Report No. 430

epa Research

Sources, Pathways and Environmental Fate of Microplastics

Authors: Roisin Nash, John O'Sullivan, Sinead Murphy, Michael Bruen, Anne Marie Mahon, Heather Lally, Linda Heerey, James O'Connor, Xiaodi Wang, Albert Koelmans and Ian O'Connor

www.epa.ie

Rialtas na hÉireann
Government of Ireland

Environmental Protection Agency

The EPA is responsible for protecting and improving the environment as a valuable asset for the people of Ireland. We are committed to protecting people and the environment from the harmful effects of radiation and pollution.

The work of the EPA can be divided into three main areas:

Regulation: Implementing regulation and environmental compliance systems to deliver good environmental outcomes and target those who don't comply.

Knowledge: Providing high quality, targeted and timely environmental data, information and assessment to inform decision making.

Advocacy: Working with others to advocate for a clean, productive and well protected environment and for sustainable environmental practices.

Our Responsibilities Include:

Licensing

> Large-scale industrial, waste and petrol storage activities;
> Urban waste water discharges;
> The contained use and controlled release of Genetically Modified Organisms;
> Sources of ionising radiation;
> Greenhouse gas emissions from industry and aviation through the EU Emissions Trading Scheme.

National Environmental Enforcement

> Audit and inspection of EPA licensed facilities;
> Drive the implementation of best practice in regulated activities and facilities;
> Oversee local authority responsibilities for environmental protection;
> Regulate the quality of public drinking water and enforce urban waste water discharge authorisations;
> Assess and report on public and private drinking water quality;
> Coordinate a network of public service organisations to support action against environmental crime;
> Prosecute those who flout environmental law and damage the environment.

Waste Management and Chemicals in the Environment

> Implement and enforce waste regulations including national enforcement issues;
> Prepare and publish national waste statistics and the National Hazardous Waste Management Plan;
> Develop and implement the National Waste Prevention Programme;
> Implement and report on legislation on the control of chemicals in the environment.

Water Management

> Engage with national and regional governance and operational structures to implement the Water Framework Directive;
> Monitor, assess and report on the quality of rivers, lakes, transitional and coastal waters, bathing waters and groundwaters, and measurement of water levels and river flows.

Climate Science & Climate Change

> Publish Ireland's greenhouse gas emission inventories and projections;
> Provide the Secretariat to the Climate Change Advisory Council and support to the National Dialogue on Climate Action;
> Support National, EU and UN Climate Science and Policy development activities.

Environmental Monitoring & Assessment

> Design and implement national environmental monitoring systems: technology, data management, analysis and forecasting;
> Produce the State of Ireland's Environment and Indicator Reports;
> Monitor air quality and implement the EU Clean Air for Europe Directive, the Convention on Long Range Transboundary Air Pollution, and the National Emissions Ceiling Directive;
> Oversee the implementation of the Environmental Noise Directive;
> Assess the impact of proposed plans and programmes on the Irish environment.

Environmental Research and Development

> Coordinate and fund national environmental research activity to identify pressures, inform policy and provide solutions;
> Collaborate with national and EU environmental research activity.

Radiological Protection

> Monitoring radiation levels and assess public exposure to ionising radiation and electromagnetic fields;
> Assist in developing national plans for emergencies arising from nuclear accidents;
> Monitor developments abroad relating to nuclear installations and radiological safety;
> Provide, or oversee the provision of, specialist radiation protection services.

Guidance, Awareness Raising, and Accessible Information

> Provide independent evidence-based reporting, advice and guidance to Government, industry and the public on environmental and radiological protection topics;
> Promote the link between health and wellbeing, the economy and a clean environment;
> Promote environmental awareness including supporting behaviours for resource efficiency and climate transition;
> Promote radon testing in homes and workplaces and encourage remediation where necessary.

Partnership and Networking

> Work with international and national agencies, regional and local authorities, non-governmental organisations, representative bodies and government departments to deliver environmental and radiological protection, research coordination and science-based decision making.

Management and Structure of the EPA

The EPA is managed by a full time Board, consisting of a Director General and five Directors. The work is carried out across five Offices:

1. Office of Environmental Sustainability
2. Office of Environmental Enforcement
3. Office of Evidence and Assessment
4. Office of Radiation Protection and Environmental Monitoring
5. Office of Communications and Corporate Services

The EPA is assisted by advisory committees who meet regularly to discuss issues of concern and provide advice to the Board.

Sources, Pathways and Environmental Fate of Microplastics

Authors: Roisin Nash, John O'Sullivan, Sinead Murphy, Michael Bruen, Anne Marie Mahon, Heather Lally, Linda Heerey, James O'Connor, Xiaodi Wang, Albert Koelmans and Ian O'Connor

Identifying pressures

As plastic production continues to increase, we are seeing significant quantities of microplastics (MPs), a contaminant of emerging concern, being recorded worldwide. MPs have been recorded in freshwater environments globally, adding additional pressure to already burdened systems. The cumulative impact of threats to water bodies will prevent authorities from achieving their environmental objective of being free from pollution. This research, by improving understanding of MP sources, pathways and environmental fate in freshwater systems in Ireland, aims to inform the development and implementation of policies such as the Water Framework Directive.

Informing policy

This research project has identified key challenges and recommendations that target MP pollution, highlighting immediate measures that could manage MP debris at known sources. This will inform the development and implementation of policies such as the Water Framework Directive by improving understanding of MP sources, pathways and environmental fate in freshwater systems in Ireland. This will, in turn, reduce the amount of marine litter reaching the sea and so contribute to OSPAR's objective in the Regional Action Plan for Marine Litter of significantly reducing amounts of marine litter.

Developing solutions

This study emphasises that river catchments are complex, with unexplained variations in the spatial and temporal distribution of MPs. Further research is required to investigate variables such as rainfall, as well as the role of atmospheric deposition, recreational river use, hydrological events, river hydrogeomorphology and land use within catchments. Similarly, pathways need to be researched further, for example to determine whether interflow is an important pathway for MPs under rainfall erosion.

An exploration of sources has provided insight into interventions that could reduce the abundance of MPs released into the environment. The implementation of a continuous monitoring framework for MPs within river catchments is deemed necessary to monitor progress towards achieving good water quality. It is recommended that this be incorporated at a minimum level into river basin management planning in Ireland. Further exploration of sediment in conjunction with benthic communities is necessary to confirm the potential of these to act as bioindicators for MPs and would allow for short- to medium-term monitoring.

EPA RESEARCH PROGRAMME 2021–2030

Sources, Pathways and Environmental Fate of Microplastics

(2016-W-LS-10)

EPA Research Report

Prepared for the Environmental Protection Agency

by

Atlantic Technological University

Authors:

Róisín Nash, John O'Sullivan, Sineád Murphy, Michael Bruen, Anne Marie Mahon, Heather Lally, Linda Heerey, James O'Connor, Xiaodi Wang, Albert Koelmans and Ian O'Connor

ENVIRONMENTAL PROTECTION AGENCY
An Ghníomhaireacht um Chaomhnú Comhshaoil
PO Box 3000, Johnstown Castle, Co. Wexford, Ireland

Telephone: +353 53 916 0600 Fax: +353 53 916 0699
Email: info@epa.ie Website: www.epa.ie

ACKNOWLEDGEMENTS

This report is published as part of the EPA Research Programme 2021–2030. The EPA Research Programme is a Government of Ireland initiative funded by the Department of the Environment, Climate and Communications. It is administered by the Environmental Protection Agency, which has the statutory function of co-ordinating and promoting environmental research.

The authors would like to acknowledge the members of the project steering committee, namely Conall O'Connor (Department of Housing, Local Government and Heritage), Garvan O'Donnell (Marine Institute), Eadaoin Joyce (Irish Water), Charlotte Picard (Irish Water), Rosa Busquets (Kingston University), Bruno Tassin (École des Ponts ParisTech, ENPC) and Lisa Sheils (EPA).

DISCLAIMER

This report is based on research carried out/data from 2017 to 2021. More recent data may have become available since the research was completed.

The EPA Research Programme addresses the need for research in Ireland to inform policymakers and other stakeholders on a range of questions in relation to environmental protection. These reports are intended as contributions to the necessary debate on the protection of the environment.

EPA RESEARCH PROGRAMME 2021–2030

Published by the Environmental Protection Agency, Ireland

ISBN: 978-1-80009-092-7 March 2023

Price: Free Online version

Project Partners

Dr Róisín Nash
Marine & Freshwater Research Centre
Atlantic Technological University, Galway
Dublin Road
Galway
Tel.: +353 91742593
Email: roisin.nash@atu.ie

Associate Professor John O'Sullivan
UCD School of Civil Engineering
UCD Dooge Centre for Water Resources Research and UCD Earth Institute
University College Dublin
Belfield
Dublin 4
Tel.: +353 1 7163213
Email: jj.osullivan@ucd.ie

Dr Sineád Murphy
Marine & Freshwater Research Centre
Atlantic Technological University, Galway
Dublin Road
Galway
Tel.: +353 74 912086
Email: sinead.murphy@atu.ie

Professor Michael Bruen
UCD School of Civil Engineering
UCD Dooge Centre for Water Resources Research and UCD Earth Institute
University College Dublin
Belfield
Dublin 4
Tel.: +353 1 7163213
Email: michael.bruen@ucd.ie

Dr Anne Marie Mahon
Marine & Freshwater Research Centre
Atlantic Technological University, Galway
Dublin Road
Galway
Email: halecium@gmail.com

Dr Heather Lally
Marine & Freshwater Research Centre
Atlantic Technological University, Galway
Dublin Road
Galway
Tel.: +353 91742484
Email: heather.lally@atu.ie

Ms Linda Heerey
UCD School of Civil Engineering
UCD Dooge Centre for Water Resources Research
University College Dublin
Belfield
Dublin 4
Email: linda.heerey@ucdconnect.ie

Dr James O'Connor
Marine & Freshwater Research Centre
Atlantic Technological University, Galway
Dublin Road
Galway
Email: james.oconnor@research.gmit.ie

Ms Xiaodi Wang
UCD School of Civil Engineering
UCD Dooge Centre for Water Resources Research
University College Dublin
Belfield
Dublin 4
Email: xiaodiwang@outlook.com

Dr Ian O'Connor
Marine & Freshwater Research Centre
Atlantic Technological University, Galway
Dublin Road
Galway
Tel.: +353 91742384
Email: ian.oconnor@atu.ie

Professor Dr Albert A. Koelmans
Aquatic Ecology and Water Quality Management
Department of Environmental Sciences
Wageningen University
6700AA Wageningen
Netherlands
Tel.: +31 317 483 898
Email: bart.koelmans@wur.nl

Contents

List of Figures

List of Tables

Executive Summary

As a result of a growing global reliance on the plastics industry, plastic-derived pollution has become a threat to global ecology and a topic of international concern. Sources of plastic pollution in freshwater catchments are directly related to a wide range of human activities that have the potential to trigger long-term irreversible changes to freshwater ecosystems and their resources.

Plastics are persistent pollutants in the environment, and those 1 µm to 5 mm in size are termed microplastics (MPs). MPs can be further refined as primary or secondary. Primary MPs are produced to have microscopic dimensions whereas secondary MPs result from the fragmentation and degradation of larger items.

Approximately 80% of marine litter derives from land-based sources, with rivers identified as one of the most important MP pathways. Plastic waste enters rivers through several natural processes influenced by wind- or rain-induced surface runoff or via direct dumping or disposal. Therefore, the identification of all MP sources in river catchments is key to management efforts to reduce the presence of MPs in freshwater and marine environments. An analysis of surface water samples in rivers has revealed that MPs are widely distributed in catchments to the point that they are considered ubiquitous. During the current study, MPs were recovered from all water samples collected, with variations in MP concentrations recorded between sampling periods as well as between sampling sites. While wastewater is a confirmed source of MPs found in rivers, previous research into the partitioning of MPs at various treatment stages in treatment plants revealed that up to 97% of MPs can be retained in sludge (biosolids) following primary treatment. However, this sludge is often spread on land. Further research has confirmed both the construction industry, through the onsite cutting of plastic materials, and artificial pitches, through wear and tear, as sources of MPs, and that smaller MPs have the potential to be transported offsite via wind- and rain-induced runoff or by boots in the case of artificial pitches.

Agriculture is one of the most important economic sectors globally, and biosolids from wastewater treatment plants (WWTPs) are spread on suitable land to meet ever-increasing demand, increase productivity and reduce costs. A new model incorporating overland flow pathways from Thomas *et al.'s* DiffuseTools project can indicate land potentially suitable for the landspreading of biosolids from WWTPs and aggregate the risk to water courses from the mobilisation of MPs from sludge spreading. The processes that govern the overland movement of MPs are complex, but the research presented here shows that the concentrations of MPs exported can be influenced by catchment slope and rainfall intensity. In addition, ploughing on agricultural land has been confirmed to contribute to the vertical migration of MPs; notably, this pathway appears to carry low risk of groundwater contamination.

Spatial and temporal changes play an important role in the abundance and bioavailability of floating MP concentrations in rivers, particularly after periods of high rainfall, when the concentrations can appear to be lower as a result of dilution factors. MPs were present in all riverine macroinvertebrate families, providing a site-specific assessment of MP abundance due to limited migration patterns. MPs were also recovered from brown trout (72%) and otter spraints (53%), in which salmonids were the most frequently identified food. Bioaccumulation and biomagnification were not evident in the food chain, with low levels of MPs recorded, although all evidence indicates that the pathway most likely to account for the presence of MPs in top predators in freshwater ecosystems is trophic transfer.

Interventions are possible at the sources identified. For example, to reduce the spread of MPs from artificial pitches, construction guidelines should include retainer (butt) walls of height > 100 mm and the installation of steel boot-cleaning grids. Reduction targets should be set for the release into the environment of MPs from effluents and any by-products (e.g. sludge) from WWTPs. Onsite waste audits of MPs should be introduced on construction sites, including an assessment of the use of building information models.

The results reveal knowledge gaps that require further research, such as the identification and quantification of all sources of MP pollution and the influencing factors within river catchments; further assessment of agricultural soils and drainage water from various types of land use before and after the application of biosolids to investigate if interflow is a significant pathway; and the accumulation, translocation and abundance of smaller MPs (< 100 µm) in biotic tissue outside the gastrointestinal tract. Only carefully designed monitoring will detect elevated levels of plastic pollution in rivers and its subsequent flow downstream to the marine environment.

1 Introduction

1.1 Overview

The ubiquity of plastic in virtually all industrial and consumer products has resulted in a continuous increase in the annual production of plastics. European and international policies support the transition towards a more sustainable economy, with the European Union (EU) Circular Economy Action Plan in place to reduce carbon budgets and support countries to become carbon neutral. Furthermore, EU Member States, through the European Green Deal, are committed to turning the EU into the first climate-neutral continent by 2050 and transforming the EU's blue economy for a sustainable future. However, despite policies and initiatives to reduce single-use plastics, embrace sustainability and circular economies and reduce reliance on petrochemicals in transport in response to climate change, the petroleum industry is expected to continue to develop new markets for plastic (Carbon Tracker Initiative, 2020).

The increase in plastic production has resulted in significant quantities of microplastics (MPs), a contaminant of emerging concern, being recorded worldwide. MPs are formed either through the degradation of larger plastic pieces, resulting in fibres, fragments or film, or through the production of resin pellets or beads intended for the manufacture of larger plastic products, and they range in size from 1 μm to 5 mm (Frias and Nash, 2019). MPs are recorded to such an extent that they have become a threat to global ecology and a topic of international political concern (Nielsen *et al*., 2020). The extent of current knowledge on the abundance and distribution of MPs worldwide has led to the development of international regulatory instruments and initiatives (da Costa *et al*., 2020) to mitigate impacts, for example the prohibition of plastic microbeads and the introduction of charges for single-use plastic bags in several countries (Xanthos and Walker, 2017). Furthermore, several initiatives and directives addressing waste management and prevention of marine litter and MPs have been developed since 2008, particularly the Waste Directive, the Marine Strategy Framework Directive, the European Strategy on Plastics in a Circular Economy and, more recently, the new European Green Deal and the directive on the reduction of the impact of certain plastic products in the environment, often referred to as the Single-Use Plastics Directive.

Water-borne plastic pollution is of particular concern because of its potential to adsorb organic contaminants from the surrounding environment (Wagner *et al*., 2014). These adsorbed pollutants, along with plasticisers and additives, specific to each polymer, can induce sublethal stresses in organisms (Haegerbaeumer *et al*., 2019). Pollutants released on ingestion by biota or through environmental degradation can present risks to priority species and habitats (Andrady, 2011). MPs are now accepted as ubiquitous in the marine environment; therefore, as an estimated 80% of marine pollution comes from land-based sources (Andrady, 2011), attention has now turned to the sources of MPs, pathways to marine environments and potential for MPs to be retained in and have an impact on freshwater ecosystems (Eerkes-Medrano *et al*., 2015).

1.2 Microplastics in Freshwater Environments

MPs have been recorded in several freshwater studies worldwide (O'Connor *et al*., 2019). MPs' dispersal in aquatic environments is thought to depend on their physical and chemical nature, the physical forces that drive their movements and the interactions between the particles and biota, or a combination of all of these (Hoellein *et al*., 2014). Although the physical processes affecting dispersal of MPs in the marine environment are known, the same processes cannot be assumed to occur in freshwater because the density of freshwater is lower than that of marine water, and this may have a significant effect on the vertical distribution of the polymer types (Ballent *et al*., 2012). However, common polymers have such a range of densities that they can be distributed throughout the water column and benthos (Morét-Ferguson *et al*., 2010), an anthropogenic pressure affecting freshwater systems that is currently unaccounted for in river basin management plans. Studies in large lakes have identified wind, surface currents and wave energy as responsible for MP distribution (Imhof *et al*., 2013). Little is known about the factors in riverine environments that influence transport and deposition. However, greater MP abundances have been found in the vicinity of river confluences (Klein *et al*., 2015), suggesting that there may be critical flow velocities at which deposition is most likely to occur.

1.3 Aims

At the European Conference on Plastics in Freshwater Environments in Berlin in June 2016, the consensus was that a systematic approach is needed to understand the movement of MPs from sources to aquatic receptors and dispersal within freshwater ecosystems. Using the Slaney catchment as a case study, this research builds on previous EPA studies (Mahon *et al*., 2017) by refining potential sources and determining factors critical for pathway attenuation. Based on these findings, a number of recommendations are provided for the monitoring of MPs in freshwater environments. Furthermore, this research will inform the development and implementation of policies such as the EU Water Framework Directive by improving the understanding of the sources of MPs, pathways and environmental fate in freshwater systems in Ireland.

2 Sources

2.1 Overview

In Europe, the plastics in greatest demand are polypropylene (PP), low-density polyethylene (LDPE), high-density polyethylene (HDPE), polyvinyl chloride (PVC), polystyrene (PS), polyurethane (PU) and polyethylene terephthalate (PET) (Plastics Europe, 2020), and, from studies to date, these account for most of the plastics found in the environment. Research has identified a number of MP sources, including the cosmetics industry (Fendall and Sewell, 2009), industry as a whole (Lechner and Ramler, 2015), urban wastewater treatment plants (UWWTPs) (Magnusson and Norén, 2014), synthetic rubber tyres (Essel *et al*., 2015), clothing and furnishings (Astrom, 2016), and the construction sector (Plastics Europe, 2020).

In this chapter we further refine the characterisation of MP sources that were previously reported in the EPA-funded GMIT desktop study and small-scale studies (Mahon *et al*., 2017) and explore the scale and overland pathway of diffuse sources by means of three case studies, namely (1) UWWTPs, (2) construction sites and (3) AstroTurf pitches. Each of these case studies is described in detail in Mahon *et al*. (2021).

2.2 Case Study 1: Urban Wastewater Treatment Plants

2.2.1 Overview

Wastewater treatment plants (WWTPs) have been identified as receptors of MPs which effectively capture large proportions of the MP loading (Magnusson and Norén, 2014). However, the proportion of MPs that do not become trapped in WWTP systems still contribute to the loading of discharge waters and, therefore, to the input to riverine systems. In this study, we investigated MPs in three WWTPs: WWTPs 1 and 2 in the River Slaney catchment, and WWTP 3 in the River Barrow catchment. These WWTPs were selected to determine the capture rates, loadings and characteristics of MPs in the Slaney and Barrow catchments.

2.2.2 Methodology

Each of the WWTPs serves a population equivalent of <2500 and uses activated sludge and settlement as a secondary treatment; WWTP 1 uses membrane filtration as a tertiary treatment. A composite sample (5 L) was taken from the inflow (post preliminary) and final effluent points of the three WWTPs by a technician on site. Sampling took place on four occasions (in April, June, August and October 2019). Once the samples were received, sodium hypochlorite was added to each container. Organic matter was digested using potassium hydroxide (at 60°C for 24 hours). The samples were then filtered using glass fibre filters (Whatman grade GF/C, 1.2 μm) and the MPs were identified, characterised and categorised by size. A subsample of MPs were analysed using Fourier transform infrared spectroscopy (FTIR) to identify the polymers present. The methods used in this case study are described in more detail in Mahon *et al*. (2021).

2.2.3 Results

2.2.3.1 Microplastic abundance and loadings

The mean (± SD) abundance of MPs from the WWTPs over the four sampling occasions ranged from 20 ± 1.83 to 45 ± 6.36 MPs L^{-1} in influent waters and from 6.55 ± 4.35 to 9.75 ± 6.40 MPs L^{-1} in effluent waters. Therefore, the MP capture rates (difference in abundance of MPs between inflow and effluent) varied between 58% and 97% in the studied treatment plants, with a mean capture rate of 65% at WWTP 1, 79% at WWTP 2 and 82% at WWTP 3. While these capture rates also varied over the sampling occasions, no major differences were recorded. Nylon polymers dominated both the inflow and the outflow (52% and 42%, respectively), while polycarbonate polymers were the next most abundant in the outflow (33%), with a variety of polymers accounting for the remaining MPs identified.

Based on average hydraulic loadings for each treatment plant (159 m^3 day^{-1} (WWTP 1), 210 m^{-3} day^{-1} (WWTP 2) and 753.6 m^{-3} day^{-1}(WWTP 3)), and despite the high capture rates of WWTP 3, the potential number of MPs being discharged from the plants is estimated to be around 6 million m^3 day^{-1}, with WWTP 1 and 2 each estimated to account for around 1 million m^3 day^{-1}. While all three WWTPs showed good retention rates, the results, based on mean abundance of MPs over the four sampling occasions and the average hydraulic loading (m^3 day^{-1}), show that WWTP 3 is more efficient at retaining MPs (estimated 34,326,480 MPs m^3 day^{-1} (influent) and 5,934,600 MPs m^3 day^{-1} (effluent)) and suggest that over 28 million MPs are trapped at this WWTP each day. The estimated MPs per m^3 passing through the three WWTPs, both influent and effluent waters, over the four sampling occasions can be seen in Figure 2.1 and also show that MP retention in WWTP 3 was considerable in all months assessed. Furthermore, there was an increase in the proportion of smaller MPs in the effluent from the WWTPs, indicating that larger MPs may be preferentially retained.

2.2.3.2 Microplastic morphologies and polymer type

Fibres were the dominant MP type across all influent waters (67–71%), followed by fragments (16–26%) and films (2–13%). Fibres showed the greatest reduction in numbers in effluent waters at all three sites (accounting for 40–46% of MPs), with fragments and films accounting for correspondingly greater proportions of MPs in effluent waters (41–46% and 8–20%, respectively). Most MPs were in the size category 600–800 µm.

2.2.4 Discussion

The results of this study were in agreement with those of a Swedish study, which found partitioning of MPs through the various treatment stages, with 99% of MPs retained in sludge following primary treatment (Magnusson and Norén, 2014). The dominance of fibres across all influent waters sampled is in line with other studies that showed high capture rates of fibres in sewage sludge (Ngo *et al*., 2019). The increased proportion of MP film detected in effluent could be attributed to the tendency for MP film to remain positively buoyant, reducing the likelihood of "fall-out" during the settlement stage. The proportion of fragments in the effluent water may also be associated with a lower likelihood of particle settling, but it is difficult to confirm this, given the variation of polymer form and type in this category. The higher abundance of smaller MPs in effluent waters is attributed to the sinking and entrapment of larger MPs in sludge. Particles < 200 µm are more difficult to handle and are, therefore, likely to be underestimated using the current methodologies.

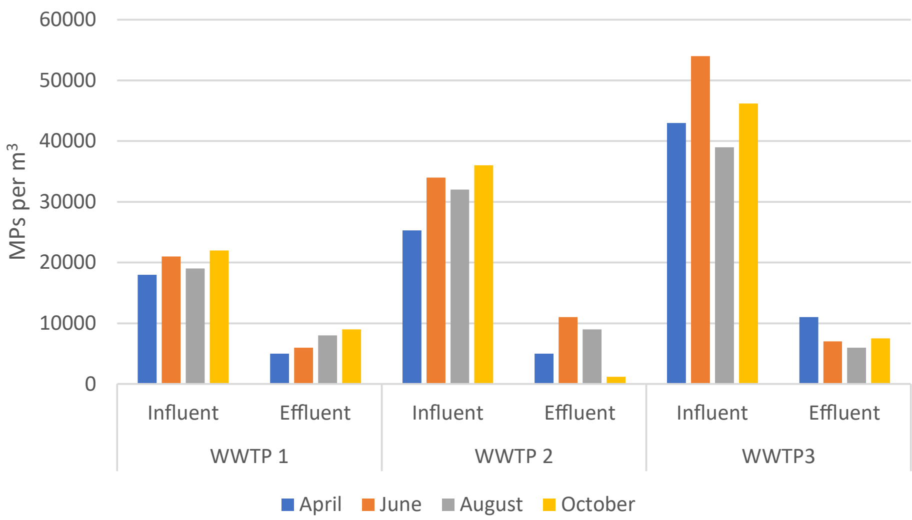

Figure 2.1. Microplastics per m^3 in influent and effluent waters of the three WWTPs over the four sampling occasions.

2.3 Case Study 2: Construction

2.3.1 Overview

The construction industry in Europe consumes 10 million tonnes of plastic each year, 20% of Europe's total, making it the largest consumer of plastics after the packaging sector. In the construction sector, plastic pipes account for over 50% of annual tonnage (Plastics Europe, 2020). During general construction activities, a variety of hand and power tools are used to cut these plastic materials, resulting in the production of MPs. However, very little is known regarding the sources and pathways of these materials during both the demolition and the construction phases of projects. This section explores the loading of MPs emanating from the construction industry with a view to assessing the quantities produced during the hand sawing of common materials on building sites and how easy it is to recapture these. Furthermore, preliminary findings from an analysis of storm water around an industrial estate are presented.

2.3.2 Methodology

Three replicate pieces were used for each of the cutting experiments: a polyvinyl chloride sewer pipe, a polypropylene foul water pipe and one piece of polyisocyanurate (PIR) insulation board (100 × 80 mm). The field experiments involved 14 cylindrical sediment traps 20 cm in diameter and 1-cm deep. The traps were placed at locations approximately 50 m from the main building during the construction phase for 2 weeks, during which time reasonable amounts of cutting would be expected. Sediment samples were taken closer to the building (approximately 20 m away). The methods used in this case study, including details of the clean room experiment, field work and the isolation and characterisation of MPs, are described in detail in Mahon *et al*. (2021). In addition, water was collected from storm runoff drains at an industrial site to allow for the preliminary investigation of MP type and abundance.

2.3.3 Results

The percentage capture rate of MPs averaged over 99% for each of the three test substrates; of these, the PP pipes had the highest loss (0.33%), followed by the insulation board (0.11%) and, finally, the PVC piping (0.03%). The results showed that cutting PVC piping using a hand saw resulted in the production of MPs ranging in size from 125 to 2000 µm, with the majority of MPs > 710 µm. An estimated average of 44,133 particles were produced during one cut. The resulting MPs from PVC were extremely brittle and tended to break easily. Based on information supplied by a Galway-based construction company, > 1.3 million PVC MPs (80 cuts each weighing 156 g) and > 1 million PP MPs were produced during the construction of a 16-classroom school. Fragments, fibres and films were all observed in the sediment traps placed at 20 m and 50 m from the building, with fragments forming the highest abundance in both cases: 86% at 20 m and 78% at 50 m.

Furthermore, the abundance of MPs was found to be greater closer to the building site (20 m) and, although MPs in all size categories were found in each sediment trap, those in the lower category (400–600 µm) were more abundant in the sediment trap furthest from the building (50 m), accounting for 80% of all MPs in this trap. Similarly, there was a lower abundance of the larger MPs in these sediment traps (25%). Overall, MP abundance recorded from around the building site ranged between 1700 MPs kg^{-1} closer to the site and 3000 MPs kg^{-1} 50 m from the building under construction. Similar to the construction site, the preliminary onsite data from the storm water in the industrial estate indicated that fragments were the dominant type (> 70%), followed by fibres and film. Storm water samples recorded abundance ranging from approximately 1000 to 30,000 MPs m^{-3}.

2.3.4 Discussion

Cutting experiments have demonstrated that, regardless of how diligent one is when cleaning up MP particles, there is still a percentage loss, and, although low in terms of mass, the abundance of MPs could still be significant. MPs from PVC, being a dense material, may be less likely to become airborne, whereas MPs from PIR (insulation boards), being light, may be more likely to do so.

The field experiment found that MP concentrations on the ground immediately outside the building site were higher, with the lower abundances further from the site being attributed to smaller MPs travelling further via airborne pathways. The authors are unaware of any research on similar construction sites and are therefore unable to compare results directly. A conceptual diagram was constructed based on what is known and anticipated regarding the pathways of MPs emanating from the construction industry (Figure 2.2). Preliminary storm water results from industrial estates, along with results from a study by Werbowski *et al.* (2021), indicate that heavy rain is a significant pathway by which MPs are transported from urban areas to aquatic ecosystems. Furthermore, there are several points along these pathways at which MPs could leak and be contaminated further by associated chemicals.

2.4 Case Study 3: AstroTurf Pitches

2.4.1 Overview

Artificial grass pitches are increasingly replacing natural pitches and play areas because their higher capacity for play and resistance to climatic conditions means that they can be used more often, with associated commercial benefits. Over the last decade, artificial pitches have become a popular choice for community activities such as hockey, basketball and Gaelic sports.

Grass turf, which is a woven carpet, comprises PP, polyamide (PA) and PU infilled with rubber granules, often derived from recycled low-cost car tyres made of material known as styrene–butadiene rubber (SBR), and in a standard artificial turf field this can amount to as much as 140 tonnes of material (Wredh, 2014). Pitches have a lifetime of approximately 8–10 years (Månsson, 2010). However, due to loss of rubber particles to the environment, artificial surfaces must be regularly topped up with rubber particles, and some estimates put these “top-ups” at between 3 and 5 tonnes per year, depending on the use intensity of the pitch and the climatic conditions to which it is

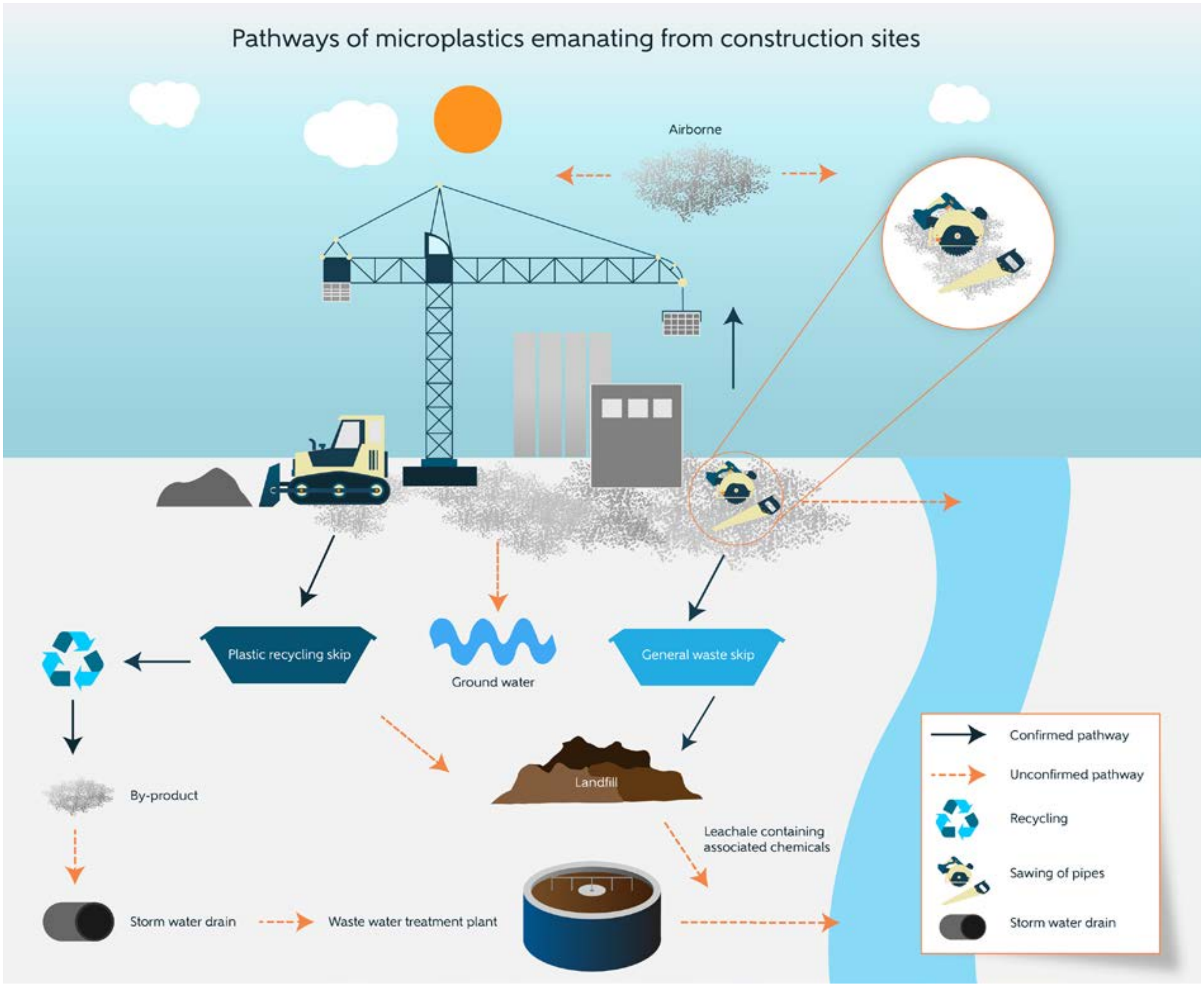

Figure 2.2. A conceptual diagram of known and anticipated MP pathways from the construction industry to aquatic waterways.

exposed. Construction standards and design guides for performance and longevity of pitches include best practice for surface regularity, pitch gradient and water infiltration rates to reduce runoff (e.g. IRFU, 2008). This study aims to look at the leakage/runoff of MPs (of < 2 mm) from AstroTurf in Ireland.

2.4.2 *Methodology*

We selected 10 artificial pitches that varied in terms of size, use (i.e. type of sport played on the surface), slope (0–15%), border height (0 to > 100 mm), drainage (none, integrated or external) and surrounding substrate (gravel, tarmac or grass). At each site, three replicate samples were taken at 1, 5, 10 and 15 m from the pitch. Where there was a grass surface, samples were obtained using a vacuum cleaner over a 5-cm-diameter ring. Where grass bordered the pitch, a core was taken of the first centimetre of the grass substrate. Where there was a noticeable slope, this was measured using a clinometer. The methods applied in this case study, including details of sampling and isolation and characterisation of MPs, are described in Mahon *et al*. (2021).

2.4.3 *Results*

2.4.3.1 *Microplastics abundance*

The abundance of MPs (SBR articles) varied between pitches and with distance from the pitches (up to 15 m). The abundance of MPs was highest (15,000 per m^2) within 1 m of the pitch and decreased with distance from the pitch (Figure 2.3). The results demonstrate that MPs can be transported outside a pitch to a distance of at least 15 m.

2.4.3.2 *Microplastics dispersal and particle size distribution*

The majority of the larger particles (> 500 µm) were found in greatest abundance within 5 m of the pitch, while the smaller particles (< 500 µm) were found to be more evenly dispersed across all sampling sites (Figure 2.4). The lowest overall abundance of MPs (mean = 8863) was recorded in the vicinity of pitches bounded by a butt wall of height > 100 mm (n = 2). The abundance of MPs was considerably higher near pitches with a butt wall of height < 100 mm

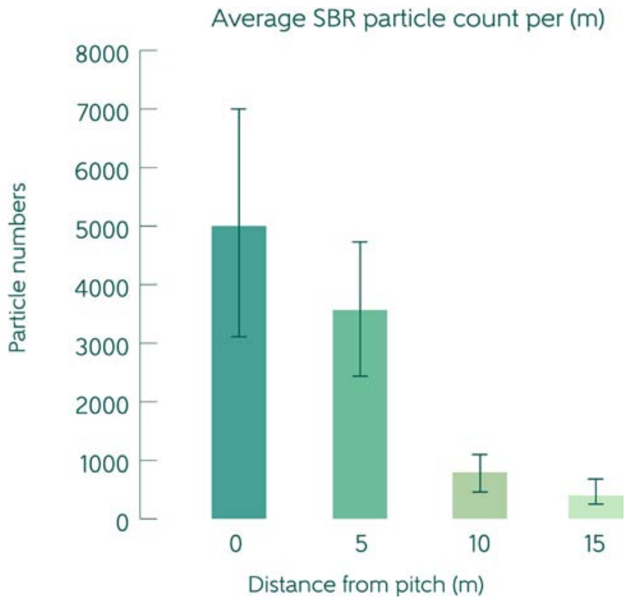

Figure 2.3. Mean abundance of MPs (SBR particles) per m^2 by distance from pitch.

(n = 3; mean 30,655 MPs) or no butt wall (n = 5; 30,154 MPs). Transferral of particles at 5 m from the pitch was greater at the site with the steepest slope (15%) than at other sites (slope 0–10%), which may suggest the potential influence of gradient on dispersal in the absence of a border. Furthermore, it was found that a surrounding substrate of grass did not appear to reduce the transferral of MPs. A site that had no integrated or external drainage system recorded the highest overall abundance of MPs (47,619). At another pitch, an external drainage system led directly into the storm water drain, and the drain itself was observed to be half-full of rubber crumb.

2.4.4 *Discussion*

This study confirms that MPs emanating from artificial grass pitches follow a horizontal pathway to the surrounding landscape extending to a distance of 15 m from the side of the pitches and shows that AstroTurf pitches are an important source of MPs found in the surrounding environment. We hypothesise that the smaller size fractions (< 500 µm) will be the most abundant at distances > 20 m from the pitches, with even smaller size fractions (< 100 µm; not recorded in this study) likely to become airborne. Furthermore, the range of size fractions recorded may be attributable both to wear and tear on the pitch and to degradation from UV rays and the influence of wetting and drying cycles (McLaren *et al*., 2012). The adhesive nature of

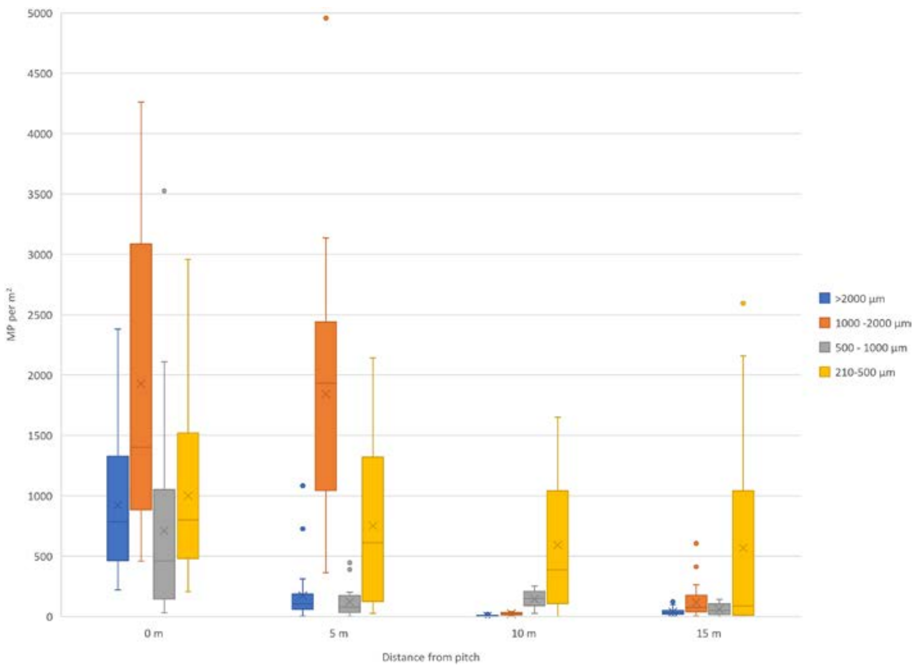

Figure 2.4. Mean abundance per m² of MPs of different size by distance from pitch.

the smallest size fraction retained on a 90-µm sieve suggests that it may interact differently with other particles in the environment. In addition, although the number of replicates taken at each site was not sufficient to enable us to obtain statistically significant results, the current study suggests that the inclusion of a butt wall of height > 100 mm is effective in retaining crumb within the boundary of the pitch.

A conceptual diagram was constructed to illustrate crumb particle transport pathways into and out of AstroTurf pitches. There are many confirmed pathways and potential pathways through which MPs could be exported from pitches and end up in the freshwater environment (Figure 2.5). In the absence of integrated drainage, external drain blockages facilitate the vertical translocation of rainwater through sand and other barrier layers into subsurface pipes, creating a direct pathway for MPs to enter the storm water drainage network.

2.5 Conclusion

The case studies presented here highlight the extent to which previously unexplored sources of MPs with the potential to travel overland (namely UWWTPs, construction sites and AstroTurf pitches) constitute further pathways of diffuse sources of MPs. Also identified here is the need to explore storm water as a potential MP pathway. Several recommendations to mitigate MP export from these settings are summarised in the final chapter of this report. However, where no mitigation actions are in place, the destination of these MPs will vary and depend on the characteristics of the areas surrounding the locations from which MPs are derived. For example, where construction sites are surrounded by agricultural land, works on site can be considered a source of MPs to the neighbouring fields. The next chapter follows the next step along the journey and explores MPs in agricultural land as an additional pathway to freshwater bodies and the abundance of MPs in a riverine system.

Figure 2.5. A conceptual diagram of inflow and outflow of rubber crumb from AstroTurf pitches.

3 Transport Pathways: From Agricultural Land to Freshwater

3.1 Overview

The environmental fate of MPs once they are released from terrestrial systems remains relatively uncertain, despite an increase in research in this area over recent years. Particular attention has been paid to agricultural soils, as they are considered a significant MP hotspot, with annual loads in these soils predicted to be between 4 and 23 times higher than those in the marine environment (Horton *et al.*, 2017). These high loadings are attributable to a number of sources. Of particular significance is the widespread use of biosolids, a by-product of wastewater treatment plant (WWTP) sludges, for the nutrient enrichment of agricultural lands. While concentrations of MPs in biosolids can vary greatly due to seasonality, treatment processes and levels of urbanisation (Mahon *et al.*, 2017; Lee and Kim, 2018; Li *et al.*, 2018), up to 99% of MP entering WWTPs can be retained in sewage sludge (Carr *et al.*, 2016). With an estimated 80% reuse rate, Ireland recycles more WWTP sludge as a biosolid fertiliser than other European countries (Eurostat, 2021). Given that around a further 16% of WWTP sludge is used in the production of agricultural compost, which is subsequently used for agricultural purposes, the actual reuse rate is considered to be around 96% (Eurostat, 2021). Additional MPs in agricultural soils derive from plastic mulching (Steinmetz *et al.*, 2016), the breakdown of both macroplastics and mesoplastics, and atmospheric deposition (Horton *et al.*, 2017; Bläsing and Amelung, 2018).

Given the heavy reliance of the Irish agricultural sector on MP-rich WWTP sludge for land treatment applications, developing an understanding of the mechanisms pertaining to the export of plastics from lands, and the potential risks that this export poses to other environmental systems, is particularly important. Published research findings in this regard remain somewhat contradictory. Corradini *et al.* (2019), for example, highlight the potential of agricultural soils to serve as MP sinks, with concentrations increasing with successive biosolid treatments. Other studies suggest that MPs can exit land systems by moving vertically through soils or horizontally via surface runoff and/or interflow or can be windswept from exposed surfaces. The physical processes governing this vertical movement are complex and subject to climate/weather influences (Allen *et al.*, 2019; Hitchcock, 2020), agricultural land use (Rillig *et al.*, 2017; Ouyang *et al.*, 2020), soil structure (Zubris and Richards, 2005; Horton and Dixon, 2018) and the characteristics of the MP particles (O'Connor *et al.*, 2019; Crossman *et al.*, 2020).

The vertical migration of MPs through soils may be enhanced by agricultural interference via ploughing, bioturbation effects and the presence of preferential flow paths from old root channels, and other interaggregate pores can also contribute to increased transport (Rillig *et al.*, 2017). The formation of surface cracks in soils due to the presence of expanding materials (for example montmorillonite (Rillig *et al.*, 2017)) or wet–dry cycles (O'Connor *et al.*, 2019) can also enhance vertical MP migration. Vertical migration in some settings has also been shown to depend on MP characteristics, such as size and shape, as smaller particles are more likely to be mobile (e.g. O'Connor *et al.*, 2019), and fibres are more likely to be retained in the soil column as a result of entanglement with soil particles (Crossman *et al.*, 2020). However, while it has been suggested that these vertical transport pathways potentially exist and may contribute to the presence of MP in groundwater resources, they have predominantly been tested under laboratory conditions, typically with the use of spherical MP particles in high-purity quartz sands (Wanner, 2021). Given that research on the role of vertical transport pathways in the overall export of MPs from field-scale terrestrial systems remains limited, further work is needed to provide insight into the actual environmental risk posed by this pathway.

Overland MP movement is less well understood, with knowledge gaps surrounding the behaviour and mobility of MPs during runoff events (Rehm *et al.*, 2021). Research has confirmed the significant export of MPs from agricultural soils (Crossman *et al.*, 2020), driven particularly by heavy rainfall events (Hitchcock, 2020). However, the erodibility of MPs from soils can be reduced when they form aggregates with soil (Rehm *et al.*, 2021), although these aggregates are typically unstable (Lehmann *et al.*, 2020) and can potentially be broken up by agricultural practices (e.g. tillage). Tillage has also been associated with the remobilisation of trapped MP particles in surface layers of agricultural fields and their subsequent release into freshwater systems; Ouyang *et al.* (2020), for example, reported increased MP concentrations from river discharges to Jiaozhou Bay, China, at times that coincide with annual tillage events.

The research presented in this chapter seeks to address some of the research gaps pertaining to the potential export pathways (vertical and overland) of MPs from agricultural soils, particularly from surface layers by vertical migration and overland movement. To examine the vertical movement of MPs through soil, two separate research elements were conducted. The first of these comprised a laboratory investigation of packed soil columns in which the potential for vertical migration of a range of MPs in different soils was tested. More specifically, this work sought to investigate the physical characteristics of MPs (specifically, particle size and polymer type) that may influence their downwards mobility through a soil matrix. The second element extended the laboratory work to a field setting in the south-east of Ireland that has an extended history of land treatment with WWTP sludges. There the significance of this vertical transport pathway was further investigated through the extraction of both deep and shallow cores and the collection of water samples from below the water table in the field and from a subsurface land drain from the field. The deep cores (2 m) were extracted to determine MP concentrations at different depths and to assess whether there were features of the soil structure that contributed to vertical MP movement. The shallow surface cores (0.01 m) were extracted at intervals along a transect orientated with the field gradient to assess if down-slope MP accumulations were evident. Water samples were obtained from boreholes following the removal of deep cores and from the subsurface field drain to assess whether groundwater was at risk of MP contamination.

MP mobility in the overland transport pathway was investigated through a suite of laboratory tests using a purpose-built, large-scale rainfall simulation rig in the UCD School of Civil Engineering. The experimental rig facilitated the examination of a key catchment parameter (slope), MP characteristics (size, shape and density), rainfall regime, timing of rainfall event after MP seeding and surface texture (bare soil to grass) on overland MP mobility.

In the following sections of this chapter we first outline the experimental methodologies underpinning our investigations of both the vertical and overland transport pathways. We then present the main results, discussions and conclusions, and consider the implications of the findings from a future policy perspective. Finally, a section exploring the River Slaney as a transport pathway is presented.

3.2 Vertical Movement of Microplastics through Porous Media

3.2.1 Laboratory investigation

Initial drainage experiments were conducted to investigate the vertical movement of MPs through porous media under extreme rainfall conditions such that the likelihood of MP movement was maximised. A purpose-built rig consisting of three columns containing porous media (to allow for experiments to be conducted in triplicates) was constructed based on guidelines outlined by Wefer-Roehl and Kübeck (2014) (Figure 3.1). Columns were established in acrylic pipes (120-cm high by 9.4 cm in diameter) with a steel mesh layer at the base of the column retaining the porous medium contained within. A 3D-printed funnel attached to the base of the column facilitated the collection of water samples. The manner in which rainfall was applied to the column was adopted from Mohanty *et al.* (2015) and involved the use of constant-head tanks (40-cm high by 14 cm in diameter) from which potable water was passed through an array of 16 hypodermic needles (21 gauge) distributed evenly over the top of each column. The vertical distance between needle tips and surfaces of porous media in each column was maintained at 0.5 m, and rainfall intensity could be varied by varying the water level in the constant-head tanks. Experiments were conducted with a range of porous media of different particle sizes to investigate the role of porosity in the downward movement of particles. These included silica sand (1.2–1.6 mm, 1.6–2 mm, 2–4 mm), glass beads (3 mm) and ceramic beads (8–10 mm). All porous media were first washed

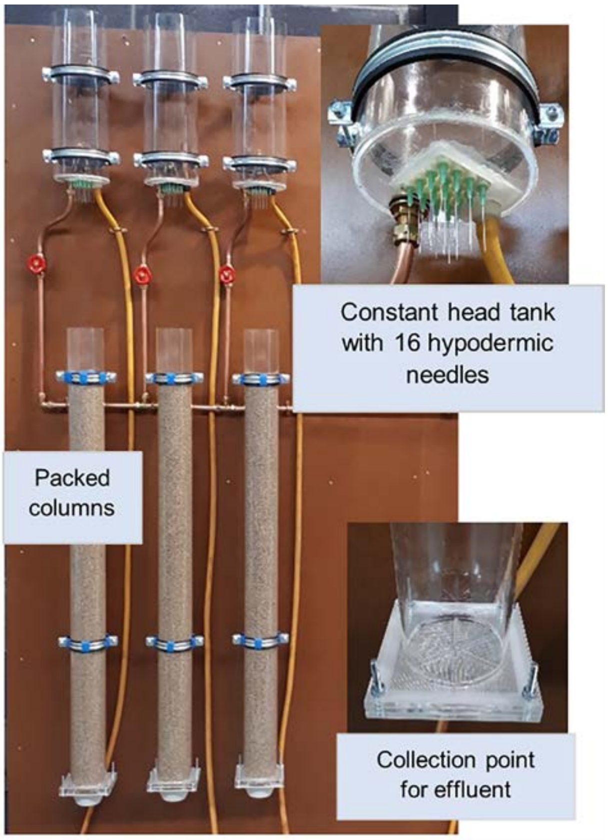

Figure 3.1. Experimental rig for porous media column tests.

using potable water, then rinsed in deionised water, and finally dried in an oven for 24 hours at 70°C. "Virgin" MP powders (< 0.3 mm) of varying polymer types (PVC, PET and LDPE) were sourced from Goodfellows (UK) and sieved to produce two size fractions (< 0.15 mm, 0.15–0.3 mm).

Columns were wet-packed following guidelines outlined by Wefer-Roehl and Kübeck (2014), with a layer of glass beads in the bottom 2 cm to allow for the free movement of water. MPs (0.75 g) were seeded on top of the porous medium and covered with an additional layer of glass beads. The rainfall regime for each experiment consisted of two blocks of 10 hours' continuous rainfall (intensity of 100 mm h^{-1}) with a dry (no rain) 10-hour period in between. Water samples were collected from the base of the columns at hourly intervals and subsequently vacuum filtered using Whatman Grade 4GF/C filter paper so that any particulate matter retained in the filter papers could be examined for the presence of MPs.

3.2.2 Results

The experiments examined the mobility of MP particles in two size ranges (< 0.15 mm and 0.15–0.3 mm) given a fixed rainfall regime and duration. The results indicated that all tested MPs were immobile in different test media (sand, glass and ceramic beads of different size), with 100% of the seeded MPs being retained in the surface layers of the porous media. Visual observations revealed the clear tendency of MPs to adhere to individual particles in the porous media, an issue that we suspected was exacerbated by the pristine ("virgin") MPs being tested in a system with a potable water solution. Our results showing that particles remained static in such systems appear inconsistent with some findings (albeit somewhat limited) in recent scientific literature, where the presence of MPs in groundwater resources has been attributed to the role of vertical transport pathways (Mintenig *et al.*, 2019; Panno *et al.*, 2019). Given these apparent contradictions, further research is warranted to test vertical MP transport pathways in porous media with "weathered" particles across a wide size spectrum (down to nanoscale).

3.3 Mobility and Migration of Microplastics: A Field-based Assessment

3.3.1 Methodology

Two study sites for the field-based investigation assessing the vertical movement of MPs through agricultural soils were carefully chosen based on several criteria, including an extensive history of biosolid treatment, comparable land use and similarity of soil texture and underlying geology (Figure 3.2). Two fields (fields A and B) in the south-east of Ireland, both with an established record of biosolid treatment, were identified for study in 2017. Land use in both fields was predominantly arable farming, and for the 20 years prior to study both fields had been treated annually with thermally dried sludge from the Ringsend (Dublin) WWTP. The Ringsend WWTP is the largest in the country and currently provides preliminary, primary, secondary and tertiary wastewater treatment for a population equivalent of 1.9 million, treating around 40% of Ireland's sewage. Furthermore, fields A and B were subjected to a single application of plastic mulching to support a crop of

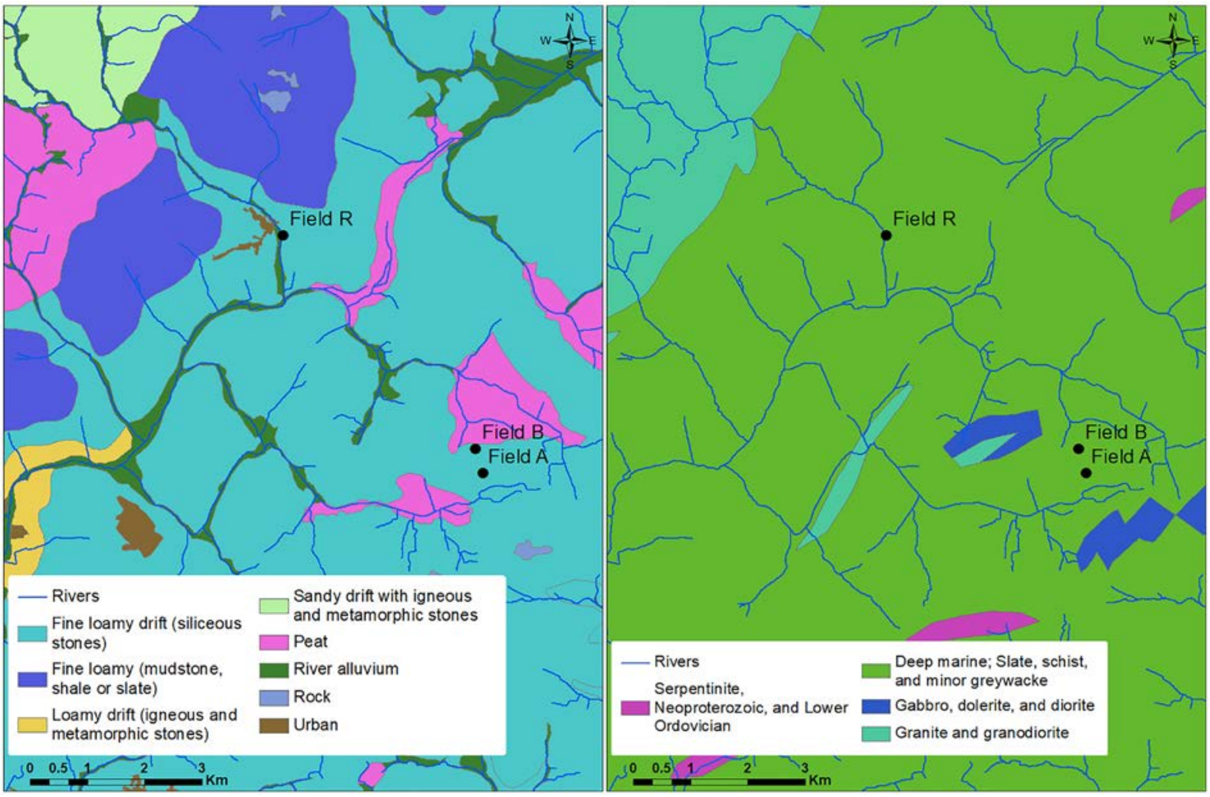

Figure 3.2. Field study locations showing soil texture (left) and geology (right).

maize in 2016 (personal correspondence, landowner, 14 December 2016). The underlying geology at both sites consisted of grey and green slate and thin siltstone (GSI), and both fields were overlain with a typical brown earth topsoil (SIS, 2014). Given that the field investigation at the two study sites sought to identify, or otherwise, evidence of down-slope MP accumulations, both tested fields were characterised by gradients: from top to bottom 4.58% for field A and 8.69% for field B. In addition to the two "impacted" field sites, an "unimpacted" reference site was included for study. This site (the playing fields of a school) was in proximity (within a 20-km radius) to fields A and B and chosen specifically because of its identical soil and similar geological characteristics.

In field A, soil sampling involved the extraction of six 2-m-deep cores (ø = 5 cm) using a Cobra TT percussion drill. Three cores (triplicates) were extracted at the top of the field and three at the bottom of the field, along transects running perpendicular to the field slope. The two transects were separated by approximately 260 m, with cores along each transect being laterally separated by approximately 1 m. In addition, 15 surface cores (10-cm depth, ø = 5 cm) were extracted at intervals of 30 m using a soil auger at five locations (three at each location) along a down-slope transect, connecting and perpendicular to the cross-field transects from which deep cores were extracted. In field B, following a similar methodology to that in field A, 15 surface cores were extracted at five locations (samples in triplicate), again separated by 30 m along a down-slope transect. Deep cores were not extracted from field B. Soil sampling from the reference field site involved the extraction of three 1-m-deep cores. Additionally, a composite groundwater sample (10.5 L) was collected in field A from the boreholes created following the extraction of deep cores, and a 20-L sample was collected from a subsurface land drain that discharged from the field to a stream at the bottom of the field.

Deep cores were subsampled by taking the uppermost 2 cm of every 5 cm of core (approximately 50 g of soil, wet weight), with a linear adjustment made for compression caused by the percussion drilling. The surface samples were also divided, with subsamples derived from the top 0–5 cm and bottom 5–10 cm of each core. There currently exists no standard method for the removal of MPs from soil, but density separation analysis remains one of the most frequently used methods and was the one adopted in this study.

Soil samples were first sieved using a 5-mm metal sieve to remove stones and large debris, followed by saturation in a solution of sodium chloride (NaCl) ($1.2\,g\,cm^{-3}$). Soil samples ($19 \pm 4.7\,g$ soil, dried weight) were added to 1-L glass beakers, to which 500 mL of the salt solution was then added. Once saturated, samples were mixed for 5 minutes at 300 r.p.m. using a Lovibond ET 740 mechanical agitator (Amesbury, UK) and allowed to settle for 24 hours. Surface water was removed using a glass pipette and vacuum filtered onto Whatman Grade 4GF/C filter paper, a process that was repeated three times for NaCl. Additionally, 25% of samples were retained for further zinc chloride ($ZnCl_2$) analysis ($1.6\,g\,cm^{-3}$) to extract high-density plastics (e.g. PVC ($1.38\,g\,cm^{-3}$) and PET ($1.38\,cm^{-3}$)). $ZnCl_2$ is both a costly and a toxic substance, and, given the significant quantities that would have been required to test all soil samples, a selection of soil samples was randomly chosen for these additional tests (25%). This density separation method followed the same protocols as for NaCl, but the process was limited to a single phase of mixing and settling. The $ZnCl_2$ tests recovered, on average, 15% more MPs than extraction with NaCl, and so an adjustment of 15% was made for the remaining 75% of MPs recovered.

All filter papers retained from density separation analysis were visually analysed under a microscope (Ash Omni), with MP identification following the guidelines outlined by Hidalgo-Ruz *et al.* (2012). MP particles were counted, characterised and measured before being transferred onto glass fibre filter papers. Polymer verification was conducted on 60% of all MPs using FTIR. FTIR analysis was conducted using a Bruker Hyperion 2000 microscope (MA, USA) together with a Bruker Tensor 27 spectrometer. The spectra were collected in absorbance mode using 128 scans (wavenumber range 4000–60 cm^{-1}) at a spectral resolution of 4 cm^{-1} and recorded using OPUS 7.8 software. For comparison, background spectra were collected on blank areas of the filter paper using the same parameters. MP concentrations found in soil were reported in particles per kilogram of soil and refer to concentrations found to a depth of 5 cm. All necessary precautions were taken to mitigate the risk of contamination, such as avoiding the use of plastic equipment, minimising samples' exposure to air by covering them all in tinfoil, testing reagent blanks, and testing air contamination weekly using filter paper samples on the workspace. Weekly contamination was minimal (mean contaiminated particles $n = 1$), which was accounted for in final MP concentrations.

Bulk soil density testing (every 10 cm) and particle size analysis (every 50 cm) were conducted to obtain an understanding of the density profile of the soil column. These data were complemented with the additional analysis of two cores (one from the top and one from the bottom of field A) using an Itrax XRF Core Scanner (Cox Analytical Systems, Sweden) mounted with a molybdenum X-ray tube, housed in the UCD School of Geography. Surface scanning and simultaneous optical imaging was carried out over 100-μm steps to produce optical images at 254 dpi. X-radiographic scanning was carried out over 200-μm steps to give 20-mm-wide X-ray images at around 125 dpi. Under uniform sample thickness, X-ray sensitivity is dependent on matrix composition, and thus the X-ray images were a surrogate for sediment density and matrix compaction, with darker shades corresponding to lower recorded X-ray intensities and denser sediments. The X-radiograph provides a supporting dataset that may reveal structures not visible in the surface optical scan.

An analysis of collected data was undertaken to examine the statistical significance of differences in MP concentrations with depth in the soil and to examine variations in MP size with depth. The distribution of the data was initially determined using the Shapiro–Wilk test. Where data were shown to be normally distributed, ANOVA (analysis of variance) tests and Pearson correlations were used. Where data were not normally distributed, the statistical analysis was underpinned by the non-parametric Mann–Whitney *U*-test, the Kruskal–Wallis test or the Spearman correlation test. Tests were performed on composite samples (i.e. all deep cores, all surface samples or all reference cores), unless otherwise stated.

3.3.2 Results

MPs were observed in all six of the deep cores extracted from field A, although evidence of MPs was limited to the top 35 cm of the cores (coinciding with the plough zone depth in the field). Within the 35-cm core depth where MPs were observed, the median concentration was 320.5 MPs kg^{-1} soil (interquartile range (IQR) 551.9 MP kg^{-1} soil). The highest concentration, 2103.1 MP kg^{-1} soil, was observed in the uppermost 5 cm (0–5 cm section) of the core and the lowest concentration (41.7 MP kg^{-1} soil) was observed in the 5 cm of core between 30 and 35 cm (Figure 3.3). The median concentration was higher (382.04 MP kg^{-1} soil) in the cores extracted at the top of the field than in the cores taken from the bottom of the field (299.28 MP kg^{-1} soil) but the difference was not statistically significant (Mann–Whitney U, $W=174$, $p=0.8738$).

Likewise, no statistically significant difference was observed when comparing the vertical distribution of MP in deep cores, assessed by considering the total (i.e. combined for all six cores) MP concentrations in 5-cm core sections (0–5 cm, 5–10 cm, etc.) for the 35-cm plough zone depth (Kruskal–Wallis $\chi^2=8.9797$, df = 6, $p=0.1747$). A weak, negative correlation (Spearman's $R=-0.39$, $p=0.054$) was observed in concentrations of MP with vertical depth in the deep cores. An Itrax X-ray image of a soil core extracted from field A (Figure 3.4) shows clearly a change in density structure between topsoil in the plough zone and the lower-level subsoils, with greater levels of compactions and higher soil densities in the latter (as evidenced by darker shading in grey). Density separation analysis was performed from soil samples extracted on both sides of the boundary layer separating the plough zone from lower-level subsoils, but no evidence of MP accumulation at this interface in the cores was observed.

Regarding the physical characteristics of MP recovered from the deep cores, the median MP particle size was 0.62 mm (across a particle size range from 0.09 mm to 3.79 mm). Smaller MPs were more prevalent, with around 40% of recovered particles less than 0.5 mm, around 30% in the range from 0.5 to 1 mm, 28% between 1 and 3 mm, and just 1.7% greater than 3 mm. No significant variation of particle size with depth was observed in the cores (Spearman's $R=0.13$, $p=0.051$). Fibres were the dominant shape found in soil samples and made up 74% of the particles recovered from the field A cores. The remaining MPs from these cores were characterised as fragments (22%) and films (4%).

An analysis of soil samples in the reference field cores indicated the presence of MPs, but only to depths of 15 cm (compared with 35 cm for the field A cores) and

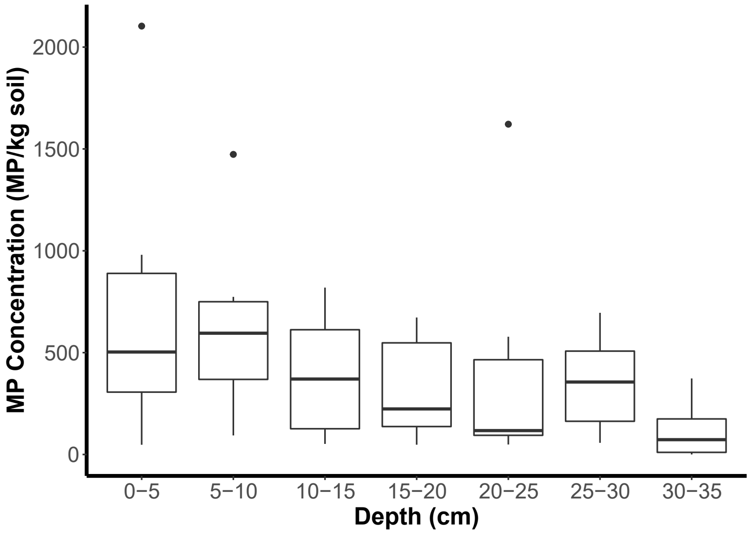

Figure 3.3. Variation of average MP concentrations with depth across all deep cores.

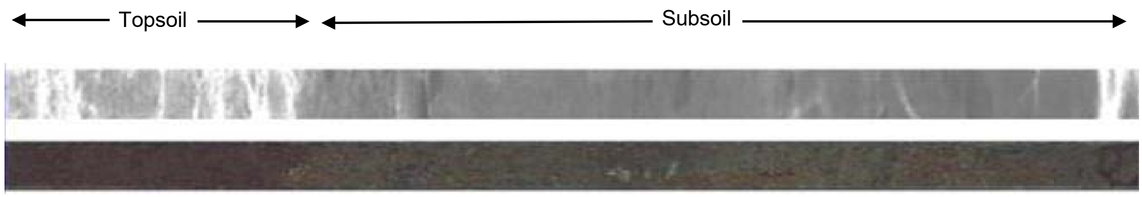

Figure 3.4. Optical image and X-ray of a soil core to a depth of 1 m.

at concentrations significantly lower (particularly for the surface samples) than those observed for field A. The median MP concentration across all three reference cores was 81.2 MP kg^{-1} soil (IQR 38.3 MP kg^{-1} soil), with maximum and minimum concentrations of 151 MP kg^{-1} soil and 44.3 MP kg^{-1} soil being found in the 0–5 cm and 10–15 cm layers, respectively. As was the case for the field A cores, fibres (87%) were the dominant MP shape in the reference cores, with fragments accounting for the remaining 13% of the MP mix (no film particles were observed in the reference cores).

All of the surface (shallow) soil samples (i.e. the 0–5 cm and 5–10 cm layers) from fields A and B were shown to contain MP. Among these samples, the median concentration was 330 MP kg^{-1} soil (IQR 144 MP kg^{-1} soil) in field A samples and 247 MP kg^{-1} soil (IQR 119.2 MP kg^{-1} soil) in field B samples. However, analysis of all samples from fields A and B combined revealed no statistically significant difference in the distribution of MPs between the 0–5 cm layer and the 5–10 cm layer (Mann–Whitney U, $w = 446$, $p = 0.959$). When considering MPs in surface soil samples at consecutive locations along the down-slope transects in fields A and B, no statistically significant correlation between MP concentration and location of sample was observed (field A: Spearman's $R = –0.049$, $p = 0.8$; and field B: Spearman's $R = –0.004$, $p = 0.82$), indicating little evidence of down-slope MP accumulations. Regarding the shape of the MPs recovered from the surface soil samples in fields A and B, 76% of MPs were fibres, 19% were fragments and 5% were films. Subsequent FTIR analysis conducted on 60% of all recovered MP confirmed that 58% of fibres were nylon, 23% were polyester (PES) and 10% were polyblend, while all MP films were polyethylene (PE). A greater variation in polymer type was observed for fragments, the majority of which were shown to be PE (40%), with PP, poly(methyl methacrylate) (PMMA), acrylonitrile butadiene styrene and polychloroprene among those constituting the remainder in percentages of 14%, 10%, 6% and 16%, respectively.

Finally, MPs were found in water samples from both the groundwater and the subsurface drainage outflow pipe discharging from field A to the adjacent stream. The concentration of MPs in groundwater at the field site was 0.29 MP L^{-1} (all fibres < 1 mm in size) and in water from the outflow pipe was 1.55 MP L^{-1} (81% fibres, 19% fragments).

3.3.3 Discussion

Although MPs were recovered in soil samples from fields A and B, and at higher concentrations than those in the reference soil samples, the abundances were lower than might have been expected given the abundances reported in the scientific literature for similar environmental settings (i.e. with extensive histories of biosolid land treatment). For example, Corradini *et al.* (2019) reported a median range of 1100–3500 MP kg^{-1} soil in agricultural fields in Chile treated with between one and five applications of biosolids. Additionally, van den Berg *et al.* (2020) reported an average of 2130 ± 950 low-density MP kg^{-1} soil, and 3060 ± 1680 MP kg^{-1} soil in biosolid-treated agricultural fields in Spain.

It is difficult to pinpoint the exact reasons why the MP concentrations observed in this study for Irish agroecosystems are lower than in other terrestrial MP research. One factor, however, may be the abundances of MP in sewage sludge itself. Mahon *et al.* (2017) reported a range of 4196–15,385 MP kg^{-1} sludge in Ireland, while significantly higher concentrations were found in Chile and Spain, with average abundances of 34,000 and 50,000 MP kg^{-1} sludge, respectively.

Although the rate of export of MPs from soil via surface runoff has not been quantified in this study, Crossman *et al.* (2020) reported a net MP loss of up to 102% following heavy rainfall, a factor that could contribute to the lower MP abundances reported here, given the climatic conditions that prevail across Ireland. Additionally, no standard method exists for removing MPs from soil, and extraction methods can vary across research programmes. For example, van den Berg *et al.* (2020) used a denser extraction liquid (NaI, 1.7 g cm^{-3}) than that used in this study ($ZnCl_2$, 1.55 g cm^{-3}), and, although the amount of additional MPs that could potentially be extracted from soil samples by using a higher-density buffer in the density separation process is not considered significant, it may contribute to higher MP recovery, particularly for dense particles.

Although a weak negative correlation was observed between MP concentrations and vertical depth in soil, no correlation between MP size and vertical depth was observed. Given that smaller MPs are more likely to be mobile, and the fact that concentration of smaller MPs was not significantly greater at greater depth, it seems most likely that annual ploughing is the main driver of vertical movement at the study site. This finding is echoed in the study by van den Berg *et al.* (2020), who suggested that ploughing and bioturbation were the likely reasons for the similarity in MP concentrations observed in soil samples extracted from agricultural fields at depths below the surface of between 0 and 10 cm and between 10 and 30 cm.

Although the reference cores in this study have experienced no land disturbance in the last 60 years, MPs have been recorded to depths of 15 cm. In this instance it is plausible that bioturbation or preferential flow paths (Zubris and Richards, 2005) underpin the processes governing movement. Given that MPs were not observed at a depth greater than the plough zone in the field A cores (biosolid-treated field), coupled with the fact that MPs were recovered from groundwater samples, albeit in low concentrations, the data suggest that vertical movement through the soil may present a relatively low-risk pathway for MPs to enter groundwater.

Further evidence to support this comes from the analysis of MP concentrations across the boundary layer between the topsoil and subsoil. The median MP concentration above and below the layer (averaged across all six cores) was 150.95 MP kg^{-1} soil and 0 MP kg^{-1} soil, respectively. X-ray scans (Figure 3.4) show a distinct change in the density structure of the soil along the plough zone that separated disturbed topsoil from deeper substrates, the darker colouring in the image reflecting a denser soil structure with reduced permeability and porosity below the plough layer. We hypothesised that MPs may accumulate at this boundary zone and potentially be unable to migrate further into the soil. However, while such accumulation was not observed in our study cores, this may be an issue in other settings, particularly where a plough-pan has formed. For this reason, varying the ploughing depths from year to year in tillage operations is recommended.

To determine if MPs exhibit overland movement down slopes, we examined MP concentrations in surface soil samples (0.1 m deep) from along the downslope transect in fields A and B. We found no statistically significant difference in MP concentrations in samples taken at different levels along these transects, that is, the distribution of MPs across surfaces was relatively even and there was no evidence of MP build-up at lower levels. There are a number of potential reasons for this, including the fact that "free" MPs (i.e. those not bound to soil particles) may already have been transported from the fields and/or that the remaining MPs had formed aggregates with the soil and were therefore less mobile (a process described by Rehm *et al.*, 2021). This is particularly likely given that biosolids had not been applied to these fields since the previous year. Additionally, samples were extracted after crops had been harvested, and so the soil was still relatively compact following the use of heavy machinery.

Another potential pathway by which MPs can leave soil systems is through interflow, which may be enhanced as a result of artificial drainage of agricultural fields. Approximately 50% of Irish land is considered poorly drained, in part because of Ireland's temperate climate but also as a result of human interaction with the soil and the historical effects of glaciation (Connolly and Holden, 2009; SIS, 2014). Artificial drainage networks in Ireland are typically managed by private landowners, who are not required to disclose information regarding drainage networks to local authorities. As a result, our knowledge of the extent to which agricultural land is artificially drained remains limited (Paul *et al*., 2018). Given the high percentage of poorly drained land in Ireland, Mockler *et al*. (2013) predicted that 44% of agricultural land in the country is likely to be artificially drained. A mean concentration of 10.5 MP L^{-1} was reported by Bigalke *et al*. (2022) in agricultural drainage water extracted near fields where extensive plastic mulching was used. A much lower concentration was observed in drain samples from field A in this study (1.55 MP L^{-1}). However, the samples analysed by Bigalke *et al.* (2022) were collected following significant rainfall, which was not the case in our study, and this may explain the difference. Further research into MP concentrations in interflow and drain flow is recommended, given the significant reliance of the Irish agricultural sector on land considered to have some degree of artificial drainage.

3.4 Export of Microplastics from Terrestrial Systems through Overland Flow Pathways

3.4.1 Overview

Rainfall simulation laboratory experiments on small-scale test catchments were undertaken to investigate the overland flow pathways by which MPs can be mobilised and exported from terrestrial systems. The use of a laboratory-based rainfall simulation rig was preferred over field-based testing as the former enables the influence of a range of variables to be individually tested under controlled conditions. Key variables that were tested in the context of understanding this overland MP transport pathway included particle shape, density and size, catchment slope, rainfall regime (intensity and duration) and timing of the rainfall event after MP seeding, together with the catchment condition in the progression of a growing cycle (soil to grass).

3.4.2 Methodology

The rainfall simulator was designed and constructed such that two Veejet pressurised nozzles would spray from a stainless-steel catch pan, and a rotating disc would disperse the rainfall (Figure 3.5). The rainfall simulator was suspended 2 m above the test catchment and fed with potable water from a tank. A plastic tarpaulin surrounded the structure to ensure minimal impact from wind. The rainfall simulator was custom designed to replicate the low rainfall intensities experienced in Ireland and controlled using individual pressure gauges connected to each shower head. Limitations due to data privacy allow for a very restricted overview of where biosolids are spread in Ireland. Given the significant east–west rainfall gradient that exists across Ireland (annual rainfall in the east of the country being between approximately 750 and 1000 mm, compared with more than 3000 mm in elevated western regions), a central location (Irish Grid: easting 203973, northing 241213) in the Irish midlands was chosen to determine average return periods for high-intensity precipitation events. From an analysis of precipitation records at this location, two rainfall intensities of 8.4 mm h^{-1} and 18 mm h^{-1}, corresponding to 2- and 30-year return periods, for 30-minute durations were selected for study.

Figure 3.5. Experimental rig for overland flow tests.

A scaled-up version of the soil box (measuring 3.3 × 1.2 × 0.1 m (length × width × depth)) outlined by Kibet *et al.* (2014) was constructed, and 15 0.05-m holes were cut in the base to allow water to drain freely. The box was constructed out of plywood coated with an epoxy resin and fitted with PVC guttering to collect the runoff from the test sample. A layer of muslin was added to the base of the box to retain soil and prevent this from escaping through the drainage holes. Each experiment was conducted using fresh soil to ensure no MP contamination from previous events/tests, and this necessitated the use of significant soil quantities. For this reason, commercially available topsoil with a loamy texture (particle size distribution: clay 21.1%, silt 40.9%, sand 37.8%) was procured (from Landscape Depot, Tallaght). A bulk density of 1.36 ± 0.2 g cm^{-3}, consistent with that in agricultural settings, was achieved in the test boxes by adding a known weight of soil in incremental layers of 2.5 cm, with each layer compacted and lightly scarified before a new soil layer was added. Care was taken to ensure that the soil surface in each test box was level prior to the commencement of each experiment. This minimised the risk of preferential flow paths forming over the surface of the test sample, which could skew the experiment results.

Table 3.1. Characteristics of MPs used in surface runoff experiments

Polymer type	Density ($g\,cm^{-3}$)	Mean size (mm)	Max. size (mm)	Min. size (mm)
PP	0.92	0.65	2.5	0.05
HDPE	0.97	0.85	3	0.15
PVC	1.38	1.5	5	0.15

The use of virgin MP in laboratory experiments has been criticised in recent scientific literature as they are not deemed sufficiently representative of MPs found in the natural environment (Waldman and Rillig, 2020). As a result, secondary MPs for experimental purposes were produced from a variety of plastic materials (see Table 3.1). Blue PP rope was manually cut with scissors to obtain fibres/cylinders. Opaque HDPE milk bottles were cut into small pieces and shredded in a blender (Tefal Blendforce II) to produce fragments. A black PVC guttering pipe was cut with a band saw, and the sawdust was collected to provide flakes/fragments (see Figure 3.6 for SEM (scanning electron microscope) images). For the remainder of the discussion pertaining to these rainfall simulation tests, PP and HDPE are grouped together as low-density MPs, and PVC is considered a high-density MP.

All rainfall simulation experiments followed the same method. Soil was saturated and remained so until the commencement of rainfall. MP were seeded evenly on the surface in a concentration of $10\,g\,m^{-2}$ following the method of Rehm *et al.* (2021), and a rainfall event was initiated either 48 or 216 hours after seeding, depending on the test. Rainfall events lasted a total of 1 hour made up of two 30-minute periods, the first with a rainfall intensity of $8 \pm 0.8\,mm\,h^{-1}$ and the second with an increased rainfall intensity of $18 \pm 0.9\,mm\,h^{-1}$. A Casella 0.5-mm tipping bucket rain gauge measured rainfall depths during the experiments, and water samples were collected at 10-minute intervals following the commencement of rainfall and subsequent runoff from the catchment.

In total, 15 experiments were performed across four test series where specific parameters were independently tested. In **series 1**, four experiments were conducted to examine the impact of increasing the time period (t) from the application of MPs to surfaces of bare soils in the test box to the commencement of rainfall (Table 3.2). Two timings were investigated: $t = 48$ hours and $t = 216$ hours. This approach followed that of Lucid *et al.* (2014), with the exception that, in the current study, the impact of varying the timing (t) was considered in stand-alone experiments, whereas Lucid *et al*. considered the two timings successively in a single experiment. The MPs tested in this test series were limited to low-density MPs, and two catchment gradients, 2.5% and 11%, were investigated for each value of t. The upper gradient of 11% reflected the maximum catchment gradient at which biosolid land treatment is permitted in Ireland.

Test **series 2** aimed to further investigate the impact of slope on the overland runoff of MPs. Series 2 experiments were conducted in triplicate in soil boxes at two slopes, 2.5% and 11% (six experiments in total). Both low- and high-density MP were investigated, with rainfall commencing 48 hours after MP seeding.

In test **series 3**, triplicate experiments were again undertaken, examining in this case the impact of a grass sward (a single-species perennial ryegrass) on MP erosion. Tests were conducted on smaller samples established in the test box (measuring $1.8 \times 1.2 \times 0.1$ m (length × width × depth)). The perennial ryegrass (Xenon) was grown in shallow

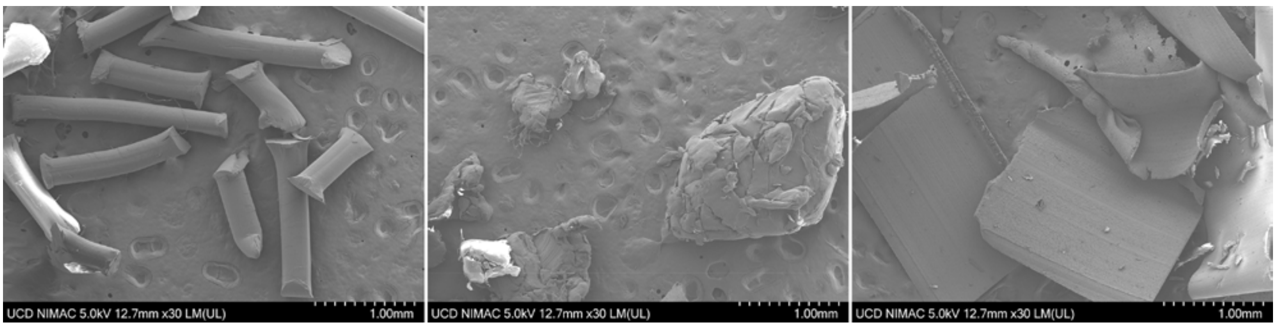

Figure 3.6. SEM images demonstrating the surface morphological characteristics of MPs used in experiments. From left to right: PP, HDPE and PVC.

Table 3.2. Overview of test series for overland runoff experiments

Series	No. of experiments	Surface condition	Timing of rainfall (h)	Gradient (%)	Total MP exported (g)	Total MP exported (%)
1A	1	Bare	48	11	0.8551	2.4
1B	1	Bare	216	11	0.5405	1.5
1C	1	Bare	48	2.5	3.7782	10.8
1D	1	Bare	216	2.5	3.086	8.8
2A	3	Bare	48	2.5	0.141	0.47
2B	3	Bare	48	11	0.019	0.06
3	3	Grass	48	2.5	0.018	0.12
4A	1	Bare	48	2.5	0.5858	1.67
4B	1	Bare	48	2.5	0.6526	1.86

soil boxes (3.5 × 1.3 × 0.05 m), seeded at a rate of 14 kg ha^{-1}. Following a growing period of 6 weeks, sods were relocated to the main soil box in sections (30 × 30 cm), with considerable care taken to ensure that the soil surface remained even, minimising the risk of preferential flow paths emerging following the commencement of rainfall. All three experiments were conducted with the soil box pitched at a gradient of 2.5%, and again the rainfall commenced 48 hours after MP seeding.

Finally, test **series 4** focused on investigating the impact of contour ploughing on MP runoff. Two experiments were conducted with "cultivation" tracks running in the first instance parallel to the field slope and subsequently perpendicular to the field slope. Cultivation tracks (not to scale) for experimental purposes were created in the sample box by standard raking such that furrows of around 10 mm in depth were formed on the soil surface. Tests were conducted on a gradient of 2.5% and for rainfall occurring 48 hours after MP seeding.

Runoff samples, collected every 10 minutes, were filtered through a 0.3-mm stainless-steel mesh sieve. Retained MPs were transferred onto a tinfoil layer and oven dried for 12 hours at 65°C. Samples were then analysed under a microscope (Ash Omni) and separated into their three polymer groups (PP, HDPE and PVC). The mean size of each polymer was recorded by taking a subsample, measuring length and determining an average value. Owing to the high volume of MPs, mass of MPs (correct to 0.0001 g) rather than number of particles was recorded. Experimental results were analysed in a manner similar to that described in section 3.3.1. The results from different test series were considered together so that the significance of certain parameters (e.g. rainfall intensity) for overland MP transport could be assessed. Given the smaller samples sizes that were tested in series 3, and to facilitate the cross-comparison of results from different test series, the results of all test series were, for the most part, calculated as a percentage of the transported MP relative to the total MP applied to soil surfaces (rather than actual masses of exported particles being reported).

3.4.3 Results

In all tests, the differences in MP concentrations exported from the test boxes at rainfall intensities of 8 mm h^{-1} and 18 mm h^{-1} were statistically significant (Mann–Whitney U-test: $p = 0.0001$). The correlation between MP export and rainfall intensity was strongest in experiments conducted with low-density MPs (series 1 and 4) (Spearman's $R = 0.87$, $p = 0.0001$). A slightly weaker positive correlation was determined for experiments that examined both low- and high-density particles (series 2 and 3) (Spearman's $R = 0.6$, $p = 0.0001$). The maximum recorded MP export was 10.8% (3.7782 g) (experiment 1C) and the minimum was 0.06% (0.019 g) (experiment 2B). A comparison of data from tests 1A and 1C with those from tests 1B and 1D showed that the timing of the rainfall event following MP seeding did not have a significant effect on MP export (one-way ANOVA, $F = 0.151969$, $p = 0.699836$, F-crit = 4.2250201). Somewhat surprisingly, the direction of cultivation, tested in series 4 (furrows perpendicular or parallel to the field slope), was not found to have a significant effect on MP overland export (Mann–Whitney U-test: $w = 20$, $p = 0.62$), but this may be attributable to scaling issues in the experiment set-up.

Slope was shown to be less influential than rainfall intensity in overland MP transport, as a moderate negative correlation was observed between MP export and catchment slope (series 2) (Spearman's $R=-0.52$, $p=0.00041$), and a slightly weaker negative correlation was observed in experiments with only low-density MP (series 1) (Pearson's $R=-0.46$, $p=0.023$). Furthermore, the presence of a grass sward was shown to reduce the export potential of MPs, with average MP export for the perennial ryegrass test configuration in series 3 being 0.12%, compared with 0.47% for the bare soil tested in series 2A (Mann–Whitney *U*-test: $W=111$, $p=0.006095$) (Figure 3.7).

The impact of rainfall intensity on MP export was reduced by the presence of grass, with a moderate positive correlation observed (Spearman's $R=0.53$, $p=0.025$), compared with a strong correlation on bare soil (series 2A) (Spearman's $R=0.8$, $p=0.0001$). In relation to MP particles, surface roughness appeared to have an impact on the mobility of MPs, as HDPE and PVC were shown to be less mobile than PP. Among the experiments examining low-density MP only (series 1 and 4), the average percentage breakdown between PP and HDPE was 85% and 15%, respectively, which changed in the experiments examining both low- and high-density MP (series 2 and 3), with breakdowns of 62%, 35% and 3% (PP, HDPE and PVC, respectively).

Finally, among the parameters tested, rainfall intensity was solely responsible for promoting variation in the average size of MP, with no statistical significance observed with slope, surface texture (grass compared with bare soil) or direction of cultivation. Smaller MP particles were more mobile across all experiments, with larger particles increasing in mobility only with an increase in rainfall intensity. Increases in MP mobility were shown to be shape specific, with the strongest correlations noted for PP (Spearman's $R=0.75$, $p<0.0001$) and HDPE (Spearman's $R=0.74$, $p<0.0001$), while a very weak correlation was found for PVC (Spearman's $R=0.11$, $p=0.58$). The average MP size during the low-intensity rainfall event was 0.89, 0.63 and 1.67 mm for PP, HDPE and PVC, respectively, increasing to 1.15, 0.89 and 1.74 mm for the same polymers during the high-intensity event.

3.4.4 Discussion

This study examined some of the key parameters considered to influence the overland movement of MPs across terrestrial soils. Among the parameters tested, rainfall intensity was found to be one of the most influential, with findings suggesting that an increase in intensity is strongly correlated with an increase in MP export. This result is consistent with other research, in that heavy rainfall is one of the key drivers mobilising MP in surface runoff events (e.g. Crossman *et al.*, 2020; Hitchcock, 2020). Although the highest rainfall intensity examined in this research had an estimated return period of 30 years, climate change predictions for Ireland point towards

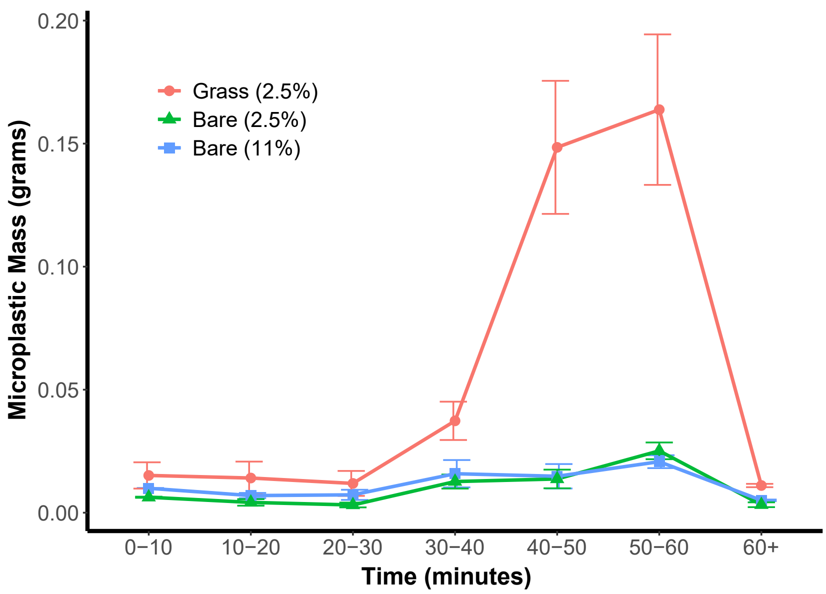

Figure 3.7. Average percentage of MPs exported over time in series 2 and 3 tests.

an increase in the frequency of heavy rainfall events, particularly during the autumn and winter months (Nolan, 2015), with the likelihood that high-intensity events may become commonplace in future years.

The impact of rainfall intensity on MP export was weaker when tested on grass sods, as the concentrations of MPs exported was lower than from bare soil. This is consistent with the findings of Pan and Shangguan (2006) regarding sediment particles, as they showed that grass lessened erosion by reducing raindrop energy due to canopy cover, increasing surface roughness and increasing the stability of soil. The results, therefore, may point towards a decrease in exportation potential of MP from fields with vegetation or crops, while other agricultural practices, such as tillage, may be representative of activities that cause MPs to become more mobile from exposure and the increase in the erosion vulnerability of field soils.

Furthermore, the addition of PVC in experimental series 2 and 3 is also considered to have contributed to an increase in surface friction. Largely immobile, partly because of their high density and larger size, PVC particles were observed to enhance surface irregularities in the soil, causing a build-up of PP and HDPE particles in localised areas. While the true impact of the increased surface friction caused by the PVC particles cannot be quantified, surface friction may account for some of the variability in total MP exported between experiments undertaken with low-density MP and those conducted with both low- and high-density particles. Physical characteristics of MPs, such as size and surface roughness, were also found to influence export potential. Smaller particles, particularly PP and HDPE, were significantly more mobile across all experiments, these findings being consistent with similar research on MP mobility (e.g. O'Connor *et al.*, 2019). However, HDPE and PVC were significantly less mobile than PP because of their surface roughness and shape rather than the polymer type. A decrease in mobility with an increase in surface roughness is most likely due to a greater level of surface friction along the contact interface between particles and the soil surface. Surface roughness is an important parameter given that MPs in the natural environment are exposed to various weathering processes that have been found to increase fragmentation, surface brittleness and, therefore, surface roughness (Liu *et al.*, 2020).

An interesting finding from this research was the negative correlation between increasing catchment slope and decreasing MP export. It had been hypothesised that an increase in slope would contribute to an increase in soil erosion (e.g. Jourgholami *et al.*, 2021) and, therefore, to an increase in MP erosion/transport. Instead, our results support recent findings by Laermanns *et al.* (2021), who noted that an increase in slope resulted in a decrease in the thickness of the film of water running over the surface. A thicker film of water may allow an MP particle to be entirely engulfed by water, reducing the impact of other processes that might otherwise impede movement, such as surface roughness and soil texture.

Although Rehm *et al.* (2021) reported that the timing of rainfall after seeding affects MP mobility, we found no statistically significant correlation between the timing of a rainfall event after MP seeding and the concentration of MP exported. However, this was perhaps unsurprising, as Rehm *et al.* (2021) investigated this relationship over markedly longer time frames (up to 1.5 years).

Although we found no statistically significant difference in overland MP export between fields cultivated parallel to the field slope and fields cultivated perpendicular to the slope, the results do not exclude the possibility that contour cultivation may contribute to reductions in MP export from agroecosystems. Given the potential for scale effects to have an impact on the cultivation experiments reported in this study, larger-scale field tests to further examine the role of sediment loss mitigation measures (e.g. contour ploughing, riparian buffers, conservative ploughing) and how these may affect the overland movement of MP are recommended. While this study offers some important insights into the characteristics of MP overland movement, further research is needed to fully quantify MP export from terrestrial systems and to understand the level of risk this poses to aquatic systems.

3.5 The River Slaney as a Transport Pathway

3.5.1 Overview

MPs are ubiquitous in river environments, which are well-known conduits of MP pollution to the marine environment. There are multiple potential sources of the MPs found in rivers, including point sources from WWTPs and diffuse sources from agricultural land, both of which are a dominant feature of Irish riverine catchments. In Ireland, little empirical evidence currently exists on the abundance of MPs found in freshwater ecosystems. This research aims to investigate temporal and spatial trends in MP abundance in the upper reaches of the River Slaney catchment over a 12-month period.

3.5.2 Methodology

Sampling was undertaken upstream and downstream of three WWTPs (Baltinglass, Rathvilly and Tullow) in the upper reaches of the River Slaney. A reference site in the headwaters of the catchment, not affected by WWTPs or licensed waste facilities, was also sampled. Sampling at all seven locations was conducted monthly between April 2018 and March 2019. Four replicate surface water samples were taken from the top 40 cm at each sampling site. A 200-L volume-reduced sample was collecting using a 10-L HDPE bucket and subsequently filtered on the riverbank using a stainless-steel sieve (150 μm), giving a 50-L composite sample. The contents of the sieve were washed into glass bottles (1 L) with metal screw-cap lids, which had been acid washed (in 0.05% nitric acid) and dried before sampling (O’Connor *et al.*, 2019). Four replicate sediment samples were collected from the top 10 cm of the riverbed at random locations and placed into sterile polyethylene sampling bags from each sampling site. The processing of water and sediment samples, including FTIR analysis and contamination control, followed Frias *et al*. (2018). Particles were placed into two size categories (100–350 μm and 350 μm to 5 mm) based on recommendations by Frais and Nash (2019) for standardising the reporting of MPs and improving comparability between studies. Particles were measured using CellSens software (QImaging Retiga 2000R digital camera), with the length and width of fibres and filaments measured and area of fragments recorded.

3.5.3 Results

MPs were reported in all surface waters and shore sediment samples from all seven sampling locations and in all 12 sampling months. A total of 76,903 MPs were processed and analysed from surface waters (29,105) and shore sediment samples (47,498). Fibres dominated both the surface waters (93%) and the sediments (77%); blue fibres were the most common, followed by clear fibres, and in both cases most of those recorded were in the 350 μm to 5 mm size range. The mean abundance in surface waters varied widely, from 236 ± 46 to 1174 ± 116 MPs m^{-3}, over the 12-month sampling period (Figure 3.8). Significant temporal variation in mean monthly abundance was observed; for example, mean abundance was significantly higher in June 2018 than in July 2018 and November 2018. Similarly, mean abundance in shore sediment samples ranged widely, from 466 ± 3 to 1798 ± 137 MPs kg^{-1} dw (dry weight) over 12 months (Figure 3.8). Significant temporal variation in median monthly abundance was also observed.

Furthermore, monthly median MP abundance in water and sediment varied significantly from month to month at each site and within a month between the sites (Baltinglass downstream, Rathvilly upstream and Tullow downstream). Monthly rainfall levels varied greatly throughout the 12-month sampling period, although water levels in the river remained steady. The MP abundance in surface waters or sediment samples was not found to correlate with water levels each month, and similarly there was no correlation between rainfall and MP abundance in sediment samples. However, a negative correlation was observed between monthly mean MP abundance in surface waters and monthly rainfall levels, indicating that as rainfall levels increase particle abundance in the River Slaney surface waters decreases.

Polymers were observed to change longitudinally, often with a significant association between polymer type and site. For example, the reference site and Baltinglass upstream were very similar, dominated by particles of polyester urethane (PEU), PE and copolymers; however, once the river passed the first WWTP outflow at Baltinglass downstream the polymer composition changed, with PES dominating, followed by polyester epoxide (PEE), PS and copolymers. Further downstream, at Rathvilly upstream and downstream and Tullow upstream, polymer composition was dominated by poly(ethyl methacrylate) (PEMA) and polyacrylic acid (PAA). Again, as the river passed Tullow downstream outflow, a different polymer composition was observed, dominated by PAA, PES, PEE and PEMA.

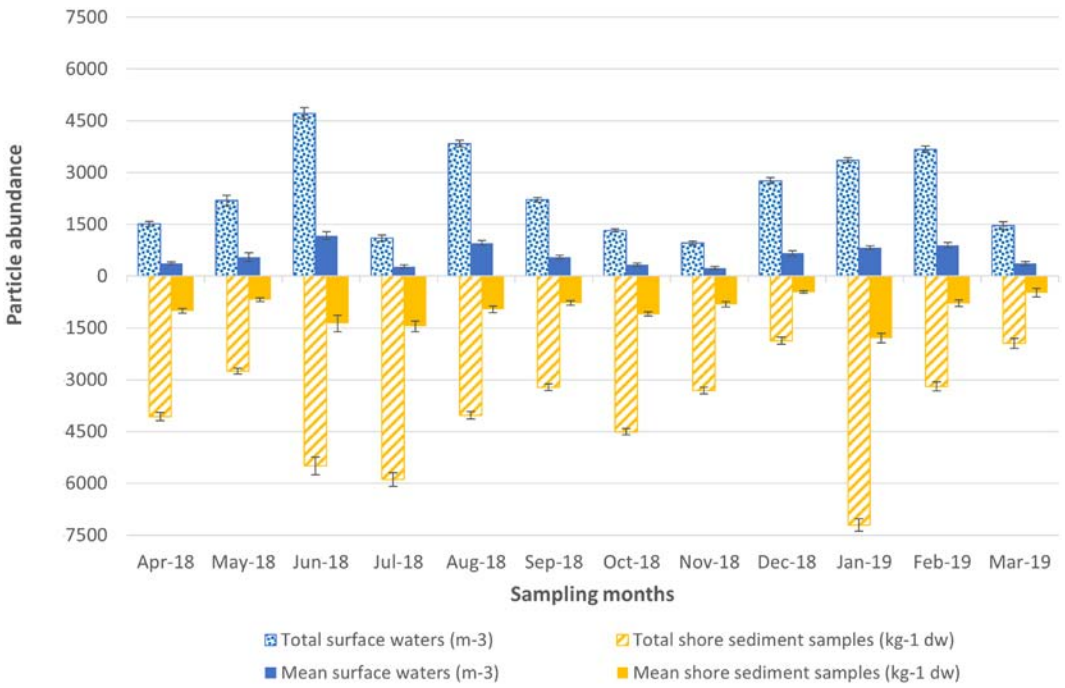

Figure 3.8. Total and mean MP abundance in surface waters (blue) and shore sediment samples (orange) in the upper reaches of the River Slaney during a 12-month sampling period (April 2018 to March 2019).

3.5.4 Discussion

MP abundances in the surface waters of the River Slaney were comparable to those found in heavily polluted and urbanised freshwater environments worldwide, such as the Nakdong River, South Korea (May: 700 MPs m^{-3}), and the Yellow River, China (March: 930 MPs m^{-3}), during the dry season (Han *et al*., 2020), while the sediment samples were comparable to those from the upper St Lawrence River, Quebec (832 MP kg^{-1} dw) (Crew *et al*., 2020), and the River Main, Germany (1077 MPs kg^{-1} dw) (Klein *et al*., 2015). As expected, increased rainfall was seen to dilute the abundance of MP in surface water. Furthermore, this study illustrates that river catchments are complex, and the spatial and temporal distribution of MPs cannot be explained by rainfall or water levels alone, with further research required to account for atmospheric deposition, the recreational use of the river, hydrological events, the hydrogeomorphology of the river, and land use within the catchment.

4 Modelling Microplastic Risks to Waterbodies

4.1 Introduction

This chapter focuses on modelling the risks from landspreading WWTP sludge containing MPs, treating it as diffuse source pollution. Published figures on the fate of MPs in WWTPs have indicated that MPs are captured mainly in the sludge itself (see section 2.2.3.1). An updated map of land potentially suitable for landspreading sludge is presented in section 4.2, while section 4.3 shows how risks to water courses may be estimated and aggregated.

4.2 Microplastics: Areas Potentially Suitable for Landspreading Wastewater Treatment Plant Sludge

The European Sewage Sludge Directive (European Council, 1986) lays down conditions for the spreading of sewage sludge on agricultural land. In Ireland, it has been implemented by S.I. No. 148/1998. This sets out the conditions under which sewage sludge may be landspread and includes factors relating to land suitability.

- The directive sets limits on the heavy metal content of the soil (Table 4.1) (depending on soil pH values).
- Landspreading of sludge is not limited in the case of some types of land use. These have been interpreted based on the Corine land use categories of pastures (231), annual crops associated with permanent crops (241), complex cultivation patterns (242) and land principally occupied by agriculture, with significant areas of natural vegetation (243) (Corine identifiers).
- Landspreading must not impair groundwater or surface water. The protection of groundwater has been interpreted in relation to the groundwater vulnerability map produced by the Geological Survey of Ireland. Any areas in this map where groundwater vulnerability is classified as extremely high (or rock or karst) are excluded from the landspreading of sludge.
- There should be a buffer zone from surface waterbodies of 20 m for gradients < 6% (larger buffer width otherwise).

Table 4.1. EU and Irish limits on metals content of soils for landspreading sludge (S.I. No. 148/1998 imposes the lower limit for all parameters)

Metal	EU limit (mg/kg dry matter)
Cadmium	1–3
Chromium	No limits set yet
Copper	50–140 (no values above 50 in current data)
Lead	50–300
Mercury	1–1.5 (no values above 1 in current data)
Nickel	30–75
Zinc	150–300

There are other considerations. For instance:

- limits on the spreading period, i.e. before grassland or forage crops are harvested (to be set by EU Member States and not to be less than 3 weeks);
- a requirement that the nutrient needs of plants be taken into account, and the later S.I. No. 267/2001 adds that spreading must be in accordance with a nutrient management plan; and
- limits on the heavy metal content of sludge.

A national wastewater sludge management plan was produced by Irish Water in 2016 and was due to be revised and upgraded in 2021. It contains (page 69) a map of Ireland showing lands potentially suitable for landspreading sewage sludge based on the restrictions on cadmium and nickel in Table 4.1, extreme groundwater vulnerability and land use (Corine 2012) limitations. Other limitations are to be assessed for proposed spreading sites on a case-by-case basis.

Here we develop a new map of areas potentially suitable for landspreading that:

- takes account of all six soil metal limitations specified in S.I. No. 148/1998 (maps are now available showing the distribution of all listed metals at a grid square resolution of 2 km);
- uses the more recent Corine 2018 land use classification map;

- builds in explicit consideration of the objective to protect surface waterbodies by (i) implementing a 20-m buffer restriction on rivers, (ii) excluding land likely to generate large volumes of runoff and (iii) excluding overland flow pathways with high accumulations of runoff from upslope areas.

The last of these is the least straightforward, as a number of considerations must be taken into account to protect surface waters when choosing land suitable for sludge landspreading. First, the provision of a buffer zone around rivers (in this case of 20 m) is intended to prevent impacts on surface waters from the application of the sludge on land close to water bodies. However, even if sludge is applied further away, overland flow can mobilise MPs from more distant sources and transport them into the water course.

The recently completed EPA-funded DiffuseTools research project (Thomas *et al*., 2021) modelled the mobilisation of phosphorus and sediment by overland flow and produced maps of areas of surface runoff and locations that concentrate surface runoff from upslope areas into overland flow pathways that can deliver particulates, and therefore MPs, to streams at specific locations, shown as red dots in Figure 4.1 (Thomas *et al*., 2021). These areas of concentration can have greater flow film thicknesses, with the potential to mobilise more and larger MPs. The yellow areas show the overland flow pathways determined by modelling the soil response to rainfall, using soil type and topography. Any sludge spread in these areas is likely to be mobilised by this overland flow, so spreading is not advisable in these areas. The areas with the highest range of overland flow accumulations have been mapped and are removed from our map of permitted spreading areas. In addition, areas that are likely to generate large amounts of surface runoff, for example because of poor soil infiltration capacity or steep slopes, should be excluded. Maps of average annual runoff production per year (determined for the 30-year period 1981–2010) are used to identify these areas. Maximum thresholds are set for runoff and upslope flow accumulations, and all areas estimated to have values higher than these thresholds are excluded from the map.

The overall procedure is shown in Figure 4.2. The boxes in blue are interpretations of S.I. No. 148/1998, while the boxes in green are factors introduced in

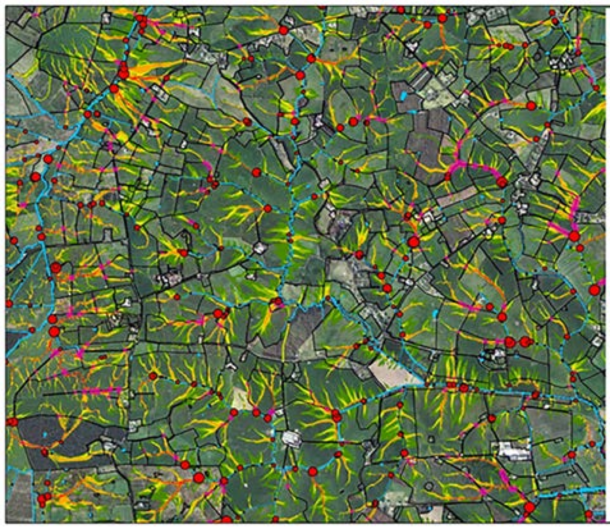

Figure 4.1. Close-up of a section of the surface runoff delivery map to streams showing main overland flow pathways (yellow), locations of breakthrough points at field boundaries (pink circles) and delivery points to waterbodies (red circles). Source: DiffuseTools project (Thomas *et al*., 2021).

this project to protect surface water courses from MPs carried by surface runoff and overland flow. A series of Python scripts has been written to use the ArcPy library (associated with ArcGIS) to generate automatically an overall potential suitability map (Figure 4.3) incorporating all the factors in Figure 4.2. The Python scripts allow easy recalculation and updating if new mapping of metals, groundwater vulnerability, land use or runoff becomes available, if the buffering distance from waterbodies is changed or if any of the limiting soil metal concentrations or runoff thresholds are changed.

Our updated map is more restrictive, showing 1.97 Mha of potential land compared with the 2.54 Mha shown in the 2016 map. Much of this reduction is due to our exclusion of high-runoff areas and areas of overland flow concentration. This is shown in the close-ups of Figure 4.4 (A and B), in which the green areas are the identified potential landspreading areas and the red areas are the exclusions because of runoff and overland flow. The linear yellow areas are excluded due to buffers around water courses and the white areas are excluded because of heavy metals, unsuitable land use or groundwater vulnerability, or any combination of these.

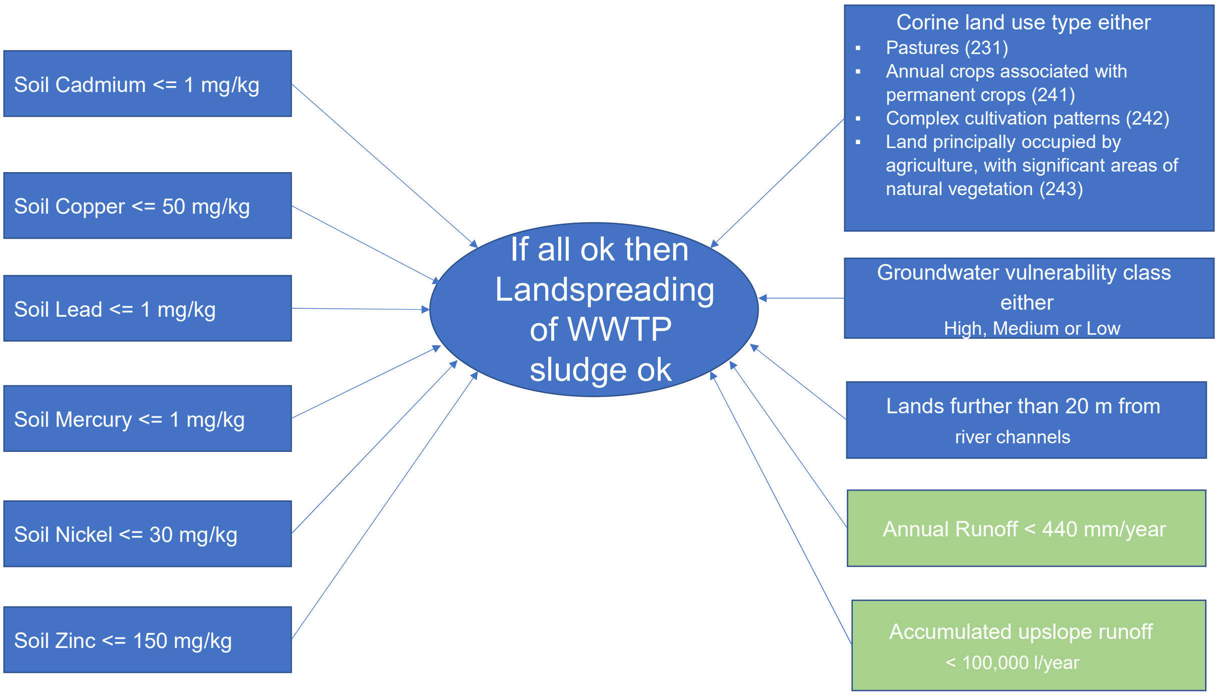

Figure 4.2. Factors determining land suitability for spreading wastewater sludge.

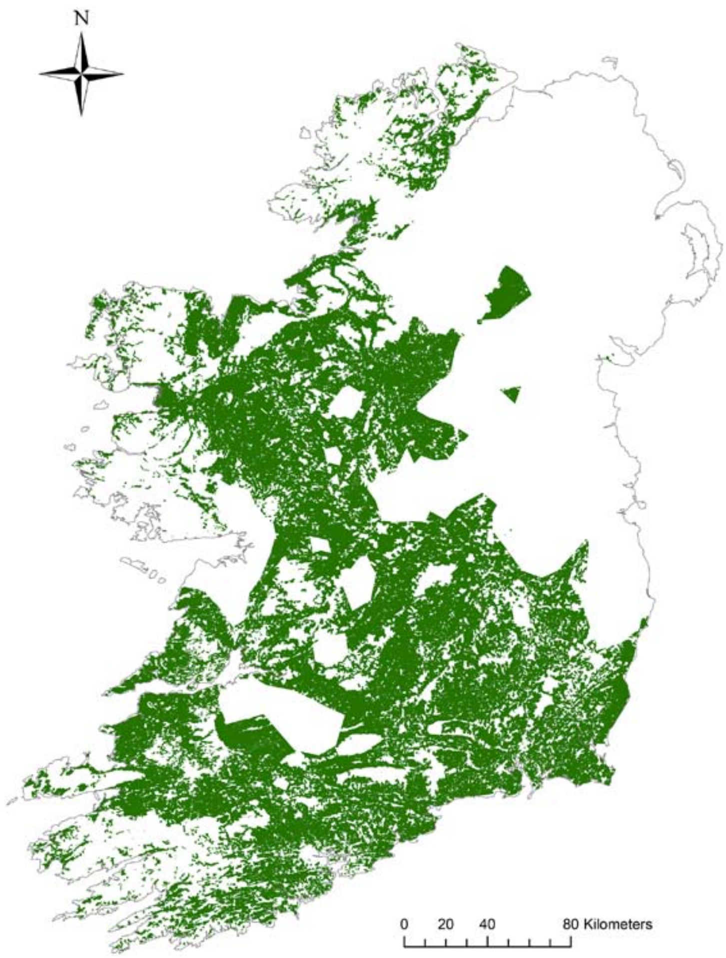

Figure 4.3. Indicative national map of lands potentially suitable for landspreading sludge (with exclusions due to surface runoff risks).

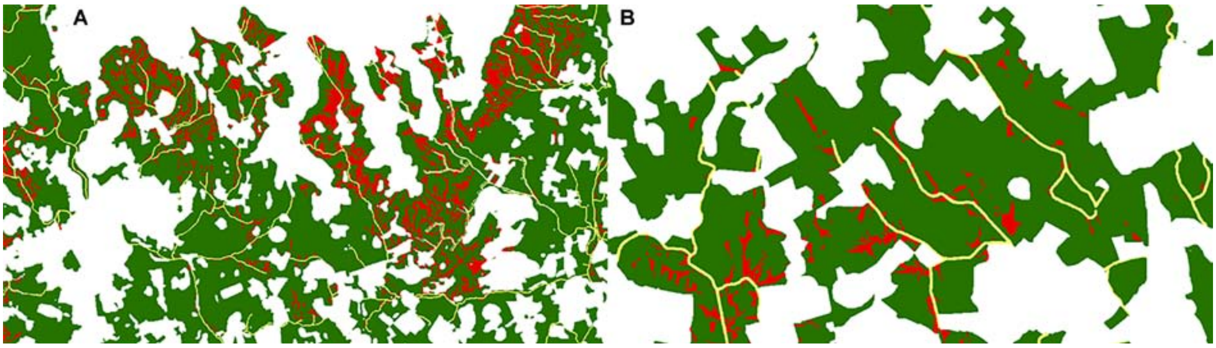

Figure 4.4. Close-up of (A) areas suitable for landspreading of sludge showing severe fragmentation (in green) and (B) areas potentially suitable for landspreading sludge with less fragmentation (in green). (Red areas are excluded because of surface runoff considerations, yellow areas are 20-m buffers around water courses and white areas are excluded due to heavy metals, unsuitable land use or groundwater vulnerability.)

Note that the red runoff/overland flow concentration exclusions tend to lead into or border the yellow river buffer areas, as might be expected. The map is available as a shapefile and thus can be used in computer tools to check proposed areas for landspreading at high resolution. Note that in some areas the potential lands are quite fragmented, as in Figure 4.4A, while in others there are substantial portions of connected potential areas, as in Figure 4.4B. Since the landspreading map is available in electronic form as a shapefile or in a geodatabase, it can be incorporated into tools to check that proposed landspreading areas conform to the map and to identify mismatches.

4.3 Diffuse Source Modelling and Risk Identification for Subcatchments

Wang (2018) developed a model to estimate the annual export amounts of MPs contained in landspread sludge from WWTPs. This is only one of the possible sources and pathways for delivering MPs to rivers, but it is likely to be a major one, as WWTPs receive wastewater from industrial and household sewage and storm runoff as well as leachate from landfills, and the MPs they receive are concentrated in the sludge they generate.

The method developed was implemented in a GIS package with spatial data on annual rainfall amounts and actual evapotranspiration (from Met Éireann), annual recharge amounts (from the GSI) and a map of locations where sludge from WWTPs might be landspread and a shapefile of EPA-delineated subcatchments. For every polygon in the spatial data, the calculation of annual MPs export per unit area is based on surface runoff estimates from information on rainfall, evapotranspiration and infiltration for local conditions. To illustrate the procedure, we used a subset of historical data about landspreading locations and amounts of sludge used in the past, compiled by A.M. Mahon (personal communication, 2 May 2020), extracted from the local authority sludge records in 2014.

These data are illustrated in Figure 4.5. We do not have access to the individual farm sizes and layouts of fields, so, for illustration, we have assumed an average farm size of 100 ha and a circular field shape, centred on the grid coordinates in the sludge record. The GIS script estimates the annual export of MPs from each farm, taking into account its applied load of MPs and annual runoff amount. It then aggregates these for all polygons in which sludge is landspread and assigns the total to the river subcatchment in which the lands lie. This gives a maximum amount of MP particles potentially exported to the river segment from each subcatchment (Figure 4.6) by aggregating the outputs of all the farms in a subcatchment.

The overall workflow is shown in Figure 4.7, and this estimates export values for each subcatchment (Figure 4.8). These values can be colour-coded with respect to their magnitude and shown as an overall subcatchment risk map for MPs from this landspreading source (Figure 4.9).

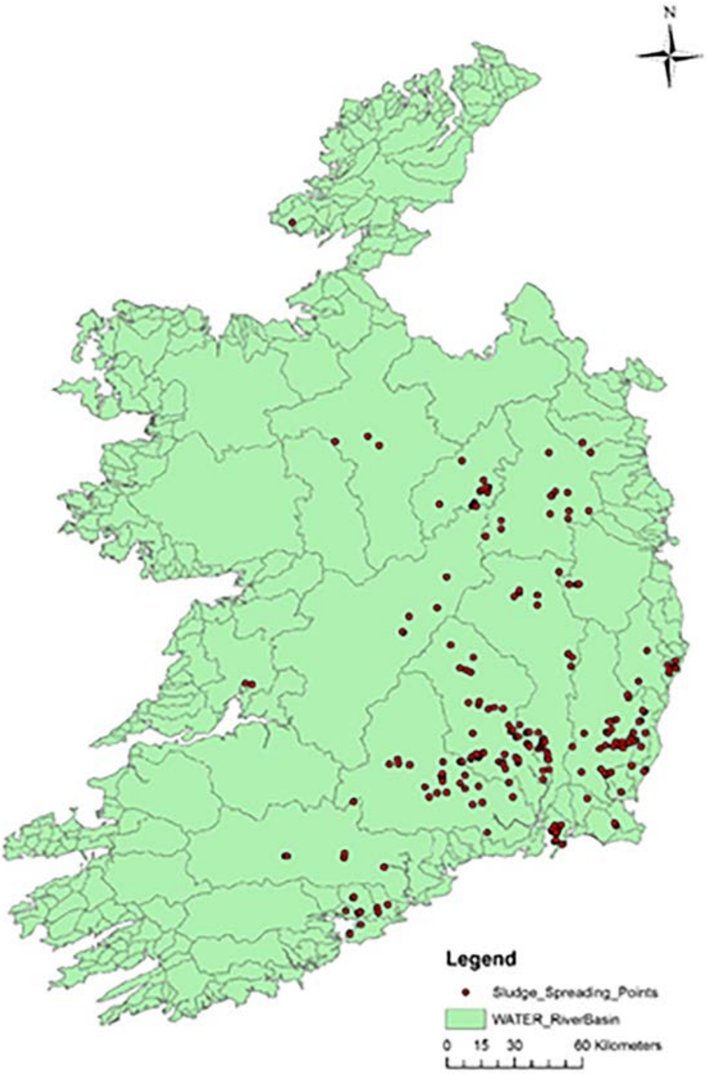

Figure 4.5. Illustrative subset of historical spreading locations.

4.4 Summary

A new map showing land suitable for landspreading WWTP slurry has been produced to take account of new modelling of overland flow pathways from the DiffuseTools project (Thomas *et al*., 2021). This can be used to indicate promising areas for landspreading WWTP sludge at small scales, such as on farms and/or in fields. At larger, subcatchment, scales, a general methodology for determining and aggregating the risk to water courses from the mobilisation of MPs from sludge spreading has been demonstrated using a subset of historical data.

Figure 4.9 shows that there is considerable variation in risk between different, even neighbouring, subcatchments, and within subcatchments (Figure 4.8), related to differences in both application amounts and hydrological mobilisation factors. As we do not have a complete dataset on current landspreading amounts and locations, Figure 4.9 should be interpreted not as the present situation, but rather as a hypothetical example of the type of analysis and outputs possible. All of this has been implemented as Python scripts using the ArcPy library associated with ArcMap GIS and can be readily implemented wherever this library is available. It would be a straightforward, albeit slightly more complex, task to rewrite the scripts for open-source GIS platforms, such as QGIS, that also support Python scripting.

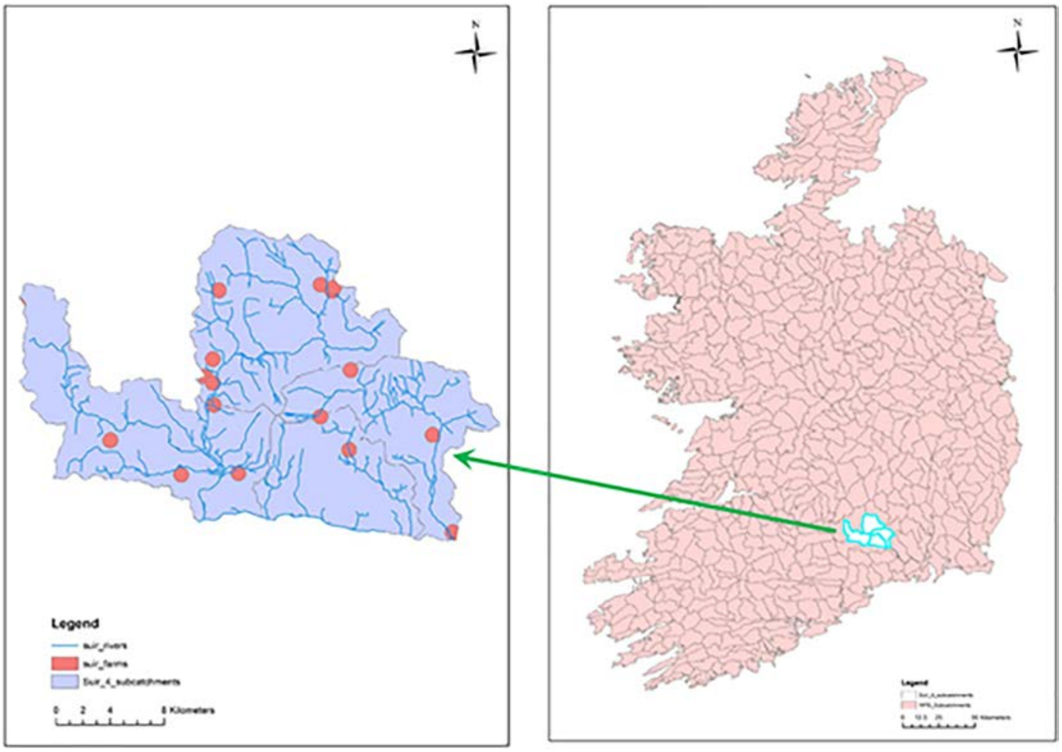

Figure 4.6. Example of aggregation from individual farms to EPA subcatchment scale.

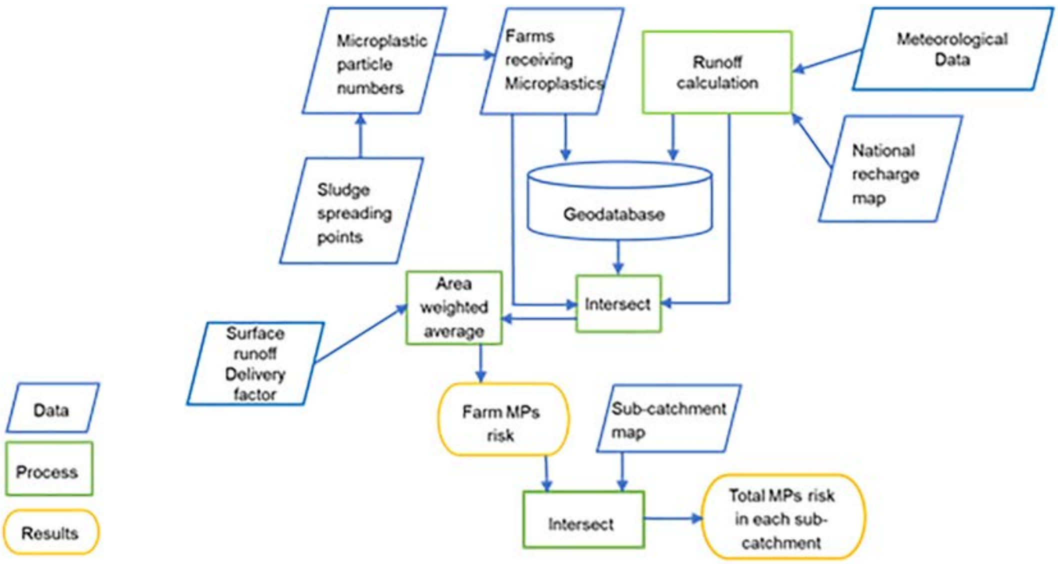

Figure 4.7. Methodology for generating indicative relative risk map for subcatchments.

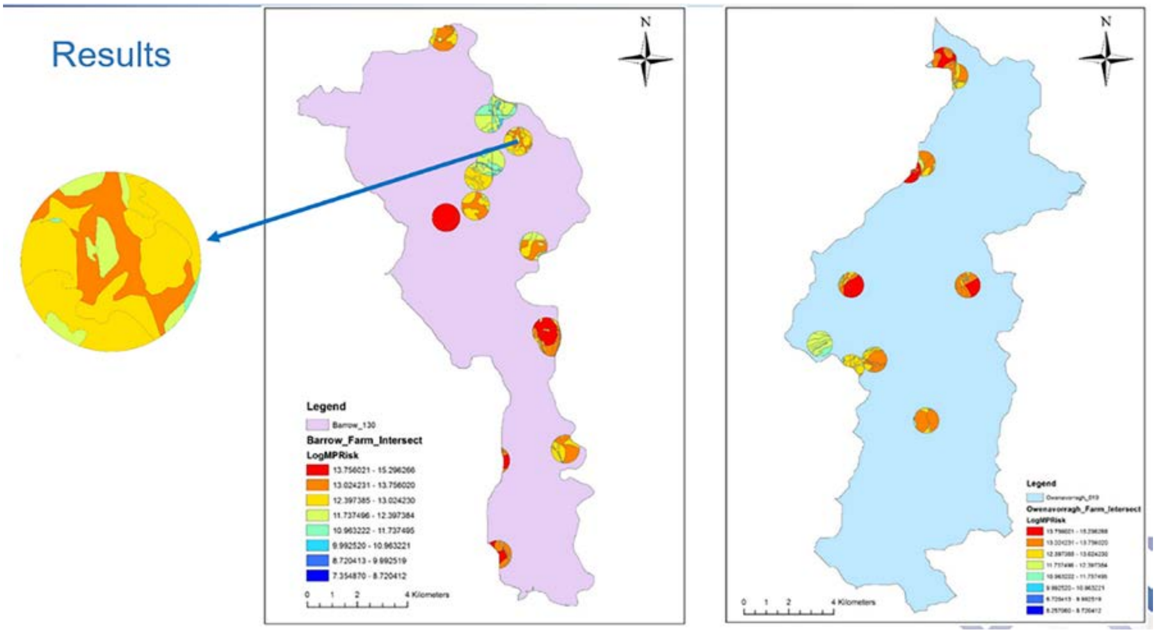

Figure 4.8. Demonstration of illustrative risk maps of a single subcatchment.

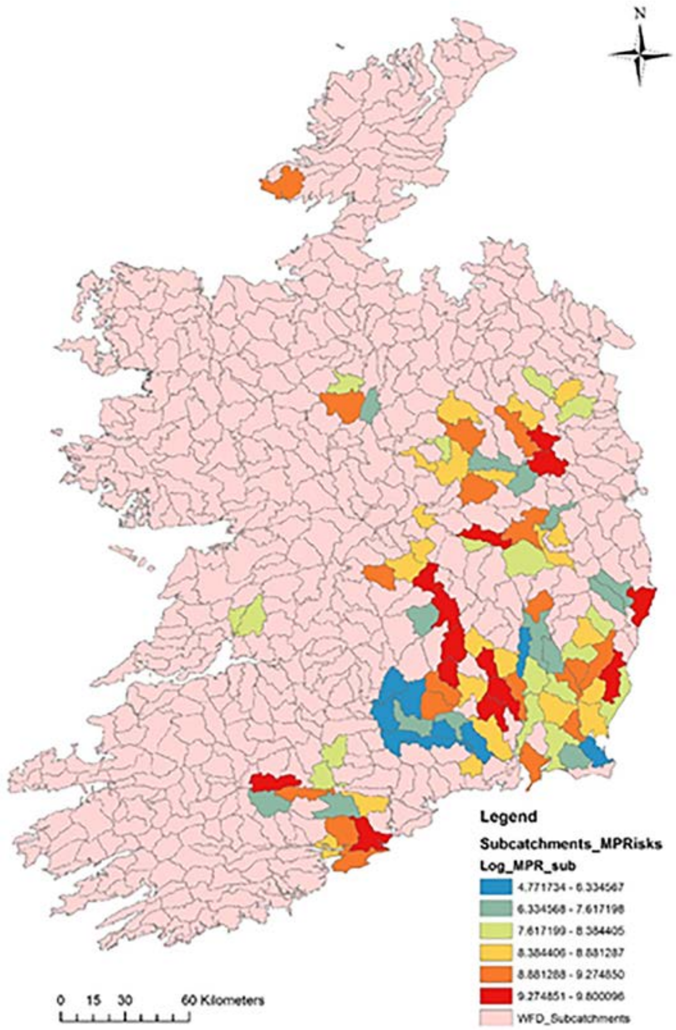

Figure 4.9. Demonstration of an illustrative risk map of subcatchments based on a subset of historic application areas (note: this map is not intended to be complete or to represent the current situation but is provided to illustrate the methodology and the type of modelling outputs possible).

5 Environmental Fate

5.1 Interactions with Species and Habitats in Ireland

5.1.1 Overview

The pervasiveness of MPs in the hydrosphere has raised much scientific and societal concern, largely because MPs are a similar size to many planktonic and benthic communities and may therefore interact with aquatic biota and transfer within aquatic food webs (Hidalgo-Ruz *et al*., 2012). Although findings from laboratory ecotoxicological studies are often conflicting, with many neutral outcomes (Foley *et al*., 2018), MPs have been observed to induce sublethal stresses in aquatic organisms (e.g. reduced growth, impaired reproduction) (reviewed in Haegerbaeumer *et al*., 2019) and may expose biota to a mixture of chemicals, including polymer additives such as phthalate plasticisers (e.g. Capolupo *et al*., 2020), as well as hydrophobic chemicals from the surrounding environment that they may absorb (e.g. Xia *et al*., 2019). However, the limited environmental relevance of many ecotoxicological studies to date means that it is difficult to accurately determine the ecological risk to aquatic biota from MPs (Provencher *et al*., 2020). Thus, a movement towards more realistic test conditions must be informed by field studies. This chapter aims to bridge the knowledge gap concerning biotic interactions with freshwater MPs. Specific emphasis is placed on the potential exposure pathways for freshwater biota, the potential transfer rates of MPs in freshwater food webs and the identification of potential bioindicators for monitoring MPs in freshwater systems.

5.1.2 Potential pathways and transfer rates for microplastics within a freshwater food web

MP exposure pathways for aquatic biota are assumed to be determined largely by the feeding strategy of the organism (Scherer *et al*., 2017), the characteristics of the particle type (e.g. size, density) (Wright *et al*., 2013) and the hydrological and environmental conditions governing MP input, transport, transformation and accumulation (Krause *et al*., 2020). The ubiquity and diversity of MP particles in aquatic systems mean that MPs may interact with a wide variety of biota (Rochman *et al*., 2019). In theory, planktivores, filter feeders and suspension feeders are likely to encounter buoyant, low-density polymers, while denser polymers are likely to become available to deposit feeders and detritivores as they sink to the benthos (Wright *et al*., 2013). However, once in the environment, MPs undergo a range of natural processes, such as biofouling (see, for example, Rummel *et al*., 2017), which may alter their behaviour, fate and subsequent bioavailability to biota over time.

In freshwaters, MPs have been reported in biota from both lentic (e.g. Driscoll *et al*., 2021) and lotic environments (e.g. Windsor *et al*., 2019). Among benthic macroinvertebrates (e.g. larval insects) it is expected that MPs are primarily obtained directly from the surrounding environment or food source, either via direct ingestion (Garcia *et al*., 2021) or externally via incorporation into external protective structures (i.e. larval cases) (e.g. Ehlers *et al*., 2021) and adherence to surface anatomy (e.g. adherence of fibres) (Kolandhasamy *et al*., 2018). In fish it is expected that MPs may be ingested directly (e.g. through drinking processes or unintentionally when foraging for food) (Roch *et al*., 2020) but also indirectly (i.e. secondary ingestion through prey). The presence of MPs in top aquatic predators (e.g. Hernandez-Milian *et al*., 2019), combined with evidence of dietary transfer of MPs in experimental (e.g. Farrell and Nelson, 2013) and semi-natural environments (Nelms *et al*., 2018), also suggests secondary ingestion to be the main pathway for species such as the Eurasian otter (*Lutra lutra*), which is a top predator in freshwater ecosystems. The ingestion of MP particles by predator biota via their prey may suggest a potential mechanism enabling bioaccumulation and biomagnification along aquatic food webs. That is the process by which a pollutant progressively increases in an organism because the rate of ingestion exceeds the rate of egestion, with higher concentrations attained at higher trophic levels (Nordberg *et al*., 2009). Although there is at present no indication that MPs biomagnify given the current level of assessment in biota, data are still limited.

To identify the exposure pathways for MPs in the River Slaney food web, biota representing three different trophic levels were analysed for MPs. Benthic macroinvertebrates and brown trout (*Salmo trutta*) were obtained from six sites in the River Slaney catchment (Figure 5.1), and Eurasian otter (*Lutra lutra*) spraints were opportunistically collected during routine sampling sessions. Dietary preferences were also verified to determine trophic linkages for brown trout and otter and to assess whether any relationship exists between prey type consumed and MP levels found. Finally, the potential of MPs to transfer and accumulate along the River Slaney food web was assessed through the implementation of a food web accumulation model.

5.2 Benthic Macroinvertebrates

5.2.1 Overview

To date, MPs have been detected among a number of freshwater biota, most of which were fish (reviewed in O'Connor *et al*., 2019). Macroinvertebrates play an important role in riverine ecosystems by providing regulatory functions and contributing to the diet of many ecologically significant species (e.g. salmonids) (O'Connor *et al*., 2020). In addition to individual effects (e.g. reduced growth) (Redondo-Hasselerharm *et al*., 2018), macroinvertebrate community response to long-term MP exposure may alter key ecosystem functions over time by modifying community

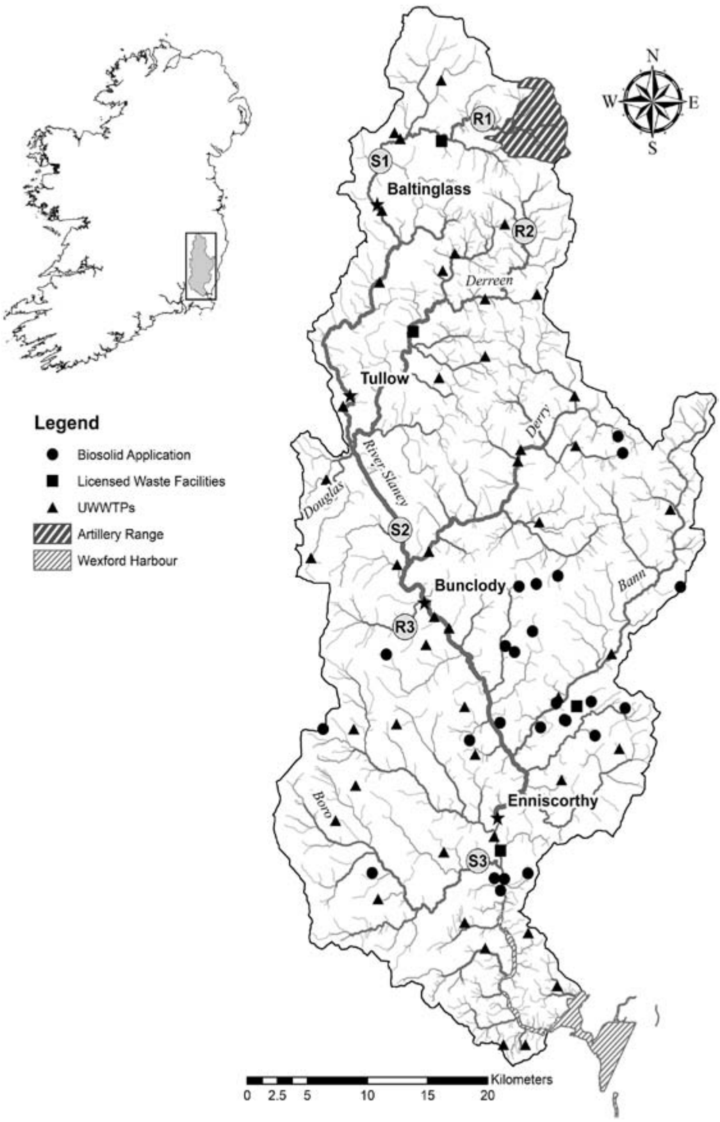

Figure 5.1. Map showing sampling locations (grey circles) on the River Slaney and its tributaries, along with potential sources of MP pollution in the catchment: UWWTPs (triangles), sites of biosolid application (circles) and licensed waste facilities (squares).

composition and reducing overall macroinvertebrate abundance (Redondo-Hasselerharm *et al*., 2020). This study looks at the spatial and temporal variability of MP concentrations in macroinvertebrate communities in the River Slaney catchment. It also assesses the variability in MP concentrations between benthic macroinvertebrate functional feeding groups (FFGs), taxa (i.e. class, subclass, order) and individual families, and examines whether the relative abundance and composition of benthic macroinvertebrates is related to community MP concentrations. To determine the distribution of MPs in the River Slaney catchment, along with the potential exposure levels for biota, surface water samples were assessed.

5.2.2 Methodology

The sampling of benthic macroinvertebrates and surface water samples took place in August 2017 and September 2018 at six sites in the River Slaney catchment (Figure 5.1). Three of the sites selected were upstream of potential MP sources (e.g. UWWTPs, licensed waste facilities, biosolid application sites) (R1, R2, R3) and classified as 'low' exposure, while three were downstream (S1, S2, S3) and classified as 'high' exposure.

Five volume-reduced water samples were collected at each of the six sampling sites. Samples were collected from the water surface using a 200 µm neuston net up to a maximum depth of 0.28 m. The net was deployed from the downstream side of road bridges and positioned in the main river channel with the aperture of the net perpendicular to the direction of flow. Upon collection of each sample, the net was raised from the river channel and the outside of the net was rinsed with a squeeze bottle containing site water.

The contents of the cod end were first transferred onto a 90-µm stainless-steel sieve, where detritus such as leaves and twigs was rinsed over the sieve and removed. The contents of the sieve were then transferred to acid-washed (0.05% nitric acid (HNO_3)) glass jars (344 mL) using pre-filtered 70% ethanol and sealed pending further processing and analysis. In the laboratory, contents from each sample were poured through a stainless-steel sieve stack (2 mm, 38 µm) and rinsed with ultrapure water (resistivity: 15.0 MΩ-cm). Larger material (> 2 mm) retained on the top sieve, such as plant debris, was carefully rinsed and discarded, and the sieve was inspected for potential MP particles under a stereo microscope (Hornet Micro Zoom 1280, Micros Austria). Contents retained on the 38-µm sieve were again rinsed with ultrapure water and, with a stainless-steel spatula, transferred to an acid-washed glass jar (344 mL) and sealed with a metal screw-cap lid. A 10% potassium hydroxide (KOH) (w/v) solution was then added to the retained material in a 3:1 ratio (solution volume : sample volume) and incubated at 60°C for 24 hours. After this time, the supernatant of each jar was transferred to a filtration apparatus using a glass pipette, filtered through glass fibre filter paper (1.2 µm particle retention) and sealed in a sterile Petri dish (ϕ 55 mm) for MP enumeration and characterisation. Lastly, a hypersaline solution (NaCl 1.2 g cm^{-3}) (3:1 ratio) was added to each jar, stirred for 1 minute using a stainless-steel spatula and allowed to settle overnight (minimum 12 hours) and the filtration process was repeated to improve MP recovery.

To collect macroinvertebrates, 10 standardised kick samples were collected at random from the riverbed of each site using a hand-held net (0.25-m frame and 250-µm mesh bag). The contents of each kick sample were transferred to a white tray, washed and transferred to an acid-washed (0.05% HNO_3) glass jar (750 mL) that was sealed with a metal screw-cap lid. Samples were preserved in pre-filtered 70% ethanol pending macroinvertebrate identification and MP isolation. Upon processing, each kick sample was emptied onto a pre-rinsed white tray, where macroinvertebrates were sorted into the lowest taxonomic classification commonly used to describe macroinvertebrate groups (i.e. class, subclass or order), hereafter referred to as taxa. Taxa were further identified to family level and assigned their main functional traits (i.e. FFG) according to Tachet *et al*. (2010) (e.g. scrapers, shredders, collector–gatherers).

All individuals of a specific macroinvertebrate family were enumerated and aggregated to form a single macroinvertebrate sample. These were then weighed on an analytical balance to the nearest 0.001 g, transferred to glass crimp vials (20 mL) and covered with aluminium foil. Although all kick samples collected in 2018 ($n = 60$) were included in the analysis, 14 kick samples collected in 2017 (R1, $n = 5$; R2, $n = 2$; S1, $n = 7$) were omitted, as they had been processed using a different isolation reagent. To isolate MPs, each macroinvertebrate sample was first placed in a laboratory oven at 60°C for a minimum of 24 hours or until such time as specimens comprising exoskeleton became brittle (typically less than 48 hours). These were then gently disaggregated within crimp vials using a stainless-steel dissection probe and digested using 5 mL of a 30% hydrogen peroxide (H_2O_2) solution heated to 60°C for 48 hours. Following digestion, samples were vacuum filtered onto glass fibre filter paper and sealed in Petri dishes.

MP characterisation was performed similarly for both sets of samples (i.e. benthic macroinvertebrates and surface water). First, particles suspected of being MPs were checked under a stereomicroscope (Olympus SZX10, ×1.6 magnification), their abundance recorded, and assigned a size category according to Frias and Nash (2019) (100–350 μm or 350 μm to ≤5 mm), while a subsample was analysed using μ-FTIR in each of the sampling years. A number of procedures were performed in the laboratory to account for and reduce background MP concentration, and these are described in detail in O'Connor *et al*. (2020). A *p*-value < 0.05 was considered significant.

5.2.3 Results

Analysis of surface water samples revealed that MPs were widely distributed in the River Slaney catchment, having been recovered in all 60 water samples at a mean concentration of 0.47 ± 0.08 MPs m^{-3} (mean ± SE) (range 0.03–4.02 MPs m^{-3}). A significant variation in the median MP concentration of surface water samples was found between sampling periods as well as between sampling sites, with samples collected in 2017 observed to have a significantly higher concentration than those collected in 2018, and site R3 found to have a significantly higher concentration than site R1 (Table 5.1). However, the latter was apparent only in 2017.

MPs were recovered from all but one kick sample assessed during 2017 (*n* = 46) and 2018 (*n* = 60), resulting in an overall mean concentration of 28.7 ± 3.0 MPs g^{-1} (wet mass) per kick sample (range 0–211 MPs g^{-1}). Of the 713 aggregated macroinvertebrate samples (i.e. macroinvertebrate families) derived from these 106 kick samples, 73% contained MPs, resulting in a mean concentration of 104.7 ± 9.5 MPs g^{-1} (range 0–2889.0 MPs g^{-1}) per sample. Fibres were the dominant MP type found (86%), followed by fragments and film (7% each). MPs recovered from macroinvertebrates ranged in size from 100 μm to 4.99 mm, and most fell within the 350 μm to ≤5 mm size category (71%). Of those particles confirmed as synthetic, the most frequently encountered were vinyl polymers (e.g. PVC) and copolymers (e.g. polyvinyl butyrate-co-vinyl acetate) (18%), polyesters (e.g. PET) (12%), polypropylene polymers and copolymers (8%) as well as polyacrylonitrile copolymers (4%).

Table 5.1. Mean (±SE) and median MP concentrations for surface water samples and benthic macroinvertebrates in each sampling site across combined years, and for 2017 and 2018 on their own

				Year			
		2017 and 2018 combined[a]		2017		2018	
Sample	Site	Mean	Median	Mean	Median	Mean	Median
Water (MPs m^{-3})	R1	0.23 ± 0.05	0.15	0.28 ± 0.08	0.22	0.18 ± 0.07	0.11
	R2	0.34 ± 0.04	0.34	0.35 ± 0.07	0.34	0.33 ± 0.06	0.35
	R3	1.40 ± 0.39	1.15	2.39 ± 0.44	1.94	0.41 ± 0.10	0.37
	S1	0.23 ± 0.02	0.25	0.22 ± 0.04	0.17	0.25 ± 0.01	0.25
	S2	0.33 ± 0.07	0.28	0.46 ± 0.10	0.46	0.19 ± 0.05	0.18
	S3	0.30 ± 0.06	0.31	0.38 ± 0.09	0.36	0.23 ± 0.05	0.22
Invertebrates (MPs g^{-1})	R1	46.4 ± 14.0	27.5	85.2 ± 33.4	45.0	27.0 ± 9.0	22.1
	R2	34.8 ± 5.0	33.6	26.1 ± 4.5	26.3	40.9 ± 7.4	40.3
	R3	51.0 ± 5.0	53.0	48.3 ± 7.2	43.3	53.8 ± 7.1	59.6
	S1	8.9 ± 2.1	7.1	6.5 ± 0.8	7.1	9.6 ± 2.7	6.4
	S2	15.7 ± 4.7	8.7	27.2 ± 8.0	18.0	4.3 ± 1.2	3.0
	S3	13.2 ± 2.9	7.4	19.0 ± 4.8	13.9	7.9 ± 2.5	5.9

Spatiotemporal variations in kick sample concentrations were also observed; samples collected in 2017 (n = 45, median = 27.1 MPs g^{-1}) had a significantly higher median concentration than those collected in 2018 (n = 60, median = 13.6 MPs g^{-1}). Furthermore, MPs were recovered in significantly higher concentrations at low-exposure sites (i.e. R1, R2, R3), and this was apparent for combined years, as well as for the years 2017 and 2018 when these were analysed independently. Overall, the low-exposure site R3 was found to have significantly higher median concentration than all high-exposure sites (i.e. S1, S2, S3), while R2 was also higher than S1. During 2017, median MP concentrations in both R1 (n = 5) and R3 (n = 10) samples were significantly higher than in site S1 (n = 3) samples located on the main river channel, while in 2018 the low-exposure site R2 had a significantly higher median concentration than site S2, also located on the main channel. Again, R3 was significantly different from S1, S2 and S3 in 2018.

Among FFGs, predators (n = 82, median = 37.0 MPs g^{-1}) were found to have a significantly higher median MP concentration than filter feeders (n = 38, median = 2.3 MPs g^{-1}), but this was apparent for only one of the years analysed (2018). Overall, Plecoptera (i.e. stonefly larvae) (n = 96, median = 47.2 MPs g^{-1}) was found to contain a significantly higher median MP concentration than Gastropoda (i.e. freshwater snails) (n = 31, median = 6.0 MPs g^{-1}) and Amphipoda (i.e. *Gammarus* spp.) (n = 47, median = 7.9 MPs g^{-1}), taxa that also had significantly lower concentrations than Ephemeroptera (i.e. mayfly larvae) (n = 108, median = 37.0 MPs g^{-1}). In addition, Ephemeroptera contained a significantly higher concentration than Trichoptera (i.e. caddisfly larvae) (n = 154, median 12.8 MPs g^{-1}), while Diptera (n = 122, median = 22.1 MPs g^{-1}) contained a significantly higher concentration than Gastropoda. The ephemeroptan family Heptageniidae and plecopteran families Nemouridae and Chloroperlidae were found to have some of the highest MP concentrations overall, while Nemouridae and Chloroperlidae also had some of the highest concentrations in 2017 and 2018, respectively. Furthermore, in 2018 the dipteran family Pediciidae had a significantly higher median concentration than its dipteran relative Tipulidae, as well as the gastropod Physidae. The relative abundance of some of these families was also observed to significantly correlate with MP concentrations, most notably in 2018 (see O'Connor, 2021).

5.2.4 Discussion

The presence of MPs in all macroinvertebrate families assessed indicates a wide level of interaction among benthic communities in the River Slaney catchment. While temporal differences in MP concentrations are difficult to explain in the present study, they could be attributed to the slight discrepancy between sampling periods (August 2017; September 2018), which may have influenced macroinvertebrate community composition, in addition to changes in hydrological conditions between sampling periods. The dominant concentrations in macroinvertebrate communities sampled upstream of potential sources could also be attributed in part to river hydrology. Additionally, as noted by Windsor *et al*. (2019), a higher ratio of river flow to UWWTP effluent in sites located along the main river channel, where the flow is often greater (e.g. S1, S2), might have obscured input from these sources and also may have affected MP bioavailability. On the other hand, higher concentrations observed in low-exposure sites may be influenced not only by point and diffuse sources (possibly of domestic and agriculture origin) but also by site characteristics overlooked during site selection. MPs were recovered among all macroinvertebrate groups assessed, comprising a number of families reported to contain MPs in previous studies (e.g. Windsor *et al*., 2019). While results from the present investigation indicate that certain groups (e.g. predators, plecopterans, ephemeropterans) may contain higher MP concentrations than others, the fact that this was not consistent throughout the study period makes it difficult to conclude that they are necessarily at greater risk of exposure. This is also true for macroinvertebrates that share a particular feeding mechanism, given the significant differences observed between families of the same FFG (e.g. shredders).

5.3 Brown Trout (*Salmo trutta*)

5.3.1 Overview

A detailed overview of this study is presented in O'Connor *et al*. (2020). Ultimately, the aims were to investigate the prevalence and burden of MPs in riverine brown trout populations sampled upstream (low exposure) and downstream (high exposure) of potential MP sources (e.g. UWWTPs), analyse possible relationships between MP burden/characteristics and biological traits of brown trout (e.g. fish length, maturity) and, finally, identify dietary contents (i.e. stomach contents), which may provide indication as to possible trophic links, at least at the time of sampling.

5.3.2 Methodology

Methods including sampling techniques and the isolation and characterisation of MPs (in this case 100 µm to 5 mm) as well the identification of dietary remains are described in detail in O'Connor *et al*. (2020), while the six sampling sites are shown in Figure 5.1.

5.3.3 Results (as described in O'Connor et al., 2020)

In total, 58 brown trout specimens were assessed for MPs, with individuals ranging in length (fork length) from 72 to 291 mm (mean: 149 mm ± 42 SD). After µ-FTIR analysis, a (final) total of 92 MP particles were recovered from 72% of fish (gastrointestinal tract (GIT) and stomach contents (SC) combined), resulting in a mean burden of 1.88 ± 1.53 (SD) MPs fish^{-1} within the GIT (66% prevalence) and 1.31 ± 0.48 MPs fish^{-1} within the SC (28% prevalence). Fibres were the dominant particle type found in both the GIT and SC (67% and 57%, respectively), while MPs in the 350 µm to ≤ 5 mm range were the main size class (73% and 71%, respectively). PS was the main polymer recovered from both components.

MPs were recovered in fish from all sites but were most prevalent in site R1 (low exposure), followed by the high-exposure sites, S2 and S1 (Figure 5.2). While no significant difference was observed between the median MP burdens in the GIT or SC of fish sampled in high- and low-exposure sites, the GIT burden of fish was found to significantly differ between the low-exposure sites of R1 and R2. Overall, MPs were most prevalent in the 0+/1+ age classification of fish (< 150 mm), where they were found primarily in the GIT (74%), and mean MP burdens were highest in the GITs of 2+ individuals (150 < 180 mm) (2.09 ± 2.28 MPs fish^{-1}). MP burdens in brown trout were independent of fish fork length and thus body size, while the proportion of MP size classes recovered from each component (i.e. GITs, SCs) was not associated with maturity status.

Finally, 38 dietary contents were identified in 54 fish (four stomachs were empty), including benthic macroinvertebrates, terrestrial invertebrates and winged adult insects, as well as plant material and sediment. Trichoptera (e.g. Limnephilidae,

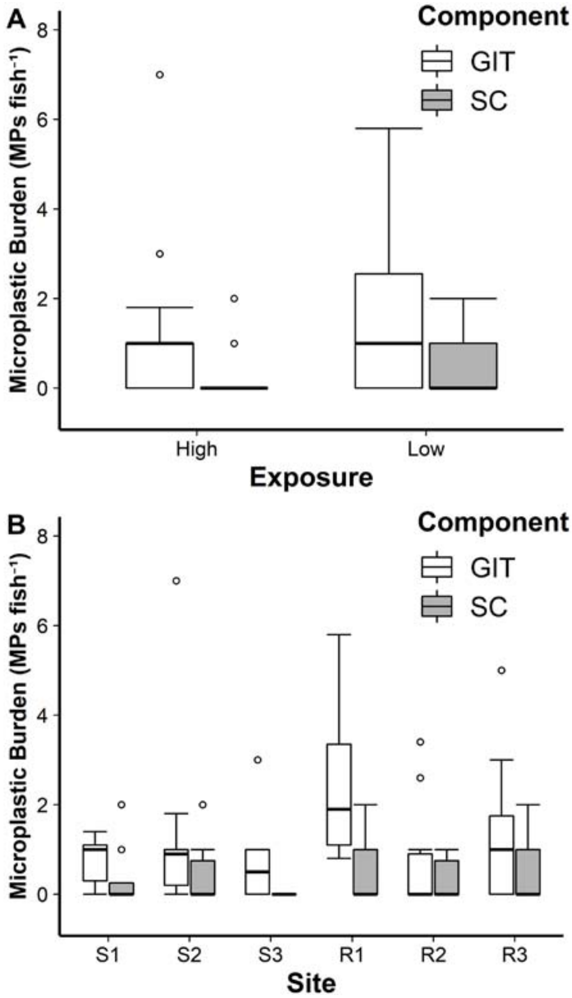

Figure 5.2. Microplastic burden of fish GIT and SC (MPs fish^{-1}) (*n* = 58) per exposure level (i.e. high and low) (a) and individual site (b). Boxplot midline shows the median, while lower and upper limits show the first quartile (Q1) and third quartile (Q3), respectively, with the box representing the IQR. The upper whisker represents Q3 + IQR × 1.5 and the lower whisker represents Q1 – IQR × 1.5, with open circles indicating the outliers. Reproduced from O'Connor *et al*. (2020); licensed under CC BY 4.0 (https://creativecommons.org/licenses/by/4.0/).

Hydropsyche spp.) (17%), gastropods such as the common bladder snail (*Physa fontinalis*) (12%) and dipterans (12%), particularly Chironomidae, were the most encountered benthic macroinvertebrates in brown trout diet, while adult insects (winged) and terrestrial invertebrates (e.g. Forficulidae) were important prey items, particularly in shaded sites (e.g. R1). No relationship was found between diet and MP burden in GITs or SCs.

5.3.4 Discussion

A full discussion concerning the significance of this study can be found in O'Connor *et al*. (2020). Briefly, this study assessed and confirmed MPs in brown trout, a fish species of considerable ecological and socioeconomic importance, serving as one of the first records of MPs in an Irish freshwater fish species and only the second record of MPs in any European salmonid. From a species perspective, the MP prevalence observed in the present study is quite comparable to those reported elsewhere in Europe (e.g. Karlsson *et al*., 2017); however, estimates from more recent European freshwater studies are much lower. It should be noted, however, that these studies used different isolation and detection procedures from the present one, and were also concerned with other fish species. Contrary to expectations, it was found that MP burden did not significantly differ between high- and low-exposure sites, and in fact the highest prevalence and burden was in an upstream, low-exposure site (R1) (Figures 5.1 and 5.2). Therefore, factors such as brown trout mobility and source of MP pollution may require greater consideration, particularly in future work. The present study found that the abundance of MPs recovered from brown trout is independent of fish fork length, indicating that larger fish do not necessarily ingest more MPs, while the lack of association between brown trout age group and MP size class ingested suggests proportionate susceptibility to all MP size classes assessed. With regard to diet, prey recovered from brown trout SCs were similar to those reported in previous dietary studies for the catchment (Ryan and Kelly-Quinn, 2015). However, the lack of association between diet and MP burden suggests that MP burdens in brown trout are not explained by the composition of prey items ingested, at least in this study.

5.4 Eurasian Otter (*Lutra lutra*)

5.4.1 Overview

A detailed description of this study is provided in O'Connor *et al*. (2022). In short, this study explored the opportunistic collection of otter spraints as a means of investigating the prevalence and abundance of MPs in otter faecal remains and assessing whether MPs are transferred to this top-level predator. Where possible, and to explore the viability of monitoring MPs in otter spraints and their applicability to different regions, samples were also obtained from a number of additional catchments in the south-west and west of the country. To determine the suitability of otter spraints for monitoring, spatial variations between regions, collection sites (e.g. higher and lower exposure), spraint condition and collection seasons were assessed. Moreover, otter diet was characterised so as to identify possible trophic links for the otter and also to assist in verifying the food web of the River Slaney. Relationships between MP levels in spraints and dietary composition were explored, and possible variations in diet were assessed for sampling region and season.

5.4.2 Methodology

Methods, including sampling techniques and the isolation and characterisation of MPs (in this case 100 µm to 5 mm), as well the identification of dietary remains are described in detail in O'Connor *et al*. (2022).

5.4.3 Results (as described in O'Connor et al.*, 2022)*

Overall, 53 spraints opportunistically collected from eight river catchments spanning three regions of Ireland were assessed for MPs, 35 of which derived from the River Slaney catchment. A total of 40 particles were identified as MPs, which included acrylic (PMMA), acrylic copolymers, nylon, PA, polyacrylonitrile (PAN), polycarbonate (PC), polyisoprene (PI), PP, polypropylene copolymers, vinyl copolymers and terpolymers (e.g. vinyl pyridine and methyl acrylate), as well as other copolymers (e.g. styrene methyl methacrylate) and synthetics. MPs were recovered in 57% of spraints at a mean abundance of 1.2 ± 0.1 MPs sprint^{-1} (mean ± SE) and a mean concentration of 3.8 ± 0.6 MPs g^{-1} dw. Fibres were the dominant MP type recovered (85%), followed by film (10%), with the range 350 μm to ≤ 5 mm the main MP size class (75%).

No spatial variation was observed either between sampling regions or depending on whether spraints were collected upstream (lower exposure) or downstream (higher exposure) of local MP sources. However, it is noted that spraints collected from the River Slaney catchment generally had the highest MP concentration overall (4.8 ± 0.8 MPs g^{-1}) (Figure 5.3). While there was no significant variation in median MP concentration across the three condition classes assigned to otter spraints (i.e. fresh, drying, dry), a significant seasonal variation was observed, with spraints collected in autumn (n = 10) found to contain a higher median MP concentration than those collected in spring (n = 28) and summer (n = 15). This was also true for the River Slaney catchment when assessed independently, where autumn spraints (n = 10) were found to have a significantly higher median concentration than spraints collected in spring (n = 14). A total of 20 dietary items were recovered from 53 spraint samples, the majority of which were fish (F% = 85%). Of the fish remains identified, salmonids were the most frequently encountered taxon (F% = 62%), having been found in 33 spraints (26 from the River Slaney catchment), followed by three-spined stickleback (*Gasterosteus aculeatus*) (F% = 25%), cyprinids (e.g. *Phoxinus phoxinus)* (F% = 19%) and European eel (*Anguilla anguilla*) (F% = 8%). Macroinvertebrates were also quite prevalent in otter diet and included a mixture of benthic macroinvertebrates, terrestrial invertebrates and winged adult insects as well as white-clawed crayfish

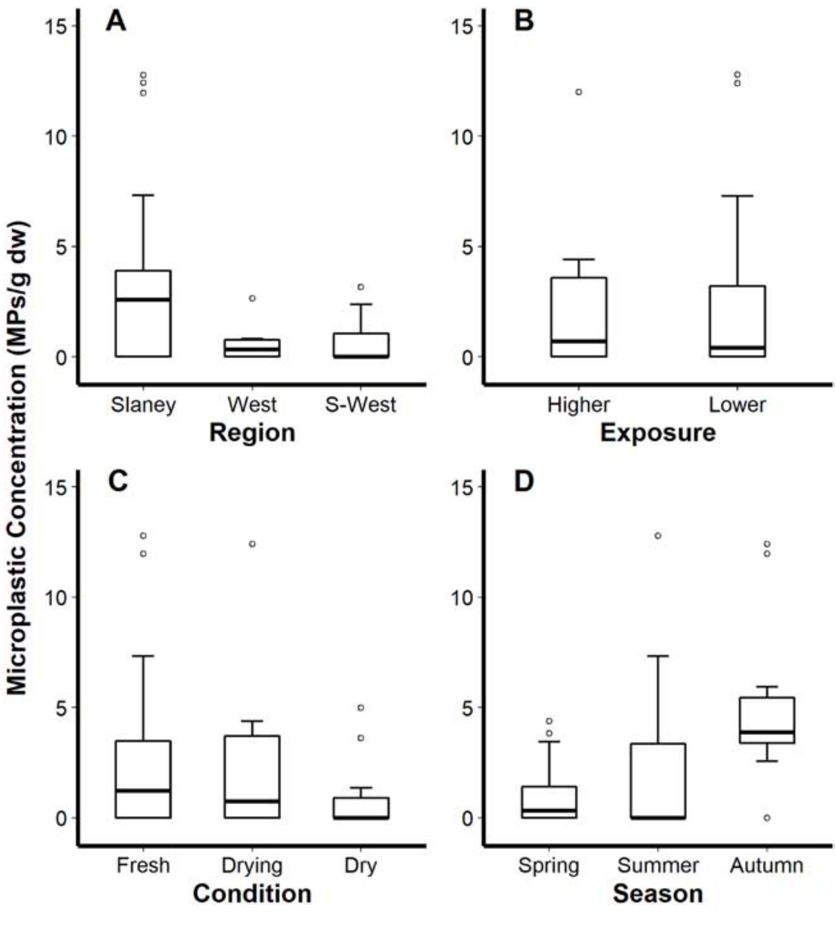

Figure 5.3. MP concentration (MPs g^{-1} dw) (n = 53) in spraint samples per region (A), exposure level (B), condition (C) and season (D). Boxplot midline shows the median, while lower and upper limits show the first quartile (Q1) and third quartile (Q3), respectively, with the box representing the IQR. The upper whisker represents Q3 + IQR × 1.5 and the lower whisker represents Q1 – IQR × 1.5, with open circles indicating the outliers. Reproduced from O’Connor *et al.* (2022); licensed under CC BY 4.0 (https://creativecommons.org/licenses/by/4.0/).

(*Austropotamobius pallipes*). No trend was observed between MP levels in spraints and otter diet, and no significant variations in diet were observed between sampling regions or sampling seasons, although it is acknowledged that the samples were small.

5.4.4 Discussion (excerpts taken from O'Connor et al., 2022)

This study confirms that the Eurasian otter ingests MPs in Ireland. Moreover, it presents evidence to support the assumption that the most likely pathway through which MPs pass between top predators in freshwater ecosystems is secondary ingestion (i.e. trophic transfer). Similarities were observed in the characteristics of MPs recovered in spraints from the River Slaney catchment and those previously found in brown trout (O'Connor *et al*., 2020), particularly in the dominance of fibres (94%) and larger particles (78%) (350 µm to ≤5 mm), although for brown trout a considerable proportion of fragments were recovered (GIT 25%, SC 24%).

Although there was no difference in dietary composition between sampling season or sampling region, or any trend in MP abundance or MP concentration, it was observed that the River Slaney catchment that had the highest MP concentration of all regions also had the highest occurrence of salmonids, although estimated fork lengths of 66–117 mm were below the mean fork length of brown trout previously assessed for MPs in this catchment (149 mm ± 42 mm; mean ± SD) (see O'Connor *et al*., 2020). The reasons for the observed differences in seasonal MP concentrations are unclear, but it may be that water courses experience a greater influx of MPs associated with overland flow during the wetter months of the year (e.g. in autumn) (Campanale *et al*., 2020). Overall, a more robust seasonal assessment of individual sites would provide more information, which, owing to its opportunistic nature, was not feasible in the present study.

No spatial variations were observed in MP concentrations, either between regions of contrasting risk (as defined by Mahon *et al*., 2017) or between higher- and lower-exposure areas, rendering it difficult to establish "reference" concentrations (i.e. in spraints in "lower"-exposure areas) for the purposes of monitoring MPs. While it is known that otters ingest MPs, it is not known how long the MPs are retained internally and what timescale they represent. Moreover, the extent of otter home ranges means that spraint contents are representative of the entire range and thus are not necessarily associated with specific marking sites. In addition, although the otter diet is dominated by fish, the presence of amphibian remains, as well as of other terrestrial and unidentified food items (possibly avian or mammalian), although low, suggests that MPs found in spraints may not be necessarily of freshwater origin. Therefore, while it is acknowledged that the criteria used to determine expected exposure levels in the present study may have been too simplistic, otter mobility and niche breadth mean that spraints are likely to provide a poor representation of site-specific MP levels.

5.5 Trophic Transfer and Accumulation of Microplastics

5.5.1 Overview

The capacity of MPs to transfer between trophic levels and along simple food chains has been demonstrated in laboratory (e.g. Setälä *et al*., 2014) and semi-natural environments (Nelms *et al*., 2018). Although MPs recovered from piscivorous fish species, birds and mammals in the field are thought to derive primarily from prey items (e.g. D'Souza *et al*., 2020), secondary ingestion could provide a potential mechanism for MPs to bioaccumulate in freshwater biota, possibly resulting in biomagnification. However, as it is challenging to replicate the inherent complexities of food webs in controlled environments, little is known about how MPs may transfer and accumulate through different predator–prey combinations, and so the potential for biomagnification is generally not yet well established (Krause *et al*., 2020). Bioaccumulation models can explore the transfer of MPs along food webs (Alava, 2020). This study aims to simulate the potential transfer, bioaccumulation and biomagnification of MPs in the River Slaney food web using steady-state solutions derived from a set of equations capable of accommodating MP uptake from multiple dietary components (Diepens and Koelmans, 2018).

5.5.2 Methodology

The model was adopted from the food web accumulation model (MICROWEB) developed and described by Diepens and Koelmans (2018), who studied the accumulation of plastics and associated contaminants (i.e. hydrophobic organic compounds (HOCs)) in the biota of an Arctic food web. The primary aim of the current analysis was to assess the potential food web transfer and biomagnification of MPs, whereby accumulation is restricted to the GIT of the organism and assumed to be a balance of uptake and loss processes (Herzke *et al*., 2016). Hence, accumulation excludes the tissue of the organism, as well as adherence to surface anatomy (e.g. adherence to integument). All modelling was performed in Microsoft Excel 2016 and verified by the manual calculation of all steps in the transfer of MPs along a hypothetical food web.

Information obtained from analyses of the diet of brown trout and Eurasian otter (O'Connor *et al*., 2020, 2022), along with information from regulatory monitoring surveys (e.g. Kelly *et al*., 2014), was used to define the River Slaney food web (Figure 5.4). Three trophic levels previously assessed for MPs in this catchment (i.e. benthic macroinvertebrates, brown trout and Eurasian otter) were included in the model, and values from the literature were used to inform ingestion rates and gut retention times (GRTs). MP uptake was simulated through the diet of benthic macroinvertebrates at the base of the food web, with no parallel uptake of MPs at higher trophic levels (i.e. direct ingestion). Environmental MP concentrations in the River Slaney obtained as part of this research project (i.e. from surface water (section 5.2.3) and sediment (section 3.5.3)) were used to inform the model, and three scenarios were simulated by modifying these fractions. Empirical concentrations observed from the biota analysed were used to assess model performance (see O'Connor (2021) for further information).

As the concentration of MPs is modelled as the weight of MP ingested per unit mass of animal (e.g. $g\,MP\,g^{-1}$ biota), MP abundances from different datasets (e.g. sediment, macroinvertebrates) were converted to mass using information on particle type, size and density. MP concentration in biota was the main output parameter of the model, which was then used to calculate the bioaccumulation metric, or biota

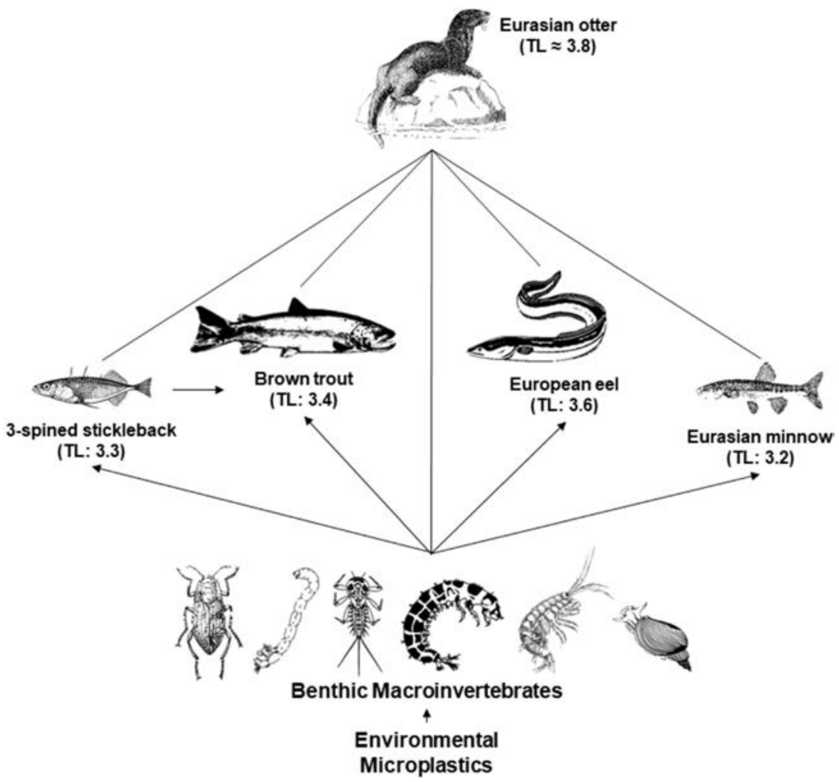

Figure 5.4. Biota in the River Slaney food web along with specific predator–prey interactions as specified by dietary analysis and in the literature. MP uptake is simulated at the base of the food web through the feeding of benthic macroinvertebrates (TL, trophic level; all images were sourced from Creative Commons archives: www.shutterstock.com; www.vectorstock.com; www.istockphoto.com).

magnification factor (BMF), for each predator–prey combination. BMF values > 1 indicate biomagnification, while values < 1 indicate trophic dilution (Diepens and Koelmans, 2018).

5.5.3 Results

Predicted concentrations in macroinvertebrates were deemed to be low based on initial uptake from environmental media (combined mean fraction in water and sediment), and large discrepancies were observed between predicted and measured concentrations. Results were most similar when the maximum fraction of MPs was applied in sediment (default scenario), resulting in a range of 2.58×10^{-7} – 7.30×10^{-6} g MPs g^{-1} biota. While predicted concentrations were still generally lower than empirical observations, many estimates were within measured ranges (i.e. minimum and maximum MP concentrations), exceptions being Ephemeroptera and Coleoptera. To match all taxa with empirical ranges, it was necessary to increase MP fractions in sediment by a factor of 2.6 (scenario 2), which corresponds to a mass fraction of approximately 1.2×10^{-5} g MPs g^{-1} sediment (dry weight). Steady-state MP concentrations in fish ranged from 2.15×10^{-8} for European eel to 6.54×10^{-8} g MPs g^{-1} biota for brown trout based on the default modelling scenario (i.e. maximum environmental fraction), and predicted concentrations were also within empirical ranges for brown trout. When the maximum MP fraction in sediment was increased by a factor of 37.8 (scenario 3), the simulated and maximum concentrations in brown trout were reduced to a difference of 0.2%. MP concentrations among fish species followed a similar pattern when environmental fractions were adjusted. BMF values of 0.05 for brown trout, 0.03 for Eurasian minnow and 0.02 for European eel and three-spined stickleback indicate that the abundance of MPs accumulating in fish is unlikely to be sufficient to lead to biomagnification. After European eel and three-spined stickleback, the lowest MP concentration was predicted to occur in the Eurasian otter, and this was consistent following all adjustments of environmental MP levels. Although it possesses the highest BMF value in the food web (BMF: 0.13), realistic biological ingestion and egestion data suggest that there is no bioaccumulation of MPs in this top predator. Therefore, biomagnification of MPs in the River Slaney food web is unlikely, based on the predator–prey interactions, the parameters specified and the size range of the MPs assessed (> 100 µm).

5.5.4 Discussion

The simulation of MP transfer within the River Slaney food web agrees with recent work suggesting that the biomagnification of MPs is not currently predicted for the main size ranges reported in aquatic biota (i.e. > 100 µm) (Diepens and Koelmans, 2018; Gouin, 2020; Covernton *et al*., 2021). This is largely due to the transitory throughput of plastics in predators (D'Souza *et al*., 2020), which would mitigate accumulation and render it insufficient to facilitate potential biomagnification. However, the MP concentrations reported here are merely a reflection of the balance between ingestion, gut retention and egestion (Diepens and Koelmans, 2018); they do not consider accumulation in tissues or organs, nor do they account for direct ingestion among fish, which is considered a significant pathway (Roch *et al*., 2020). Furthermore, as data used to inform the current model did not consider size ranges < 100 µm, the same conclusions regarding biomagnification may not necessarily apply to smaller MPs or particles in the submicron range. The discrepancies between predicted and empirical concentrations in invertebrate taxa, using realistic environmental MP fractions, suggest that the environmental MP data used here may not represent the true extent of biota exposure to MPs. However, it is noted that sediments from the River Slaney (section 3.5) were not collected from the same sampling sites as biota, and nor were they collected from the main river channel or during the same sampling period. It is also possible that many smaller particles were overlooked in water and sediment samples because of the isolation methods employed, and so MPs that are bioavailable to biota might not have been effectively recovered. Additional explanations for the discrepancies observed in macroinvertebrates may be related to the MP exposure pathways for these biota. Garcia *et al*. (2021) reported that fish species with a higher proportion of allochthonous carbon also contained larger numbers of MPs, which may imply that MPs are ingested either through terrestrial invertebrates or through aquatic invertebrates consuming allochthonous detritus (e.g. shredders). Further information on the possible MP inputs associated with these resources (e.g. leaf litter), as well as the extent to which MPs interact with plants in freshwater systems, will improve the accuracy of the model. Finally, as the GRT of MPs is considered especially important for the accumulation and translocation as well as dietary transfer of MPs, it is imperative that further research is conducted to inform accumulation models such as this, which are a valuable tool for exploring the transfer of MPs in complex food webs.

5.6 Bioindicators Suitable for Monitoring Microplastics in Freshwater Systems

5.6.1 Overview

A bioindicator can be defined as an organism, part of an organism or a community of organisms that contains information about the quality of the environment (Markert *et al.*, 1999). Bioaccumulation indicators are characterised as organisms that accumulate and concentrate pollutants from their surrounding environment or food, so that analysis of their tissues provides an estimate of the bioavailable concentrations of pollutants in their environment (Gerhardt, 2002). As a result, bioindicators may be used for the purposes of biomonitoring, providing a long-term set of observations of a particular water quality parameter and allowing a pollutant to be measured over a given period of time (CleanSea Project, 2016). However, there are requirements that need to be fulfilled before an organism can be considered for the purposes of biomonitoring (e.g. widespread distribution, moderate tolerance to exposure) (CleanSea Project, 2016; Gerhardt, 2002). In addition, the selection of bioindicators depends on the specific aims of the monitoring programme proposed, the scale of that programme (e.g. site-specific, regional level), whether it intends to assess short- or long-term exposure and how species traits fit those criteria (e.g. distribution, range).

Biomonitoring concepts for aquatic contaminants may also apply to plastic pollution, and a number of marine biota have been assessed as potential indicators of plastics in the environment, including invertebrates (e.g. Li *et al.*, 2019), fish (e.g. Garcia-Garin *et al.*, 2019), birds (e.g. Acampora *et al.*, 2016) and large marine mammals (e.g. Fossi *et al.*, 2020). However, only a handful of studies have assessed biota for the purposes of monitoring MPs in freshwater ecosystems, with benthic macroinvertebrates the main group proposed for this (e.g. Nel *et al.*, 2018). This chapter also evaluates the suitability of biota suitable for monitoring MPs in freshwater systems by assessing MP concentrations in their body or associated residues and appraising their ecological traits.

5.6.2 Benthic macroinvertebrates

Because of their limited migration patterns, benthic macroinvertebrates can offer a site-specific assessment of MP levels. Moreover, given that they spend all, or most, of their life cycle in water and often live for more than 1 year, macroinvertebrates are well suited to medium-term observations of pollution (Perera *et al.*, 2012). A possible shortcoming of the use of benthic macroinvertebrates as bioindicators for MPs is that selective ingestion and egestion may bias the particle size distributions recovered, thereby providing a poor representation of MPs in the environment (Ward *et al.*, 2019). However, as observed in the present study (see section 5.2), MPs within the entire size range assessed (100 µm to 5 mm) were recovered in benthic macroinvertebrate communities from the River Slaney, with a higher frequency of larger particles (350 µm to 5 mm). Although the higher concentrations observed in certain families (i.e. plecopterans) may suggest that these would be suitable for monitoring MPs in the present study, sensitivities to other forms of pollution (e.g. organic pollution) may impede their use as bioindicators of MPs, and so further exploration should be performed at community level. Relative ease of sampling and pre-existing platforms, such as that employed for the ecological assessment of water quality in Ireland as part of the EU Water Framework Directive, could provide a good framework for simultaneous assessments of MP burdens in macroinvertebrate tissues.

*5.6.3 Brown trout (*Salmo trutta*)*

Although the migratory patterns of resident brown trout in the River Slaney are not fully understood, a general habitat preference among adult brown trout for deeper, slow-flowing habitats (Höjesjö *et al.*, 2007) could render them useful for assessing the abundances and characteristics of MPs in depositing habitats where MPs are likely to accumulate over time. As fish may ingest MPs both directly and indirectly (i.e. secondary ingestion through prey), the burdens observed in brown trout are likely to be reflective of MPs in both abiotic compartments. However, brown trout are unlikely to retain detectable particles in their GIT, and so conventional assessments for MPs are likely to provide only a snapshot of recent conditions (i.e. short term). An analysis of other tissues where MPs may accumulate could provide a better indication of overall exposure levels.

5.6.4 Eurasian otter (Lutra lutra)

Although this study has demonstrated that the collection of otter spraints is a non-invasive method of assessing MP ingestion among the species, large home ranges and a wide niche breadth (Ó Néill *et al*., 2009) mean that, while otter spraints may be applicable for monitoring MPs at a regional scale, they are not likely to be good indicators of local or catchment conditions. As biomonitoring in the context of this study was concerned with indicating local MP levels, the otter was not considered further.

5.6.5 Conclusions

For the purposes of monitoring short- to medium-term MP exposure, it is believed that benthic macroinvertebrates, because of their site specificity and level of interaction with sedimentary MPs, are the biota most suited for biomonitoring, based on the biota analysed in this study. However, to evaluate the discriminatory power of macroinvertebrates, and their ability to represent the MPs that occur in their habitats, sediment samples and possibly food resources (e.g. periphyton) should be assessed in tandem. Other notable biota not assessed in this study but reported to interact with MPs in freshwater systems include bird species, such as the white-throated dipper (D’Souza *et al*., 2020), which are known to exhibit a reasonably high site fidelity and are heavily reliant on benthic macroinvertebrates as a food source.

6 Recommendations

The EU plastics strategy is a key element of Europe's transition towards a carbon-neutral and circular economy. The implementation of a continuous monitoring framework for microplastics (MPs), as a pollutant of emerging concern, in riverine catchments is necessary for monitoring progress in achieving this goal and needs to be incorporated at a minimum level of river basin management plans in Ireland. This monitoring framework could be applied through a defined set of measures including calibrated sampling techniques, facilitating comparisons of the abundance, occurrence and characteristics of MPs in freshwater systems. This would allow major transport routes and hotspots of MP pollution to be identified, quantifying MP fluxes and discharges from riverine catchments to marine environments and, ultimately, providing a clearer understanding of the risks posed by plastic pollution and the complex interlinkages between the atmosphere, hydrosphere and biosphere. This research project has identified key challenges and recommendations that target MP pollution, highlighting immediate measures that could manage MP debris at known sources, reducing the amount of marine litter reaching the sea and contributing to that objective as part of OSPAR's Regional Action Plan for Marine Litter (OSPAR Commission, 2022).

6.1 Potential Interventions at Source

Through the three case studies of sources, insight was gained into both research and potential interventions that could reduce the abundance of MPs being released in the environment.

6.1.1 Wastewater treatment plants

- Collect further empirical data on MP removal rates by treatment level (e.g. primary, secondary and tertiary) from WWTPs serving different population equivalents, including grid sizes used to retain particles, and MPs in storm overflows and in biosolids.
- Introduce policies aimed at reducing microfibres entering WWTPs; for example, filters could be incorporated into washing machines.

6.1.2 Construction sites

- An onsite waste audit of MPs, including an assessment of the use of building information models to reduce the need for cutting and how this may reduce MP waste production.

6.1.3 AstroTurf pitches

- All pitches should be built with an internal drainage system including sand, which reduces the transport of microparticles into the drainage systems.
- Installation of a steel boot-cleaning grid to clean crumb from boots, reducing crumb transfer from pitch.
- Retainer walls should be built along all sides to reduce the loss of crumb.

6.2 Recommendations for Future Research

The results of this research project highlight several data and knowledge gaps that, when addressed, will build on current knowledge and enhance future monitoring frameworks. These topics range from MP sources to pathways within river catchments.

- Identify all sources of MP pollution in river catchments, e.g. atmospheric deposition, recreation, hydrological events, hydrogeomorphology, septic tanks and agricultural runoff.
- Collect and incorporate into models data relating to influencing factors such as atmospheric fallout, wind and rain (particularly during extreme events).
- Quantify the export of MP from Irish catchments to freshwater systems to assess the risk this poses.
- Conduct an in-depth study of urban storm water runoff as a pathway for anthropogenic particles to urban receiving waters.
- This study has investigated the concentration of MP in a particular agricultural setting; further studies are necessary to extend this to other settings and land uses. Furthermore, there exists a paucity of longitudinal MP studies in

the scientific literature; to properly investigate the capacity for MP to accumulate in these environments, longer-term studies should be funded.
- Further sample agricultural drainage water to investigate if interflow is a significant pathway, particularly following heavy rainfall events.
- Carry out further research to test the vertical transport of weathered MPs in porous media.
- Carry out research on the accumulation, translocation and abundance of MP in lower size ranges (< 100 μm) in tissues outside the gastrointestinal tract in fish and mammals.
- Conduct further research into how biosolids can be treated to reduce MP content.
- Conduct research on assessing the feasibility of using cutting zones to capture more MPs and developing suitable capture devices for cutting equipment.
- Carry out further studies on the different types of cutting tools and their associated MP production.
- Conduct further research to investigate the vertical movement of rubber crumb to storm drains.

6.3 Future Monitoring

River systems are highly heterogeneous, with many variables affecting the transport of MPs throughout riverine catchments. Ultimately, the best monitoring strategy for MPs will largely depend on the aims of the monitoring programme and the periods for which information is sought. Furthermore, it will depend on the timescale the various abiotic and biotic compartments represent and what constitutes short-, medium- and long-term exposure in the context of MPs. Existing methods adopted for the collection, isolation and quantification of MPs from riverine systems, including sediments, are often time-consuming, expensive and laborious. Thus, biota may serve as a proxy for determining MP variations, given their relative ease of processing. Furthermore, more extensive water sampling will assist in quantifying MP inputs derived from terrestrial-based diffuse sources (i.e. surface runoff), thus providing information on short-term exposure.

6.3.1 Catchment level

- Introduce fine-scale, long-term monitoring of riverine waters, sediment and biota.
- Align sampling and isolation protocols for biotic and abiotic components as closely as possible to reduce methodological biases (e.g. MP characteristics) and provide a better understanding of the bioavailable fractions of MPs in the environment.

6.3.2 Abiotic matrices

- Undertake sediment sampling, which gives the best indication of overall pollution levels in river catchments and provides information on medium- to long-term exposure to MPs.
- Monitor sediments, in addition to biota, to determine site-specific exposure levels and hence exposure pathways for benthic biota within these sites.
- Carry out bulk water sampling, which is recommended for improving representativeness and increasing the detection of larger particles.

6.3.3 Biotic matrices

- Use an ecosystem-based approach to monitoring, with the employment of multiple environmental matrices, including water, sediment and biota, to develop our understanding of factors affecting the presence and distribution of MPs in rivers.

References

Acampora, H., Lyashevska, O., Van Franeker, J.A. and O'Connor, I., 2016. The use of beached bird surveys for marine plastic litter monitoring in Ireland. *Marine Environmental Research* 120: 122–129.

Alava, J.J., 2020. Modeling the bioaccumulation and biomagnification potential of microplastics in a cetacean foodweb of the Northeastern Pacific: a prospective tool to assess the risk exposure to plastic particles. *Frontiers in Marine Science* 7: 1–23.

Allen, S., Allen, D., Phoenix, V.R., Le Roux, G., Durántez Jiménez, P., Simonneau, A., Binet, S. and Galop, D., 2019. Atmospheric transport and deposition of microplastics in a remote mountain catchment. *Nature Geoscience* 12: 339–344.

Andrady, A.L., 2011. Microplastics in the marine environment. *Marine Pollution Bulletin* 62: 1596–1605.

Astrom, L., 2016. Shedding of synthetic microfibers from textiles. University of Gothenburg, Sweden.

Ballent, A., Purser, A., de Jesus Mendes, P., Pando, S. and Thomsen, L., 2012. Physical transport properties of marine microplastic pollution. *Biogeosciences Discussions* 9: 18,755–18,798.

Bigalke, M., Fieber, M., Foetisch, A., Reynes, J. and Tollan, P., 2022. Microplastics in agricultural drainage water: a link between terrestrial and aquatic microplastic pollution. *Science of the Total Environment* 806: 150709

Bläsing, M. and Amelung, W., 2018. Plastics in soil: analytical methods and possible sources. *Science of the Total Environment* 612: 422–435.

Campanale, C., Stock, F., Massarelli, C., Kochleus, C., Bagnuolo, G., Reifferscheid, G. and Uricchio, V.F., 2020. Microplastics and their possible sources: the example of Ofanto river in southeast Italy. *Environmental Pollution* 258: 113284.

Capolupo, M., Sørensen, L., Jayasena, K.D.R., Booth, A.M. and Fabbri, E., 2020. Chemical composition and ecotoxicity of plastic and car tire rubber leachates to aquatic organisms. *Water Research* 169: 115270.

Carbon Tracker Initiative, 2020. The future's not in plastics – why plastics demand won't rescue the oil sector. Available from https://carbontracker.org/reports/the-futures-not-in-plastics (accessed 20 November 2021).

Carr, S.A., Liu, J. and Tesoro, A.G., 2016. Transport and fate of microplastic particles in wastewater treatment plants. *Water Research* 91: 174–182.

CleanSea Project, 2016. *Testing Indicators for Biological Impacts of Microplastics*. Institute for Environmental Studies, VU University, Amsterdam, Netherlands.

Connolly, J. and Holden, N.M., 2009. Mapping peat soils in Ireland: updating the derived Irish peat map. *Irish Geography* 42: 343–352.

Corradini, F., Meza, P., Eguiluz, R., Casado, F., Huerta-Lwanga, E. and Geissen, V., 2019. Evidence of microplastic accumulation in agricultural soils from sewage sludge disposal. *Science of the Total Environment* 671: 411–420.

Covernton, G.A., Davies, H.L., Cox, K.D., El-Sabaawi, R., Juanes, F., Dudas, S.E. and Dower, J.F., 2021. A Bayesian analysis of the factors determining microplastics ingestion in fishes. *Journal of Hazardous Materials* 413: 125405.

Crew, A., Gregory-Eaves, I. and Ricciardi, A., 2020. Distribution, abundance, and diversity of microplastics in the upper St. Lawrence River. *Environmental Pollution* 260: 113994.

Crossman, J., Hurley, R.R., Futter, M. and Nizzetto, L., 2020. Transfer and transport of microplastics from biosolids to agricultural soils and the wider environment. *Science of the Total Environment* 724.

da Costa, J.P., Mouneyrac, C., Costa, M., Duarte, A.C. and Rocha-Santos, T., 2020. The role of legislation, regulatory initiatives and guidelines on the control of plastic pollution. *Frontiers in Environmental Science* 8: 104.

Diepens, N.I.J. and Koelmans, A.A., 2018. Accumulation of plastic debris and associated contaminants in aquatic food webs. *Environmental Science & Technology* 52: 8510–8520.

Driscoll, S.C., Glassic, H.C., Guy, C.S. and Koel, T.M., 2021. Presence of microplastics in the food web of the largest high-elevation lake in North America. *Water* 13: 264.

D'Souza, J.M., Windsor, F.M., Santillo, D. and Ormerod, S.J., 2020. Food web transfer of plastics to an apex riverine predator. *Global Change Biology* 26: 3846–3857.

Eerkes-Medrano, D., Thompson, R.C. and Aldridge, D.C., 2015. Microplastics in freshwater systems: a review of the emerging threats, identification of knowledge gaps and prioritisation of research needs. *Water Research* 75: 63–82.

Ehlers, S.M., Ellrich, J.A. and Gestoso, I., 2021. Plasticrusts derive from maritime ropes scouring across raspy rocks. *Marine Pollution Bulletin* 172: 112841.

Essel, R., Arens, R.H., Engel, L. and Carus, M., 2015. *Sources of Microplastics Relevant to Marine Protection in Germany*. Report to the Federal Environment Agency, Heurth, Germany.

European Council, 1986. Council Directive 86/278/EEC of 12 June 1986 on the protection of the environment, and in particular of the soil, when sewage sludge is used in agriculture. OJ L 181, 4.7.1986, pp. 6–12. European Union, Brussels, Belgium.

Eurostat, 2021. Sludge production and disposal. Available online: http://appsso.eurostat.ec.europa.eu/nui/show.do?lang=en&dataset=env_ww_spd (accessed 17 November 2021).

Farrell, P. and Nelson, K., 2013. Trophic level transfer of microplastic: *Mytilus edulis* (L.) to *Carcinus maenas* (L.). *Environmental Pollution* 177: 1–3.

Fendall, L.S. and Sewell, M.A., 2009. Contributing to marine pollution by washing your face: microplastics in facial cleansers. *Marine Pollution Bulletin* 58: 1225–1228.

Foley, C.J., Feiner, Z.S., Malinich, T.D. and Höök, T.O., 2018. A meta-analysis of the effects of exposure to microplastics on fish and aquatic invertebrates. *Science of the Total Environment* 631: 550–559.

Fossi, M.C., Baini, M. and Simmonds, M.P., 2020. Cetaceans as ocean health indicators of marine litter impact at global scale. *Frontiers in Environmental Science* 8: 255.

Frias, J. and Nash, R., 2019. Microplastics: finding a consensus on the definition. *Marine Pollution Bulletin* 138: 145–147.

Frias, J., Pagter, E., Nash, R., O'Connor, I., Carretero, O., Filgueiras, A., Viñas, L., Gago, J., Antunes, J., Bessa, F., Sobral, P., *et al*., 2018. *Standardised Protocol for Monitoring Microplastics in Sediments*. JPI-Oceans BASEMAN project.

Garcia, F., de Carvalho, A.R., Riem-Galliano, L., Tudesque, L., Albignac, M., Ter Halle, A. and Cucherousset, J., 2021. Stable isotope insights into microplastic contamination within freshwater food webs. *Environmental Science & Technology* 55: 1024–1035.

Garcia-Garin, O., Vighi, M., Aguilar, A., Tsangaris, C., Digka, N., Kaberi, H. and Borrell, A., 2019. *Boops boops* as a bioindicator of microplastic pollution along the Spanish Catalan coast. *Marine Pollution Bulletin* 149: 110648.

Gerhardt, A., 2002. Bioindicator species and their use in biomonitoring. In Inyang, H.I. and Daniels, J.L. (eds), *Environmental Monitoring*. EOLSS, Oxford, UK, pp. 77–123.

Gouin, T., 2020. Toward an improved understanding of the ingestion and trophic transfer of microplastic particles: critical review and implications for future research. *Environmental Toxicology and Chemistry* 39: 1119–1137.

Haegerbaeumer, A., Mueller, M.-T., Fueser, H. and Traunspurger, W., 2019. Impacts of micro- and nano-sized plastic particles on benthic invertebrates: a literature review and gap analysis. *Frontiers in Environmental Science* 7: 17.

Han, M., Niu, X., Tang, M., Zhang, B.T., Wang, G., Yue, W., Kong, X. and Zhu, J., 2020. Distribution of microplastics in surface water of the lower Yellow River near estuary. *Science of the Total Environment* 707: 135601.

Hernandez-Milian, G., Lusher, A., MacGabban, S. and Rogan, E., 2019. Microplastics in grey seal (*Halichoerus grypus*) intestines: are they associated with parasite aggregations? *Marine Pollution Bulletin* 146: 349–354.

Herzke, D., Anker-Nilssen, T., Nøst, T.H., Götsch, A., Christensen-Dalsgaard, S., Langset, M., Fangel, K. and Koelmans, A.A., 2016. Negligible impact of ingested microplastics on tissue concentrations of persistent organic pollutants in northern fulmars off coastal Norway. *Environmental Science & Technology* 50: 1924–1933.

Hidalgo-Ruz, V., Gutow, L., Thompson, R.C. and Thiel, M., 2012. Microplastics in the marine environment: a review of the methods used for identification and quantification. *Environmental Science & Technology* 46: 3060–3075.

Hitchcock, J.N., 2020. Storm events as key moments of microplastic contamination in aquatic ecosystems. *Science of the Total Environment* 734: 139436.

Hoellein, T., Rojas, M., Pink, A., Gasior, J. and Kelly, J., 2014. Anthropogenic litter in urban freshwater ecosystems: distribution and microbial interactions. *PLOS ONE* 9: e98485.

Höjesjö, J., Økland, F., Sundström, L., Pettersson, J. and Johnsson, J., 2007. Movement and home range in relation to dominance; a telemetry study on brown trout *Salmo trutta*. *Journal of Fish Biology* 70: 257–268.

Horton, A.A. and Dixon, S.J., 2018. Microplastics: an introduction to environmental transport processes. *Wiley Interdisciplinary Reviews: Water* 5: e1268.

Horton, A.A., Walton, A., Spurgeon, D.J., Lahive, E. and Svendsen, C., 2017. Microplastics in freshwater and terrestrial environments: evaluating the current understanding to identify the knowledge gaps and future research priorities. *Science of the Total Environment* 586: 127–141.

Imhof, H.K., Ivleva, N.P., Schmid, J., Niessner, R. and Laforsch, C., 2013. Contamination of beach sediments of a subalpine lake with microplastic particles. *Current Biology* 23: R867–R868.

IRFU, 2008. *Artificial Grass Pitches for Rugby. Performance Standards and Design Guides for Community Use Pitches and Training Areas*. Irish Rugby Football Union. Available online: https://tinyurl.com/2wret2s7.

Jourgholami, M., Karami, S. and Tavankar, F., 2021. Effects of slope gradient on runoff and sediment yield on machine-induced compacted soil in temperate forests. *Forests* 12: 1–19.

Karlsson, T.M., Vethaak, A.D., Almroth, B.C., Ariese, F., van Velzen, M., Hassellöv, M. and Leslie, H.A., 2017. Screening for microplastics in sediment, water, marine invertebrates and fish: method development and microplastic accumulation. *Marine Pollution Bulletin* 122: 403–408.

Kelly, F., Connor, L., Matson, R., Feeney, R., Morrissey, E., Coyne, J. and Rocks, K., 2014. *Water Framework Directive Fish Stock Survey of Rivers in the South Eastern River Basin District, 2013*. Inland Fisheries Ireland, Dublin, Ireland.

Kibet, L.C., Saporito, L.S., Allen, A.L., May, E.B., Kleinman, P.J., Hashem, F.M. and Bryant, R.B., 2014. A protocol for conducting rainfall simulation to study soil runoff. *Journal of Visualized Experiments* 51664.

Klein, S., Worch, E. and Knepper, T.P., 2015. Occurrence and spatial distribution of microplastics in river shore sediments of the Rhine-Main area in Germany. *Environmental Science & Technology* 49: 6070–6076.

Kolandhasamy, P., Su, L., Li, J., Qu, X., Jabeen, K. and Shi, H., 2018. Adherence of microplastics to soft tissue of mussels: a novel way to uptake microplastics beyond ingestion. *Science of the Total Environment* 610: 635–640.

Krause, S., Baranov, V., Nel, H.A., Drummond, J., Kukkola, A., Hoellein, T., Smith, G.S., Lewandowski, J., Bonnet, B. and Packman, A.I., 2020. Gathering at the top? Environmental controls of microplastic uptake and biomagnification in freshwater food webs. *Environmental Pollution* 268: 115750.

Laermanns, H., Lehmann, M., Klee, M., J. Löder, M.G., Gekle, S. and Bogner, C., 2021. Tracing the horizontal transport of microplastics on rough surfaces. *Microplastics and Nanoplastics* 1.

Lechner, A. and Ramler, D., 2015. The discharge of certain amounts of industrial microplastic from a production plant into the River Danube is permitted by the Austrian legislation. *Environmental Pollution* 200: 159–160.

Lee, H. and Kim, Y., 2018. Treatment characteristics of microplastics at biological sewage treatment facilities in Korea. *Marine Pollution Bulletin* 137: 1–8.

Li, J., Lusher, A.L., Rotchell, J.M., Deudero, S., Turra, A., Bråte, I.L.N., Sun, C., Hossain, M.S., Li, Q. and Kolandhasamy, P., 2019. Using mussel as a global bioindicator of coastal microplastic pollution. *Environmental Pollution* 244: 522–533.

Li, X., Chen, L., Mei, Q., Dong, B., Dai, X., Ding, G. and Zeng, E.Y., 2018. Microplastics in sewage sludge from the wastewater treatment plants in China. *Water Research* 142: 75–85.

Liu, P., Zhan, X., Wu, X., Li, J., Wang, H. and Gao, S., 2020. Effect of weathering on environmental behavior of microplastics: properties, sorption and potential risks. *Chemosphere* 242: 125193.

Lucid, J.D., Fenton, O., Grant, J. and Healy, M.G., 2014. Effect of rainfall time interval on runoff losses of biosolids and meat and bone meal when applied to a grassland soil. *Water, Air, and Soil Pollution* 225.

Magnusson, K. and Norén, F., 2014. *Screening of Microplastic Particles in and Down-stream a Wastewater Treatment Plant*. IVL Swedish Environmental Research Institute.

Mahon, A.M., Officer, R., Nash, R. and O’Connor, I., 2017. *Scope, Fate, Risks and Impacts of Microplastic Pollution in Irish Freshwater Systems*. Environmental Protection Agency, Johnstown Castle, Ireland.

Mahon, A.M., O’Connor, I., Bruen, M., O’Sullivan, J., Heerey, L., Murphy, S., Lally, H.T., O’Connor, I., Koelmans, A.A. and Nash, R. 2021. *Pathways of Diffuse Sources of Microplastics*. Environmental Protection Agency, Johnstown Castle, Ireland.

Månsson, C., 2010. Gräsligt eller gräslikt? En jämförelse mellan naturgräs och konstgräs på svenska idrottsarenor (Hideous or grassy? A comparison between natural grass and artificial grass in Swedish sports stadiums). MSc Thesis. Swedish University of Agricultural Sciences, Uppsala, Sweden.

Markert, B., Wappelhorst, O., Weckert, V., Herpin, U., Siewers, U., Friese, K. and Breulmann, G., 1999. The use of bioindicators for monitoring the heavy-metal status of the environment. *Journal of Radioanalytical and Nuclear Chemistry* 240: 425–429.

McLaren, N., Fleming, P. and Forrester, S., 2012. Artificial grass: a conceptual model for degradation in performance. *Procedia Engineering* 34: 831–836.

Mintenig, S.M., Löder, M.G.J., Primpke, S. and Gerdts, G., 2019. Low numbers of microplastics detected in drinking water from ground water sources. *Science of The Total Environment* 648: 631–635.

Mockler, E., Bruen, M., Desta, M. and Misstear, B., 2013. *Pathways Project Final Report Volume 4: Catchment Modelling Tool*. Environmental Protection Agency, Johnstown Castle, Ireland.

Mohanty, S.K., Bulicek, M.C.D., Metge, D.W., Harvey, R.W., Ryan, J.N. and Boehm, A.B., 2015. Mobilization of microspheres from a fractured soil during intermittent infiltration events. *Vadose Zone Journal* 14.

Morét-Ferguson, S., Law, K.L., Proskurowski, G., Murphy, E.K., Peacock, E.E. and Reddy, C.M., 2010. The size, mass, and composition of plastic debris in the western North Atlantic Ocean. *Marine Pollution Bulletin* 60: 1873–1878.

Nel, H.A., Dalu, T. and Wasserman, R.J., 2018. Sinks and sources: assessing microplastic abundance in river sediment and deposit feeders in an Austral temperate urban river system. *Science of the Total Environment* 612: 950–956.

Nelms, S.E., Galloway, T.S., Godley, B.J., Jarvis, D.S. and Lindeque, P.K., 2018. Investigating microplastic trophic transfer in marine top predators. *Environmental Pollution* 238: 999–1007.

Ngo, P.L., Pramanik, B.K., Shah, K. and Roychand, R., 2019. Pathway, classification and removal efficiency of microplastics in wastewater treatment plants. *Environmental Pollution* 255: 113326.

Nielsen, T.D., Hasselbalch, J., Holmberg, K. and Stripple, J., 2020. Politics and the plastic crisis: a review throughout the plastic life cycle. *Wiley Interdisciplinary Reviews: Energy and Environment* 9: e360.

Nolan, P., 2015. *Ensemble of Regional Climate Model Projections for Ireland*. Environmental Protection Agency, Johnstown Castle, Ireland.

Nordberg, M., Templeton, D.M., Andersen, O. and Duffus, J.H., 2009. Glossary of terms used in ecotoxicology (IUPAC recommendations 2009). *Pure and Applied Chemistry* 81: 829–970.

O'Connor, D., Pan, S., Shen, Z., Song, Y., Jin, Y., Wu, W.M. and Hou, D., 2019. Microplastics undergo accelerated vertical migration in sand soil due to small size and wet–dry cycles. *Environmental Pollution* 249: 527–534.

O'Connor, J.D., 2021. Potential pathways, trophic transfer and bioindicators of microplastics in freshwater systems. PhD thesis. Galway-Mayo Institute of Technology, Galway, Ireland.

O'Connor, J.D., Mahon, A.M., Ramsperger, A.F., Trotter, B., Redondo-Hasselerharm, P.E., Koelmans, A.A., Lally, H.T. and Murphy, S., 2019. Microplastics in freshwater biota: a critical review of isolation, characterization, and assessment methods. *Global Challenges* 4: 1800118.

O'Connor, J.D., Murphy, S., Lally, H.T., *et al*., 2020. Microplastics in brown trout (*Salmo trutta* Linnaeus, 1758) from an Irish riverine system. *Environmental Pollution* 267: 115572.

O'Connor, J.D., Lally, H.T., Mahon, A.M., *et al*., 2022. Microplastics in Eurasian otter (*Lutra lutra*) spraints and their potential as a biomonitoring tool in freshwater systems. *Ecosphere* 13: e3955.

Ó Néill, L., Veldhuizen, T., de Jongh, A. and Rochford, J., 2009. Ranging behaviour and socio-biology of Eurasian otters (*Lutra lutra*) on lowland mesotrophic river systems. *European Journal of Wildlife Research* 55: 363–370.

OSPAR Commission, 2022. *The Second OSPAR Regional Action Plan on Marine Litter*. Available online: https://www.ospar.org/work-areas/eiha/marine-litter/regional-action-plan/rap2 (accessed 22 December 2022).

Ouyang, W., Zhang, Y., Wang, L., Barceló, D., Wang, Y. and Lin, C., 2020. Seasonal relevance of agricultural diffuse pollutant with microplastic in the bay. *Journal of Hazardous Materials* 396: 122602.

Pan, C. and Shangguan, Z., 2006. Runoff hydraulic characteristics and sediment generation in sloped grassplots under simulated rainfall conditions. *Journal of Hydrology* 331: 178–185.

Panno, S.V., Kelly, W.R., Scott, J., Zheng, W., McNeish, R.E., Holm, N., Hoellein, T.J. and Baranski, E.L., 2019. Microplastic contamination in karst groundwater systems. *Groundwater* 57: 189–196.

Paul, C., Fealy, R., Fenton, O., Lanigan, G., O'Sullivan, L. and Schulte, R.P.O., 2018. Assessing the role of artificially drained agricultural land for climate change mitigation in Ireland. *Environmental Science & Policy* 80: 95–104.

Perera, L., Wattavidanage, J. and Nilakarawasam, N., 2012. Development of a macroinvertebrate–based index of biotic integrity (MIBI) for Colombo-Sri Jayewardenepura canal system. *Journal of Tropical Forestry and Environment* 2: 10–19.

Piehl, S., Leibner, A., Löder, M. G. J., Dris, R., Bogner, C. and Laforsch, C., 2018. Identification and quantification of macro- and microplastics on an agricultural farmland. *Scientific Reports* 8: 17950.

Plastics Europe, 2020. *Plastics – The Facts 2020: An Analysis of European Plastics Production, Demand and Waste Data*. Available at: https://www.plasticseurope.org/en/resources/publications/1804-plastics-facts-2019.

Provencher, J.F., Covernton, G.A., Moore, R.C., Horn, D.A., Conkle, J.L. and Lusher, A.L., 2020. Proceed with caution: the need to raise the publication bar for microplastics research. *Science of the Total Environment* 748: 141426.

Redondo-Hasselerharm, P., Gort, G., Peeters, E. and Koelmans, A., 2020. Nano- and microplastics affect the composition of freshwater benthic communities in the long term. *Science Advances* 6: eaay4054.

Redondo-Hasselerharm, P.E., Falahudin, D., Peeters, E.T. and Koelmans, A.A., 2018. Microplastic effect thresholds for freshwater benthic macroinvertebrates. *Environmental Science & Technology* 52: 2278–2286.

Rehm, R., Zeyer, T., Schmidt, A. and Fiener, P., 2021. Soil erosion as transport pathway of microplastic from agriculture soils to aquatic ecosystems. *Science of the Total Environment*, 795: 148774.

Reid, N., Thompson, D., Hayden, B., Marnell, F. and Montgomery, W.I., 2013. Review and quantitative meta-analysis of diet suggests the Eurasian otter (*Lutra lutra*) is likely to be a poor bioindicator. *Ecological Indicators* 26: 5–13.

Rillig, M.C., Ingraffia, R. and de Souza Machado, A.A., 2017. Microplastic incorporation into soil in agroecosystems. *Frontiers in Plant Science* 8: 1805.

Roch, S., Friedrich, C. and Brinker, A., 2020. Uptake routes of microplastics in fishes: practical and theoretical approaches to test existing theories. *Scientific Reports* 10: 3896.

Rochman, C.M., Brookson, C., Bikker, J., Djuric, N., Earn, A., Bucci, K., Athey, S., Huntington, A., McIlwraith, H. and Munno, K., 2019. Rethinking microplastics as a diverse contaminant suite. *Environmental Toxicology and Chemistry* 38: 703–711.

Rummel, C.D., Jahnke, A., Gorokhova, E., Kühnel, D. and Schmitt-Jansen, M., 2017. Impacts of biofilm formation on the fate and potential effects of microplastic in the aquatic environment. *Environmental Science & Technology Letters* 4: 258–267.

Ryan, D. and Kelly-Quinn, M., 2015. Effects of riparian canopy cover on salmonid diet and prey selectivity in low nutrient streams. *Journal of Fish Biology* 86: 16–31.

Scherer, C., Brennholt, N., Reifferscheid, G. and Wagner, M., 2017. Feeding type and development drive the ingestion of microplastics by freshwater invertebrates. *Scientific Reports* 7: 17006.

Setälä, O., Fleming-Lehtinen, V. and Lehtiniemi, M., 2014. Ingestion and transfer of microplastics in the planktonic food web. *Environmental Pollution* 185: 77–83.

SIS, 2014. *Irish Soil Information System*. Available from: http://gis.teagasc.ie/soils/map.php (accessed 14 November 2021).

Steinmetz, Z., Wollmann, C., Schaefer, M., Buchmann, C., David, J., Tröger, J., Muñoz, K., Frör, O. and Schaumann, G.E., 2016. Plastic mulching in agriculture. Trading short-term agronomic benefits for long-term soil degradation? *Science of the Total Environment* 550: 690–705.

Tachet, H., Bournaud, M., Richoux, P. and Usseglio-Polatera, P., 2010. *Invertébrés d'eau douce – systématique, biologie, écologie*. CNRS Editions, Paris, France. Available from: https://www.freshwaterecology.info (accessed 28 November 2017).

Thomas, I., Bruen, M., Mockler, E., *et al*., 2021. *Catchment Models and Management Tools for Diffuse Contaminants (Sediment, Phosphorus and Pesticides): DiffuseTools Project*. Environmental Protection Agency, Johnstown Castle, Ireland.

van den Berg, P., Huerta-Lwanga, E., Corradini, F. and Geissen, V., 2020. Sewage sludge application as a vehicle for microplastics in eastern Spanish agricultural soils. *Environmental Pollution* 261: 114198.

Vollertsen, J. and Hansen, A.A. (eds), 2017. *Microplastic in Danish Wastewater: Sources, Occurrences, and Fate*. Danish Environmental Protection Agency, Odense, Denmark.

Wagner, M., Scherer, C., Alvarez-Muñoz, D., *et al*., 2014. Microplastics in freshwater ecosystems: what we know and what we need to know. *Environmental Sciences Europe* 26: 1–9.

Waldman, W.R. and Rillig, M.C., 2020. Microplastic research should embrace the complexity of secondary particles. *Environmental Science and Technology*, pp. 7751–7753.

Wang, X., 2018. Using geographical information systems (GIS) to determine critical source areas for diffuse source microplastic contamination of rivers. MScEng Dissertation, University College Dublin, Ireland.

Wanner, P., 2021. Plastic in agricultural soils – a global risk for groundwater systems and drinking water supplies? – a review. *Chemosphere* 264: 128453.

Ward, J.E., Zhao, S., Holohan, B.A., Mladinich, K.M., Griffin, T.W., Wozniak, J. and Shumway, S.E., 2019. Selective ingestion and egestion of plastic particles by the blue mussel (*Mytilus edulis*) and eastern oyster (*Crassostrea virginica*): implications for using bivalves as bioindicators of microplastic pollution. *Environmental Science & Technology* 53: 8776–8784.

Wefer-Roehl, A. and Kübeck, C., 2014. *Guidelining Protocol for Soil-column Experiments Assessing Fate and Transport of Trace Organics*. Available online: https://demeau-fp7.eu/sites/files/D123a%20Guidelines%20Column%20experiments.pdf (accessed 1 November 2022).

Werbowski, L.M., Gilbreath, A.N., Munno, K., Zhu, X., Grbic, J., Wu, T., Sutton, R., Sedlak, M.D., Deshpande, A.D. and Rochman, C.M., 2021. Urban stormwater runoff: a major pathway for anthropogenic particles, black rubbery fragments, and other types of microplastics to urban receiving waters. *ACS ES&T Water* 1: 1420–1428.

Windsor, F.M., Tilley, R.M., Tyler, C.R. and Ormerod, S.J., 2019. Microplastic ingestion by riverine macroinvertebrates. *Science of the Total Environment* 646: 68–74.

Wredh, G., 2014. Miljö-och hälsorisker med konstgräsplaner (Environmental and health risks with artificial turf fields). MSc Thesis. Stockholm University, Sweden.

Wright, S.L., Thompson, R.C. and Galloway, T.S., 2013. The physical impacts of microplastics on marine organisms: a review. *Environmental Pollution* 178: 483–492.

Xanthos, D. and Walker, T.R., 2017. International policies to reduce plastic marine pollution from single-use plastics (plastic bags and microbeads): a review. *Marine Pollution Bulletin* 118: 17–26.

Xia, B., Zhang, J., Zhao, X., Feng, J., Teng, Y., Chen, B., Sun, X., Zhu, L., Sun, X. and Qu, K., 2019. Polystyrene microplastics increase uptake, elimination and cytotoxicity of decabromodiphenyl ether (BDE-209) in the marine scallop *Chlamys farreri*. *Environmental Pollution* 258: 113657.

Zubris, K.A.V. and Richards, B.K., 2005. Synthetic fibers as an indicator of land application of sludge. *Environmental Pollution* 138: 201–211.

Abbreviations

ANOVA	Analysis of variance
BMF	Biota magnification factor
dw	Dry weight
FTIR	Fourier transform infrared spectroscopy
HDPE	High-density polyethylene
LDPE	Low-density polyethylene
MP	Microplastic
PA	Polyamide (nylon)
PAA	Polyacrylic
PAN	Polyacrylonitrile
PE	Polyethylene
PEE	Polyester epoxide
PEMA	Poly(ethyl methacrylate)
PES	Polyester
PET	Polyethylene terephthalate
PEU	Polyester urethane
PIR	Polyisocyanurate
PMMA	Poly(methyl methacrylate) (acrylic)
PP	Polypropylene
PS	Polystyrene
PU	Polyurethane
PVC	Polyvinyl chloride
SBR	Styrene–butadiene rubber
SEM	Scanning electron microscope
UWWTP	Urban wastewater treatment plant
WWTP	Wastewater treatment plant

An Ghníomhaireacht Um Chaomhnú Comhshaoil

Tá an GCC freagrach as an gcomhshaol a chosaint agus a fheabhsú, mar shócmhainn luachmhar do mhuintir na hÉireann. Táimid tiomanta do dhaoine agus don chomhshaol a chosaint ar thionchar díobhálach na radaíochta agus an truaillithe.

Is féidir obair na Gníomhaireachta a roinnt ina trí phríomhréimse:

Rialáil: Rialáil agus córais chomhlíonta comhshaoil éifeachtacha a chur i bhfeidhm, chun dea-thorthaí comhshaoil a bhaint amach agus díriú orthu siúd nach mbíonn ag cloí leo.

Eolas: Sonraí, eolas agus measúnú ardchaighdeáin, spriocdhírithe agus tráthúil a chur ar fáil i leith an chomhshaoil chun bonn eolais a chur faoin gcinnteoireacht.

Abhcóideacht: Ag obair le daoine eile ar son timpeallachta glaine, táirgiúla agus dea-chosanta agus ar son cleachtas inbhuanaithe i dtaobh an chomhshaoil.

I measc ár gcuid freagrachtaí tá:

Ceadúnú

- Gníomhaíochtaí tionscail, dramhaíola agus stórála peitril ar scála mór;
- Sceitheadh fuíolluisce uirbigh;
- Úsáid shrianta agus scaoileadh rialaithe Orgánach Géinmhodhnaithe;
- Foinsí radaíochta ianúcháin;
- Astaíochtaí gás ceaptha teasa ó thionscal agus ón eitlíocht trí Scéim an AE um Thrádáil Astaíochtaí.

Forfheidhmiú Náisiúnta i leith Cúrsaí Comhshaoil

- Iniúchadh agus cigireacht ar shaoráidí a bhfuil ceadúnas acu ón GCC;
- Cur i bhfeidhm an dea-chleachtais a stiúradh i ngníomhaíochtaí agus i saoráidí rialáilte;
- Maoirseacht a dhéanamh ar fhreagrachtaí an údaráis áitiúil as cosaint an chomhshaoil;
- Caighdeán an uisce óil phoiblí a rialáil agus údaruithe um sceitheadh fuíolluisce uirbigh a fhorfheidhmiú
- Caighdeán an uisce óil phoiblí agus phríobháidigh a mheasúnú agus tuairisciú air;
- Comhordú a dhéanamh ar líonra d'eagraíochtaí seirbhíse poiblí chun tacú le gníomhú i gcoinne coireachta comhshaoil;
- An dlí a chur orthu siúd a bhriseann dlí an chomhshaoil agus a dhéanann dochar don chomhshaol.

Bainistíocht Dramhaíola agus Ceimiceáin sa Chomhshaol

- Rialacháin dramhaíola a chur i bhfeidhm agus a fhorfheidhmiú lena n-áirítear saincheisteanna forfheidhmithe náisiúnta;
- Staitisticí dramhaíola náisiúnta a ullmhú agus a fhoilsiú chomh maith leis an bPlean Náisiúnta um Bainistíocht Dramhaíola Guaisí;
- An Clár Náisiúnta um Chosc Dramhaíola a fhorbairt agus a chur i bhfeidhm;
- Reachtaíocht ar rialú ceimiceán sa timpeallacht a chur i bhfeidhm agus tuairisciú ar an reachtaíocht sin.

Bainistíocht Uisce

- Plé le struchtúir náisiúnta agus réigiúnacha rialachais agus oibriúcháin chun an Chreat-treoir Uisce a chur i bhfeidhm;
- Monatóireacht, measúnú agus tuairisciú a dhéanamh ar chaighdeán aibhneacha, lochanna, uiscí idirchreasa agus cósta, uiscí snámha agus screamhuisce chomh maith le tomhas ar leibhéil uisce agus sreabhadh abhann.

Eolaíocht Aeráide & Athrú Aeráide

- Fardail agus réamh-mheastacháin a fhoilsiú um astaíochtaí gás ceaptha teasa na hÉireann;
- Rúnaíocht a chur ar fáil don Chomhairle Chomhairleach ar Athrú Aeráide agus tacaíocht a thabhairt don Idirphlé Náisiúnta ar Ghníomhú ar son na hAeráide;
- Tacú le gníomhaíochtaí forbartha Náisiúnta, AE agus NA um Eolaíocht agus Beartas Aeráide.

Monatóireacht & Measúnú ar an gComhshaol

- Córais náisiúnta um monatóireacht an chomhshaoil a cheapadh agus a chur i bhfeidhm: teicneolaíocht, bainistíocht sonraí, anailís agus réamhaisnéisiú;
- Tuairiscí ar Staid Thimpeallacht na hÉireann agus ar Tháscairí a chur ar fáil;
- Monatóireacht a dhéanamh ar chaighdeán an aeir agus Treoir an AE i leith Aeir Ghlain don Eoraip a chur i bhfeidhm chomh maith leis an gCoinbhinsiún ar Aerthruailliú Fadraoin Trasteorann, agus an Treoir i leith na Teorann Náisiúnta Astaíochtaí;
- Maoirseacht a dhéanamh ar chur i bhfeidhm na Treorach i leith Torainn Timpeallachta;
- Measúnú a dhéanamh ar thionchar pleananna agus clár beartaithe ar chomhshaol na hÉireann.

Taighde agus Forbairt Comhshaoil

- Comhordú a dhéanamh ar ghníomhaíochtaí taighde comhshaoil agus iad a mhaoiniú chun brú a aithint, bonn eolais a chur faoin mbeartas agus réitigh a chur ar fáil;
- Comhoibriú le gníomhaíocht náisiúnta agus AE um thaighde comhshaoil.

Cosaint Raideolaíoch

- Monatóireacht a dhéanamh ar leibhéil radaíochta agus nochtadh an phobail do radaíocht ianúcháin agus do réimsí leictreamaighnéadacha a mheas;
- Cabhrú le pleananna náisiúnta a fhorbairt le haghaidh éigeandálaí ag eascairt as taismí núicléacha;
- Monatóireacht a dhéanamh ar fhorbairtí thar lear a bhaineann le saoráidí núicléacha agus leis an tsábháilteacht raideolaíochta;
- Sainseirbhísí um chosaint ar an radaíocht a sholáthar, nó maoirsiú a dhéanamh ar sholáthar na seirbhísí sin.

Treoir, Ardú Feasachta agus Faisnéis Inrochtana

- Tuairisciú, comhairle agus treoir neamhspleách, fianaise-bhunaithe a chur ar fáil don Rialtas, don tionscal agus don phobal ar ábhair maidir le cosaint comhshaoil agus raideolaíoch;
- An nasc idir sláinte agus folláine, an geilleagar agus timpeallacht ghlan a chur chun cinn;
- Feasacht comhshaoil a chur chun cinn lena n-áirítear tacú le hiompraíocht um éifeachtúlacht acmhainní agus aistriú aeráide;
- Tástáil radóin a chur chun cinn i dtithe agus in ionaid oibre agus feabhsúchán a mholadh áit is gá.

Comhpháirtíocht agus Líonrú

- Oibriú le gníomhaireachtaí idirnáisiúnta agus náisiúnta, údaráis réigiúnacha agus áitiúla, eagraíochtaí neamhrialtais, comhlachtaí ionadaíocha agus ranna rialtais chun cosaint chomhshaoil agus raideolaíoch a chur ar fáil, chomh maith le taighde, comhordú agus cinnteoireacht bunaithe ar an eolaíocht.

Bainistíocht agus struchtúr na Gníomhaireachta um Chaomhnú Comhshaoil

Tá an GCC á bainistiú ag Bord lánaimseartha, ar a bhfuil Ard-Stiúrthóir agus cúigear Stiúrthóir. Déantar an obair ar fud cúig cinn d'Oifigí:

1. An Oifig um Inbhunaitheacht i leith Cúrsaí Comhshaoil
2. An Oifig Forfheidhmithe i leith Cúrsaí Comhshaoil
3. An Oifig um Fhianaise agus Measúnú
4. An Oifig um Chosaint ar Radaíocht agus Monatóireacht Comhshaoil
5. An Oifig Cumarsáide agus Seirbhísí Corparáideacha

Tugann coistí comhairleacha cabhair don Ghníomhaireacht agus tagann siad le chéile go rialta le plé a dhéanamh ar ábhair imní agus le comhairle a chur ar an mBord.

EPA Research

Webpages: www.epa.ie/our-services/research/
LinkedIn: www.linkedin.com/showcase/eparesearch/
Twitter: @EPAResearchNews
Email: research@epa.ie

www.epa.ie