imputation approach has demonstrated an accuracy of 65–90% for the most

prevalent polymer categories, which typically constitute 80–90% of microplastic particles found in urban environments within the 300 µm–5 mm size range.

In contrast, all particles within the smaller 50–300 µm fraction were subjected to direct polymer identification using scanning µFTIR, ensuring comprehensive characterization of this size class.

7.2.2.3. Changes in Manta sample analyses

Differentiation Between Technical and Textile Fibers in Manta Net Samples

In the Manta net samples collected from Lake Mjøsa and the Oslofjord, textile fibres were systematically excluded from analysis, whereas technical fibres were retained. The distinction between these two fibre types was based on morphological and physical characteristics, as outlined below:

Technical fibres—also referred to as filaments—are typically composed of monofilament strands or consist of twisted or braided structures. These fibres exhibit a relatively uniform thickness and may appear either straight or bent. In aquatic samples, they behave similarly to microplastic fragments. Common examples include materials derived from fishing lines and nets.

Textile fibres, in contrast, are generally thinner (approximately 15–20 µm in diameter) and exhibit greater flexibility upon manipulation. Due to their pliability and small diameter, textile fibres are more likely to pass through the mesh of sampling nets. Technical fibres, being more rigid, tend to be retained even when their diameter is smaller than the mesh size.

It is anticipated that forthcoming international guidelines will adopt a similar distinction between textile and technical fibres, thereby enhancing consistency in microplastic sampling and classification methodologies.

Laboratory methods Manta samples

The pre-analyses were performed as described in Alling et al. (2024), with a modification in the oxidizing/digesting chemical. Instead of 10% KOH, 30% H_2O_2 was used to remove organic matter. This change was made because 30% H_2O_2 has proven to be a significantly more effective oxidizer based on previous experience.

Following visual inspection, suspected microplastic particles were isolated. It became evident that many particles were identical in shape and morphology, suggesting a high likelihood of shared polymer composition. The most frequently observed groups were polystyrene foam and grey varnish (see **Figure 51** and **Figure 52**). These figures present microscope images at 20× magnification, illustrating morphologically similar particles from two different samples.

Polystyrene particles were the most abundant identified microplastic particles (~35%). In samples containing numerous visually similar polystyrene particles, only a subset (3–5 particles) was analyzed using µFTIR. The remaining particles were classified as the same polymer type based on these results. A similar approach was applied to other particle types; when multiple morphologically identical particles were present, only a few were analyzed, and the rest were assumed to be of the same polymer composition. Since we always verified multiple particles of the same type using µFTIR, we would have detected any misjudgments. However, we never found discrepancies between our expectations and the actual polymer type. With extensive experience in distinguishing microplastic particles, we rely on visual classification where necessary due to budget constraints and time limitations. In total 339 of 648 particles were analyzed using µFTIR (approx 52%) ensuring a robust verification for polymer identification.

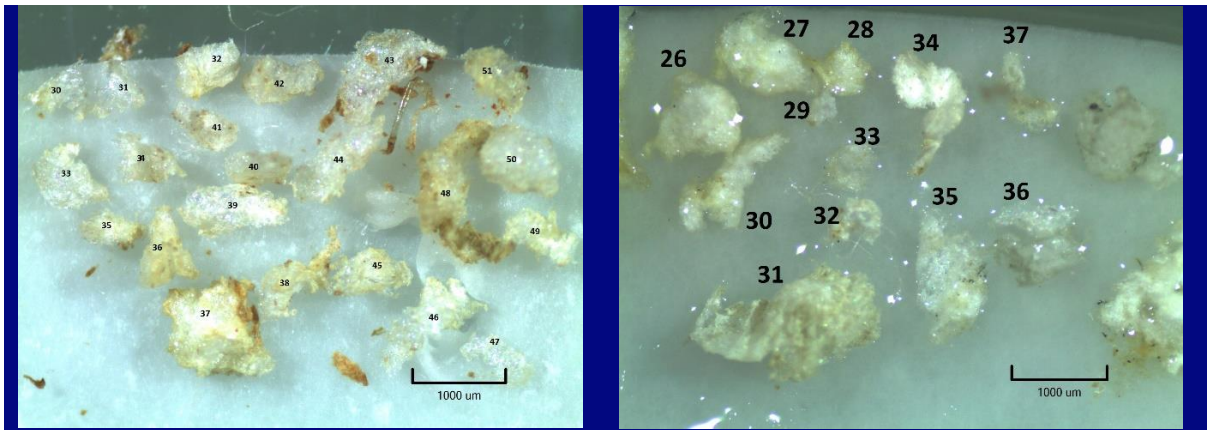


Figure 51. Polystyrene foam found in multiple different samples.

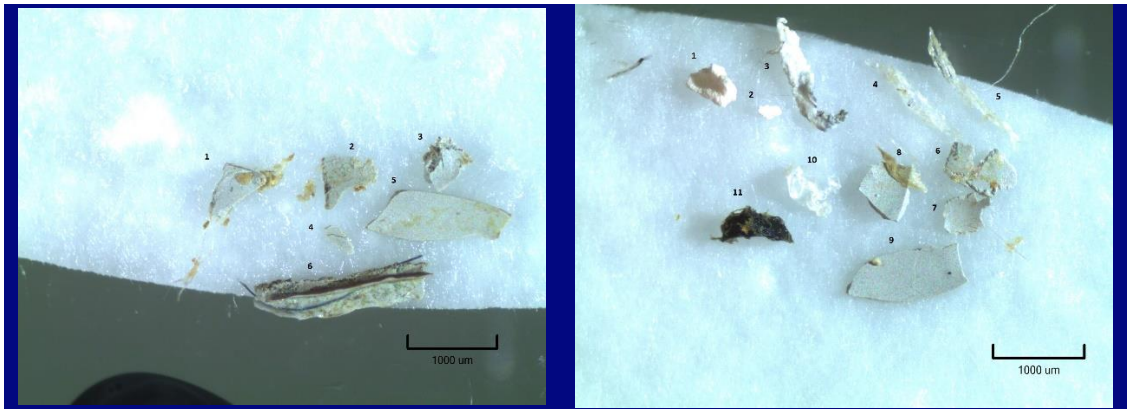


Figure 52. Varnish found in multiple different samples.

7.2.2.4. Measuring microplastics in air samples: pyrolysis-GC/MS

For full method description, see Alling et al., 2023. Briefly, deposition samples were filtered onto stainless steel filters. These were then suspended in ethanol, sonicated (30 min.) and left overnight. The ethanol extract was then filtered onto a glassfiber filter (GF/F), which after drying was injected into pyrolysis-GC/MS. All laboratory work was conducted in a laminar flow clean cabinet.

In this study, 10 polymer types were measured in atmospheric samples:

Table 12. Polymer types analysed in air samples with Pyr-GC/MS and abbreviations used in this report.

Polymer type	Abbreviation
Poly(methyl methacrylate)	PMMA
Polypropylene	PP
Polyvinylchloride	PVC
Polyamide/ Nylon	PA
Polyurethane	PU
Polystyrene	PS
Polyethylene	PE
Polyethylene terephthalate	PET
Polycarbonate	PC
Styrene-butadiene rubber	SBR/TWP

The Frontier Laboratories Microplastics Calibration Standard mixed with $CaCO_3$ was used for a one-point calibration. SBR was calibrated using an in-house standard consisting of a mix of tyre tread from 20 different tyres, mixed with sand (Foscari et al., 2023). As this approach directly simulates the chemical composition of tyre wear particles (TWP), the abbreviation TWP is used to describe the results throughout the report. Note that the same calibration standard has also been used in previous reports. To investigate potential sources, air mass backwards trajectories were simulated using the FLEXPART model for atmospheric particle transport, allowing to track particles forward and backward in time, influenced by Eulerian wind fields generated by 3D meteorological models.

Measuring UV compounds and tyre-related chemicals in air samples

UV stabilizers and tyre-related chemicals were extracted from the ethanol phase (see description of sample preparation of deposition samples) of air samples through liquid-liquid extraction using hexane. After evaporation and a solvent change to acetonitrile, extracts were injected into GC/MS (UV compounds) and LC/MS (tyre-related compounds). Tyre-related compounds were also measured in the water phase of deposition samples. For this, 300 mL sub-samples were collected after the stainless-steel filter filtration step, extracted using solid phase extraction (200 mg HLB columns) and eluted using methanol. After concentration through evaporation, extracts were injected into LC/MS (method adapted from Rauert et al., 2022).

Measuring tyre wear particles in environmental samples with pyrolysis GC/MS

Biota samples and high-volume water samples for TWP analysis were pretreated as described in Alling et al. 2023, results for FTIR analyses are presented in this report chapter 3.2.2 and 3.5, and filters (15 mm silver filters, 5µm pore size) were inserted in pyrolysis cups. Each cup were spiked with 25 µg/cup of internal standard (d6-Polybutadiene, Polymer Source Inc, Canada) and analyzed with a Multi-Shot Pyrolyzer (EGA/PY-3030D) equipped with an Auto-Shot Sampler (AS-1020E) (Frontier lab Ltd., Fukushima, Japan) coupled to gas chromatography mass spectrometer (GC/MS) (5977B MSD with 8860 GC, Agilent Technologies Inc., CA, USA). All samples were pyrolyzed at 700C and in 25:1 split or 50:1 split, following the same methods described in Alling et al., 2023.

For the 50:1 split calibration curve, calibration points were created by adding 1, 2, 5, 10, 20, 40, 60, 100, 140 µg SBR (SBR1500, Polymer Source Inc., Canada) per cup and 25µg d6-PB/cup for all levels. For the 25:1 split calibration curve, calibration points were created hy adding 0.1 (three replicates of the same level), 0.5,1, 2, 10, 15, 20 (two replicates), 40 and 60 µg/cup of SBR (SBR1500, Polymer Source Inc., Canada) per cup and 25µg d6-PB/cup for all levels.

Seven different marker compounds for SBR and BR rubber were monitored: m/z 78 for benzene (B), m/z 118 for α-methylstyrene (MS), m/z 117 for ethylstyrene (ES), m/z 91 for butadiene trimer (Bt), m/z 54 for 4-Vinylcyclohexene (4-VCH), m/z 104 for styrene butadiene dimer (SB) and m/z 91 for styrene butadiene trimer (SBB). Quantification of SBR+BR was performed using the combined peak heights of the four markers (M4) B, MS, ES and Bt, normalized by the internal standard (d6-Pb). For some the sediment cores, B was not used as a marker due to influencing sources and the quantification was performed with three markers (M3) MS, ES and Bt.

All water and sediment samples were analyzed using a 50:1 split and the biota samples using 25:1 split. Due to analysis performed over a long time scale, each analysis batch has different calibration curves and calculated LOD and LOQs. The signal to noise ratio (S/N) is determined by the Agilent Masshunter software for each of the selected markers and then summarised to represent the sum of markers. The S/N level was plotted against the concentration level of SBR to determine the S/N vs concentration relationship following the method by Donovan (2016), using power of regression. The calculated LOD (3 x S/N) and LOQ (10 x S/N) for each calibration can be seen below.

Calibration curve 1

Samples include atmospheric blanks. Urban runoff lab blanks, urban runoff water samples, sediment grab samples and sediment core samples

	M4 µg/cup	M3 µg/cup
LOD	0.076	0.300
LOQ	0.338	2.065

Calibration curve 2

Samples include sediment core samples, sediment grab samples, stormwater sediment samples.

Note, this curve was only using calibration points 1, 2, 5 10, 20, 40, 60 and 100 µg/cup SBR.

	M4 µg/cup	M3 µg/cup
LOD	0.024	0.135
LOQ	0.149	1.405

Calibration curve 3

Samples include stormwater sediments and core sediment samples

	M4 µg/cup	M3 µg/cup
LOD	0.001	0.071
LOQ	0.010	0.610

Calibration curve 4

Samples include blue mussel samples and lab blanks related to blue mussels

	M4 µg/cup	M3 µg/cup
LOD	0.006	0.045
LOQ	0.055	0.776

Calculations from SBR to TWP

The concentration of TWP were further calculated based on the SBR concentrations, following the method described in (Rødland et al., 2022), using measured SBR levels in Norwegian tyres (n=32) and Monte Carlo prediction modelling.

7.3 Results

7.3.1. Sediment Cores

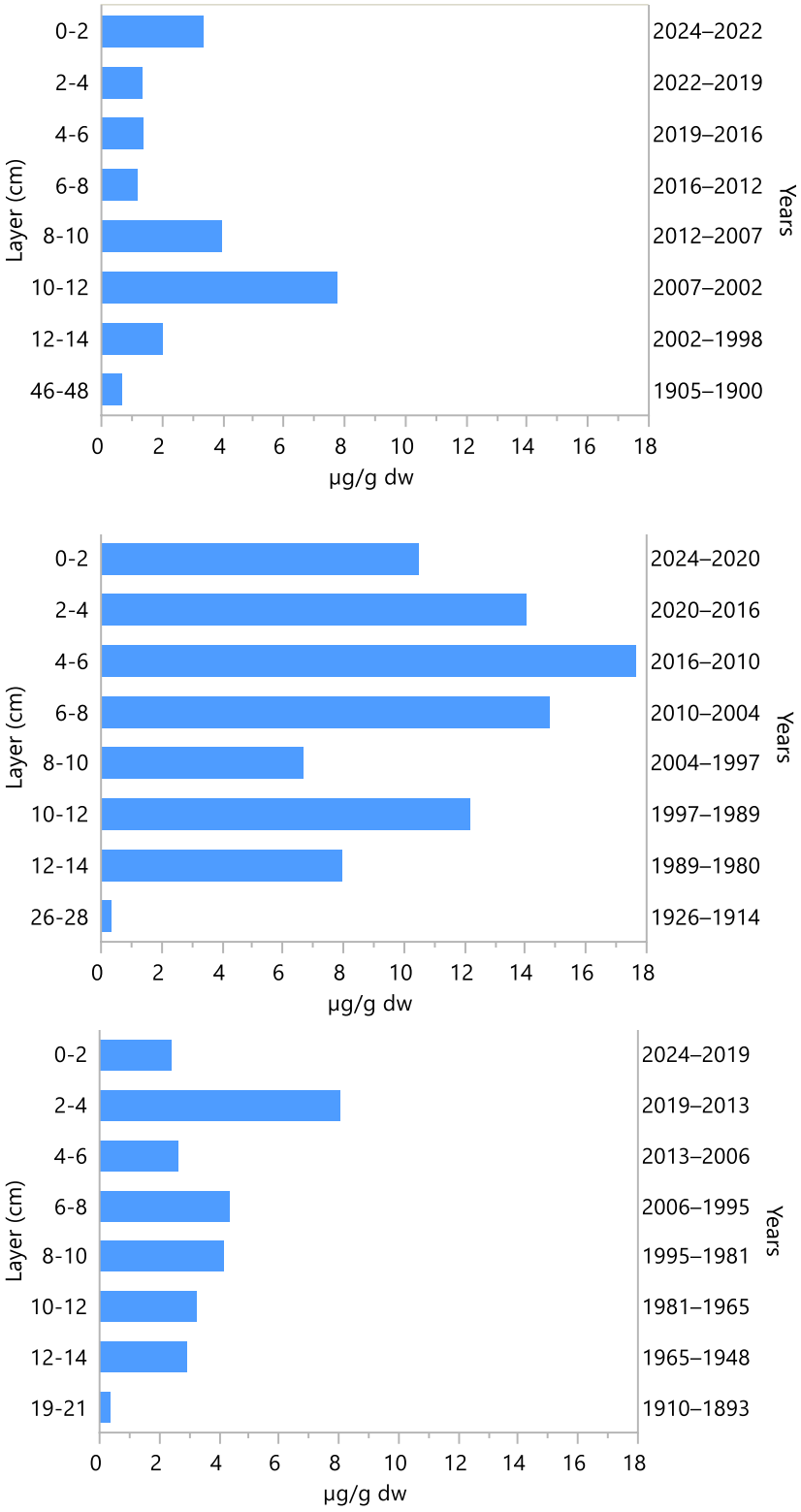


Figure 53. Mass (µg/d dw) of MP in sediment cores. Upper: Oslo, Middle: Bergen, Lower: Tromsø.

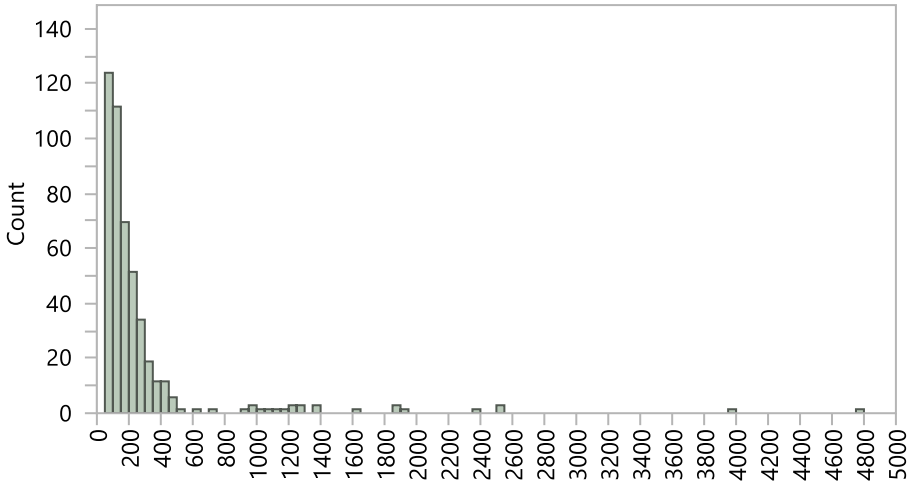


Figure 54. Distribution of microplastic particle sizes (µm) in sediment core from Oslo.

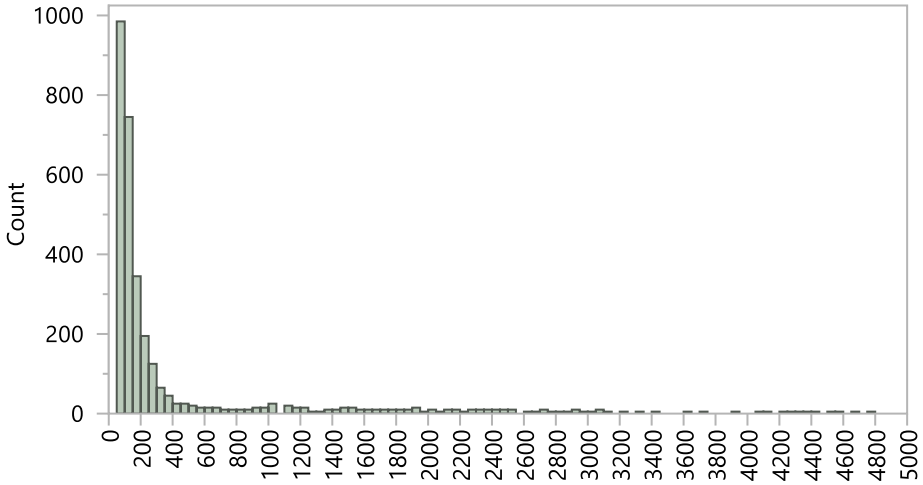


Figure 55. Distribution of microplastic particle sizes (µm) in sediment core from Bergen.

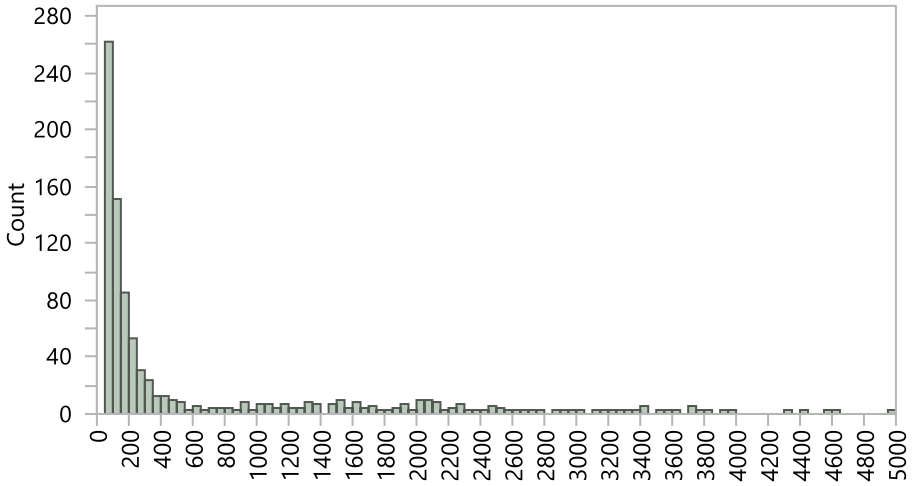


Figure 56. Distribution of microplastic particle sizes (µm) in sediment core from Tromsø.

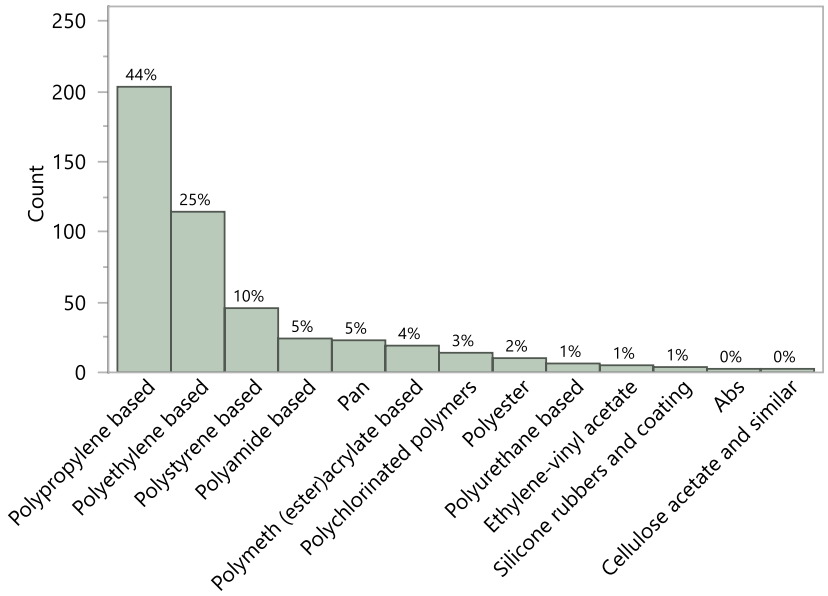


Figure 57. Distribution of polymer categories of microplastic particles in sediment core from Oslo.

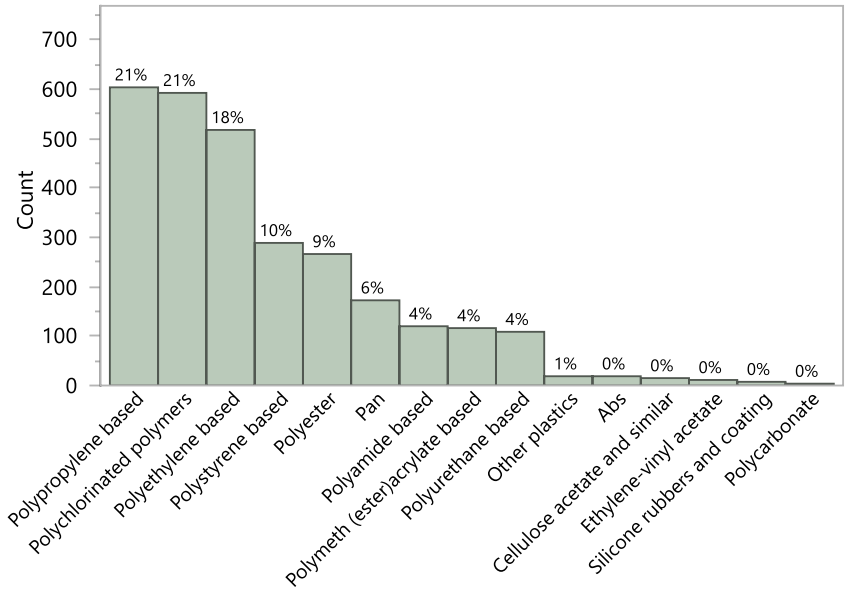


Figure 58. Distribution of polymer categories of microplastic particles in sediment core from Bergen.

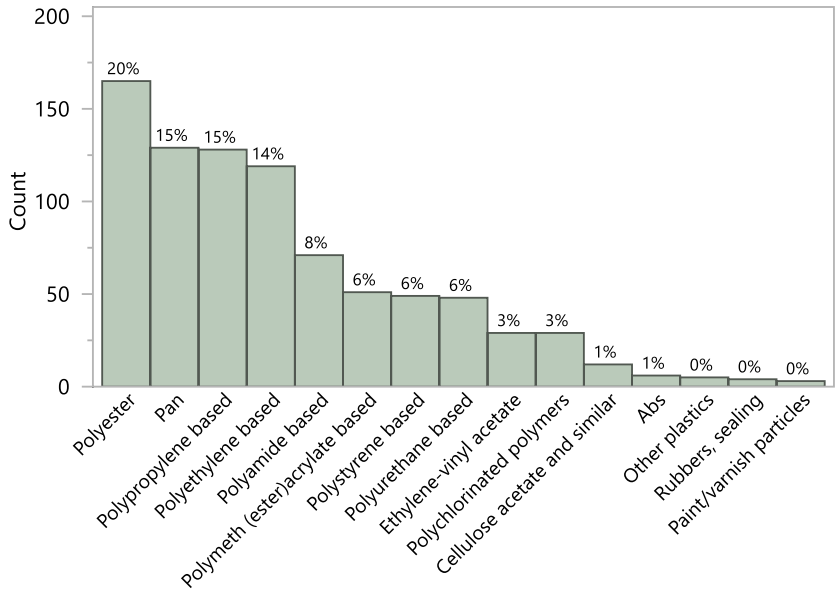


Figure 59. Distribution of polymer categories of microplastic particles in sediment core from Tromsø.

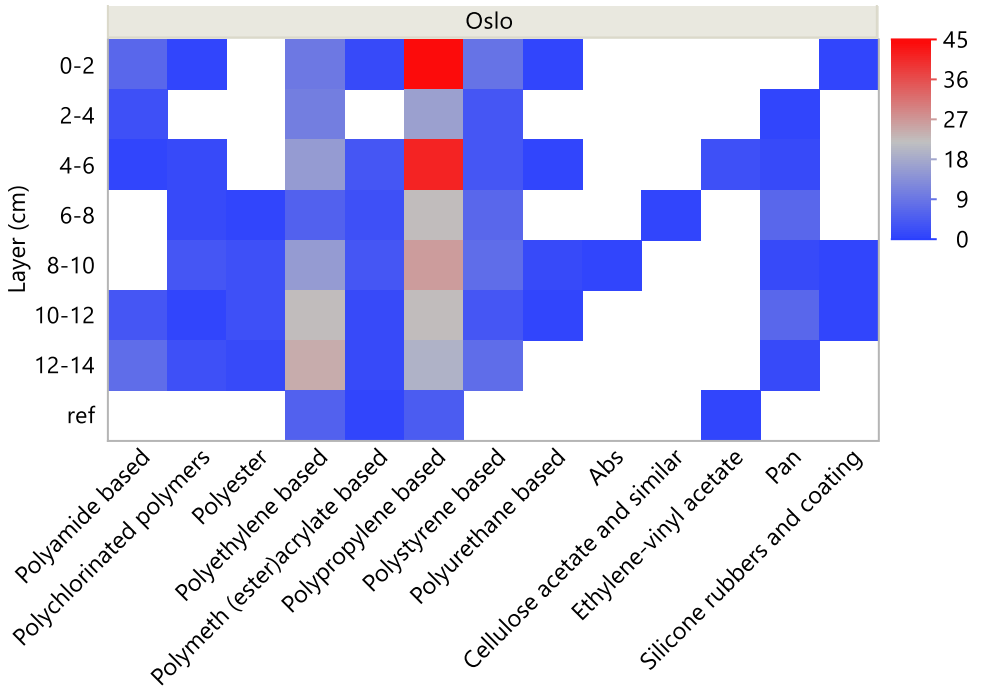


Figure 60. Heatmap polymer categories Oslo

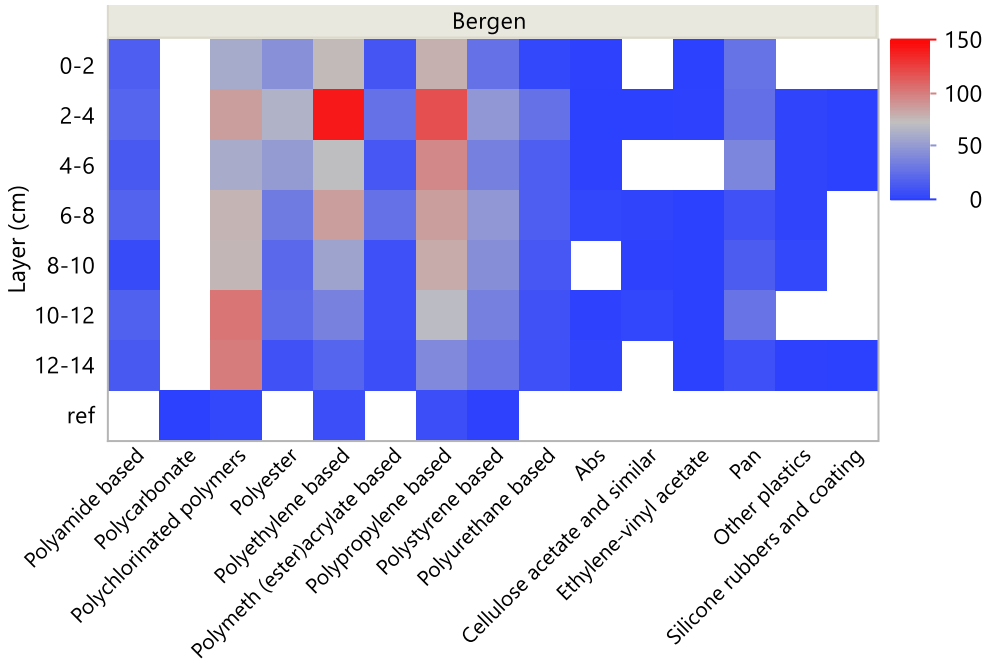


Figure 61. Heatmap polymer categories Bergen

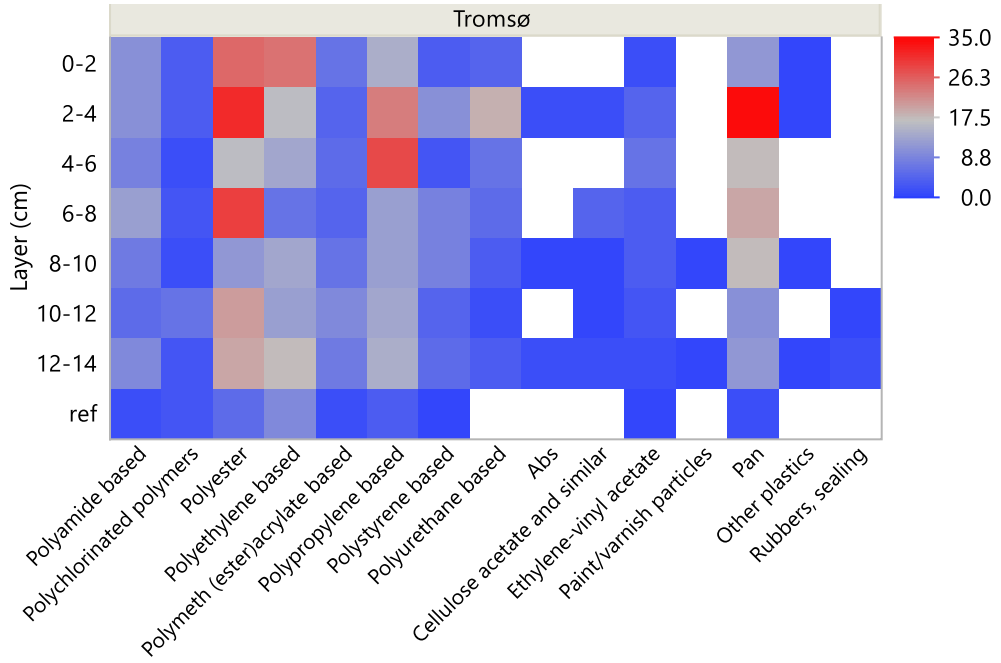


Figure 62. Heatmap polymer categories Tromsø

7.3.2. Manta samples

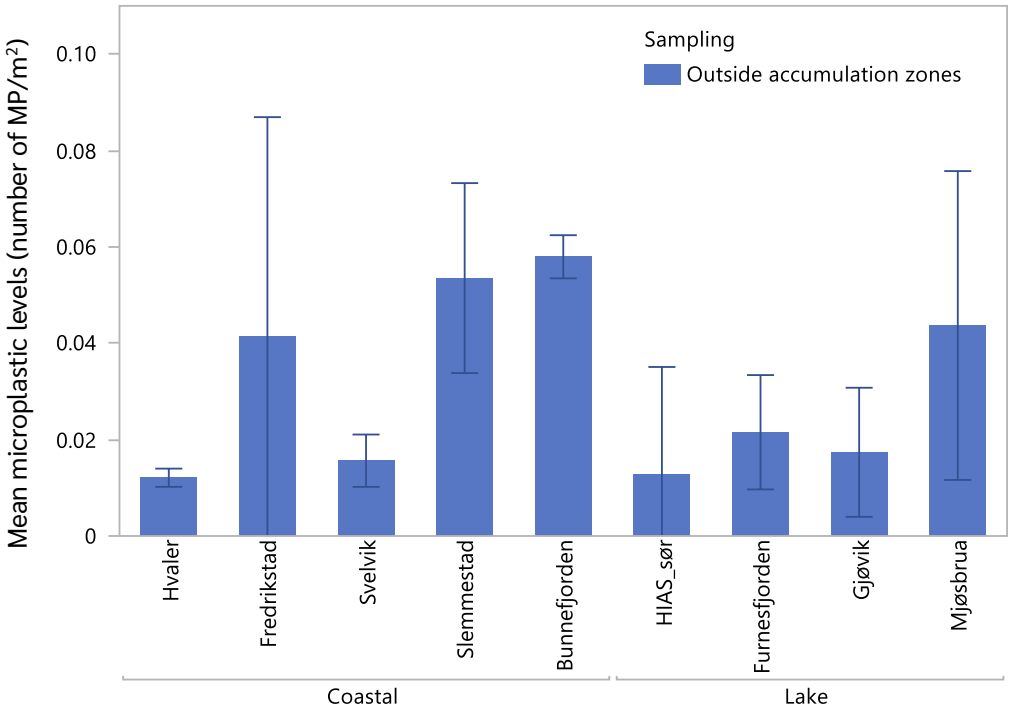


Figure 63. Mean microplastic levels (MP/m²) in coastal and lake (Mjøsa) manta water samples outside accumulation zones. Number of samples per station n=3 except Fredrikstad and Bunnefjorden where n=2. Manta 2024:

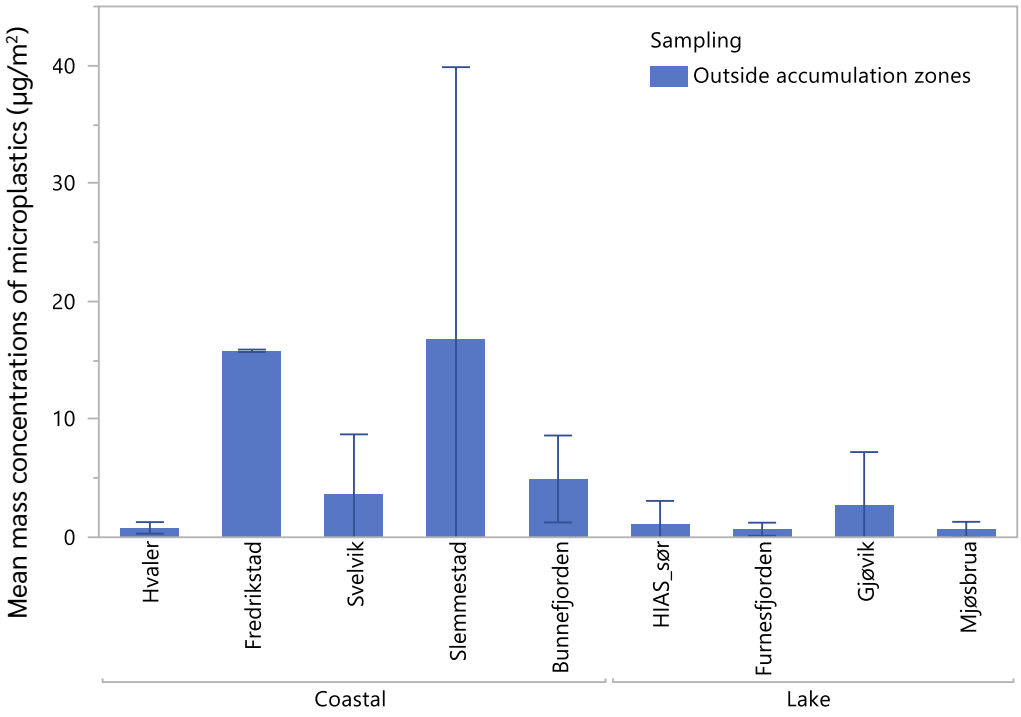


Figure 64. Mean mass concentrations of microplastics (µg/m²) in coastal and lake (Mjøsa) manta water samples outside accumulation zones. Number of samples per station n=3 except Fredrikstad and Bunnefjorden where n=2.

7.3.3. Atmospheric deposition samples

7.3.3.1. Reflections and limitations concerning the analysis of tyre-related chemicals in air samples

In 2024, the analysis of eleven tyre-related organic chemicals in atmospheric deposition samples was offered and chosen as an option within the MIKRONOR program. The objective was to investigate whether these compounds were detectable in this yet understudied sample type, whether a correlation with TWP fluxes could be observed and whether the data might indicate substances which should be prioritized in future monitoring campaigns. Hereby, samples from Zeppelin showed lowest concentrations, followed by Birkenes and Sofienbergparken, the latter of which also showed the highest diversity of detected compounds (**Table 13**). It could further be observed that some compounds were only detected in the water phase, while others were only detected in the ethanol extracts. For example, the highly hydrophilic HMMM (hexamethoxymethylmelamine) was only present in the water phase, while the hydrophobic CPPD (N,N'-dicyclohexyl-p-phenylenediamine) was mainly detected in the ethanol extracts of the particles. However, while the data allowed to illustrate the general differences in sampling sites (i.e., remote vs. semi-remote vs. urban) and chemical properties of target compounds, most compounds were ≤ LOQ and no obvious correlation with TWP fluxes could be observed. This can partly be explained by the instability of certain target chemicals and their high potential for oxidative transformation, yielding an unknown number of transformation products instead (e.g. N-(1,3-dimethylbutyl)-N′-phenyl-p-phenylenediamine, commonly known as 6PPD; Hu et al., 2022). Particularly in atmospheric deposition samples this is aggravated by varying (partly high) levels of UV irradiation during the prolonged sampling periods. The relevance of monitoring tyre-related target chemicals in atmospheric deposition samples thus seems limited at the current stage. Recommendations for the future would include prior and comprehensive spike-and-recovery experiments of target compounds in water in controlled light conditions (e.g., using an UV irradiation chamber) and possibly to prioritize non-target analysis above targeted analysis in order to better understand the transformation processes and investigate the number and occurrence of transformation products.

Table 13. Data table listing tyre-related chemicals (in ng/sample) detected in the water phase (left columns) or ethanol extracts (right columns) of atmospheric deposition samples. Only compounds detected >LOQ are listed.

Sample ID	DPG		HMMM		IPPDQ		DPPD		6-PPD		6-PPDQ		CPPD	
ZD01	<LOQ	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	<LOQ
ZD02	<LOQ	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	<LOQ
ZD03	<LOQ	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.
ZD04	<LOQ	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.
ZD05	<LOQ	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.
ZD06	<LOQ	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.
ZD07	<LOQ	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.
ZD08	<LOQ	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	<LOQ

Sample ID	DPG		HMMM		IPPDQ		DPPD		6-PPD		6-PPDQ		CPPD	
ZD09	<LOQ	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.
ZD10	<LOQ	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.
ZD11	<LOQ	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.
ZD12	<LOQ	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.
ZD13	<LOQ	n.d.	n.d.	n.d.	n.d.	0.13	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.
ZD14	<LOQ	<LOQ	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.
ZD15	<LOQ	n.d.	0.73	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.
ZD16	<LOQ	<LOQ	n.d.	n.d.	n.d.	<LOQ	n.d.	n.d.	n.d.	n.d.	<LOQ	n.d.	n.d.	0.15
ZD17	<LOQ	n.d.	n.d.	n.d.	n.d.	1.45	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	0.16
ZD18	<LOQ	n.d.	n.d.	n.d.	n.d.	0.06	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	0.16
ZD19	<LOQ	<LOQ	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	0.29	n.d.	19.5
ZD20	<LOQ	<LOQ	<LOQ	n.d.	n.d.	0.18	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	0.36
ZD21	<LOQ	<LOQ	n.d.	n.d.	n.d.	<LOQ	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	<LOQ	0.50
ZD22	<LOQ	n.d.	n.d.	n.d.	n.d.	<LOQ	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	1.16
BD01	<LOQ	<LOQ	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.
BD02	<LOQ	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.
BD03	8.66	<LOQ	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.
BD04	12.82	<LOQ	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.
BD05	65.04	<LOQ	n.d.	n.d.	<LOQ	<LOQ	n.d.	n.d.	n.d.	0.41	n.d.	n.d.	n.d.	1.55
BD06	47.95	<LOQ	n.d.	n.d.	n.d.	0.27	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	0.72
BD07	27.38	<LOQ	n.d.	n.d.	n.d.	0.21	n.d.	n.d.	n.d.	1.46	n.d.	n.d.	n.d.	1.77
BD08	68.85	<LOQ	n.d.	n.d.	n.d.	<LOQ	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	0.72
BD09	59.55	<LOQ	n.d.	n.d.	n.d.	<LOQ	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.

Sample ID	DPG		HMMM		IPPDQ		DPPD		6-PPD		6-PPDQ		CPPD	
BD10	45.33	<LOQ	n.d.	n.d.	n.d.	0.71	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.
BD11	10.95	<LOQ	n.d.	n.d.	n.d.	21.1	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	0.67
BD12*	-	-	-	-	-	-	-	-	-	-	-	-	-	-
BD13	85.68	<LOQ	n.d.	n.d.	n.d.	4.73	n.d.	n.d.	n.d.	0.16	n.d.	n.d.	n.d.	0.12
BD14	70.21	<LOQ	<LOQ	n.d.	n.d.	1.22	n.d.	n.d.	n.d.	0.18	n.d.	n.d.	n.d.	1.09
BD15	14.09	<LOQ	0.11	n.d.	n.d.	0.20	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.
BD16	13.59	<LOQ	n.d.	n.d.	n.d.	51.0	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	3.14
BD17	9.47	<LOQ	n.d.	<LOQ	n.d.	0.46	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	0.55
BD18	10.00	<LOQ	n.d.	n.d.	n.d.	1.67	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	0.79
BD19	9.80	n.d.	n.d.	n.d.	n.d.	12.8	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	0.79
BD20	14.99	<LOQ	n.d.	n.d.	n.d.	5.98	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	0.60
BD21	19.09	n.d.	n.d.	n.d.	n.d.	0.67	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.
BD22	14.53	n.d.	n.d.	n.d.	n.d.	1.83	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	1.01
BD23	<LOQ	<LOQ	n.d.	n.d.	n.d.	2.01	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	0.66
SD01	23.5	<LOQ	4.09	n.d.	n.d.	0.44	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	46.9
SD02	54.8	<LOQ	2.16	<LOQ	n.d.	0.37	n.d.	16.3	n.d.	n.d.	n.d.	n.d.	n.d.	16.2
SD03	<LOQ	<LOQ	0.74	n.d.	n.d.	0.04	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	15.3
SD04	53.5	<LOQ	1.83	n.d.	n.d.	1.66	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	0.23	99.7
SD05	37.8	<LOQ	0.76	n.d.	n.d.	0.18	n.d.	n.d.	n.d.	n.d.	n.d.	<LOQ	n.d.	29.4
SD06	13.6	<LOQ	1.10	n.d.	n.d.	1.53	n.d.	n.d.	n.d.	n.d.	n.d.	0.19	0.13	77.0
SD07	20.4	<LOQ	1.79	n.d.	n.d.	0.90	n.d.	1.19	n.d.	n.d.	n.d.	n.d.	<LOQ	24.4
SD08	15.6	<LOQ	1.10	n.d.	n.d.	14.18	n.d.	0.13	n.d.	n.d.	n.d.	0.96	0.10	31.4

Sample ID	DPG		HMMM		IPPDQ		DPPD		6-PPD		6-PPDQ		CPPD	
SD09	78.2	<LOQ	1.03	n.d.	n.d.	16.30	n.d.	n.d.	n.d.	n.d.	<LOQ	2.93	n.d.	2.01
SD10	229	<LOQ	0.29	n.d.	n.d.	0.14	n.d.	n.d.	n.d.	0.20	n.d.	n.d.	n.d.	0.90
SD11	208	<LOQ	0.54	n.d.	n.d.	12.27	0.07	n.d.	n.d.	2.10	n.d.	0.37	<LOQ	30.0
SD12	517	<LOQ	1.25	n.d.	n.d.	9.15	n.d.	0.14	n.d.	17.0	<LOQ	0.97	0.60	36.8
SD13	91.4	<LOQ	13.4	n.d.	n.d.	0.14	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	1.57	17.9
SD14	330	<LOQ	4.89	n.d.	n.d.	0.11	n.d.	n.d.	n.d.	15.9	n.d.	0.49	0.88	32.5
SD15	259	<LOQ	0.65	n.d.	n.d.	0.50	n.d.	0.32	n.d.	30.6	n.d.	0.81	<LOQ	52.9
SD16	40.5	<LOQ	0.16	n.d.	n.d.	0.20	n.d.	n.d.	n.d.	1.42	n.d.	0.25	<LOQ	28.6
SD17	497	<LOQ	1.63	n.d.	n.d.	1.53	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	0.19	77.1
SD18	96.5	<LOQ	0.53	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	<LOQ	92.2
SD19	212	<LOQ	1.72	n.d.	n.d.	0.21	n.d.	n.d.	n.d.	n.d.	n.d.	0.68	1.53	253
SD20	239	<LOQ	n.d.	n.d.	n.d.	0.00	n.d.	n.d.	n.d.	n.d.	<LOQ	0.27	n.d.	65.9
SD21	26.4	<LOQ	4.65	n.d.	n.d.	1.45	n.d.	n.d.	n.d.	n.d.	0.12	1.86	1.31	282
SD22	192	<LOQ	4.51	<LOQ	n.d.	0.93	n.d.	n.d.	n.d.	n.d.	1.23	0.54	2.16	116

*Note that sample BD12 was lost due to instrumental issues.

Data table concerning atmospheric microplastic deposition

Table 14. Data table listing all microplastic deposition fluxes (in µg/m²/d) measured in samples from Zeppelin, Birkenes and Sofienbergparken. Values ≤ LOD are highlighted in light grey, values ≤ LOQ are highlighted in dark grey.

Sample ID	Sampling start	Sampling end	PMMA	PP	PVC	TWP	PA	PU	PC	PE	PS	PET	Sum
Zeppelin station													
ZD01	01.02.2024	15.02.2024	0.89	0.30	2.63	n.d.	0.21	n.d.	n.d.	n.d.	n.d.	n.d.	**4.03**
ZD02	15.02.2024	28.02.2024	1.37	1.89	5.70	n.d.	0.04	n.d.	n.d.	n.d.	0.21	1.02	**10.2**
ZD03	28.02.2024	13.03.2024	0.74	0.25	2.70	n.d.	n.d.	n.d.	n.d.	n.d.	0.05	0.24	**3.98**
ZD04	13.03.2024	27.03.2024	0.57	0.88	2.99	n.d.	n.d.	n.d.	n.d.	n.d.	0.01	n.d.	**4.44**
ZD05	27.03.2024	10.04.2024	1.60	6.53	11.4	n.d.	0.17	0.01	n.d.	n.d.	0.37	3.76	**23.8**
ZD06	10.04.2024	25.04.2024	0.81	8.95	11.9	n.d.	0.11	0.02	n.d.	n.d.	0.06	0.62	**22.4**
ZD07	25.04.2024	08.05.2024	0.34	0.02	3.87	n.d.	1.00	n.d.	n.d.	n.d.	n.d.	n.d.	**5.23**
ZD08	08.05.2024	21.05.2024	0.94	0.88	34.0	n.d.	0.03	0.10	0.29	n.d.	0.15	1.14	**37.6**
ZD09	21.05.2024	05.06.2024	0.76	2.45	20.8	13.7	n.d.	0.18	n.d.	2.90	0.79	4.70	**46.3**
ZD10	05.06.2024	19.06.2024	0.12	0.40	4.96	n.d.	0.58	0.03	n.d.	2.03	0.17	0.65	**8.94**
ZD11	19.06.2024	03.07.2024	0.15	4.91	3.29	29.43	n.d.	0.16	n.d.	2.81	0.12	0.50	**41.4**
ZD12	03.07.2024	17.07.2024	0.08	3.86	2.10	n.d.	n.d.	0.10	n.d.	n.d.	0.17	n.d.	**6.30**
ZD13	17.07.2024	31.07.2024	0.07	3.26	3.17	n.d.	n.d.	0.09	n.d.	n.d.	0.16	n.d.	**6.75**
ZD14	31.07.2024	14.08.2024	0.84	3.18	14.6	n.d.	0.70	0.32	n.d.	6.48	0.14	n.d.	**26.2**
ZD15	14.08.2024	28.08.2024	0.56	5.21	6.74	n.d.	0.40	0.27	n.d.	n.d.	n.d.	n.d.	**13.2**
ZD16	28.08.2024	11.09.2024	1.35	7.65	43.2	n.d.	0.90	0.53	n.d.	n.d.	0.17	0.46	**54.3**

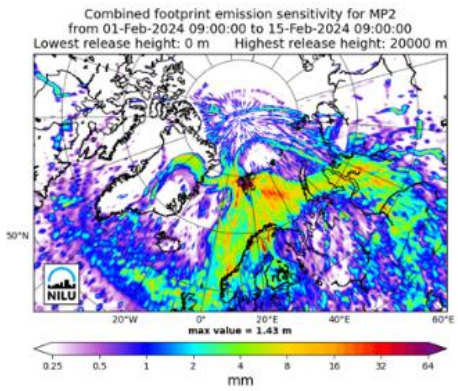
Sample ID	Sampling start	Sampling end	PMMA	PP	PVC	TWP	PA	PU	PC	PE	PS	PET	Sum
ZD17	11.09.2024	25.09.2024	3.39	6.15	35.2	n.d.	0.43	0.30	n.d.	n.d.	0.01	n.d.	**45.4**
ZD18	25.09.2024	09.10.2024	1.36	5.37	20.2	n.d.	0.41	0.10	n.d.	n.d.	1.04	5.36	**33.8**
ZD19	09.10.2024	23.10.2024	2.22	17.4	136	n.d.	0.38	0.28	2.26	2.79	0.69	3.44	**165**
ZD20	23.10.2024	07.11.2024	24.8	5.73	55.9	n.d.	0.60	0.13	n.d.	n.d.	1.55	7.74	**96.4**
ZD21	07.11.2024	20.11.2024	1.05	7.28	12.1	n.d.	0.14	0.12	n.d.	1.65	0.07	0.21	**22.6**
ZD22	20.11.2024	04.12.2024	1.76	13.05	46.5	n.d.	0.95	0.38	n.d.	2.81	2.24	13.1	**80.7**
Birkenes station													
BD01	18.01.2024	31.01.2024	1.31	2.08	85.5	8.50	0.07	0.14	13.0	6.54	0.80	6.09	**124**
BD02	31.01.2024	14.02.2024	1.42	0.46	63.2	n.d.	0.05	0.22	1.36	14.1	1.02	10.0	**91.8**
BD03	14.02.2024	28.02.2024	0.35	0.80	13.5	15.3	0.19	0.06	0.45	1.81	0.15	n.d.	**32.6**
BD04	28.02.2024	13.03.2024	0.88	0.71	35.4	30.0	0.04	0.23	1.17	10.5	0.24	1.78	**80.9**
BD05	13.03.2024	27.03.2024	0.42	n.d.	10.2	68.9	n.d.	0.05	n.d.	n.d.	n.d.	n.d.	**79.5**
BD06	27.03.2024	10.04.2024	0.71	0.21	16.1	88.6	n.d.	0.08	n.d.	0.56	0.88	5.16	**112**
BD07	10.04.2024	24.04.2024	0.84	n.d.	3.49	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	**4.34**
BD08	24.04.2024	08.05.2024	0.33	n.d.	16.9	114	n.d.	0.01	3.87	n.d.	0.08	n.d.	**135**
BD09	08.05.2024	22.05.2024	n.d.	n.d.	41.8	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	**41.8**
BD10	22.05.2024	05.06.2024	0.19	n.d.	16.1	147	n.d.	0.01	n.d.	n.d.	n.d.	n.d.	**163**
BD11	05.06.2024	19.06.2024	0.88	0.54	52.9	49.8	1.20	0.37	n.d.	7.25	n.d.	2.86	**116**
BD12*	19.06.2024	03.07.2024	-	-	-	-	-	-	-	-	-	-	-
BD13	03.07.2024	17.07.2024	4.64	n.d.	53.7	71.4	n.d.	0.22	n.d.	n.d.	n.d.	n.d.	**130**
BD14	17.07.2024	31.07.2024	1.40	n.d.	36.2	29.5	n.d.	0.24	32.1	9.32	0.87	9.21	**119**

Sample ID	Sampling start	Sampling end	PMMA	PP	PVC	TWP	PA	PU	PC	PE	PS	PET	Sum
BD15	31.07.2024	14.08.2024	3.86	n.d.	47.4	n.d.	1.23	0.15	11.4	3.78	n.d.	n.d.	**67.8**
BD16	14.08.2024	28.08.2024	2.79	0.32	73.5	n.d.	0.55	0.30	n.d.	6.75	n.d.	4.15	**88.3**
BD17	28.08.2024	11.09.2024	3.04	0.25	33.8	n.d.	0.16	0.10	n.d.	4.91	0.62	n.d.	**42.9**
BD18	11.09.2024	25.09.2024	0.63	0.46	16.1	12.3	0.33	0.14	n.d.	3.19	n.d.	2.78	**35.9**
BD19	25.09.2024	09.10.2024	1.10	n.d.	14.8	10.2	0.59	0.14	n.d.	3.56	n.d.	2.76	**33.2**
BD20	09.10.2024	23.10.2024	2.79	n.d.	33.4	36.7	7.67	0.40	n.d.	2.90	0.13	7.28	**91.3**
BD21	23.10.2024	06.11.2024	1.25	0.02	43.8	21.9	2.22	0.24	1.22	4.20	0.12	2.02	**77.1**
BD22	06.11.2024	20.11.2024	1.72	n.d.	37.1	4.34	2.72	0.45	0.36	0.73	0.12	6.20	**53.8**
BD23	20.11.2024	04.12.2024	1.53	0.13	27.5	n.d.	2.94	0.14	3.81	1.61	0.14	2.00	**39.8**
Sofienbergparken station													
SD01	17.01.2024	31.01.2024	15.9	3.15	121	4.07	5.17	0.20	69.3	7.91	1.98	20.3	**249**
SD02	31.01.2024	14.02.2024	5.28	3.30	88.4	49.1	0.94	0.26	10.5	7.47	2.16	20.4	**188**
SD03	14.02.2024	28.02.2024	0.69	0.10	12.8	9.30	0.17	0.14	n.d.	0.57	0.34	8.29	**32.4**
SD04	28.02.2024	13.03.2024	2.69	5.49	232	173	5.90	0.30	92.8	6.22	7.26	117	**642**
SD05	13.03.2024	27.03.2024	1.42	2.73	54.1	75.0	1.35	0.28	5.46	5.02	1.92	18.8	**166**
SD06	27.03.2024	10.04.2024	4.12	5.75	347	159	0.71	0.32	37.6	12.5	9.18	194	**770**
SD07	10.04.2024	24.04.2024	3.10	3.55	82.4	43.7	0.57	0.13	9.63	8.13	1.21	15.8	**168**
SD08	24.04.2024	08.05.2024	2.99	3.62	82.7	329	0.34	0.10	20.3	11.2	2.33	54.5	**507**
SD09	08.05.2024	22.05.2024	n.d.	33.7	90.8	725	n.d.	0.33	n.d.	n.d.	3.46	n.d.	**854**
SD10	22.05.2024	05.06.2024	0.69	0.49	23.0	n.d.	0.30	0.10	n.d.	5.92	n.d.	2.35	**32.8**
SD11	05.06.2024	19.06.2024	2.39	3.67	40.2	347	0.21	0.10	16.1	10.6	0.83	14.2	**435**
SD12	19.06.2024	03.07.2024	2.96	4.94	89.7	740	2.03	0.29	n.d.	7.17	1.98	17.3	**867**

Sample ID	Sampling start	Sampling end	PMMA	PP	PVC	TWP	PA	PU	PC	PE	PS	PET	Sum
SD13	03.07.2024	17.07.2024	4.76	4.32	50.4	290	2.59	0.27	n.d.	6.25	0.84	5.12	**365**
SD14	17.07.2024	31.07.2024	1.57	4.23	48.3	269	3.65	0.30	n.d.	6.40	1.54	8.59	**343**
SD15	01.08.2024	15.08.2024	2.59	3.94	74.7	470	1.33	0.34	n.d.	10.8	2.02	12.4	**578**
SD16	15.08.2024	28.08.2024	1.22	0.59	42.3	340	3.38	0.25	n.d.	4.65	1.75	9.51	**403**
SD17	28.08.2024	11.09.2024	4.10	2.29	68.1	608	124	0.38	n.d.	13.6	2.20	14.2	**837**
SD18	11.09.2024	25.09.2024	2.55	2.66	49.8	573	0.70	0.30	n.d.	6.80	1.28	8.31	**645**
SD19	25.09.2024	09.10.2024	1.76	7.16	56.1	729	224	0.32	n.d.	6.78	1.57	8.71	**1035**
SD20	09.10.2024	23.10.2024	1.68	4.28	42.2	519	4.29	0.32	n.d.	7.74	0.96	6.35	**586**
SD21	23.10.2024	06.11.2024	3.72	5.52	199	1261	6.74	0.50	n.d.	11.8	2.45	23.2	**1514**
SD22	06.11.2024	20.11.2024	2.08	5.74	114	1148	118	0.51	n.d.	13.4	3.85	21.9	**1429**

*Note that sample BD12 was lost due to instrumental issues.

ZD01



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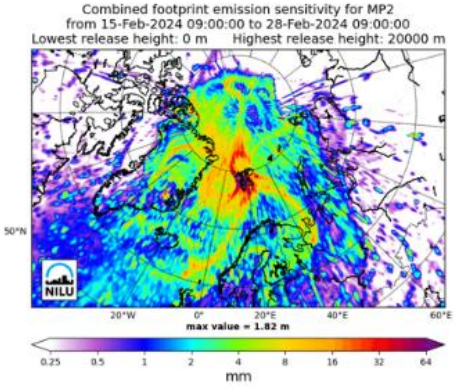

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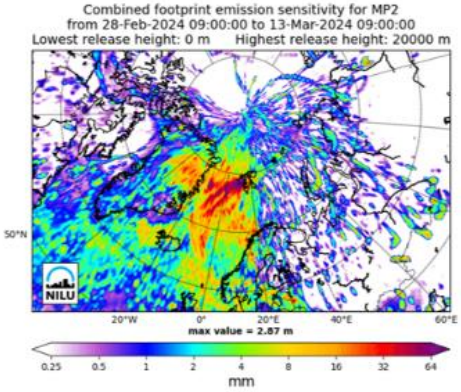

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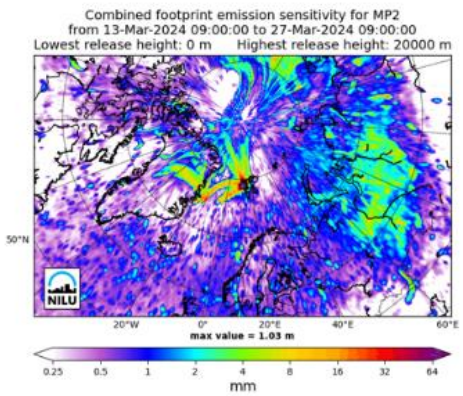

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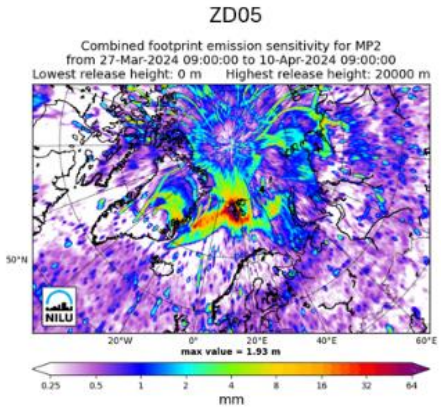

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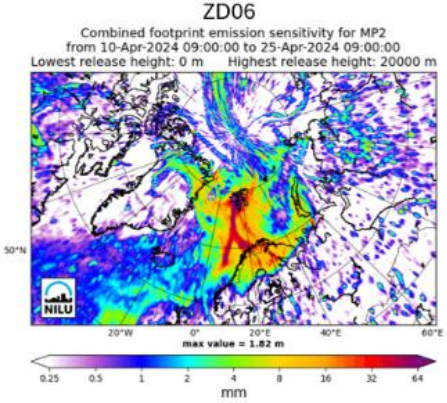

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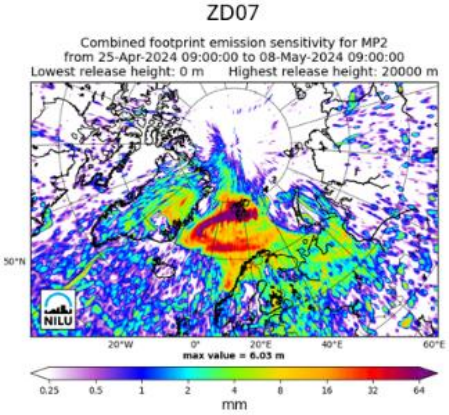

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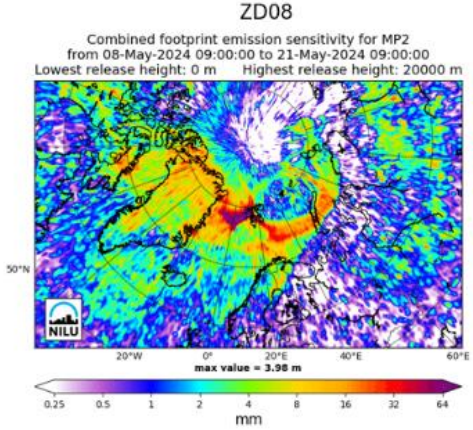
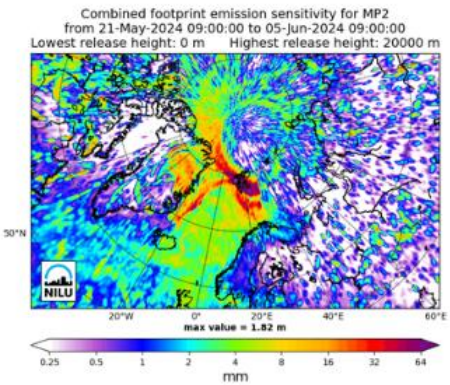


ZD09



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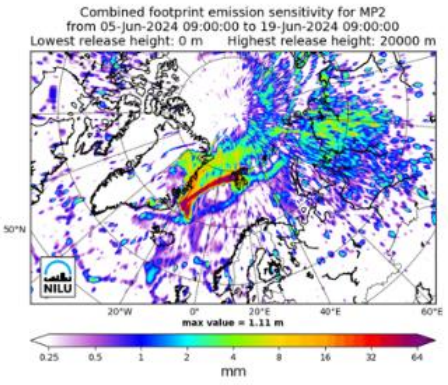
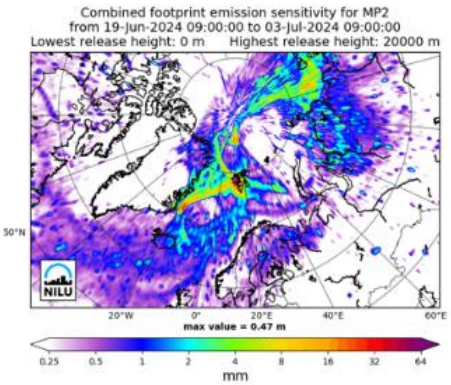
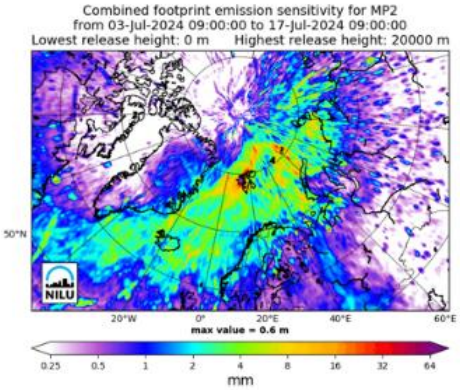


ZD11



ZD12



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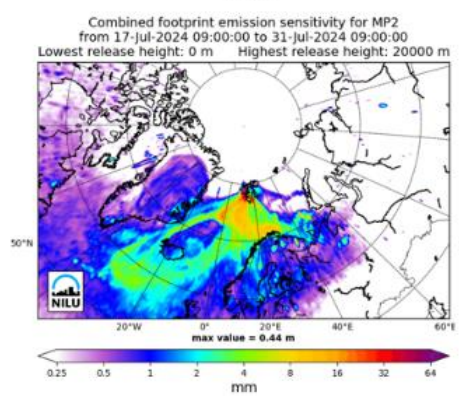
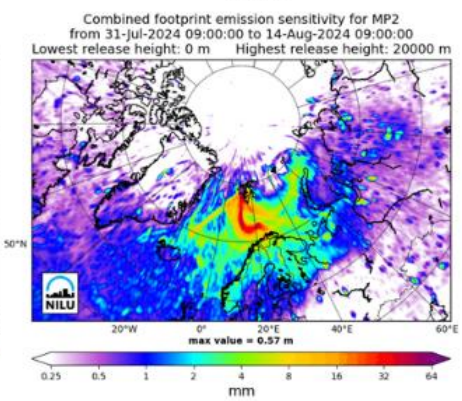
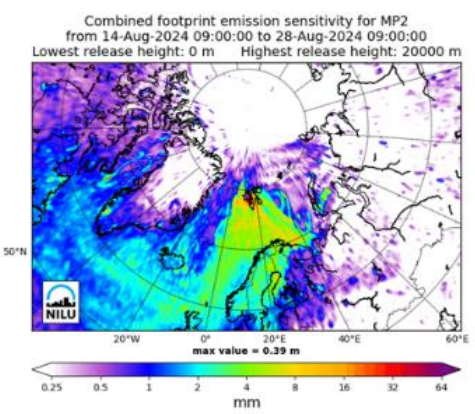


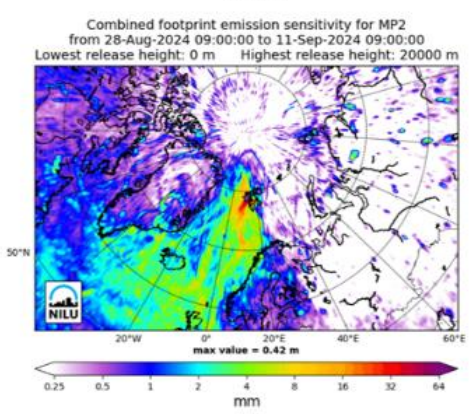
ZD14



ZD15



ZD16



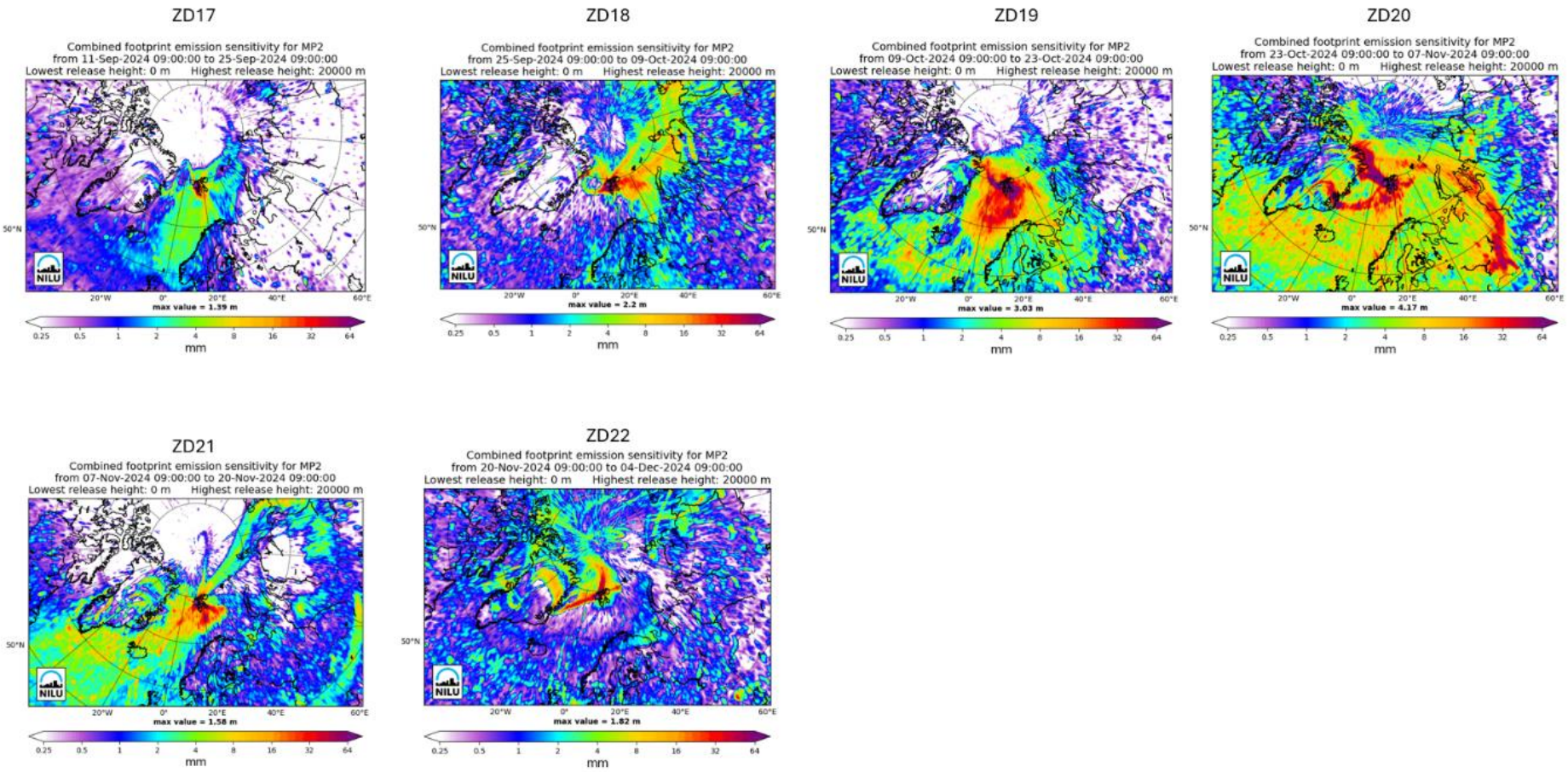
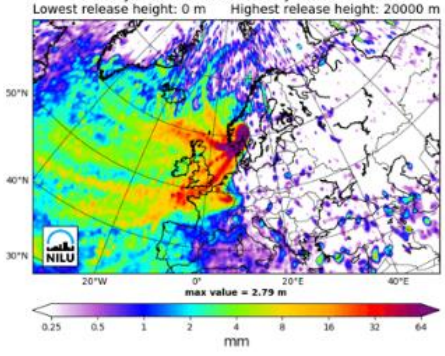
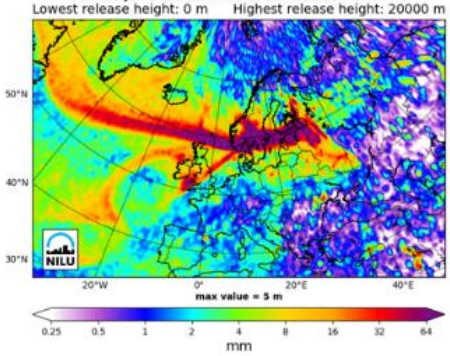


Figure 65: Emission footprint of microplastics observed in Zeppelin station during the 22 sampling periods. Simulations were made using FLEXPART and developed by N. Evangeliou (ne@nilu.no) and S. Eckhardt (sec@nilu.no) (NILU) under ATMO-ACCESS, EU grant agreement No 101008004.

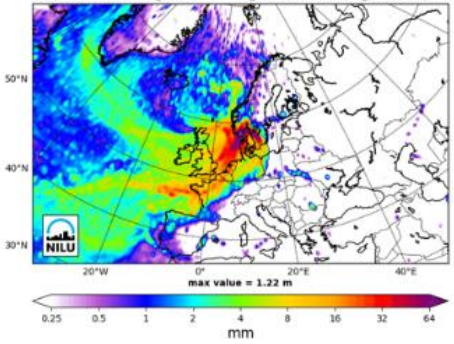
BD01



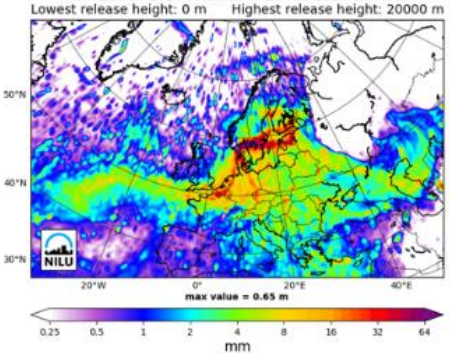
BD02



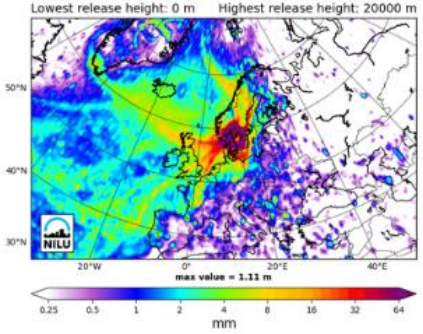
BD03



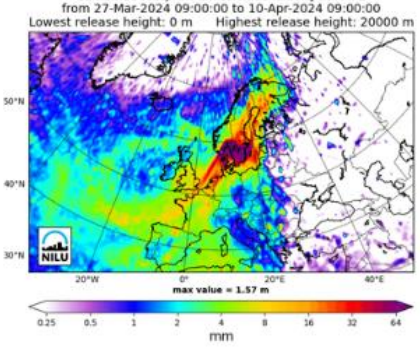
BD04



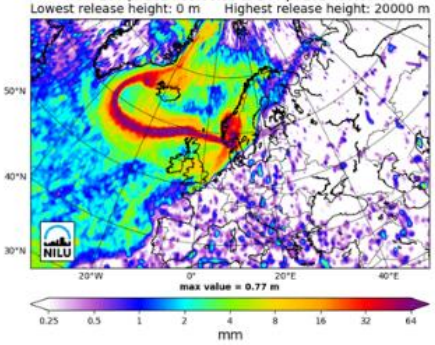
BD05



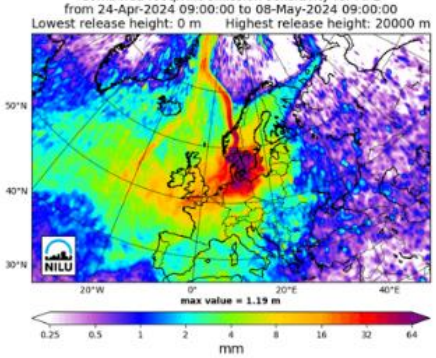
BD06



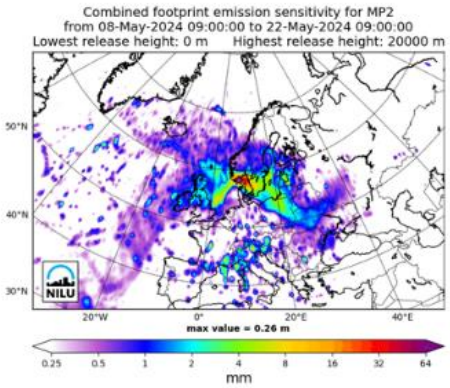
BD07



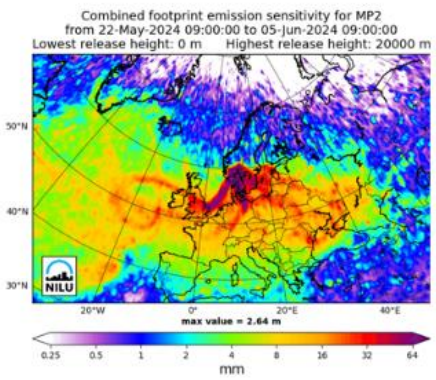
BD08



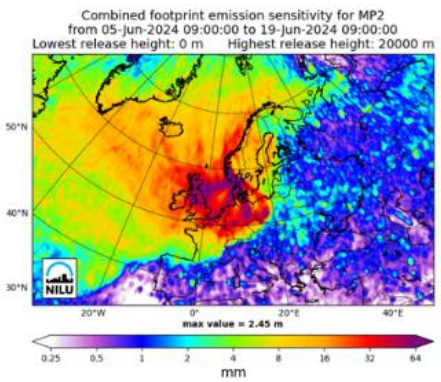
BD09



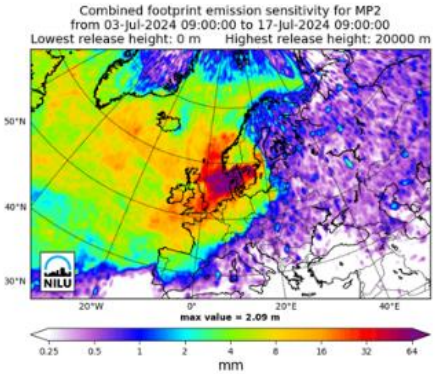
BD10



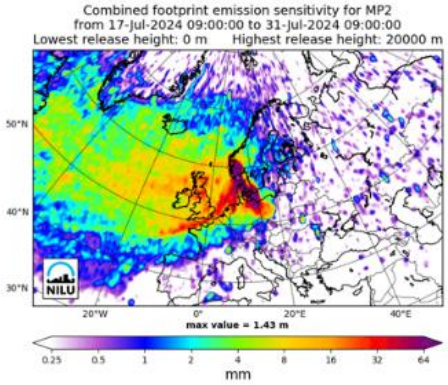
BD11



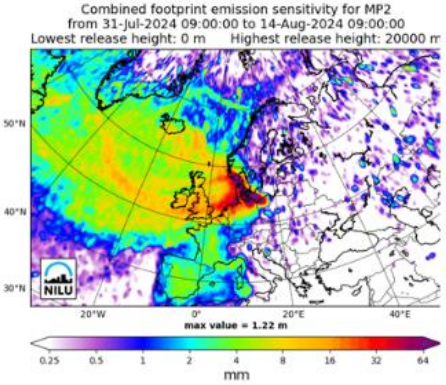
BD13



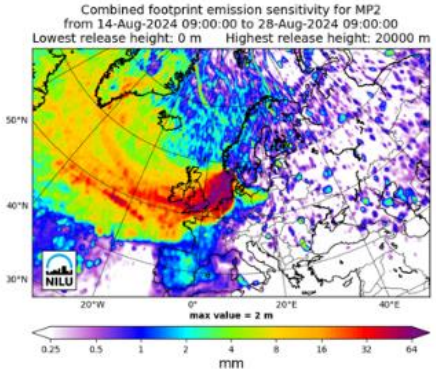
BD14



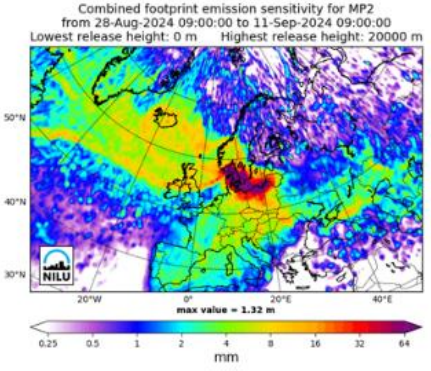
BD15



BD16



BD17



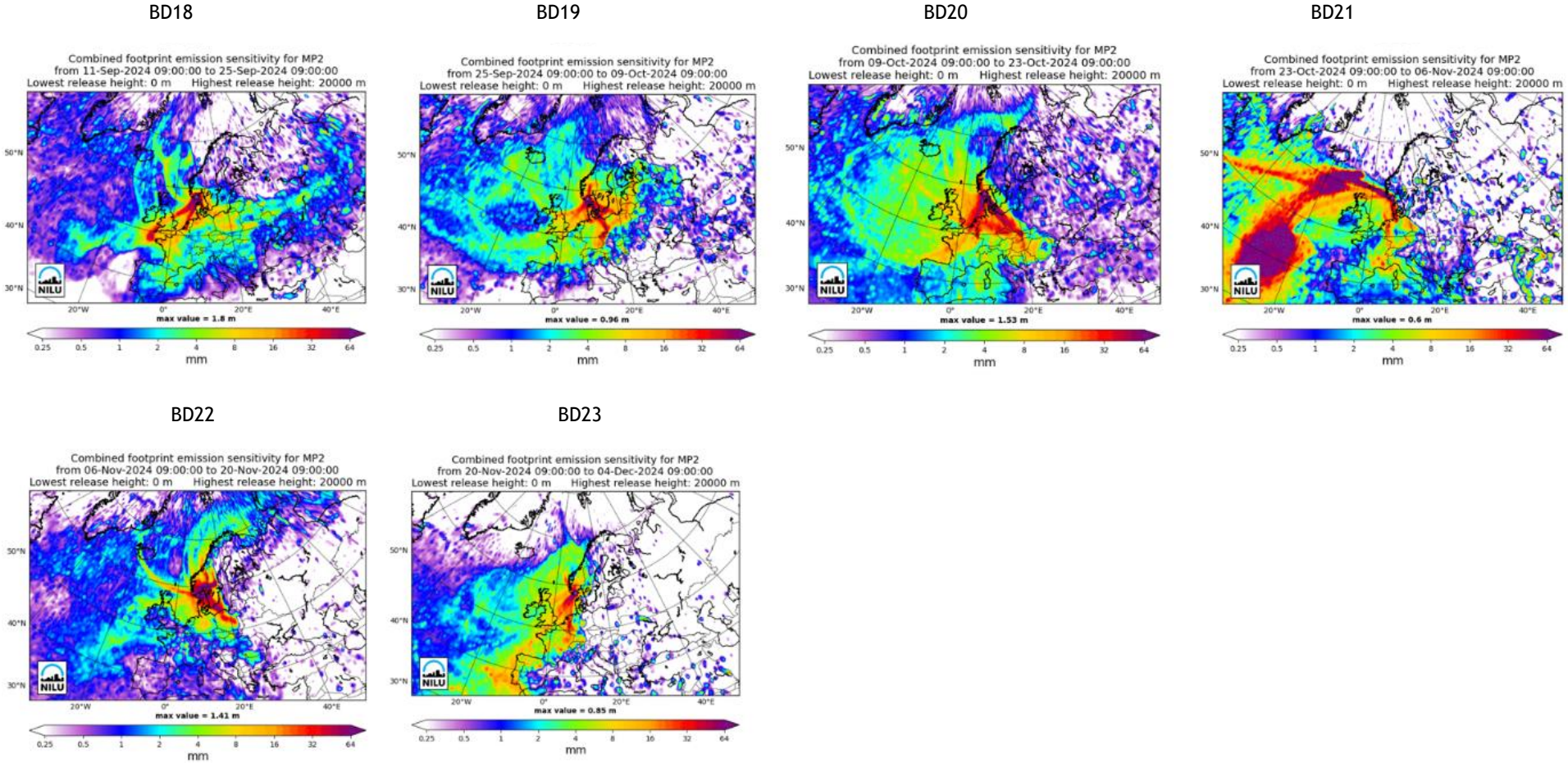


Figure 66. Emission footprint of microplastics observed in Birkenes station during the 22 sampling periods. Simulations were made using FLEXPART and developed by N. Evangeliou (ne@nilu.no) and S. Eckhardt (sec@nilu.no) (NILU) under ATMO-ACCESS, EU grant agreement No 101008004.

7.3.4. Beaches

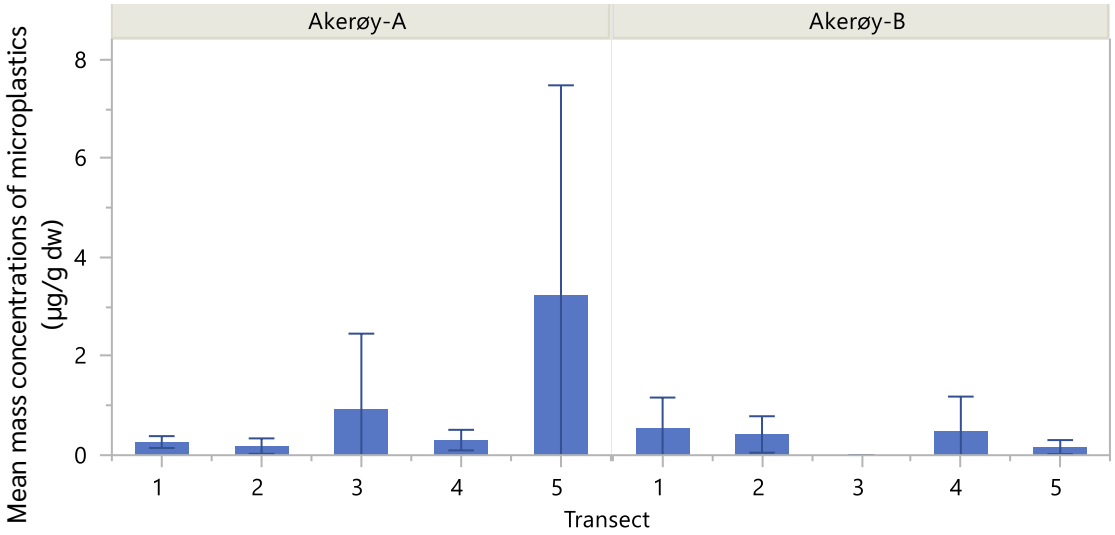


Figure 67. Mean microplastic mass (µg/g dw) per transect for Akerøya-A and Akerøya-B in 2024.

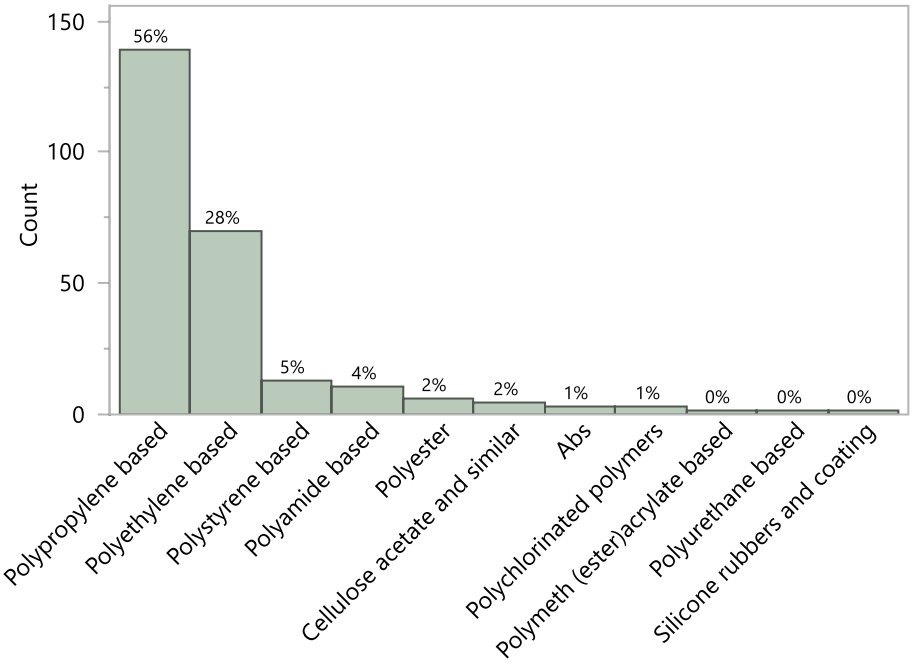


Figure 68. Akerøy-A: polymer types in beach samples.

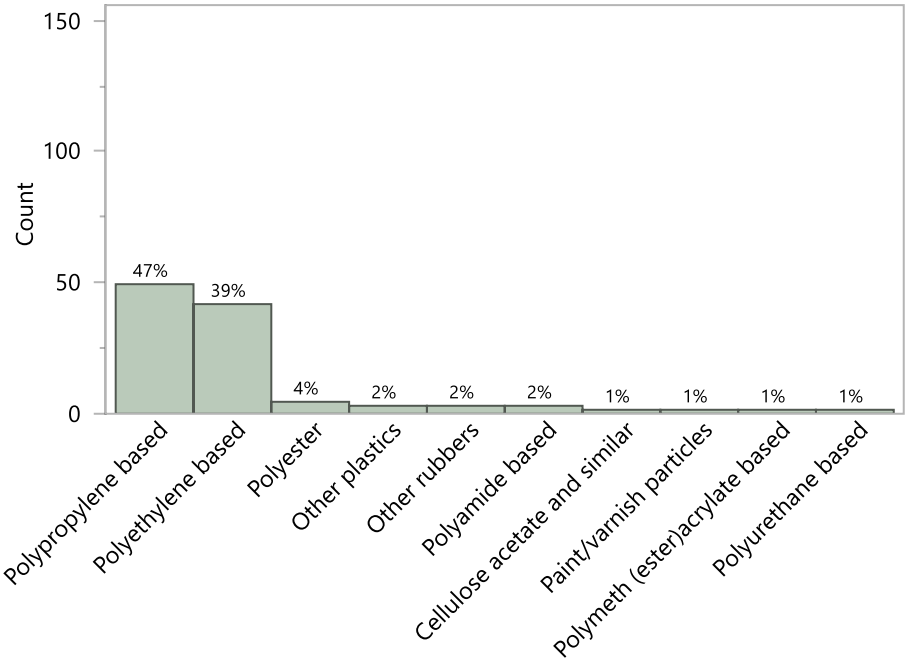


Figure 69. Akerøy-B: polymer types in beach samples.

7.3.5. Blue mussels

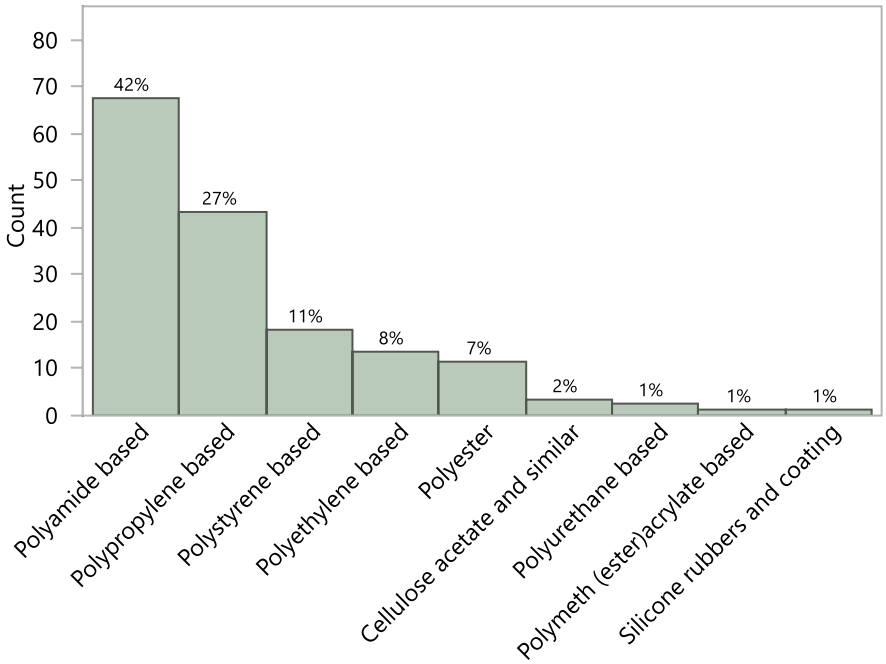


Figure 70. Distribution of polymer categories of microplastic particles in blue mussel in 2024.

7.3.6. Power analyses

An analysis of statistical power for sediment sampling in the Mikronor 1 (Bavel, et al., 2022) was performed in 2024, in order to determine the sample size required to detect statistically significant spatial and temporal differences of tire wear particle (TWP) concentrations with 80% and 95% power.

7.3.6.1. Background

Sediment sampling and analysis is expensive due to the complex sample preparation and analysis, and the high cost of specialists who can process them.

In previous Mikronor sampling in 2021 and 2022, three grab samples were collected at each sediment station, but only one of these was analysed for TWP.

A Monte Carlo simulation was used to estimate the distribution of concentrations of tyre wear particles (TWP) in the sample, based on the rubber concentration found in the sample by pyrolysis.

Questions to answer about TWP in sediments

How many samples are required to be able to say if geographical areas have statistically significant differences in concentrations?

How many samples are required to be able to say if there has been a statistically significant change over time for a given geographic location?

7.3.6.2. Approach

To answer question 1, we have considered each set of three transect samples taken in Oslofjord as separate geographical locations. So, we have estimated the sample size required to detect difference between Akershuskaia and Bekkelaget. For freshwater sediments, we have done the same for the transects taken in Mjøsa, comparing Mjøsbru and Hamar. We have calculated the required sample size to say with 80% and 95% confidence that the concentrations of TWP at these locations are significantly different from one other.

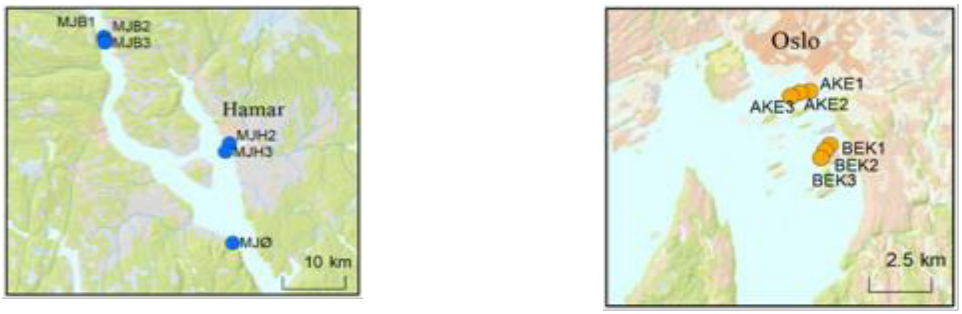


Figure 71. Left: Locations in Lake Mjøsa used for freshwater sediment spatial variation analysis. The most southerly location labelled 'MJØ' was not included. **Right**: Locations in Oslofjord used for marine sediment spatial variation analysis.

We don't yet have TWP results for the same locations in multiple years. So, to answer question 2, we have assumed that the distributions from the Monte Carlo simulations of TWP concentrations are representative of that site. As a result, sites with very low concentrations are difficult to do a sensible analysis with, due to the high standard deviation relative to their mean. With more data from these locations, we will be able to improve on this.

For each station where TWP was detected, we have calculated the required sample size to say with 80% and 95% confidence that there has been a change in TWP concentrations of 10% or 30% if we were to sample them again.

All analysis used the Python package TTestIndPower from statsmodels, a well-maintained and industry standard Python package.

7.3.6.3. Results

Marine Sediments

Detecting Differences Between Locations

Required sample size at Bekkelaget and Akershuskaia to say they are significantly different from one another:

	80% power	95% power
Required sample size	3.89	5.01

While the total quantities of TWP in the samples within the Oslofjord are generally higher than other marine sediment stations we have measured, we will assume the variability of their distributions to be representative of other marine sediment stations.

Detecting Changes over Time

The required sample sizes to see changes over time of 10% and 30%, with a statistical power of 80% and 95%, in each of the marine sediment sampling locations we have visited in 2021 and 2022, are illustrated by the following charts.

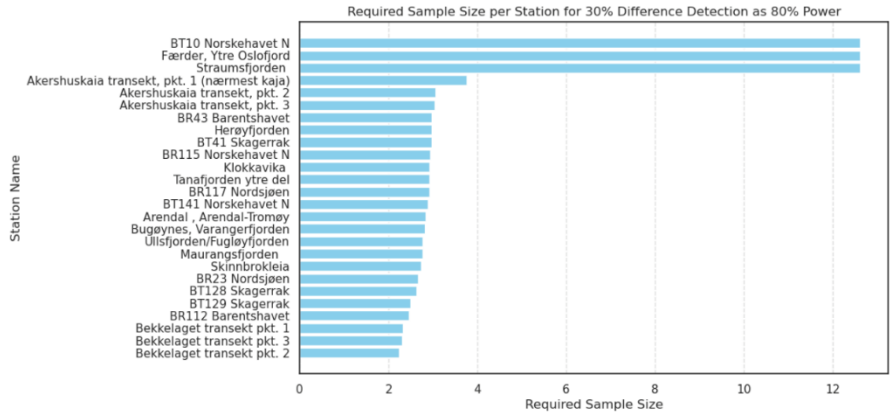


Figure 72. Required sample size to detect a 30% change over time with 80% statistical power for marine sediment monitoring stations. The much higher sample sizes required at Norskehavet N, Færder and Straumsfjorden, are due to these samples being below the detection limit for TWP. These have been excluded for the 10% difference detection.

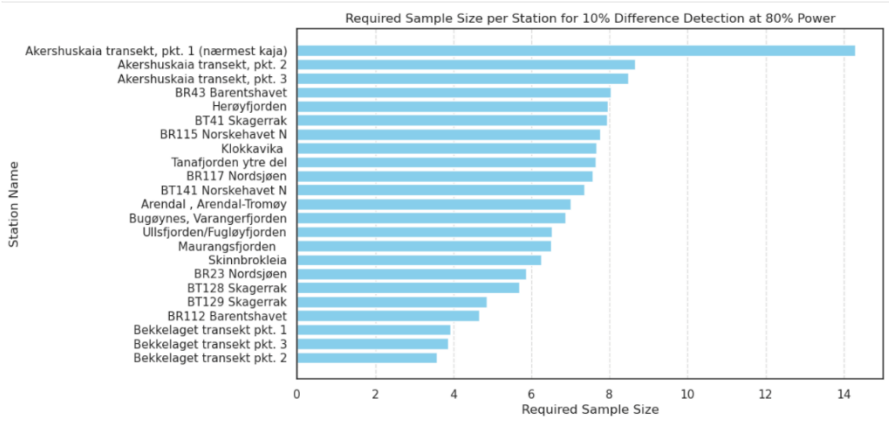


Figure 73. Required sample size to detect a 10% change over time with 80% statistical power for marine sediment monitoring stations.

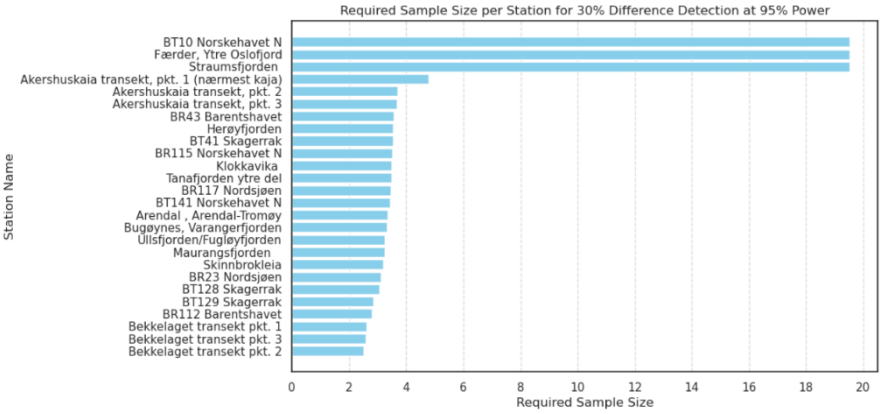


Figure 74. Required sample size to detect a 30% change over time with 95% statistical power for marine sediment monitoring stations.

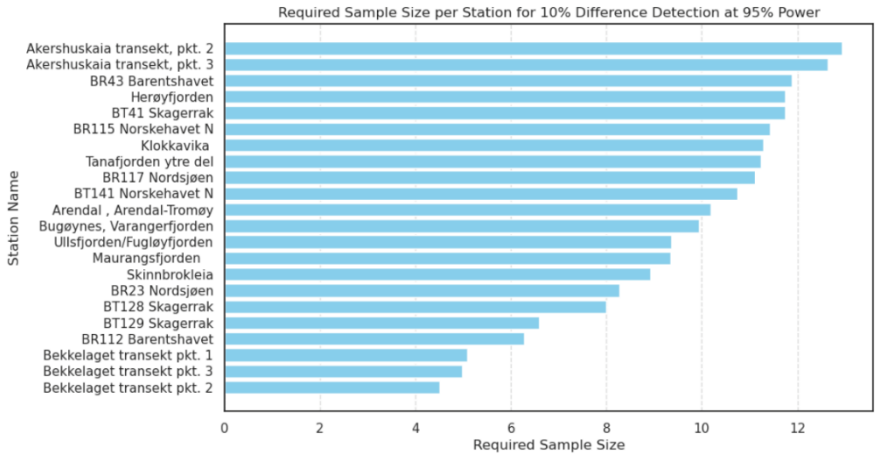


Figure 75. Required sample size to detect a 10% change over time with 95% statistical power for marine sediment monitoring stations.

Freshwater Sediments

Detecting Differences Between Locations

Required sample size at Hamar and Mjøsbru to say with 80% and 95% power that they are different from one another:

	80% power	95% power
Required sample size	2.98	3.57

We will assume the variability of the distributions within Mjøsa to be representative of other freshwater sediment stations.

Detecting Changes over Time

The required sample sizes to see changes over time of 10% and 30%, with a statistical power of 80% and 95%, in each of the freshwater sediment sampling locations we have visited in 2021 and 2022, are illustrated by the following charts.

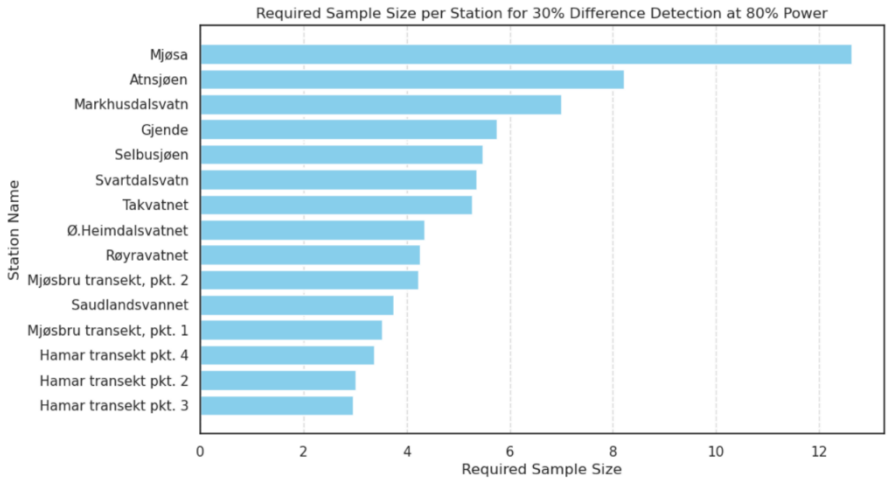


Figure 76. Required sample size to detect a 30% change over time with 80% statistical power for freshwater sediment monitoring stations.

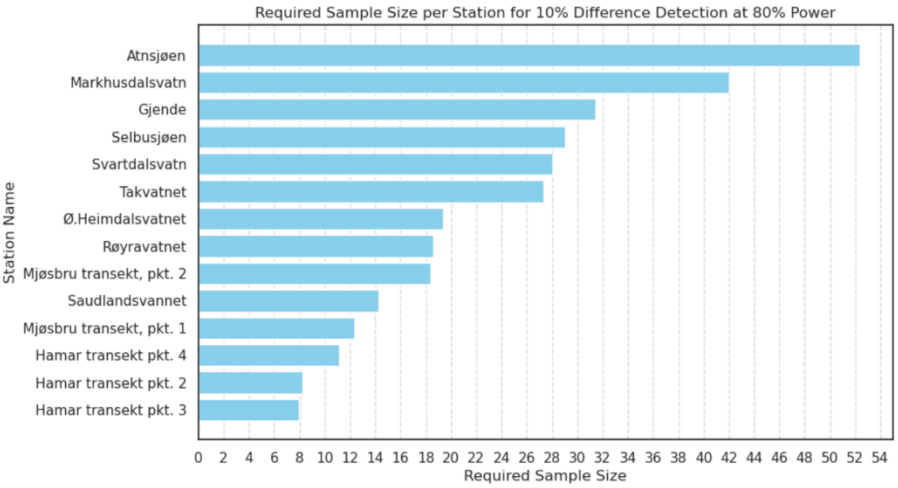


Figure 77. Required sample size to detect a 10% change over time with 80% statistical power for freshwater sediment monitoring stations.

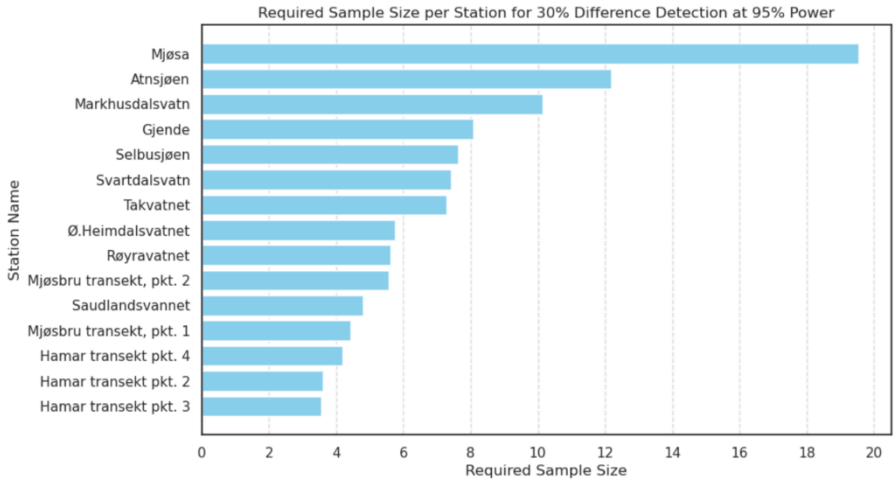


Figure 78. Required sample size to detect a 30% change over time with 95% statistical power for freshwater sediment monitoring stations.

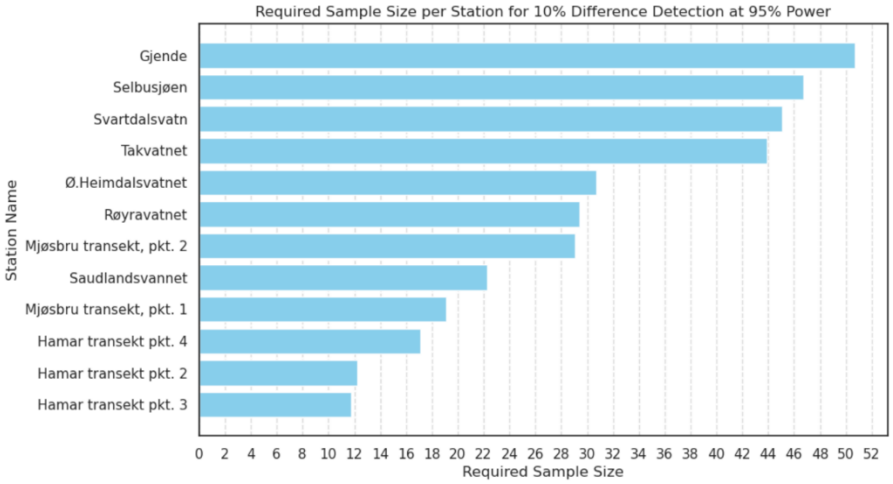


Figure 79. Required sample size to detect a 10% change over time with 95% statistical power for freshwater sediment monitoring stations.

7.3.6.4. Discussion/Conclusion

Due to the very limited data from which to estimate the actual distribution of concentrations of TWP, there are large uncertainties in the data. This means that when the concentrations are low, as we have seen in some freshwater lakes, effect sizes used in the power equation have been very low, and so unrealistic sample sizes are seen for these stations. We also do not have a lot of knowledge yet of the behaviour and distribution of TWP in the environment. This limits our ability to use prior knowledge to inform these analyses. Until more data is available, we will make the assumptions we can in order to inform sampling strategies for further work. We want to reiterate the importance of replication in sampling strategies in the future for Mikronor. However, despite the limited historical data and sample sizes, there are some clear signals from those locations where enough material was gathered to generate a sensible statistical distribution.

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