An index-based risk assessment approach for accidental contaminant release from waste management facilities during flood events.

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Abstract

Natural hazards can trigger technological disasters in installations such as waste management facilities, and chemical and processing plants, leading to explosions, fires, and/or the release of dangerous substances. The likelihood of these 'Natech' incidents can be exacerbated by climate change intensifying very wet weather patterns leading to floods. Rising sea levels may further contribute to heightened flood risk. Consequently, the number of 'at risk' installations to flooding is increasing, and this trend is expected to continue as our climate warms. The Seveso Directive provides guidelines for identifying installations handling dangerous substances and mandates safety measures to minimise the likelihood and impact of accidents. However, the risk of contaminant release during floods is not limited to installations falling under the Seveso Directive. Various small and medium-sized facilities, such as waste management facilities, often located near residential areas, also handle hazardous waste and pose a threat to both human health and the environment in the event of accidental release.

This research assesses the vulnerability of waste management facilities to flooding. We use an adapted form of the Water Risk Index (WRI), originally designed for large-scale industrial facilities, to estimate risk of flooding to waste facilities on a facility-by-facility basis. The initial application of the WRI to waste management facilities revealed significant gaps such as the absence of detailed georeferenced areas representing the spatial extent of the waste management facilities, the neglect of the spatial context, and the lack of consideration for waste materials that can degrade into smaller particulates such as microplastics. Here we address these gaps to enhance the evaluation methods for understanding the impacts of flooding on waste management facilities and the potential consequences on the environment and community resilience.

Three primary methodologies have been developed and tested in Great Britain (GB) to address the knowledge gaps. The first methodology determines the spatial extent of waste management facilities, providing a comprehensive understanding of their footprint. In testing their vulnerability to inundation, the results indicate that a decrease in flood likelihood corresponds to an increase in the number of affected waste management facilities and the severity of the impact. Specifically, out of the 1,049 facilities tested, 10% (23 sites) displayed more than 40% of their footprint at risk from high flood likelihood (with a 10% annual probability). These percentages rise to 33% (88 sites) and 35% (111 sites) for medium (0.5%) and low likelihoods (0.1%), respectively.

The second methodology assesses the vulnerability of waste facilities to flooding at the national scale by considering contextual factors from physical and human geography. These factors form a new multi-index-based assessment considering hazard, vulnerability, and exposure. The aim was to identify hotspots that necessitate additional analysis at the local level to efficiently mitigate the risk. The overall risk index (categorised as low, medium, and high) is estimated for a total of 7,292 facilities across GB. Approximately 15% (1,094 sites) classified with a high-risk index are located in areas at high risk of pluvial flood likelihood. Medium and low flood risks increase these figures to 37% (2,697 sites) and 44% (3,204 sites), respectively. We show that facilities with a high-risk index outweigh those with medium and low risks, particularly in scenarios with a high likelihood of floods, whether fluvial or pluvial. These results indicate that for flood-affected waste management facilities, the vulnerability of receptors is frequently triggered at the full potential.

Finally, the third methodology establishes a framework to assess the plastic mobilisation potential from waste management facilities by estimating the location and quantity of waste materials capable of releasing synthetic micro-components into floodwaters. The term Microplastic Releasers (MPRs) is introduced to describe waste materials capable of degrading into synthetic microplastic components. MPRs include plastic, synthetic textile, and rubber waste. When applying the method to waste management facilities across GB, the results indicate a significant amount of MPRs at high risk of fluvial flooding, totalling nearly 1 million tonnes. However, the impact of pluvial flooding is even more severe: in the case of flood events ranging from a 5-year to a 1,000-year return period, the exposure of MPRs to floodwaters increases tenfold, from 1 to 11 million tonnes.

By integrating the methodologies developed in this research, hotspots for further research on risk management and mitigation at the local level can be identified. Stakeholders and policymakers may reconsider the placement of waste facilities to non-flood-prone areas. If relocation is not possible, mitigation measures such as the implementation of flood defences as well as site-specific containment systems designed to minimise the release of synthetic micro components during a flood event can be introduced. The results have significant implications not only for waste management practices but also for broader discussions on environmental management, risk assessment, and the resilience of industries in the face of climate change.

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1 Introduction

1.1 Setting the scene

As global temperatures rise, the risk of extreme rainfall events and subsequent floods increases significantly, according to the Intergovernmental Panel on Climate Change (IPCC) Sixth Assessment Report (IPPC, 2022). A warmer climate intensifies both very wet and very dry weather patterns, leading to a higher likelihood of flooding in various regions worldwide. Additionally, rising sea levels caused by climate change further contribute to increased flood risk (IPPC, 2021). Recent years have witnessed extreme rainfall events in Europe, including severe flooding in 2019 in north Wales and central/northern England during the summer (Sefton *et al.*, 2021). These summer floods paved the way for extended and widespread flooding in late winter. The impact was intensified by intense storms, resulting in record-breaking river flows and significant damage to numerous properties, with some regions observing the highest peak flows ever recorded (Sefton *et al.*, 2021). In July 2021 severe flooding occurred in Germany and Belgium causing significant damage and loss of life. In Germany, heavy rainfall led to flooding in several states, including Rhineland-Palatinate and North Rhine-Westphalia, where over 130 people died and thousands of homes were destroyed or damaged (Truedinger *et al.*, 2023). The flooding also disrupted transportation, electricity, and communication networks. With precipitation return times varying from 1 in 300-year to 1 in 1,000-year, and peak flow return times of 1 in more than 500-year (Kreienkamp *et al.*, 2021), floods were described as some of the worst in several decades. Recovery efforts are expected to take years. In Belgium, flooding caused significant damage to several industrial zones, particularly in the provinces of Liège and Namur. Several sewage treatment plants in Altenahr, Mayschoss and Sinzig have been largely destroyed (Koks *et al.*, 2021), leading to concerns about environmental contamination due to the potential release of hazardous substances.

Many of the installations that process, store or handle hazardous substances are vulnerable to natural hazards such as floods. Potentially vulnerable infrastructure includes for example chemical and process plants, oil and gas production and transport, textile manufacturing and treatment, waste treatment and disposal sites, etc. The impact of natural hazards on hazardous installations is referred to as Natural Hazard Triggering Technological Disasters, or Natech, and it may cause explosions, fires and damage of safety systems potentially resulting in dangerous substances release. Natech incidents are difficult to identify promptly and contain during flooding because of the size and complexity of the areas involved. Recurrence is also a problem. Future estimates suggest that the severity of natural hazards associated with climate change is expected to rise in the future decades (IPPC, 2022). Additionally, there is a possibility of these hazards manifesting in previously unaffected regions, which could potentially lead to an increased number of sites that handle hazardous materials at risk of flooding.

As a means of controlling the risks posed by sites that handle large quantities of hazardous substances (referred to hereafter as industrial sites), the Seveso Directive was introduced in 1976 in response to a major industrial accident that occurred in the Italian town of Seveso where a chemical reactor at a pesticide manufacturing plant overheated and released a cloud of toxic dioxin gas into the surrounding area, exposing thousands of people to the dangerous chemical. The Directive establishes a framework for the identification of industrial sites handling dangerous substances and mandates specific safety measures to be taken by operators to minimize the likelihood and impact of accidents. It emphasises risk assessment, emergency planning, information sharing, and involvement of relevant stakeholders to ensure the protection of human health and the environment. Initially adopted in Europe in 1985 (82/501/EEC), the Directive has since undergone two revisions: one in 1996 and another in 2012 with the Seveso III Directive (2012/18/EU).

In the UK, the Control of Major Accident Hazards (COMAH) Regulation 2015 was implemented to adhere to the Seveso III Directive. Approximately 950 sites are subject to the regulation in Great Britain, representing 12% of all Seveso sites in the EU (UK Government, 2013). The location of many of these COMAH establishments can be attributed to the history and development of the chemical sector in the UK as well as subsequent local housing planning decisions. As a result, many of these facilities are situated in close proximity to residential populations and environmentally sensitive locations such as estuaries and sites of special scientific interest (UK Government, 2013). The risk posed by industrial sites to vulnerable receptors is amplified by the pressure of climate change on hydraulic systems. Over the past decade (2012-2021), summers in the UK have been approximately 6% wetter than the period between 1991-2020 and 15% wetter than the period between 1961-1990 (Kendon *et al.*, 2021). UK winters have experienced an increase in precipitation as well, with a 10% and 26% increase compared to those same periods, respectively. Additionally, the UK has seen an acceleration in the rate of sea-level rise, with selected locations recording a range of 3.0 ± 0.9 to 5.2 ± 0.9mm per year^{-1} over the past 30 years when adjusted for vertical movement, compared to the 1.5 ± 0.1mm per year^{-1} rate observed since the 1900s (Kendon *et al.*, 2021). The severity of the issue is highlighted by the level of investment made by the government towards flood and coastal defences: by the end of 2019 the UK government was committed to investing £2.6 billion in capital funding, which doubled in March 2020 to £5.2 billion for the period 2021-2027 (UK Government, 2021).

As the risk of flooding continues to put industrial sites under increasing pressure, it is important to address the potential exposure of communities and environmental areas to industrial pollution through a comprehensive risk assessment analysis at the national scale. The Water Risk Index (WRI) (ICPDR, 2001) was selected as a preferred quantitative method to identify disposal sites at the national/regional scale that may pose a significant risk of hazardous releases during inundation events for further investigations at the local scale. Unlike the Seveso III Directive (2012/18/EU), the WRI specifically focuses on the classification of hazardous substances based on their potential reaction when exposed to water. The WRI can be estimated on a per-site basis and summarised by area of interest, and it requires only a minimal amount of information for national-scale analyses. The latter consists in the location of industrial facilities (geographic coordinates or addresses), type (given by the CLP Regulation for classification and labelling of hazardous substance) and quantity of substances handled on site (e.g., annually). Although the Seveso Directive was developed to prevent major accidents resulting from the release of hazardous substances, it poses certain challenges to the application of the WRI to COMAH installations:

- There are concerns regarding sites that handle hazardous substances in quantities or of a type that do not fall under the COMAH Regulation, but which still pose a safety risk if released into the environment (Krausmann *et al.*, 2016, Mcintyre, 2016). Most business activities that have the potential to cause pollution or pose risks to the environment are subject to other licenses and permits. For instance, landfills can have a significant environmental impact if they become flooded, as has been demonstrated by several studies on the subject (Laner *et al.*, 2009, Arrighi *et al.*, 2018a, Brand *et al.*, 2018, Nicholls *et al.*, 2021, Neuhold, 2012);
- The location of COMAH sites in the UK is not publicly available due to security concerns. Although activity and geographical coordinates can be obtained with a request under the Freedom of Information Act, no information can be collected on quantity of materials handled on site and typology.

The limitations reveal gaps in the assessment of risks posed by industrial sites located in flood-prone areas. Firstly, the current approach falls short in providing adequate coverage for sites that do not fall under the COMAH Regulation but may still pose safety risks if hazardous releases occur during flooding events. This is the case of waste management activities that have been traditionally overlooked in favour of other industrial facilities. Despite the almost 50 million tonnes of hazardous waste that is handled annually in the UK (SEPA, 2019c, EA, 2019a, Natural Resources Wales, 2019, NIEA, 2019), the majority of disposal sites are not covered by COMAH Regulation. Secondly, without the access to publicly available information on COMAH installations it becomes challenging to estimate the potential hazards effectively. On the contrary, datasets on waste are freely available for most of the countries in Europe.

1.2 Preliminary analysis and key research questions

To overcome the limitations posed by the applicability of the WRI to COMAH installations, a preliminary set of analysis were conducted on waste management facilities located in Scotland. This early work played an essential role in laying the foundation for the subsequent stages of the research. Its primary objectives were to validate the applicability of the WRI methodology to the European Waste Catalogue (EWC) classification system, evaluate the comprehensiveness of publicly accessible waste datasets, identify the disposal sites located in flood-prone areas, and estimate the WRI per each waste facility and Local Authority to prioritise further risk assessment and mitigation strategies in such locations. The findings from this pivotal early work aimed not only to increase awareness regarding the hazards associated with disposal sites located in flood-prone areas, but also to identify further areas for research to enhance the application of the methodology on a larger scale in the UK.

In 2001, the International Commission for the Protection of the Elbe River (ICPDR) developed a regional-scale risk assessment methodology to identify industrial facilities that could potentially release hazardous substances in the Elbe River basin (ICPDR, 2001). Unlike the Seveso III Directive, this approach by ICPDR explicitly incorporated considerations of floods and other natural hazards. The methodology focused on the classification of hazardous substances based on their reactivity with water, leading to the development of the WRI. By analysing data on the types and quantities of hazardous substances present in each facility, the WRI was used to assess the risk of accidental pollution in the event of a flood. Substances were classified according to their chemical properties based on the German Ordinance on facilities for handling substances that are hazardous to water (Federal Environmental Agency, 1999). The German classification system has four water risk classes (WRCs): non-hazardous or generally hazardous to water (class = 0); slightly hazardous (class = 1); moderately hazardous (class = 2); and highly hazardous (class = 3). The WRI can be estimated with the following equations:

$$WRI = Log\sqrt{WRC} \qquad \textit{Eq. 1-1}$$

where:

$if\ class\ 0,$	$if\ class\ 1,$	$if\ class\ 2,$	$if\ class\ 3,$
$WRC = \frac{kg}{1{,}000}$	$WRC = \frac{kg}{100}$	$WRC = \frac{kg}{10}$	$WRC = kg$

Eq. 1-2

Table 1-1. Example of WRI estimation per substance and per site (ICPDR, 2001).

Substances handled within a facility	Class of risk (Federal Environment al Agency, 1999)	Substance amount (Kg)			Water Risk Index per substance
Triglycerides	0	1,0000	kg / 1,000	10	log√10 = 1
Magnesium chloride	1	1,0000	kg / 100	100	log√100 = 2
Ammonia	2	1,0000	kg / 10	1,000	log√1,000 = 3
Benzene	3	1,0000	kg	1,0000	log√1,0000 = 4
Water Risk Index per facility:				TOT = 11,110 kg	log√11,110 = 4.045

To apply the WRI methodology to waste management activities in the UK, the sorting of each substance into the German risk classes (0, 1, 2, or 3) based on its chemical composition is needed. However, the classification of discarded materials is a complex and dynamic process. The List of Waste (LoW), also known as EWC, was established in 2000 under the European Commission Decision 2000/532/EC and has undergone revisions in 2014 and 2017. Unlike the straightforward legislation on chemicals, the LoW does not rely on the chemical components of waste for classification. Instead, it employs alternative criteria such as (1) the waste source, (2) the waste type, and (3) the identification of waste not otherwise specified due to its mixed or undifferentiated nature. In the LoW the identification and classification of waste is provided by a six-digit code, while the level of hazard is determined by a four-type classification system. Absolute hazardous (AH) waste exhibits one or more of the fifteen hazardous properties outlined in Annex III to the Waste Framework Directive (2008/98/EC), such as being explosive, ecotoxic, mutagenic, or infectious; absolute non-hazardous (ANH) waste lacks any hazardous component; and mirror entries correspond to mixed substances, which require further assessment to classify the waste as mirror hazardous (MH) or mirror non-hazardous (MNH). The adaptation of the WRI to the EWC was therefore performed as shown in Table 1-2.

Table 1-2. An example of discarded materials is reported under the LoW code and description to show the proposed adaptation of the German classes of risk to the EWC. The asterisk next to the six-digit code represents an additional indicator for hazardous materials.

LoW code and description	Waste entry classification	Class of risk (Federal Environmental Agency, 1999)
03 01 01 waste bark and cork	AN	0
17 03 02 bituminous mixtures other than those mentioned in 17 03 01	MN	1
17 03 01* bituminous mixtures containing coal tar	MH	2
13 07 01* fuel oil and diesel	AH	3

The WRI methodology was applied to waste received by waste management facilities in Scotland (SEPA, 2019b) accordingly to the LoW codes. The results were then summarised for each of the 697 operative disposal sites in 2019, with each facility assigned a unique WRI value. The resulting WRI values ranged from a minimum of 0.17 to a maximum of 17.53, with an average of 10.91 and a standard deviation of 2.23. The WRI values were divided into equal intervals to evaluate the risk levels associated with these facilities (low, medium, and high). To assess the susceptibility of disposal sites to flooding events, a distance of 150m between the facility location and nearby rivers, wetlands, and/or the ocean was selected based on previous studies on landfills (Neuhold and Nachtnebel, 2011, Neuhold, 2012). Among the 697 sites, 182 (26%) were found to be located within 150m of freshwater (145/182), wetlands (8/182), and/or the ocean (34/182). The full list of results is available as electronic format (Ponti, 2019). Of the 182 sites potentially at risk of flooding, 70 were estimated to have a high WRI, 94 a medium, and 18 a low WRI. In relation to the annual waste intake, the 70 facilities characterised by high WRI values collectively received 218,730 tonnes of waste, averaging at 3,124 tonnes per facility. In contrast, sites with medium WRI values received an average of 5,530 tonnes, while those with low WRI values received 52 tonnes. The disparity arises from the proportion of hazardous waste within the total quantity, which is encapsulated by the WRI methodology.

With 26% of the disposal sites being potentially at risk of flooding in Scotland and, among them, 70 sites at high risk of releasing hazardous substances in flood waters, these initial findings underscored the significance of evaluating the risk associated with waste facilities situated in flood-prone areas. Moreover, the initial testing of the method identified additional gaps in applying the WRI to disposal sites that are crucial for improving its scalability across the UK:

- The existing waste datasets in the UK only provide georeferenced coordinates referring to a general point within licensed waste facilities, lacking detailed information on the actual footprint of these sites. To conduct spatial analysis on the potential impact of flooding, it is essential to recreate the precise physical area occupied by the waste facilities. The latter is particularly significant for disposal sites that encompass substantial land areas, such as landfills. For example, in Scotland the largest recorded landfill has a maximum surface area of 375,000m^2;
- The WRI applied to waste activities provides an initial estimation of potential risks based on waste quantity and type. However, it does not account for the spatial context of waste facilities. Factors like physical geography (e.g., landforms, bodies of water) and human geography (e.g., protected environmental areas, proximity to communities) must be considered. These factors help evaluate the impact of hazards and the vulnerability of receptors, including both direct and indirect consequences for people and the environment, to pollution events; and
- The EWC classifies plastic waste primarily as non-hazardous solid waste, but it is important to acknowledge that plastic consists of polymers and various additives. During degradation, plastic can release emerging contaminants, which can pose risks to ecosystems and human health. Additionally, plastic is not the only substance that can break down into micro components. Materials such as textiles and rubber also have the potential to release contaminants, thereby posing risks to ecosystems and human health (Andrady *et al.*, 2022).

To address the identified gaps, six key research questions were established leading to the formulation of three primary methodologies that form the core scientific component of the Thesis (Table 1-3).

Table 1-3. Overview of the six key research questions identified during the preliminary analysis, along with the three primary methodologies developed and their corresponding Chapter numbers.

Research questions (RQ)	Primary methodologies	Chapter no.
RQ1: considering the necessity of assessing the exposure of sites to flood risk, what can be used to represent the footprint of waste management facilities?	Estimating the footprint of waste management facilities.	2
RQ2: how can the methodology of the Water Risk Index be expanded to consider the spatial context of waste facilities when estimating the risk posed to receptors? How can this be expressed simply to enable the comparison between sites?	Developing an index-based approach to measure and compare the risk posed by disposal sites based on context factors.	3
RQ3: how is the risk distributed? Which sites and Local Authorities require further investigation at the local scale for risk mitigation and management?		
RQ4: apart from well-known plastic waste, which other type of waste classified within the EWC possess the potential to degrade and release microplastics (MPs)?	Assess the location and quantity of waste materials able to release synthetic micro-components in flood waters.	4
RQ5: what quantity of waste received by disposal sites in the UK is at risk of flooding, potentially leading to the mobilisation of MPs in floodwaters?		
RQ6: which sites and Local Authorities warrant further localised studies to evaluate the level of risk associated with the release of MPs from waste management facilities?		

1.3 Structure of the Thesis

The Thesis is organised into six Chapters and three Appendices, with each Chapter dedicated to addressing specific research questions. The content of each Chapter is outlined as follows:

Chapter 1 sets out the context of the Thesis, defines the Water Risk Index (WRI) that is used across all Chapters, and presents preliminary findings from the application of the WRI to a specific set of facilities. These findings play a crucial role in shaping the research questions that will be addressed throughout the thesis.

Chapter 2 addresses RQ1 by considering the absence of georeferenced polygons representing waste activities in publicly waste datasets, which poses a challenge for conducting spatial analysis using flood extent maps in GIS environment. The Chapter introduces and examines three methodologies aimed at recreating the spatial extent of disposal sites, referred to as the facility footprint. The first methodology investigates the use of the INSPIRE Index Polygons spatial dataset (European directive 2007/2/EC), which is tested independently and found to have several limitations. To overcome these limitations, the other two methodologies focus on estimating buffers as an alternative to using polygons. In GIS-based studies, buffers—areas surrounding a specific point measured by distance as detailed by Guo et al. (2020)—are commonly employed for numerous purposes. These include assessing landslide risks (Saha *et al.*, 2005), determining land-use effects on river water purity (Sliva and Williams, 2001), and general geographic data tasks (Liu *et al.*, 2015). Such analyses have further been utilised in inspecting facilities from the EPA's 1994 Toxic Release Inventory to understand industrial pollution impact on the environment (Sheppard *et al.*, 1999, Chakraborty and Armstrong, 2013). However, specific buffer metrics for medium to small industrial activities, such as waste management facilities, remain absent. Hence, this Chapter concentrates on formulating a strategy to determine the best buffer sizes for recreating waste site boundaries. Based on the outcomes, a preferred methodology is selected and subsequently applied in the following Chapters.

Chapter 3 is dedicated to addressing RQ2 and RQ3. Building on the WRI methodology introduced in Chapter 1, the methodology for risk assessment is further investigated by considering the core variables of risk defined by the literature as the product of hazard, vulnerability, and exposure. Research has explored these variables extensively, as evidenced by works like Kron (2005), De León (2006), Cardona et al. (2012), Cannon (2006), and Cutter and Finch (2008). Hazards indicate the potential of a certain event happening at a specific place, while vulnerability underscores the possible susceptibility to damage of the receptors in question. Exposure, on the other hand, highlights the receptors (for instance, human populations, the economy, and both natural and man-made environments) that might be impacted. Numerous methodologies for describing the core variables through designated weighted indicators and parameters have been investigated in the literature. This chapter is influenced by three pivotal studies selected for their pertinence. The DRASTIC model related to groundwater vulnerability (Linda *et al.*, 1987) employs a comparative ranking system to produce a value named the DRASTIC index. This model primarily examines the geological and hydrological elements that influence aquifer contamination. Similarly, the FIGUSED approach (Nerantzis *et al.*, 2015) was developed to identify flood-prone areas requiring intervention, factoring in aspects like geological composition and flow accumulation. Arrighi et al. (2018a) present the only study that applies a multi-index approach to a selection of waste management sites and polluted lands situated in flood-prone areas. Their regional-level study identifies specific areas where additional local-level investigations are required to prevent pollution risks. While the aforementioned studies are designed for application at the river catchment/regional scale, there's a noticeable gap in assessments for the national scale. This gap is particularly pronounced when considering how flooding might impact disposal sites. Although the criteria and parameter choices explored in this chapter draw inspiration from the multi-index approaches mentioned earlier, they are adapted to fit available datasets. Furthermore, they are optimised for the practicality of national-scale data processing, especially focusing on the risks related to waste management facilities in areas susceptible to flooding.

Chapter 4 addresses RQ4, RQ5 and RQ6 by exploring the presence of substances regularly handled/stored by waste management facilities that have the potential to deteriorate and fragment into synthetic micro-component, despite not being currently recognised as hazardous in the EWC. The fragmented substances (MPs) can then be mobilised and dispersed in case of flooding.

Flood-induced plastic debris mobilisation has yet to be fully understood although recent studies have increasingly identified terrestrial sources as primary contributors to ocean plastics (Hurley *et al.*, 2018). Rivers, in particular, have been recognised as significant conduits for the transit of plastics from inland regions to the oceans (van Emmerik *et al.*, 2019, He *et al.*, 2021, Wang *et al.*, 2021). Hurley et al. (2018) conducted a comprehensive study on the impact of flooding on plastic loads in rivers. By examining 40 rivers in northwest England both before and after the severe floods in winter 2015/2016, they observed the presence of microplastics (MPs) in every riverbed analysed. Furthermore, they determined that floods could displace and transport about 70% to 100% of the total MPs previously residing in these riverbeds.

In recent years, the effects of flooding on managed waste have been sparingly examined. A standout study is the already mentioned research by Arrighi et al. (2018a) that identified wastewater treatment plants, waste handling facilities, and contaminated locations as environmental concerns due to the risks associated with contaminants in flood-vulnerable zones. Comparatively, the possible flooding of solid waste landfills has gained more academic interest (Laner *et al.*, 2009, Neuhold, 2012, Neuhold and Nachtnebel, 2011), but many aspects remain unexplored. Notably, Nicholls et al. (2021) conducted an extensive study on European coastal landfills in the context of rising sea levels, underscoring a data shortage and a prevalent underestimation of the hazards associated with potential solid and liquid waste releases from coastal landfills.

Beyond the wear and tear from transport, treatment, and weathering—especially when waste is stored outside in containers or piled on rigid surfaces—flood waters can further impact the degradation and fragmentation of materials. This is largely due to the water acting as a mechanical force capable of breaking particles (Zhang *et al.*, 2021). In fact, during flooding discarded materials can be exposed to quickly changing flow conditions and the presence of suspended sediments. This can induce turbulent mixing, resulting in mechanical abrasion, wear, and collisions with debris and built structures.

This chapter seeks to enhance the understanding of terrestrial sources of MPs by examining the likelihood of waste management facilities releasing microplastics during flood events. To fulfil this aim, the Chapter outlines four distinct objectives: (1) identify the LoW codes, beyond well-known plastic waste, referring to discarded materials at risk of releasing MPs (defined as Microplastic Releasers); (2) estimate quantity of waste located in disposal sites that are at risk of flooding in the UK, which could result in the mobilisation of MPs in floodwaters; (3) highlight the danger posed by landfills located in flood-prone areas by considering the quantities of Microplastic Releasers received since 2007, and (4) identify spatial patterns where the concentration of risk requires further localised studies for risk monitoring and mitigation. This work was published in the Journal of Hazardous Materials Advances and is included in its entirety in Appendix D. The supplementary material can be found in Appendix C.

Chapter 5 critically examines the datasets, methodologies, and analyses presented in previous Chapters, highlighting their limitations and providing recommendations for further risk assessment of waste management facilities. Special emphasis is placed on recommendations for local-scale risk assessment.

Chapter 6 highlights final remarks and conclusions on the research methodologies developed to answer the initial research questions.

Appendix A provides additional information related to Chapter 3, including the comprehensive list of criteria, attributes, and values used for estimating hazard, vulnerability, and exposure. It also includes detailed sources for all the datasets used to populate the vulnerability criteria specifically for Great Britain.

Appendix B presents the vulnerability index values estimated per waste facility, summarised per Local Authority, and normalised per capita. Each vulnerability criterion is represented using a map of Great Britain, allowing for easy comparison among the different criteria. This visual approach provides a clear and concise way to analyse and understand the spatial distribution of vulnerability associated with waste facilities.

Appendix C contains the supplement materials associated to the published paper "A framework to assess the impact of flooding on the release of MPs from waste management facilities" (Ponti *et al.*, 2022).

Appendix D reports the published paper: "A framework to assess the impact of flooding on the release of microplastics from waste management facilities" (Ponti *et al.*, 2022).

2 Estimating the footprint of waste management facilities for the spatial analysis of risk

2.1 Introduction

In the United Kingdom, datasets pertinent to waste management spatially delineate licensed waste processing facilities through georeferenced coordinates. Notably, these coordinates represent a randomly positioned point within the confines of the respective facility. The use of points instead of polygons to locate geographic features can result in several limitations:

- Spatial information loss: point data only indicates a specific location and do not capture the spatial extent or boundaries of the land/facility considered. Consequently, crucial spatial details may be overlooked resulting in less accurate analysis;
- Impossibility to capture changes in boundaries: point data cannot capture the extent of the change overtime and make sure the boundaries of facilities are up to date;
- Inability to measure the area: because the points reported in the waste facilities dataset do not include information on the size of the installations, the use of polygons is the only feasible solution to estimate the area of sites;
- Limitation on the performance of spatial analysis: point data lack the spatial detail to conduct advanced spatial analyses, such as spatial interpolation, spatial relationships between features, or spatial pattern analysis. This limits the range of analysis that can be performed resulting in less informative outcomes.

The limitations are particularly relevant when investigating the spatial relationships between waste facilities and flooding extent maps in GIS environment (i.e., QGIS, ArcMap, ArcGIS Pro). While facilities are represented by point locations, the absence of georeferenced polygons representing the true footprint of the sites – intended as the physical area occupied by the facility itself (buildings and infrastructures), as well as any associated structures such as access roads, parking lots, or storage areas – hinders precise spatial analysis. Consequently, the spatial data reported in waste management datasets is insufficient to establish the risks posed by installations exposed to flooding.

To fill the gap, this chapter evaluated three distinct methodologies to approximate the footprint area of waste management sites. Initially, the study utilises the INSPIRE Index Polygons spatial dataset (European directive 2007/2/EC), which represents the sole publicly accessible dataset delineating the position and indicative extent of registered titles in Great Britain. Due to the unavailability of INSPIRE Index Polygons for Northern Ireland, assessments were confined to waste facilities in England, Scotland, and Wales. Subsequently, the second and third methodologies were centred around formulating two distinct 'buffer' techniques to overcome the limitations of the INSPIRE polygons, designed to approximate the footprint area of waste management facilities.

The use of buffers, defined as the zone around a point feature measured in units of distance (Guo *et al.*, 2020), is well-known in GIS-based analysis for many applications including landslide susceptibility (Saha *et al.*, 2005), land use impact on river water quality (Sliva and Williams, 2001), and geographic data processing (Liu *et al.*, 2015). Buffer analyses have also been applied to facilities listed in the EPA's 1994 Toxic Release Inventory database to investigate the exposure to industrial pollution for environmental equity assessment purposes (Sheppard *et al.*, 1999, Chakraborty and Armstrong, 2013). This work aimed to develop a methodology to estimate buffer sizes to best represent facility footprint. A simplistic circular shape was adopted. Evaluation of different shapes of buffer was considered outside the scope of this Chapter and left for future studies.

2.2 Methodology

2.2.1 The INSPIRE Index Polygons spatial dataset

The only dataset freely available in Great Britain (GB) to represent the footprint of waste facilities at the time of writing was the polygons dataset, which is also available for other countries in Europe. The INSPIRE Cadastral Parcels dataset or INSPIRE Index Polygons spatial dataset (for the UK) complies with the European INSPIRE directive 2007/2/EC and contains information on registered property represented by polygons. Polygons are shapes that show the position and indicative extent of registered titles represented by unique identification numbers called the Land Registry-INSPIRE ID.

The applicability of the polygons for the scope of this Chapter was investigated by looking at the datasets for Scotland (Registers of Scotland, 2021), England and Wales (HM Land Registry, 2021). The datasets (in the format of shapefiles), openly available files divided per city, were downloaded with the help of the programming language R. Out of the 7,292 waste facilities active in 2019 in GB (SEPA, 2019b, EA, 2019d, Natural Resources Wales, 2019), the sites matching with a polygon were identified in Scotland (171), England (833), and Wales (50) for a total of 1,054 facilities (14%). In addition to the limited coverage, the dataset presented two limitations for this research: (1) polygons frequently overestimate the actual footprint of waste facilities by including green areas, residential properties, and others and (2) the frequency at which the dataset is updated is unknown. Therefore, the polygons may not represent the most recent extents of the sites.

Because a substantial number of waste facilities did not have polygons (86% of total waste management facilities in GB), and the perimeter of the polygons may over represent the actual footprint of waste facilities, two buffer methodologies were introduced to investigate the use of buffers to compensate for the uncertainty in the property delineation of polygons for waste facilities (area, shape). All three approaches were compared.

2.2.2 Annual waste intake buffer methodology

The first methodology developed to establish reliable buffers is based on a population sample of polygons (manually inspected), representing waste facilities in Scotland, and information on their annual waste intakes (tonnes of waste received per year). Linear regression analysis among the two sets of data were performed to (1) understand the strength and direction of the linear relationship, and (2) use the annual waste intake variable to predict buffer areas representing the footprint of disposal sites for future analysis.

To select a sample of sites to be manually inspected, the Scottish dataset for annual waste received by waste management facilities (SEPA, 2019b), which contains geographic coordinate points for each facility, was overlapped with the polygons dataset in GIS to identify disposal sites with a unique footprint associated to them (171). Subsequently, polygons were manually inspected with the use of Google Earth and corrected to match the actual perimeter of waste facilities, leaving 154 footprints for further analysis. Polygons were found to be at least 3.5-times larger than the actual footprints.

The strength and direction of the linear relationship between the area (m^2) of the footprints and the quantity (tonnes) of annual waste intake was determined with the Pearson correlation coefficient per each category of waste facility (incineration, landfill, transfer, treatment, metal treatment, on/in land including landfill, and storage; Table 2-1).

Table 2-1. Name and description of categories for the waste management facilities considered in the study based on the classification included in the dataset "Waste

Permit Returns Data Interrogator 2019" for England and Wales available at https://data.gov.uk/data.

Category of waste facility	Sub-categories
Incineration	Animal by-products incinerator; biomass; clinical waste incinerator; co-incinerator; co-incinerator (hazardous); EFW incinerator; hazardous waste incinerator; municipal waste incinerator; pet crematorium
Landfill	Hazardous merchant LF; hazardous restricted LF; inert LF; non-hazardous (SNRHW) LF; non-hazardous LF; restricted LF
Transfer	Competent authority (CA) site; clinical waste transfer; hazardous waste transfer; inert waste transfer; non-hazardous waste transfer
Treatment	Anaerobic digestion; biological treatment; chemical treatment; clinical waste treatment; composting; hazardous waste treatment; material recycling facility; mechanical biological treatment; non-hazardous waste treatment; physical treatment; physical-chemical treatment; recovery of waste; WEEE treatment facility; animal and food waste; non-ferrous metal re-processing; paper and pulp re-processing; paper recycling
MRS (metal treatment)	Car breaker; metal recycling; vehicle depollution facility; ferrous metal re-processing; metal re-processing
On/In Land	Deposit of waste to land (recovery); lagoon; mining waste management (non- hazardous)
Storage	In-house storage; storage – A/D; storage – dredging; storage – incinerator; storage - metal reprocessing; storage – oils; temporary storage installation

To predict buffer values based on the annual waste intake for disposal sites, the least-squares regression line (Eq. 2-1) was estimated by using the correlation value, mean and standard deviation of the 154 footprints and their annual waste intake.

$$Predicted\ y = a + b * x \qquad Eq.\ 2\text{-}1$$

By knowing the mean, standard deviation (S) for x and y, and the correlation (r), the slope (b) and the starting value (a) were calculated with the Eq. 2-2.

$$b = \frac{r *\ s_y}{s_x} \quad and \quad a = \ \bar{y}\ - b\ \bar{x} \qquad Eq.\ 2\text{-}2$$

The buffer for each site was therefore estimated and the means (per category of waste facility) visually compared to footprints to determine the accuracy of the projected values. Although the estimated buffers were found acceptable for few categories of waste facilities, the size of the population sample was determined too low to produce statistically meaningful results. Therefore, a second approach to estimate buffers for waste facilities was investigated.

2.2.3 *Buffer sizing from INSPIRE Index Polygons and exposure to flooding*

The second methodology developed to establish reliable buffers to represent the footprint of waste facilities focuses on the interaction between polygons and estimated buffers in relation to flood risk extent. The rationale is that regardless the shape and size of polygons and buffers, if the level of exposure to flood risk is similar, then the estimated buffers can be used to overcome the limitations with the polygon's dataset. Compared to the annual waste intake buffer methodology, the size of the population sample was expanded to include datasets from England and Wales. Because of the volume of national scale sites, manual inspection of all polygons was not conducted and the process to estimate buffers was based on the average size of polygons (means) separated by category of waste facility and per country (Scotland, England and Wales).

Different Fathom-UK fluvial flood map extents were used to represent the extent of flood risk for the low (1 in 1,000-year), medium (1 in 200-year), and high (1 in 10-year) flood risk return periods (SSBN UK Limited, 2021a). Further information on the Fathom-UK maps is available in Section 4.2.2. The study acknowledges the limitation inherent in the Fathom-UK flood maps, which do not account for the effect of buildings and infrastructure on flooding dynamics. These return periods associated to the flood risk likelihood were selected after comparing the trend in flood risk perception for different countries in GB (Table 2-2). Although the perception of flood risk differs depending on the country, Scotland's approach was considered the best fit scenario and adopted for the low, medium, and high flood likelihood. Only fluvial (flooding from rivers) flood extents taking in account flood defences were used for the purpose of this Chapter. The exposure of waste facilities to pluvial (surface water) and coastal flooding is left for future studies.

Table 2-2. Different perception of flood risk likelihood for Scotland, England, and Wales. The likelihoods selected for this research Thesis are 1 in 1,000-year for the low, 1 in 200-year for the medium, and 1 in 10-year for the high return period. Scotland: SEPA Flood Maps available at https://map.sepa.org.uk/floodmap/map.htm; England: Flood Map for Planning Risk (EA, 2018); Wales: Properties at Risk of Flooding (Stats Wales).

	Likelihood of flooding	Fluvial (year)	Pluvial (year)	Coastal (year)
Scotland	High	10	10	10
	Medium	200	200	200
	Low	1,000	1,000	1,000
England	High	>= 100	>= 100	> 200
	Medium	100 - 1,000	100 - 1,000	200 - 1,000
	Low	<1,000	<1,000	<1,000
Wales	High	>=30	>=30	>=30

	Medium	30 - 100	30 - 100	30 - 200
	Low	100 - 1,000	100 - 1,000	200 - 1,000

The selection of waste facilities in GB was determined by the coverage of the polygon's dataset, which represents an increase of number of sites compared to the 3 annual waste intake buffer methodology (2.2.2) but is still limited. Only 14.5% of waste facilities in GB have polygons associated to them. The simplified procedure to estimate buffers investigated the use of the average size (mean) of available polygons in GB, separated by category of waste facility and per country (Scotland, England and Wales). Because the size of polygons was found to be consistently bigger than the actual footprint, and the information about the annual waste intake was determined to not be suitable for the purpose of this study (Section 2.3.1), a maximum size threshold was adopted with the goal of filtering out polygons too big to realistically represent the footprints of waste facilities. The threshold was established by determining the maximum size of footprint among 50 random landfills located between Scotland and Wales, which are generally bigger than other category of waste facility. This maximum size of footprint identified was 375,000 m^2. Polygons above this threshold were removed from the dataset. For the remaining 1,049 sites (of the initial 1,054), the average area (mean) and the buffer radius $r = \sqrt{(A / \pi)}$, were assessed based on country of location and category of waste facilities.

The estimated buffers and polygons were then overlapped by flood extents in GIS environment to (1) estimate the correlation coefficient between the overlap (%) for polygons and buffers with flood extents, (2) assess the overlap difference by distributing values between best and worst (overlap) quintiles, and (3) from the worst quintile, understand cases where only the polygon or the buffer were overlapped with flood risk extents. Analysing the differences in behaviour between the estimated buffers and the polygons is key to evaluate the method's potential to recreate a valid approximation of footprints for waste facilities. Because the buffers are created per each category of waste facility, the same method could be adapted to other countries' cadastral parcel datasets, or even when polygons are not available by simply utilising the estimated buffers from this Chapter. Further analysis of polygon sizes by waste facility category in different countries would be beneficial to better understand variability and uncertainty when applying this approach more broadly.

2.2.4 Additional analysis on the size of polygons vs actual footprints

The visual inspection of the annual waste intake buffer methodology (Section 2.2.1) determined that the polygons were significantly bigger than the actual footprint of waste facilities, meaning that this methodology was always likely to overestimate facility sizes. To further analyse the difference between polygons and actual footprints, additional waste management facilities were visually inspected. The population considered was the total number of sites with polygons that received plastic waste in 2019 (3,681). To obtain a representative sample, the minimum number of polygons to manually check was established with the Cochran's formula (*Eq. 2-3*) and modified Cochran's formula (Eq. 2-4) (Cochran, 1977). The latter is appropriate for small datasets with known populations (in this case 3,681 sites). In Cochran's formula, the recommended sample size (n_0) is represented as:

$$n_0 = \frac{Z^2\sigma(1-\sigma)}{e^2} \qquad \textit{Eq. 2-3}$$

where Z is assumed to 1.96 for a confidence level of 95%, σ is the population standard deviation, and e is the confidence interval. In the modified Cochran's formula, the modified recommended sample size (n) is determined from Cochrane's sample size (n_0 in *Eq. 2-3*) as the ratio of n_0 to 3,681:

$$n = \frac{n_0}{1 + \frac{(n_0 - 1)}{N}} \qquad \textit{Eq. 2-4}$$

where N is the known population.

The standard deviation must be assumed as the dataset has not yet been created, a value of 0.5 is used, which is considered an acceptable standard deviation to ensure the determination of a sample size that represents the population (The Qualtrics XM Platform, 2022). A confidence level of 95% which correlates to a Z number of 1.96, and a confidence interval of 5% are used as the inputs, resulting in a minimum sample size of 348. Therefore, 348 polygons were checked manually, corrected to represent the actual footprint of sites (size and shape), and results were compared to the original polygons. The average size (km^2) and standard deviation were estimated for polygons and actual footprints. An average scaling (down) factor was established per facility category to decrease the difference in sizes between the two sets of data. The scaling factor represents a possible solution to increase the potential of buffers to represent the actual footprint of waste facilities.

2.3 Results and discussion

2.3.1 Suitability of annual waste intake buffer representation

To estimate buffer sizes per facility, spatial footprints of 154 waste management facilities in Scotland were visually estimated and compared to the polygons dataset. Annual intake in 2019 across all sites was approximately 4 million tonnes of waste (mean = 27,033 tonnes and st. dev. = 49,704). The mean footprint area was 23,394m^2 with sites ranging from 297m^2 to 500,000m^2. Footprint versus annual waste intake is shown in Figure 2-1. A sample of 154 manually inspected footprints of waste facilities is plotted showing the annual waste intake (log scale) on the x-axis versus the footprint size (m^2) on the y-axis (log scale) resulting in a moderate correlation value (0.377).

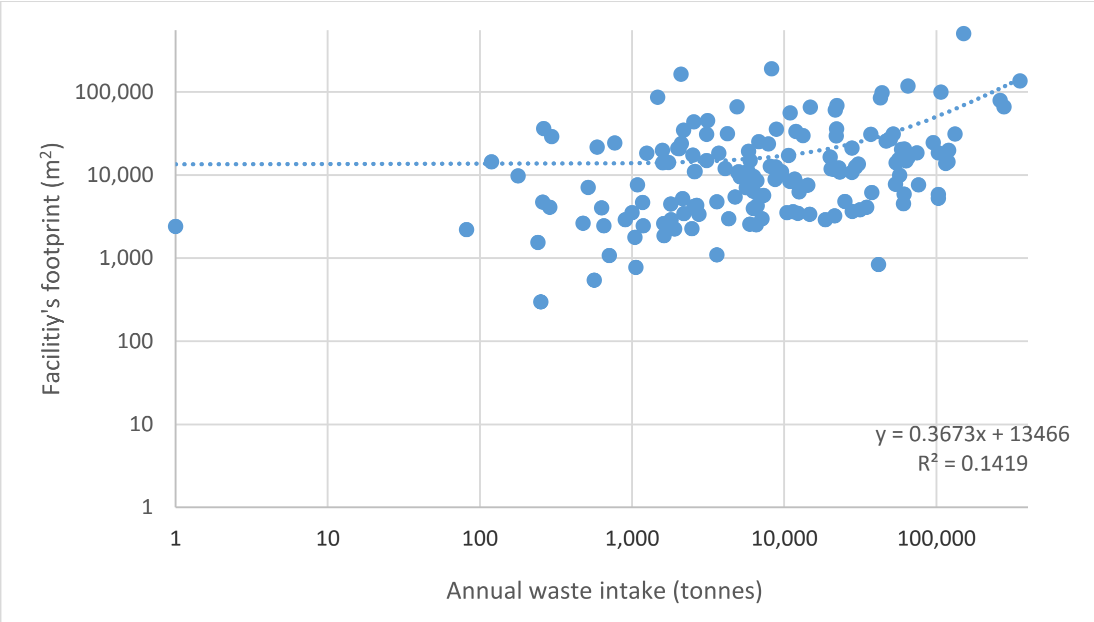


Figure 2-1. A sample of 154 manually inspected footprints of waste facilities is plotted showing the annual waste intake (log scale) on the x-axis versus the footprint size (m^2) on the y-axis (log scale) resulting in a moderate correlation value (0.377).

Although the trend line in Figure 2-1 is slightly positive, the R-squared value measuring the trend line reliability (0.141) is significantly far from the best fit (1). To further investigate the relationship between the two sets of data, the Pearson's correlation coefficient was estimated (Table 2-3). Results show higher degrees of correlation for the category of facilities with a fewer number of sites, while a lower degree of correlation applies to transfer stations that represent the highest number of sites among the sample of facilities analysed (107/154, 69%) (Table 2-2). The relatively large number of transfer stations leads to an overall low degree of correlation (0.377). The number of sites used as a sample for the total population may be too limited, particularly for incineration facilities and landfills. In addition, the extremely low degree of correlation resulting with the waste transfer stations (0.138), is highlighting the unpredictability of those facilities' size due to the many variables that can influence site footprint even within the same category of waste management facility. Tangible factors such as site location, population catchment scale, temporary storage typology, and capacity may have influence. Other intangible factors such site history may also have influence. Nevertheless, for the sake of fully developing the methodology, the buffer size was predicted from the annual waste intake based on Eq. 2-1 and Eq. 2-2.

Table 2-3. Estimation of the Pearson's correlation coefficient between the annual waste intake and footprint for selected waste facilities.

Waste category	Number of facilities	Pearson's correlation coefficient
Treatment centre	19	0.523
Transfer station	107	0.138
MRS (metal treatment)	18	0.788
Landfill	5	0.979
Incineration	5	0.915
Total	154	0.377

The buffer for each site was therefore estimated and visually compared to the actual footprint for each category of waste facilities (Figure 2-2), and for the total number of the selected sites (Figure 2-3). Although the projected buffers behaved similarly to footprints for the categories of treatment centres, metal recyclers, landfills, and incineration (Figure 2-2), they differ significantly when looking at categories with a higher number of facilities in them (i.e., transfer stations, as expected). Different behaviours are also evident in Figure 2-3 when looking at the overall comparison between footprints and predicted buffers (without the distinction of categories). Polygons appears significantly larger than the actual footprint of waste facilities and the population sample is too small, especially for some category of facilities. As a result, using the annual waste intake dataset to estimate buffer sizes was deemed infeasible even though there are some categories where the correlations seem reasonable. Even with refinements, this approach could never be used for all types of waste management facilities and therefore, it was not developed further. Instead, polygons were used to develop buffer sizes. The results of that approach are described in the following Sections.

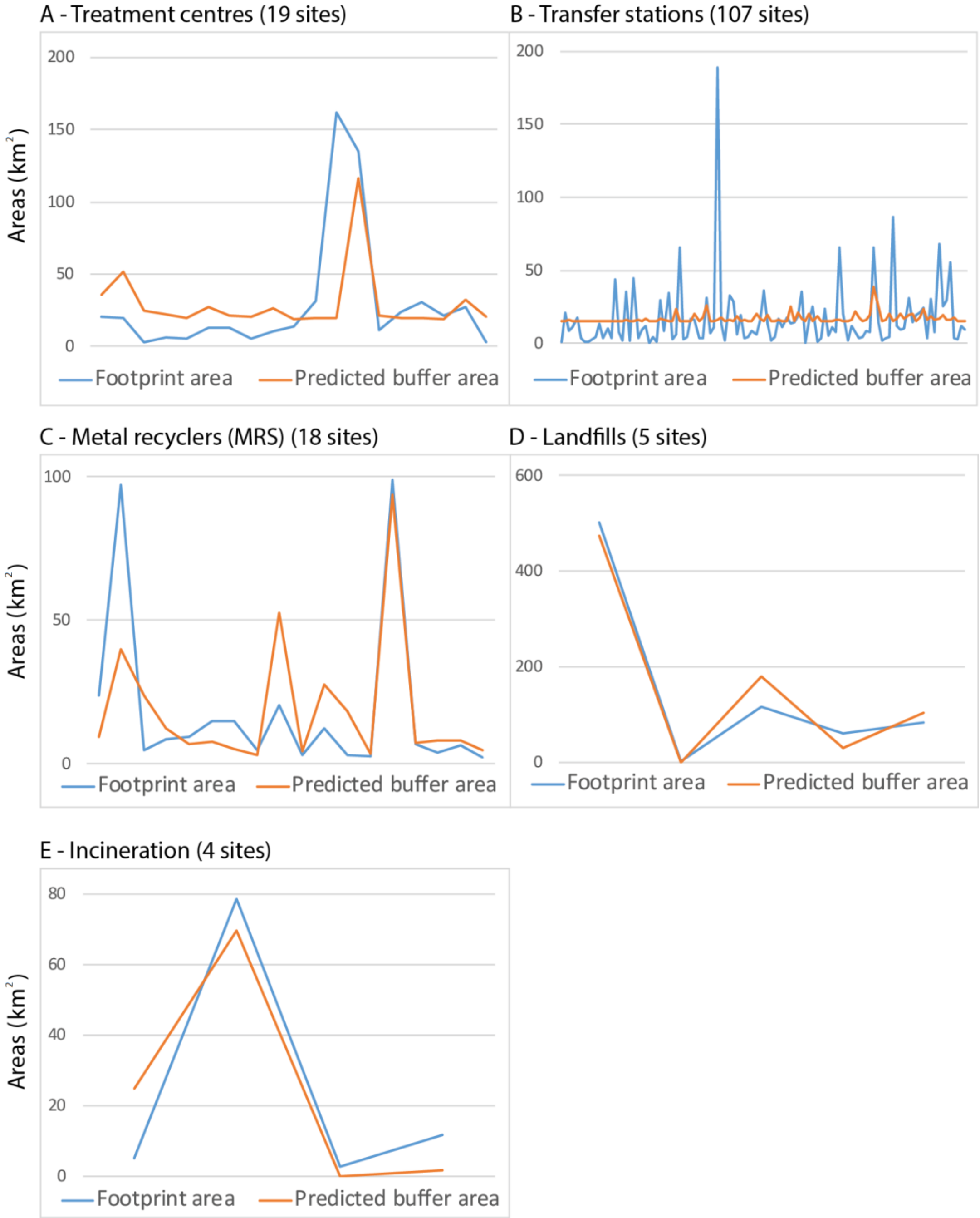


Figure 2-2. Comparison of areas between the real footprints and the predicted buffers estimated with Eq. 2-1 and Eq. 2-3 *for (A) waste treatment sites, (B) waste transfer sites, (C) metal treatment sites, (D) landfills, and (E) incinerators. The x-axis represents the number of facilities per category and the y-axis the area in km2.*

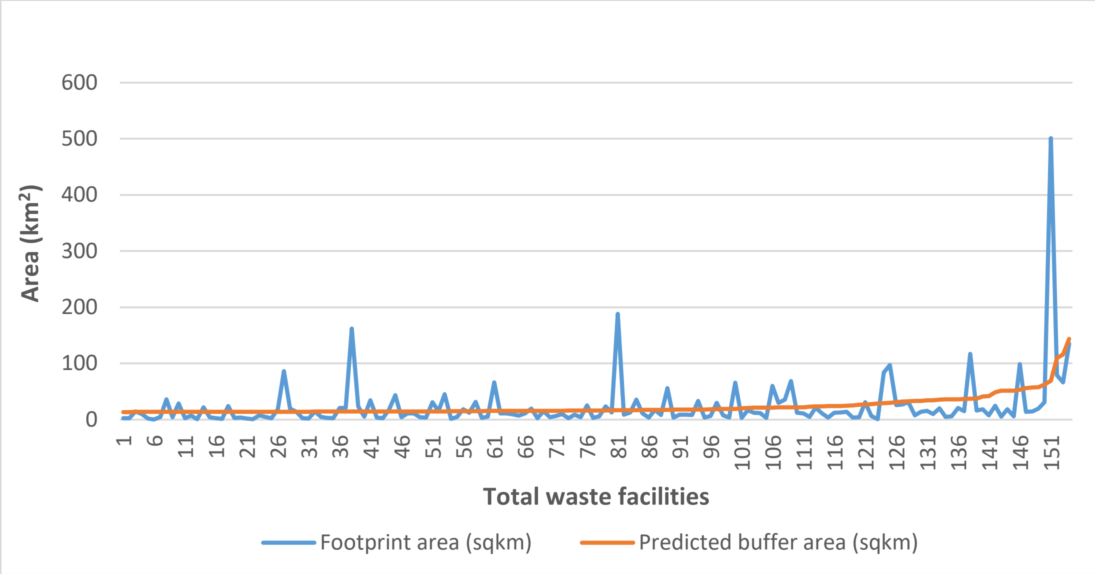


Figure 2-3. Comparison between real footprints area (in blue) and the predicted buffer values (in orange) estimated with Eq. 2-1 and Eq. 2-2 for 154 selected sites.

2.3.2 Results from the exposure of polygons to the risk of flooding

Buffers were estimated based on the polygon's datasets available for Scotland, England, and Wales, 1,049 of 7,292 (14%) of waste management facilities in the UK. The division per country and category (Table 2-4) highlights the differences based on waste facility location and, at the same time, increases the accuracy of the estimated buffers. Results show some similarities between Scotland and England where incinerators, transfer, and treatment stations are analogous in terms of average size despite the difference in number of facilities in the two countries. Facilities in Wales are generally smaller. This is particularly evident for landfills, which have an average size of approx. 31,000m^2 versus the 134,000m^2 in Scotland and 154,000m^2 in England. The difference highlights the importance in arranging similar assessments when applying the methodology to other countries outside the UK.

Table 2-4. Selected waste facilities (1,049) divided per country and category, average area (mean) and average buffer radius were estimated from the INSPIRE Index Polygons.

Country	Category description	Number of sites	Average area (m^2)	Buffer radius (m^2) per facility
Scotland (173)	Incineration	2	83,216	163
	Landfill	12	134,093	207
	Transfer Station	123	31,913	101
	Treatment	20	43,949	118
	MRS	16	31,206	100
	On/In Land	0	-	-
	Storage	0	-	-

England (826)	Incineration	10	86,505	166
	Landfill	18	153,561	221
	Transfer Station	316	31,387	100
	Treatment	262	58,430	136
	MRS	188	18,901	78
	On/In Land	19	116,005	192
	Storage	13	19,313	78
Wales (50)	Incineration	0	-	-
	Landfill	3	30,954	99
	Transfer Station	36	20,662	81
	Treatment	9	56,785	134
	MRS	1	17,640	75
	On/In Land	0	-	-
	Storage	1	42,083	116
Total		1,049		

Three flood risk return periods (1 in 1,000-year (low), 1 in 200-year (medium), and 1 in 10-year (high); described in Section 2.2.3) were overlapped with polygons and buffers for 1,049 facilities. Among these facilities, 228 sites (22%) had polygon and/or buffer impacted by high likelihood flood risk, 61 additional sites (289 sites total or 28%) impacted by medium likelihood flood risk, and, finally, 46 additional sites (335 sites or 32%) for the low likelihood flood risk. The increase in numbers reflects the proportional increase in flood map extents moving from high likelihood events with low intensity to extreme events with low likelihood but very high intensity.

The correlation coefficient values for the tested flood risk return periods show increasing correlation between the methodologies (polygons vs buffers) as flood likelihood decreases from high to low: the high likelihood has the weakest correlation (71%) followed by medium (80%) and then low (81%). A high level of correlation means that the proportion of variance in the estimated buffer overlap of areas (dependent variable) can be explained by the overlap of polygons (independent variable) data. As flood extent increases, the number of sites affected and the extent by which each site is impacted also increases, which is expected moving from the high flood likelihood (low impact) to the low likelihood (high impact). The extent of the flood impact on facilities (or severity of impact), can be observed by considering the 40% of overlap as a threshold. The vast majority of sites affected by the high likelihood are impacted below the 40% (91 and 90% of 228 sites for polygons and buffers, respectively). In the medium likelihood event, 70% polygons and 67% of buffers out of 289 sites are also below the 40% threshold, and data seems more evenly distributed around the correlations line. Finally, in the low flood likelihood, 67% of polygons and 65% of buffers result affected below the 40% out of 335 sites, confirming the trend of an increasing severity of impact obtained by lower flood likelihoods. In the high frequency low impact event, the buffer methodology seems more likely than the polygon method to show some inundation: 76 sites present only the buffer as affected (impact on polygons = 0%), compared to 49 sites with only polygons impacted. Similar results were estimated for the medium likelihood with 98 sites reporting only buffers affected by flood water vs 41 facilities with only polygons impacted. Finally, for the low likelihood, 89 sites show inundation only on buffers vs 39 facilities reporting only polygons as affected. The trend seems to show a predominance of buffers affected by flood waters compared to polygons.

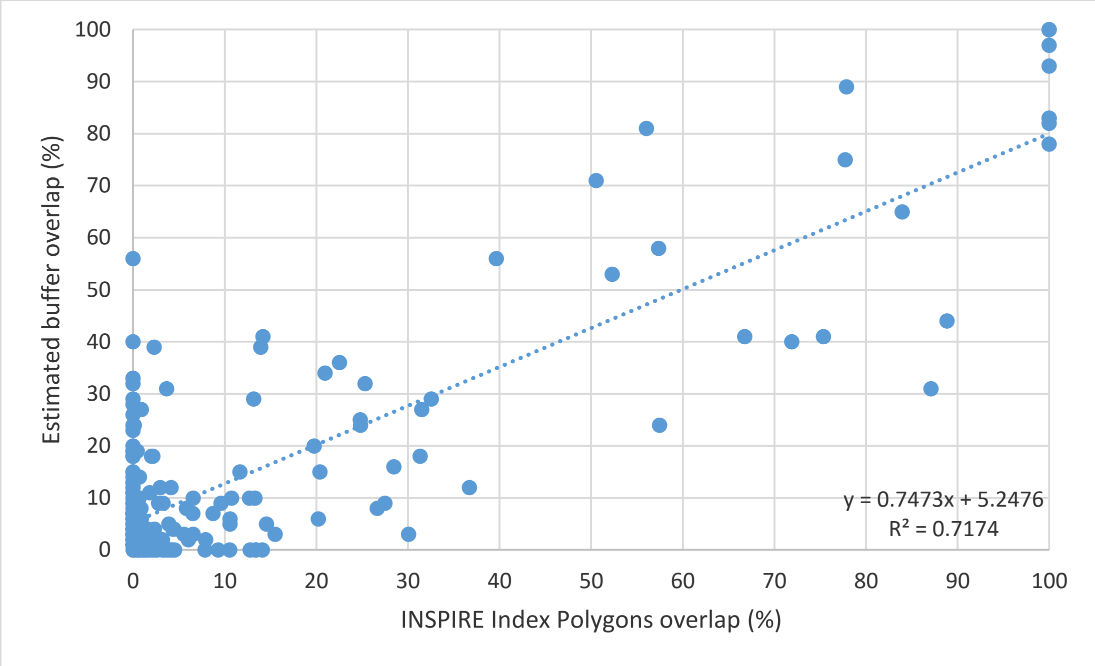


Figure 2-4. Polygons and estimated exposure of buffers to high likelihood flood extent (1 in 10-year flood) estimated by the overlap of areas (%).

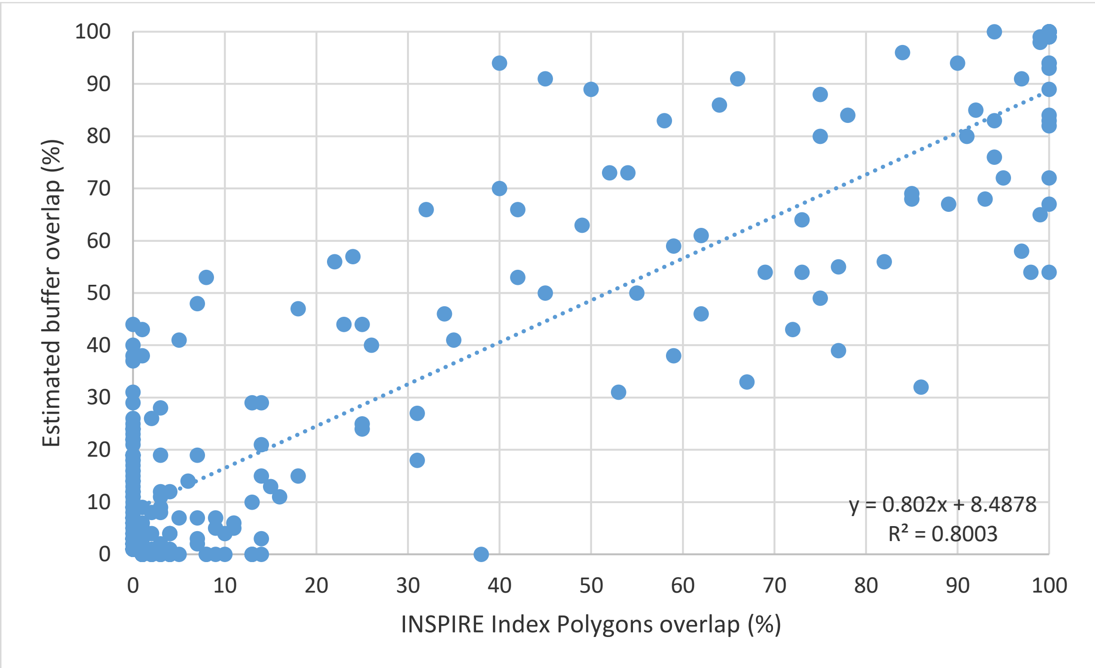


Figure 2-5. Polygons and estimated exposure of buffers to medium likelihood flood extent (1 in 200-year flood) estimated by the overlap of areas (%).

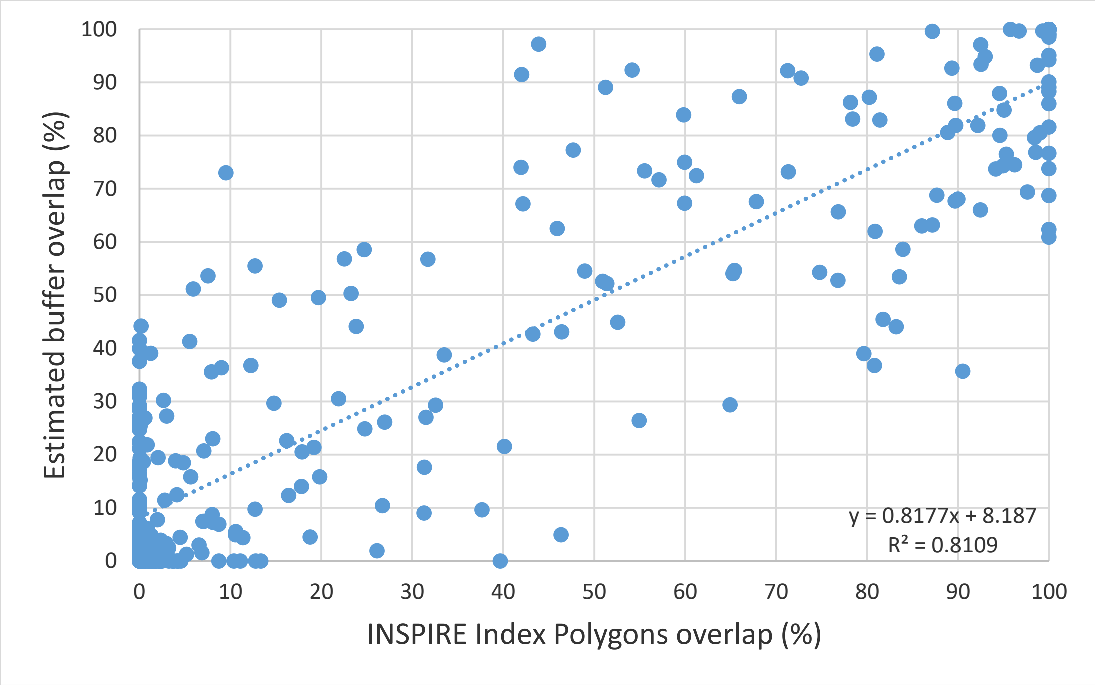


Figure 2-6. Polygons and estimated exposure of buffers to low likelihood flood extent (1 in 1,000-year flood) estimated by the overlap of areas (%).

Figure 2-7 and Figure 2-8 provide illustrative examples of consistent cases of flood exposure to site polygons and buffers. Despite the different exposure to the low flood likelihood, the overlap in Figure 2-7 with polygon and buffer has 1% of difference, which raises to 16% in Figure 2-8.

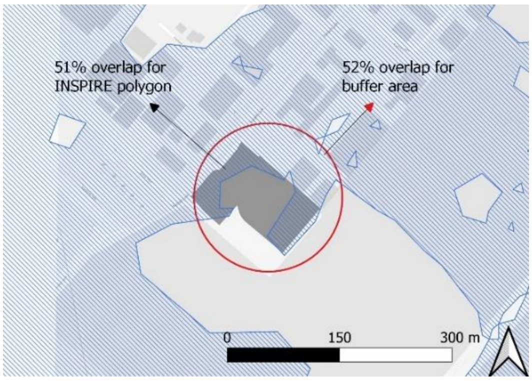


Figure 2-7. Example of transfer station overlapped by 1 in 1,000-year return period flood risk. The exposure of the Polygon (in grey) and the estimated buffer (in red) are very similar (51 and 52%).

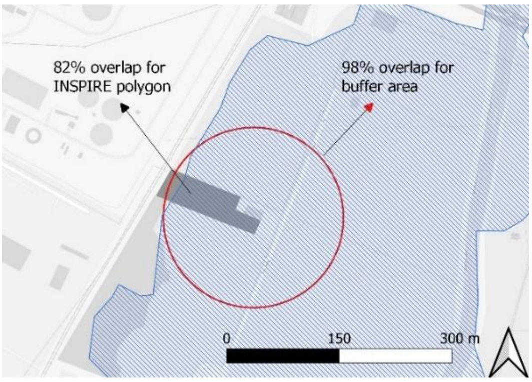


Figure 2-8. Example of a metal treatment facility overlapped by 1 in 1,000-year return period flood risk. Although the estimated buffer is visibly bigger than the Polygon, the exposure to flooding is similar (82 and 98%).

Overall, the three flood likelihoods show similar impacts using both polygons and buffers, showing that despite the differences in shape and size, the use of the mean polygons value to estimate circular buffer is appropriate to represent the footprint of waste facilities for the scope of this research Thesis. The resulting buffer sizes may be used to evaluate flood impacts to the other waste facilities for which polygon data are not available, as long as waste categories and countries are taken into account.

2.3.3 Factors affecting differences in flood inundation

To further analyse the applicability of the methodology and to better understand cases of exposure inconsistency, the overlap difference between polygons and estimated buffers was obtained for each facility (as absolute values) for each flood return period. The difference represents how shape and size of polygons and estimated buffers affect their interaction with flood maps. A low overlap difference means that the buffer is close to behave as the polygon when exposed to flooding. As mentioned before, in Figure 2-7 the overlap difference is 1% (52 vs 51%) which means that polygon and buffer have a very similar potentially flooded area. On the contrary, high value of difference values represent inconsistency in the behaviour of the estimated buffers in representing the actual footprint. The biggest difference was recorded as 64, 54 and 56%, for the low, medium and high flood likelihood, respectively.

Next, to further understand the distribution of the differences in overlap, values were divided in quintiles to examine the differences between ranges (Figure 2-9). Differences are small for the first and second quintiles, which represent 40% of the waste sites impacted by flooding. Waste facilities in these quintiles range from 0 to 4% differences in flood exposure between the polygons and buffers across low, medium, and high likelihood events. The difference increases slightly if looking at the 80% of the total of waste facilities, which show differences in flood exposure of between 0 and 22% between polygons and buffers. For the same percentage of sites (80%), the high likelihood flood shows the lowest difference in flood exposure (7 – 14%), followed by the medium and low likelihoods (9 – 21% and 10 – 22%, respectively).

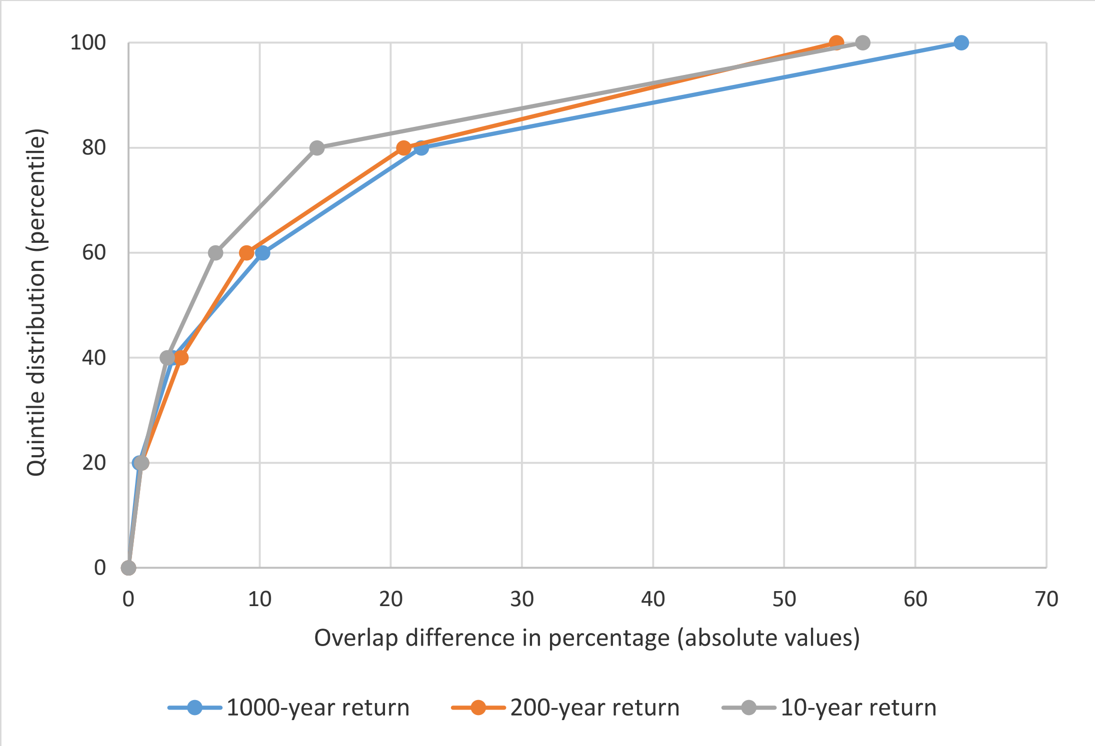


Figure 2-9. Quintile distribution of overlap difference (%) between polygons and estimated buffers for different flood likelihoods.

In the worst overlap scenario (5th quintile; 20% of facilities), polygons and estimated buffers have significantly different spatial relationships with flood maps with 14 – 56% difference in high likelihood events (mean = 21.54, st. dev. = 10.18), 21 – 54% in medium likelihood events (mean = 25, st. dev. = 7.87), and 22 – 64% in low likelihood events (mean = 32.89, st. dev. = 8.70). These differences may be determined by the shape and size of polygons, original location of the point coordinates used to generate the buffer, or a combination of both factors. The most important difference was found when, for the same facility, only the polygon or the buffer was impacted by flooding. Two such examples are illustrated in Figure 2-10 and Figure 2-11.

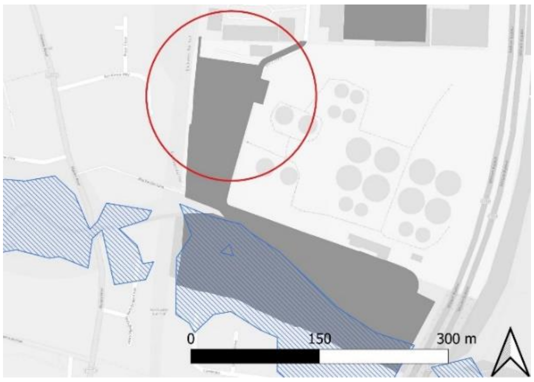


Figure 2-10. An example of exposure inconsistency: the polygon (in grey) is overlapped by the flood extent but the estimated buffer (in red) is not.

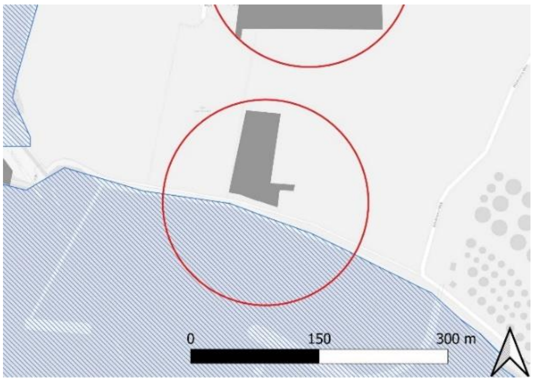


Figure 2-11. An example of exposure inconsistency: the estimated buffer (in red) is overlapped by the flood extent but the polygon (in grey) is not.

The most frequent difference between the polygon and buffer methods is illustrated in Figure 2-11: the buffer has some overlap with flood extent whereas the polygon has no overlap. The scenario in Figure 2-11 corresponds to the 39% of fifth quintile sites for the high flood likelihood, decreasing to 26% and 25% for the medium and low flood risk, respectively. In comparison, the scenario in Figure 2-10 was found for the 2.2, 3.5, and 1.5% for the high, medium and low likelihood, respectively. The extreme difference in flood exposure extents is due to complex polygon shape likely representing a multi-purpose site (Figure 2-10); poor location of point coordinate in the waste return datasets relative to the centre of the polygon (also Figure 2-10); and under/overestimation of the original site's footprint by the buffer. The prevalence of cases where only buffers are affected by flooding has highlighted a possible limitation of the methodology. Due to the circular shape of the buffer and the inclusion of permanent water in flood map extents, waste facilities situated near water bodies like wastewater treatment plants are inevitably more prone to being identified as impacted, even in the absence of an actual flooding event.

2.3.4 Final considerations from additional manually checked INSPIRE Index Polygons

Table 2-5 lists the mean footprint and polygon sizes by category for the 348 facilities (87 in Scotland, 216 in England, and 45 in Wales) that were identified manually, Including correcting in GIS when necessary. The overestimation presented by polygons is very strong, particularly for the incineration category. Polygons are at least 3.5-times bigger than footprints. Table 4 also reports the scaling down factor (average area for polygon/footprint) estimated per each waste category to include a possible way to reduce the error coming from the polygons. For example, to reduce the error in the buffers, the estimated value per category of waste facility can be multiplied by the scaling factor to be more in line with the actual footprint of facilities.

Table 2-5. Average area and standard deviation estimated for polygons and actual sites' footprint divided per waste category. The table includes the estimated scaling factor.

Waste site category	No. of sites	Polygon		Actual footprint		Scaling factor	
		mean (km^2)	σ	mean (km^2)	σ	Value	σ
Incineration	5	503.02	1,086.13	34.70	27.93	0.07	0.39
Landfill	18	1,148.17	2,406.49	422.93	862.71	0.37	0.66
MRS	39	69.85	223.45	26.03	42.75	0.37	0.52
Storage	4	16.42	18.81	16.42	18.81	1.0	0
Transfer	198	144.64	782.47	15.10	34.03	0.1	0.7
Treatment	84	245.69	865.77	25.25	37.70	0.1	0.63
Grand Total	348	216.23	939.10	40.16	213.81	0.25	3.5

Because the extended dataset of manually checked waste facilities' footprint was made available at a later date[1], the above results were not applied to the analysis carried out in Chapter 3 and 4. Information on the footprints manually checked compared to the estimated buffers is available in electronic format (Ponti and Endley, 2022). However, the outcomes highlight the presence of potential errors when using the polygon dataset to recreate the footprint of waste facilities, which lead to conditions most likely to overestimate the impact of flood on waste facilities. An attenuating aspect can be found when considering that the coastal flooding and the forecast for future flood risk due to climate change were not available at the time of writing. Therefore, although the use of the scaling factor estimated in Table 2-5 is recommended for further studies, for the scope of this research Thesis, the overestimation of the number of facilities exposed to flooding is likely to be counterbalanced by the lack of availability of coastal flood map extent and future projections for all sources of flooding.

[1] The manual inspection of footprints for waste facilities in GB was conducted in GIS environment by Stanley Endley, a MSc Hydrogeology student at the university of Strathclyde.

2.4 Conclusions

Three methodologies were introduced and evaluated to recreate the footprints of waste management facilities in GB. The INSPIRE Index Polygons spatial dataset showed a limited coverage for waste facilities (14%), and the physical areas included in the polygons were often not pertinent to waste management activities, causing an over representation of the actual size of footprints. Therefore, two buffer methodologies were investigated.

The first approach looked into establishing reliable buffers based on a limited population sample of polygons (manually corrected to represent the actual footprint), and information on their annual waste intakes. Results were non-statistically significant. The second approach expanded the population sample, created buffers based on the averaging the area of polygons per category of waste facility and country of residence, and focused on the behaviour of buffers vs polygons during inundation. A total of 1,049 waste facilities were overlapped against the high, medium, and low flood likelihood extent. As flood likelihood decreases, the number of sites affected increases (22%, 28%, and 32% of 1,049 sites impacted by the high, medium and low likelihood, respectively). The same trend was reported by the severity of impact represented by the percentage of polygons and/or buffers overlapped by flood waters. Considering the 40% of overlap as a threshold, only 9% of polygons and 10% of buffers were above the threshold with the high flood likelihood, while numbers significantly grew for the medium likelihood with 30 and 33% of polygons and buffers affected, respectively, and 33% of polygons and 35% of buffers for the low likelihood.

Buffers are more likely to show inundation than polygons across the different flood likelihoods. This is particularly evident when considering facilities where only polygons or buffers are impacted by flood waters (exposure of buffer > 0% and polygon = 0% or vice versa). Lower flood likelihoods (medium and low) showed the exposure of buffers (with impact on polygons = 0%) between 2.3 and 2.4-times higher than polygons (with impact on buffers = 0%), compared to 1.5-times in the high likelihood. The predominance of inundation on buffers is also confirmed within the worst quintile of overlapping differences where an average of 30% of the total amount of sites impacted by each likelihood reported only the buffers as affected by flood waters. In comparison, polygons were the only shape impacted for an average of only 2.4% of sites per each likelihood. Overall, the difference in flood inundation between polygons and buffers is low: 80% of sites impacted by different likelihoods of flooding have a maximum of 22% of overlap difference. The number drops to a maximum of 4% of difference for the

40% of sites. The similar behaviour against inundation proved the methodology estimating the size of buffers by averaging the area of polygons per category of waste facility and country as fit to represent the physical area occupied by waste facilities for spatial analysis in GIS.

The difference between polygons and actual footprint of waste facilities was further investigated by comparing polygons with a bigger set of manually checked footprints. Polygons were found significantly larger and a scaling factor was introduced to reduce the error. Further studies should apply the scaling factor to estimated buffers per waste facility category and rerun the inundation analysis with different flood likelihoods to assess the improvement on results.

3 An index-based assessment to estimate the risk posed by waste management facilities when exposed to flooding

3.1 Introduction

In Section 1.2, the Water Risk Index (WRI) was introduced as the chosen methodology from existing literature to determine hazardous substance concentrations in flood-prone areas. The WRI was applied to waste management facilities to underscore the risks posed by disposal sites, which often receive less attention than other industrial locations. This approach utilised open-source datasets detailing the annual quantity and type of waste received by UK facilities. While the WRI offers an initial risk estimate based on waste quantity and type, it does not account for the spatial context of waste facilities or other defining risk variables.

Risk, as defined in the literature, comprises three core components: hazard, vulnerability, and exposure. Various studies have delved into these components (Kron, 2005; De León, 2006; Cardona et al., 2012; Cannon, 2006; Cutter and Finch, 2008). A hazard represents the likelihood of a specific event occurring in a particular location, vulnerability refers to the potential damage susceptibility of those receptors, and exposure denotes the receptors (e.g., people, economy, natural and built environment) likely to be affected. These components can be evaluated using weighted parameters and indicators.

Several multi-criteria methodologies with different parameters and indicators for spatial risk identification are present in the literature. This chapter draws from three critical studies chosen for their relevance. The DRASTIC groundwater vulnerability model (Linda *et al.*, 1987) uses a relative ranking scheme to generate a numerical value known as the DRASTIC index. This model assesses mainly hydrological and geological factors impacting aquifer pollution. The FIGUSED method (Nerantzis *et al.*, 2015) was created to highlight flood hazard zones in need of mitigation measures, considering parameters like flow accumulation and geology. Another study by Arrighi et al. (2018a) studied flood-prone areas, especially targeting specific waste management facilities and contaminated lands. In contrast to these methods, this chapter focuses on national-scale risk assessment. The selection of criteria and parameters, while inspired by the multi-index methods, is adjusted for dataset availability and feasibility of national-scale data processing.

This chapter seeks to fill the gap in national-scale risk assessments by integrating spatial context factors tailored for waste management facilities. These factors encompass elements from both physical geography, like landforms and water bodies, and human geography, including protected environmental zones and land use patterns. This Chapter aims to (1) define the hazards that could lead to contaminant release from waste facilities, such as flooding likelihood and magnitude, and (2) evaluate the vulnerability of receptors, like people and the environment, to pollution incidents. The underlying assumption is that flooding impacts on facilities in industrial zones might be less consequential than those near protected natural areas or in remote regions. In these areas, pollution events can be challenging to manage and might have lasting consequences.

Additionally, to determine when receptors are activated, a set of circular buffers was established to outline risk proximity. This proximity, or the physical distance between a potential harm source and receptors, is essential in determining interactions between potential pollution sources, flooding risk, possible receptors, and factors amplifying contamination release or pollution spread. The CDOIF guideline provides a framework for determining environmental risk tolerability, recommending a 10km screening radius, especially in cases of significant environmental accident potential. This radius may be extended in river contexts. Recent research by Arrighi et al. (2018a, 2018b) employed a 5km buffer to study pollutant dispersion from environmental hotspots. The discrepancy in buffer sizes highlights the importance of the study conducted in this Chapter, which investigates the suitable dimensions to address the proximity of risk.

The Chapter is structured as follows: Section 3.2 introduces the concept of risk and its main components (hazard, exposure, and vulnerability), including the key formulas to estimate the risk index per waste management facility. In Section 3.3, hazard is defined as the probability of contaminants being released from waste facilities as a result of flooding. This is determined by evaluating two criteria: the flood depth intervals and the debris factor. Exposure is reported in Section 3.4 and represents the selected threshold to establish if a facility is considered as flooded or not based on the extent and the depth of flooding. Vulnerability is investigated in Section 3.5 by analysing different groups of receptors and their potential to be adversely affected by an event of flooding. Vulnerability is assessed using a set of criteria with their respective attributes and values, which are comprehensively described in Section 3.5.1 and summarised in a concise list provided in Appendix A. The most significant results obtained by the application of the methods to waste management facilities in Great Britain (GB) are reported in Section 3.6, which is divided into sub-Sections. 3.6.1 shows the results gathered from the application of the vulnerability criteria (the same outcomes are further analysed through a sensitivity analysis 3.6.2), 3.6.3 reports the hazard and exposure outputs, and 3.6.4 outlines the overall risk index estimated for each waste facility and summarised per LA. The full list of results is available in electronic format (Ponti, 2023). Finally, Section 3.7 highlights the main conclusions.

3.2 The concept of risk as the combination of hazard, exposure and vulnerability, and its application within an index-based approach

The definition of "risk" given by the United Nations Office for Disaster Risk Reduction (UNISDR) is the *"probability of harmful consequences, or expected losses (…) resulting from interactions between natural or human-induced hazards and vulnerable conditions" (UNISDR, 2004)*. The concept of risk, extensively explored in the literature (Kron, 2005, De León, 2006, Cardona *et al.*, 2012, Cannon, 2006, Cutter and Finch, 2008), is based on three components: hazard, vulnerability and exposure. Hazard is the probability of occurrence of a certain event (i.e., the physical event, phenomenon or human activity) in a specific location, vulnerability is the susceptibility of those receptors to damage, and exposure represents the receptors (such as people, the economy, built and natural environment) prone to be affected (Cannon, 2006, Cutter and Finch, 2008). The concept of risk (R) can therefore be determined as the product of hazard (H), vulnerability (V) and exposure (E):

$$R = HVE \qquad \textit{Eq. 3-1}$$

In this Chapter, the concept of risk (Eq. 3-1) is applied to assess the potential risk posed by waste facilities in terms of contaminant release during flooding. This estimation considers the severity of the hazard (flood depth and debris factor), the vulnerability of disposal sites to flooding (facility vulnerability), and the vulnerability of receptors to pollution (environmental and social vulnerability). To determine if facilities are considered flooded, an exposure threshold based on flood depth for different flood likelihood and sources is introduced. Additionally, Figure 3-1 explains the terminology used, including variables, sub-variables, and criteria, which are described by attributes such as permeability levels (very low, low, moderate, high, and very high).

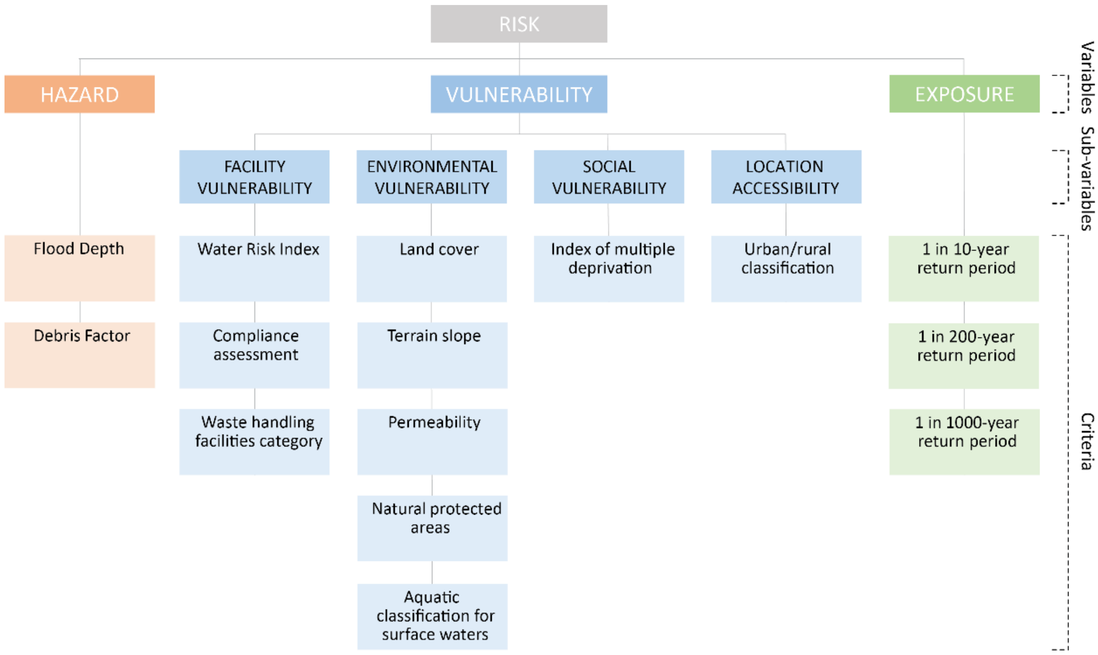


Figure 3-1. Risk diagram representing the concept of risk used in this Chapter. Hazard, vulnerability and exposure are the main variables, the vulnerability is divided into sub-variables (facility, environmental, social vulnerability, and the location accessibility), and criteria (e.g., land cover) that are describing the variables/sub-variables.

The selection of criteria and their attributes to define variables is influenced by the datasets available in GB (the data for Northern Ireland does not meet the requirements for this analysis). The approach builds upon previous approaches such as the identification of environmental hotspots for selected waste management facilities (Arrighi *et al.*, 2018a), the assessment of flood hazard areas at a regional scale using an index-based approach (FIGUSED) (Nerantzis *et al.*, 2015), and the DRASTIC groundwater vulnerability model (Linda *et al.*, 1987). The same previous studies are used as a reference also in determining weights and values for criteria and attributes, which are necessary to estimate the risk per each waste management facility. Weights and values are assigned based on the degree of importance. For example, within the environmental vulnerability, the natural protected areas criterion has a higher relevance compared to terrain slope, therefore the weights associated are 2 and 1 respectively. The assumption is that the contamination of natural areas may have a bigger impact on the overall environmental vulnerability compared to the potential risk of underground water pollution that can be facilitated or not by the degree of slope of the terrain. The Risk Index (RI), which represents the threat posed by waste facilities to people and the environment, can then be estimated for each facility (y) based on the following:

$$HI_y \ and \ VI_y = \sum_{z=1}^{N} (W_z V_z) \qquad \textit{Eq. 3-2}$$

$$RI_y = HI_y VI_y E_y \qquad \textit{Eq. 3-3}$$

where (HI) indicates the Hazard Index, (VI) the Vulnerability Index, (N) represents the total number of parameters within the variable or sub-variable, (Wz) the weight of the criterion, and (Vz) the numerical value associated to the attribute. Finally, (E) is the Exposure. To give an example, to calculate the hazard index variable for each waste facility (HI_y), the value of the attributes based on flood depth and debris factor (between 1 and 5) is multiplied by the weight of each criterion (flood depth = 2, debris factor = 1), and then summarised between criteria (Eq. 3-2). The final RI per disposal site (RI_y) is given by the multiplication of the variables (Eq. 3-3).

3.2.1 The spatial proximity of risk

The proximity, i.e., the physical distance between a potential source of harm and receptors, is a key factor to establish vulnerability thresholds. Because the location of waste facilities in GB indicates sites as point coordinates, a spatial proximity factor is investigated to establish relationships between facilities and their spatial context. This is particularly important in the spatial GIS environment. The topic was firstly introduced and assessed in Chapter 2 to determine the footprint of waste management sites with the use of the buffer areas. In this Chapter, the same principle is applied to establish the interactions (overlap) between the potential source of pollution (disposal sites), the extent of the risk of flooding, the location of potential receptors (i.e., natural protected area, land cover), and the presence of factors that can increase the risk of contaminant release and/or the spreading of pollution (i.e., terrain slope, soil permeability).

The size of the buffer is key to determine which receptor is likely to be affected by an event of pollution. The Chemical and Downstream Oil Industries Forum (CDOIF) guideline on *"Environmental Risk Tolerability for COMAH Establishments"* provides a screening methodology to help operators and the competent authorities in identifying environmental risk tolerability (CDOIF, 2010). For risks having Major Accident to the Environment (MATTE) potential, the guideline states that is reasonable to screen within 10km of the establishment, which may be for longer distances in case of rivers. More recently, Arrighi *et al.* (2018a) and (2018b) adopted a 5km buffer from each facility when looking at the potential spread of pollutants stored in environmental hotspots such as wastewater treatment plants, waste handling facilities, and contaminated sites. In this research Thesis, because waste facilities aren't under the COMAH legislation, a buffer distance of 5km is adopted for the majority of the criteria with some exceptions. The full list of the selected buffer areas and the reason for the chosen size are listed in Table 3-1. When the buffer is applied in GIS environment to identify spatial interactions with receptors, if multiple values are present, the highest one is selected to simulate the worst-case scenario. Which means that, for example, when looking at the land cover within 1km from each waste facility, multiple typologies can be present but only the most vulnerable one is considered for the scope of this Chapter.

Table 3-1. In order to establish vulnerability indices, a list of buffer areas was employed for spatial analysis within a GIS environment. These buffer areas were derived from the geographic coordinates of waste activities and were used to delineate the specific range that identifies the receptors potentially exposed to risk.

Variables/sub-variables	Criterion	Buffer area	Reason for chosen size
Hazard	Flood depth	Footprint	Footprint area as established in Chapter 2 to simulate the direct impact of flooding on waste facilities
	Debris factor	Footprint	As above
Exposure	Flooded/not-flooded	Footprint	As above
Waste facility vulnerability	Water Risk Index	NA	
	Compliance assessment scheme	NA	
	Waste facilities categories	NA	
Environmental vulnerability	Land cover	1K	Due to the volume of data, the buffer area had to be decreased from 5 to 1 km due to the proximity of facilities and buffer overlapping
	Terrain slope	Footprint	The degree of terrain slope is calculated within the waste site footprint because it directly influences the severity of the impact of flooding on industrial installations
	Permeability	5K	Saturated hydraulic conductivity (Arrighi *et al.*, 2018a)
	Natural protected areas	1K	European Directive 92/43/CEE and 2009/47/CE on the conservation of natural habitats and of wild fauna and flora

	Aquatic classification	5K	The Safety Report Assessment Manual (SRAM) for Major Accidents instalments indicates as reasonable to screen within 10km of the establishment, which may be for longer distances in case of linear pathways (rivers). Because of the type of installations considered and the datasets availability reporting only the coordinates of water samples undertaken, a distance of 5km was chosen to best represent real conditions
Social vulnerability	Index of multiple deprivation	5K	The same approach as per the aquatic classification was applied
Accessibility	Rural/urban classification	NA	The geographic coordinates for points indicating waste facilities are used

3.3 Hazard

In the context of this Chapter, hazard is intended as the severity of impact of flooding on waste management facilities that can cause a chemical reaction, pressure and other forces that can affect storage tanks and/or the structural integrity of containment systems and generate spills. The latter can be released into the environment, including deposition, or presence in standing and or moving waters. Polluted flood water can then reach other facilities, residential communities, natural protected areas, etc., and become a hazard on its own. For the scope of this Chapter, the level of risk posed by flooding (hazard) is given by the identification of flood depth intervals representing the danger posed by flooding to disposal sites, and the consideration of the debris factor that is responsible for increasing the severity of impacts.

The assessment of depth-damage probability of flooding on above-ground tanks, pipelines, and complex industrial sites has been extensively investigated (Antonioni *et al.*, 2009, Landucci *et al.*, 2012, Landucci *et al.*, 2014, Cozzani *et al.*, 2014, Antonioni *et al.*, 2015). Existing studies and methodologies rely on accessible (or easily estimated) information regarding flooding conditions (depth, velocity, extension, duration), and the size, capacity, containment, and fluid levels of the tank. Such detailed data may be available if working at the local scale, especially when focussing on one industrial complex at the time. On the contrary, this Chapter is developing a methodology applicable at the national scale with minimum information (often freely available) required. In addition the complexity and variety of waste facilities makes it hard to find common aspects that can be used to establish flood depth-damage functions (such as the presence of tanks and containment systems that are mandatory in licenced industrial sites).

The challenges presented by waste facilities can be summarised in three points: (1) disposal sites have different systems for treating/storing waste that can involve disparate number/size/typology of tanks, and the storage of waste can be indoor/outdoor/underground, etc. The inconsistency makes it arduous to find a rule for flood impact assessment that applies to all sites; (2) in some facilities there are no physical barriers between flooding and discarded materials: metal recycling centres and treatment/storage facilities treat/temporarily store waste on paved surfaces outdoor, which are directly exposed to rain and surface flooding (Figure 3-2, Figure 3-3, Figure 3-4). The absence of a net separation between flooding and waste excludes the possibility of developing depth-damage relationships that are commonly used for building structures; and (3) the structure of landfills is entirely below ground that makes it an exception that should have tailored flood depth-damage functions. Although the topic has been raised by several authors (Hao *et al.*, 2008, Young *et al.*, 2004, Brand *et al.*, 2018), studies specifically addressing the consequences of flooded landfills are rare and no correlation has been established so far between flood depth and consequences on the pollution potential of landfills. Therefore, although this Chapter is introducing a methodology to be applied to waste facilities including landfills, specific tests and research studies should be delivered in the future to address a clear gap in the literature.

Figure 3-2. An example of waste-management service located next to the Clyde in Glasgow. Waste is stored in both in tanks without ceiling and directly on piles on the ground. Tanks can be lifted/moved/flipped by flooding and the containment can be dispersing in flood waters. Imagery ©2022 Google, Imagery ©2022 CNES / Airbus,

Getmapping plc, Infoterra Ltd & Bluesky, Maxar Technologies, The GeoInformation Group. Geographic coordinates: 55.870955, -4.340831.

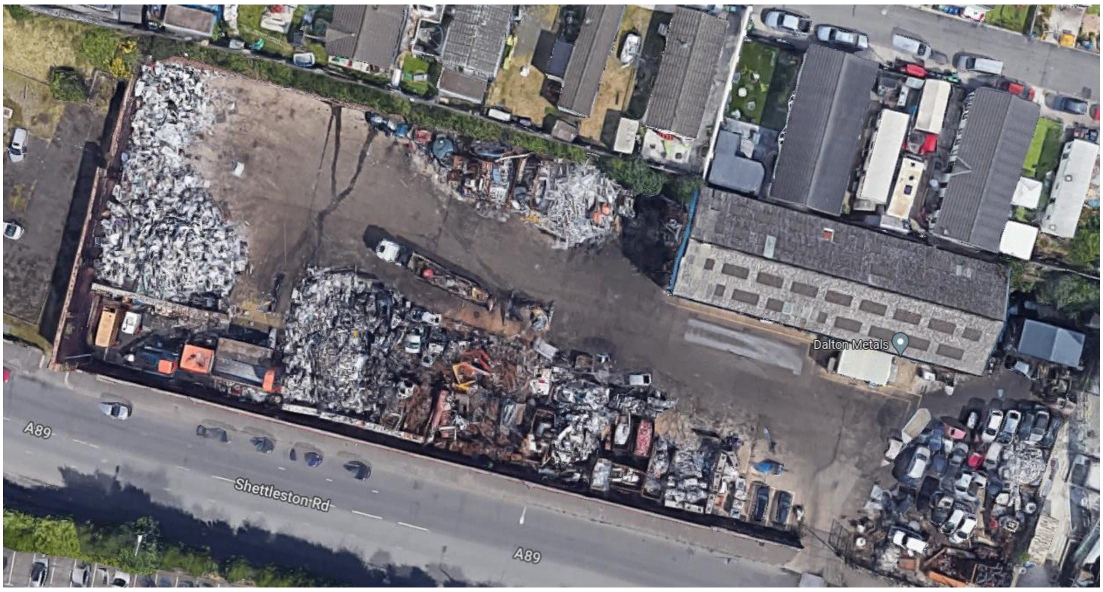


Figure 3-3. An example of a metal recycler facility. Metal waste is stored directly on the ground; no apparent containment systems are visible apart from a perimeter wall. Imagery ©2022 Google, Imagery ©2022 CNES / Airbus, Getmapping plc, Infoterra Ltd & Bluesky, Maxar Technologies, The GeoInformation Group. Geographic coordinates: 55.854481, -4.194083.

Figure 3-4. A 3D view of a recycling centre. On the left we can see a tank for liquid storage that could float/collapse under floods/debris impact and release the containment. The storing of waste is mostly directly on the ground, and the rain coverage on the right of the picture for selected types of waste won't protect against flooding. ©2022 Google, Imagery ©2022 CNES / Airbus, Getmapping plc, Infoterra

Ltd & Bluesky, Maxar Technologies, The GeoInformation Group. Geographic coordinates: 55.883620, -4.454454.

Because of the challenges listed above, reliable flood-depth damage functions could not be established, instead the flood depth intervals introduced by the Environmental Agency (EA) were implemented as a valuable tool for addressing flood risk. These intervals provide a standardized and consistent framework for assessing and categorizing flood risk based on depth measurements. In the national report on the risk of flooding from surface water (EA, 2019e), the flood depth intervals are identified based on the feedback from the Lead Local Flood Authorities (LLFAs), and span from <15cm to >120cm. Table 3-2 reports the flood water depth bands adopted in 2019 by EA and the relative likelihood of impacts. Although the bands are referring to residential properties, they indicate depth at which properties are likely to be flooded (from 15cm to 60cm) with structural damage likely to happen from >60cm. Flood waters reaching waste facilities can affect solid waste materials stored outside and inside buildings. The flooding of waste can accelerate existing degradation mechanisms such as mechanical stress, photo-oxidation and weathering processes (Golwala *et al.*, 2021), and mobilise smaller particles that could easily escape the boundaries of facilities with flood waters. Storage tanks for liquids can also be impacted by flooding (Antonioni *et al.*, 2015, Cozzani *et al.*, 2010), especially if small/medium in size: they can float/collapse under floods/debris impact and release the containment.

Table 3-2. Table adapted from "What is the Risk of Flooding from Surface Water map?", where the Environmental Agency defines flood depth intervals and likely consequences for residential properties (EA, 2019c).

<table>
<tr><th>Flood water depth bands selected in 2019 by the Environmental Agency (cm)</th><th>Likely impacts</th></tr>
<tr><td>< 15</td><td>Floodwaters are likely to be contained in any present surface water management systems such as kerbs and gullies</td></tr>
<tr><td>15 – 30</td><td>Floodwaters would typically exceed kerb height (standard kerb height is 125mm), likely exceed the level of a damp-proof course, and cause property flooding in some areas</td></tr>
<tr><td>30 – 60</td><td>Floodwaters are likely to cause property flooding</td></tr>
<tr><td>60 – 90</td><td>Property-level flood resilience measures are likely to be much less effective and structural damage is more likely to occur</td></tr>
<tr><td>90 – 120</td><td rowspan="2">Floodwaters is likely to exceed the maximum flood depth where property-level flood resilience measures are still effective</td></tr>
<tr><td>> 120</td></tr>
</table>

Because of the challenges presented by disposal sites and the national scale of the assessment described in this Chapter, EA flood water depth bands are considered appropriate to indicate the severity of flood impact for waste management facilities at the national scale, when minimum information is available. Future studies will have the task of improving the methodology to better target different categories of disposal sites. Weights and values are then associated to the flood depth criterion and its intervals to meet the hazard index estimation requirement (Eq. 3-2). Flood depth <15cm is considered without consequences (since that's equal to the height of kerbs), higher intervals have increasing values representing the potential risk posed by floods (i.e., flood depth band 15-30cm has value equal to 1, 30-60cm band has value 2, until >120cm with the highest value (5)).

To complete the hazard index estimation, the presence of debris in flood waters is also considered. Kelman and Spence (2004) identified debris actions as static, dynamic, and erosion. In addition because debris refer to solids in flood water, Kelman and Spence (2004) also included chemical, nuclear, and biological actions. Debris are therefore an important element that could increase the damage caused by flooding to the built environment and to people. A debris factor is extrapolated by the guidance on debris factors for different flood depths, velocities and dominant land uses provided by the *"Supplementary note to reconcile information provided in the 'Flood Risks to People Methodology' (FD2321/TR11) and the 'Framework and Guidance for Assessing and Managing Flood Risk for New Development' (FD2320/TR22) reports about the Flood Hazard Rating."* (EA and HR Wallingford, 2008). The debris factor associated to the dominant land use for waste facilities (urban land cover) is selected for different flood depths: <25cm (0), 25-75cm (1), and >75cm (1), and the values associated to them for the hazard index estimation are respectively 1, 2, and 2.

Flooding maps for GB (Fathom-UK flood maps) were given by SSBN UK Limited in raster format representing flood extent and water depth (a detailed description of Fathom-UK maps is reported in Section 4.2.2) (SSBN UK Limited, 2021a). The flood likelihoods manipulated for the scope of this Chapter are the low (1 in 1,000-year), medium (1 in 200-year), and high (1 in 10-year) fluvial (river) and pluvial (surface water) flood risk likelihoods (SSBN UK Limited, 2021). Further information on the flood risk likelihood chosen is available in Section 2.2.3. In GIS the raster files representing various flood likelihoods and sources underwent reclassification, a process that modifies the values of the raster pixels. This was achieved by assigning new values derived from water depth intervals and the debris factor. Subsequently, the reclassified raster files were converted into vector files and spatially joined with buffers (representing the waste facility footprints), allowing for the determination of water depth and debris factor associated with each disposal site. In case of an overlap between buffer and different depth intervals or debris values for the same flood likelihood, only the higher value was considered to reflect the worst-case of impact scenario.

3.4 Exposure

The exposure to flooding is given by the extent and the depth of flooding estimated for low, medium and high risk for fluvial and pluvial flooding. Waste facilities are considered flooded when two conditions are present at the same time: the extent of flooding intercepts the facility footprint (as established in Chapter 2), and the depth of water is higher than 15 cm that is the threshold (kerb height) normally used to distinguish flooded versus non flooded property (EA, 2019e). Therefore, to estimate the exposure index, the value of 1 is associated to water depth higher than 15 cm, while 0 is given to values from 0 to 15 cm to exclude facilities that are not at risk of flooding.

3.5 Vulnerability

The analysis of vulnerability accounts for four sub-variables:

(1) Vulnerability of disposal sites to flooding assesses the impact of flooding on disposal sites and the risk generated by the potential release of contaminants in flood waters. To estimate the level of risk per waste facility, three criteria are considered: (i) the amount and type of waste handled (estimated with the WRI); (ii) the level of compliance of sites against their environmental licence, and (iii) the category of waste handling facility that is affecting the waste treatment/storing capacity.

(2) Environmental vulnerability involves identifying receptors that are particularly susceptible to pollution events and evaluating specific geophysical factors that can influence the severity of the impact. Five criteria were selected in this Chapter (land cover, terrain slope, permeability, natural protected areas, and aquatic classification for surface waters) based on three key contributions from the literature and accessible open data for GB. These contributions include the DRASTIC system for groundwater pollution potential assessment (Linda *et al.*, 1987), the FIGUSED-S method for defining flood hazard areas (Nerantzis *et al.*, 2015), and an Italian study recognizing waste management facilities as environmental hotspots (Arrighi *et al.*, 2018a).

(3) Vulnerability to people refers to the ability of communities to handle and adapt to disruptive events. Flooding events can have diverse impacts on the population, which vary in space and time. Direct impacts, which result from the physical interaction between flooding and people, buildings, and cultural heritage, are relatively easier to identify. However, there are also indirect consequences that are more challenging to determine and quantify, such as the disruption of natural areas like parks and protected areas, as well as long-term remediation costs, particularly when floodwaters carry contaminants and debris (Hammond *et al.*, 2013, Bubeck *et al.*, 2017).

(4) The positioning of waste management facilities can impact the effectiveness of emergency responders in assisting the population during flooding events and in containing or minimizing potential spills of hazardous waste caused by flooding. The location of these facilities, whether in urban or rural areas, can either increase or decrease the vulnerability of residents involved, presenting various challenges for communities, local governments, and emergency preparedness and response. Some of the key challenges include the geography of remote areas, access to healthcare, infrastructure and communication issues, limited resources such as equipment and supplies, insufficient training for preparedness and response, and limited staffing for emergency responders and healthcare personnel (Federal Office of Rural Health Policy, 2022).

The criteria and attributes selected for each sub-variable are extensively discussed in Section 3.5.1. Detailed information about the sources of the datasets used can be found in Appendix A. Additionally, an example of spatial representation of attributes and their corresponding values for a selected set of criteria is provided in Figure 3-5.

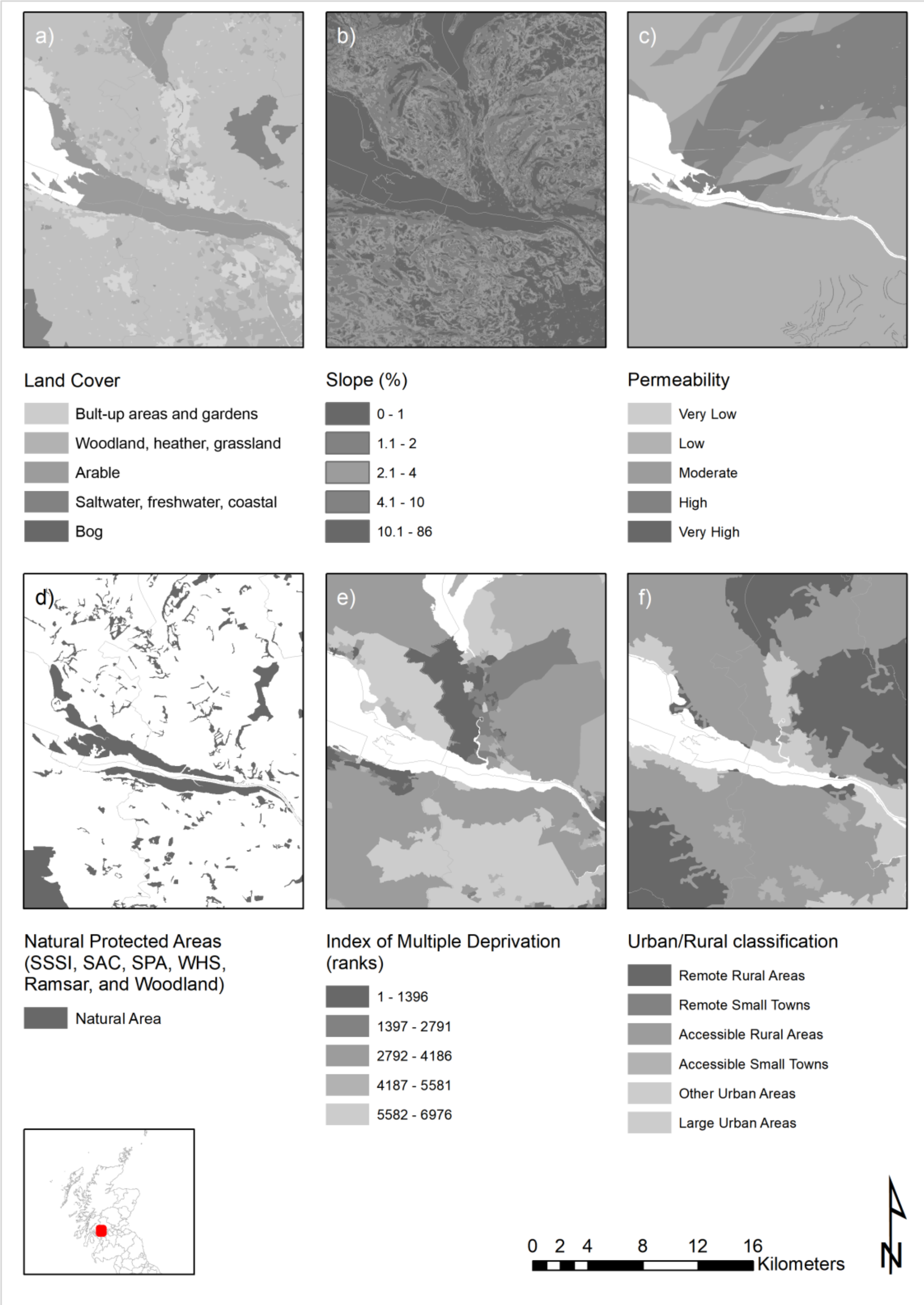


Figure 3-5. An example of spatial extent representation of attributes and values associated to a selection of criteria. The values span from 1 to 5, where 1 is assigned to the attribute that is less likely to impact on the overall criteria (lightest shade of grey),

and 5 is given to attributes that are likely to influence the overall results (darkest shade of grey).

3.5.1 Criteria and attributes: a comprehensive description

(1) The Water Risk Index (WRI)

The WRI, introduced in Chapter 1, was established as a methodology to define a priority among industrial activities located in the Elbe River basin in danger of releasing hazardous substances in case of flooding by the International Commission for the Protection of the Elbe River (ICPE) in 1995. It has since been applied more widely. The WRI is the result of two factors: (1) the classification of hazardous substances based on their reaction when in contact with water; and (2) data about the typology and quantity of hazardous substances present in each facility. The two factors represent the accumulation of risk (hotspots) in specific locations that, in case of flooding, may cause major pollution release. Once the hotspots are identified, actions can be taken to mitigate the risk. In 2001 the same methodology was adopted by the German Ordinance on Facilities Handling Substances that are Hazardous to Water (AwSV), which is based on the substance's chemical properties, published in April 2017 and coming into force on the 1st of August 2017 in Germany.

The WRI methodology was selected for this study to highlight areas where the concentration of hazardous substances, within waste management facilities, is particularly high and therefore in need of mitigation to limit potential pollution release during flood events. The WRI, originally designed for chemical substances, was therefore adapted to the hazardous properties within the waste classification system (Section 1.2), and subsequently applied to the quantity of waste and typology received annually per waste facility.

(2) The licence compliance assessment

Licence compliance assessment datasets are designed to evaluate the extent to which industries comply with their environmental licences. These datasets vary across different countries in the UK, but the overarching goal is to minimize potential harm to the environment resulting from industrial activities over time. These schemes serve as tools to monitor and ensure that industries adhere to the conditions and regulations outlined in their environmental licences. By assessing compliance levels, these datasets contribute to environmental protection efforts and promote sustainable practices within the industrial sector. The dataset for Scotland was obtained from the Compliance Assessment Scheme (CAS) Section on SEPA website (SEPA, 2019a), while the dataset for England was obtained from the gov.uk website as *"Waste operations and installations: assessing and scoring environmental permit compliance."* (EA, 2019b). No openly available dataset was found for Wales, although there is a Compliance Classification Scheme (CCS) in place.

For the scope of this Chapter the compliance licence was added as a criterion to assess the facility vulnerability to help identifying the operators and facilities that pose a higher risk to the environment. The methods behind the datasets for Scotland and England are different to each other and the datasets are therefore treated separately. SEPA's Compliance Assessment Scheme (CAS) rates an operator's environmental performance against its licence conditions, though no information is available on the method used to assign ranks. Comparatively, England's Compliance Rating Dataset is based on the Compliance Classification Scheme (CCS) to classify permit breaches. Non-compliances are identified and recorded in the course of a calendar year. The information is used to work out a compliance rating based on a points system. Permit breaches are converted into a points system by adding the points from each breach to calculate an annual total of non-compliance points (Table 3-3).

Table 3-3. Compliance Classification Scheme (CCS) applied in England to licenced waste management activities.

CCS Category	Description	Points
breach 1	most serious	60
breach 2	serious	31
breach 3	less serious	4
breach 4	minor	0.1
Band	Cumulative points (annual)	Percentage of change on subsistence charge
A	0	Discount of 5%
B	0.1-10	No impact
C	10.1-30	10% increase
D	30.1-60	25% increase
E	60.1-149.9	50% increase
F	>150	200% increase

The total number of waste facilities investigated in 2019 for Scotland and England was ~5,000 out of ~7,000 sites in total, ~4,500 in England and ~500 in Scotland. Table 3-4 shows the distribution of waste facilities that were investigated in 2019 in respect to the Scottish and English scoring legislation and the number of facilities per each scoring level. A good scoring (i.e., excellent/good or band A, B) may represent a business run with care, with maintenance performed regularly and therefore less likely to generate incidents that could affect the environment and people. On the contrary, sites at risk/poor/very poor and band D, E, and F are more likely to cause an environmental accident.

Table 3-4. Compliance to licence for waste management facilities in England and Wales (2019).

Compliance to licence in Scotland	Compliance to licence in England	Number of waste facilities
Excellent/Good	Band A,B	4500
Broadly Compliant	Band C	380
At risk	Band D	115
Poor	Band E	108
Very Poor	Band F	29

(3) Waste handling facility categories

The openly available datasets containing information on the annual waste received by waste facilities in Scotland, England, and Wales in 2019 were analysed (EA, 2019d, EA, 2019a, Natural Resources Wales, 2019). From each dataset, operational facilities were selected, and main categories were established to ensure data harmonization. Table 3-5 reports the main facility category on the left and the sub-categories on the right. The dataset for Scotland can report multiple categories on the same site that can be attributed to the presence of different type of licences, in this case the first category reported is utilised.

Table 3-5. Name and description of categories for the waste management facilities considered in the study based on the classification included in the dataset "Waste Permit Returns Data Interrogator 2019" for England and Wales available at https://data.gov.uk/datase.

Category of waste facility	Sub-categories
Incineration	Animal by-products incinerator; biomass; clinical waste incinerator; co-incinerator; co-incinerator (hazardous); EFW incinerator; hazardous waste incinerator; municipal waste incinerator; pet crematorium
Landfill	Hazardous merchant LF; hazardous restricted LF; inert LF; non-hazardous (SNRHW) LF; non-hazardous LF; restricted LF
Transfer	Competent authority (CA) site; clinical waste transfer; hazardous waste transfer; inert waste transfer; non-hazardous waste transfer
Treatment	Anaerobic digestion; biological treatment; chemical treatment; clinical waste treatment; composting; hazardous waste treatment; material recycling facility; mechanical biological treatment; non-hazardous waste treatment; physical treatment; physical-chemical treatment; recovery of waste; WEEE treatment facility; animal and food waste; non-ferrous metal re-processing; paper and pulp re-processing; paper recycling
MRS (metal treatment)	Car breaker; metal recycling; vehicle depollution facility; ferrous metal re-processing; metal re-processing

On/In Land	Deposit of waste to land (recovery); lagoon; mining waste management (non- hazardous)
Storage	In-house storage; storage – A/D; storage – dredging; storage – incinerator; storage - metal reprocessing; storage – oils; temporary storage installation

(4) Land cover

Land cover maps represent spatial information on different types of physical coverage of the Earth's surface (e.g., forests, grasslands, croplands, lakes, wetlands). The vulnerability of different land cover types is based on the negative effects that can be triggered by a pollution accident. Natural protected areas are the most vulnerable to contamination but the concept applies also, for example, to the corruption of agricultural lands vs industrial areas, which can have different consequences if the production of food is compromised.

The Land Cover Map 2020 (UKCEH, 2020) was downloaded through the EDINA Digimap online service with a scale of 1: 250,000 and a resolution in between 10 and 25m. The shapefile comprises 21 UKCEH Land Cover Classes based on Biodiversity Broad Habitats (Morton *et al.*, 2021). Table 3-6 presents the association between the 21 UKCEH classes with the attribute values for the scope of this Chapter. The highest index value (5) is given to areas that, if negatively impacted, would represent an important ecological loss. This is the case of bog/wetland areas that are ecologically important because they absorb great amounts of precipitation. Peatland also play a key role in helping to mitigate the effects of climate change by storing carbon. Saltwater, Freshwater and Coastal classes, which can also be easily contaminated in case of pollution into surface water, are given a value of 4. In comparison, urbanised lands such as built-up areas and gardens have the lowest value (1) because a potential contamination would be easier to contain and remediate compared to other areas.

Table 3-6. Land cover (LC) 2020 classes identifier and description (Morton et al., 2021) with associated attribute values.

UKCEH Land Cover Class	LC Identifier	Attribute value
Deciduous woodland	1	2
Coniferous woodland	2	2
Arable	3	3
Improve grassland	4	2
Neutral grassland	5	2
Calcareous grassland	6	2
Acid grassland	7	2
Fen	8	2

UKCEH Land Cover Class	LC Identifier	Attribute value
Heather	9	2
Heather grassland	10	2
Bog	11	5
Inland rock	12	2
Saltwater	13	4
Freshwater	14	4
Supralittoral rock	15	4
Supralittoral sediment	16	4
Littoral rock	17	4
Littoral sediment	18	4
Saltmarsh	19	4
Urban	20	1
Suburban	21	1

Due to the size of the land cover dataset, the devolved nations across GB were considered individually. Where buffer areas overlapped different countries (for facilities located, for example, on the border between England and Wales), waste disposal sites were designated to the country where the actual facility (onsite action/building/centre point) is located. The surrounding land and buffer were cut in some situations, however, this was not considered to represent any issue with regards to risk-hazard assessment as only one LA, the one in which the facility is registered and primarily located, could legally enforce jurisdiction and would respond to any emergency.

(5) Terrain slope

The terrain slope is reported in degrees from zero (horizontal) to 90 (vertical). For the scope of this Chapter, the degree of slope has a double function: it is assumed as the indicator of the potential degree of infiltration/stagnation of contaminated water, and at the same time, it can impact on the velocity of flooding water. The latter, together with the flooding depth, can increase the damage to the built infrastructure with consequent loss of containment. Therefore, the values for the terrain slope intervals are the highest (5) at both ends of the scale: when the percentage of slope is very low (which means more chances for contaminated water to reach the groundwater) and when the percentage of slope is very high for influencing the severity of flood impact.

The slope raster was obtained from the Copernicus European Digital Elevation Model (EU-DEM) version 1.1, in GIS environment. The dataset is openly available in Geotiff 32 bits format. It is a contiguous dataset divided into 1,000 x 1,000 km tiles, at 25m resolution with vertical accuracy: +/- 7 meters RMSE. To be able to process the slope raster, which has a prohibited size if considered at the national scale, a mask was created in GIS representing the extension of the waste facilities' footprint areas in GB. The slope raster was then clipped with the use of the mask, reclassified by dividing pixel slope percentage into intervals, and finally transformed in polygons to perform a spatial join with the waste facilities dataset.

(6) Permeability

The permeability, also called hydraulic conductivity, is the capacity of a rock to transmit a fluid. In the context of this study, the level of permeability can impact on the potential contamination of groundwater because it represents the infiltration of surface water in the soil. Permeability is often used in studies of groundwater and in particular during investigations of pollution or aquifer contamination. The permeability dataset was downloaded from the EDINA Digimap online service, based on the 1: 50,000 Digital Geological Map of GB (DiGMapGB) (Geological Map Data BGS © UKRI 20(2022)). The dataset reports both the maximum permeability and minimum permeability indicating the range of flow rates likely to be encountered in the unsaturated zone for each rock unit and lithology combination. For this study, the values related to the minimum permeability were considered because representing the minimum, and in some cases more normal, bulk rate of vertical movement likely to be encountered (Geological Map Data BGS © UKRI 20(2022)). For associating values to the permeability ranges, five classes were identified from the worst to the best possible condition: very high (5), high (4), moderate (3), low (2), and very low (1) permeability.

(7) Natural protected areas

Natural protected areas are natural habitats of recognized importance at international level, they are also areas that could be the most affected and suffer long-term consequences in case of an event of contamination. Datasets were obtained from different sources (Table 3-7), additional information on the sources of datasets is available in Appendix A. Natural areas under different legislations (i.e., SSSI, SAC, etc.) were considered as equal in an event of pollution, and the waste facilities located within 1km from any protected area were marked with a value of 5 to signify the potential danger posed by disposal sites.

Table 3-7. Natural protected areas considered in this study.

Natural protected areas	
Dataset name	Description
Sites of Special Scientific Interest (SSSI)	Areas of land and water that best represent the national natural heritage
Special Areas of Conservation (SAC)	Areas designated by Scottish Ministers under the EC Habitats Directive
Ancient Woodland Inventory	Protected woodlands
Special Protection Areas (SPA)	Areas of the most important habitat for rare regularly occurring migratory birds within the European Union
World Heritage Site (WHS)	GIS spatial data for World Heritage Sites and their Buffer Zones, where existing, as inscribed by the World Heritage Committee of UNESCO
Wetlands of international importance (Ramsar)	Globally important wetland areas and may extend into the marine environment up to a depth of 2m.
National Nature Reserves (NNR)	NNRs contain examples of some of the most important natural and semi-natural terrestrial and coastal eco-systems in GB

(8) Aquatic classification

Water bodies with good chemical and ecological conditions have good self-recovery capabilities due to their natural undisturbed equilibrium (Rosgen, 2011). Datasets about the overall status of water bodies in GB has proven to be hard to harmonise. For Scotland the Water Classification Hub shows the status of the surface waters, ground waters and protected areas classified under the Water Framework Directive (WFD) scheme. The dataset *"Water Body / Protected Area General Information"* (containing the location of surface water sample points) was downloaded for the year 2020 (SEPA, 2020c) and joined with the *"Classification Data By Water Body",* containing information on water overall status, by the river unique ID number to obtain a dataset with sample points and overall status associated to each of them. The classification of the overall status (very low, low, moderate, high, very high) was adapted to a 3-fold interval to meet the dataset available for Wales (good, moderate, poor).

For England, the most recent river quality dataset (2020) was obtained from the Water Quality Archive which is part of the Defra Data Services Platform (EA, 2020). The Water Quality Archive provides data on water quality measurements carried out by the EA. Samples are taken from sampling points around the country and then analysed to measure aspects of the water quality or the environment. Unfortunately the classification system for England isn't based on a 3 or 5-fold system, instead the dataset is reporting if the sampling is compliant or not in relation to permits or general monitoring. Therefore, only non-compliant samples were assigned the value 5 while the rest was assigned the neutral value of 1. In Wales the dataset on Surface Water Transfer (SWT) waterbodies under the Water Framework Directive was unavailable for download due to corruption. Instead the river quality dataset for the year 2018 was obtained from the Natural Resources Wales website (Natural Resources Wales, 2018).

In Table 3-8, the number of samples conducted in 2020 in Great Britain is presented, categorized by overall status classification outcomes. The majority of samples from Scotland showed high/very high quality, while 24% resulted in very low/low quality. In Wales, nearly 50% of the samples were classified as moderate. Notably, in England, a significant number of samples (over 400,000) were reported in 2020 indicating non-compliance with permits or general monitoring requirements. Waste facilities located within 5km from a water sample were given the attribute value of 1 for high/good/compliant samples, 3 for moderate, and 5 for poor/bad/non-compliant water quality samples.

Table 3-8. Number of samples conducted in 2020 in GB categorized by overall status classification outcomes.

	Scotland	England	Wales
Overall status	288 (high/good)	226,821 (compliant)	82 (good)
	221 (moderate)	-	202 (moderate)
	162 (poor/bad)	415,103 (non-compliant)	99 (poor)
Total values	671	641,924	427

(9) Index of Multiple Deprivation

The impact of flooding on a community is influenced by the existing vulnerabilities of its members. Different communities and individuals within those communities possess varying capacities, knowledge, experiences, and barriers that affect their ability to reside in a flood-free area, obtain insurance coverage, prepare for and cope with floods. As a result vulnerability to floods is not evenly distributed across society. Socioeconomically disadvantaged individuals, the elderly, disabled individuals, and marginalized societal groups often face greater vulnerability to the consequences of flooding due to limited access to social, human, and financial resources needed to cope with such events (Bubeck *et al.*, 2017).

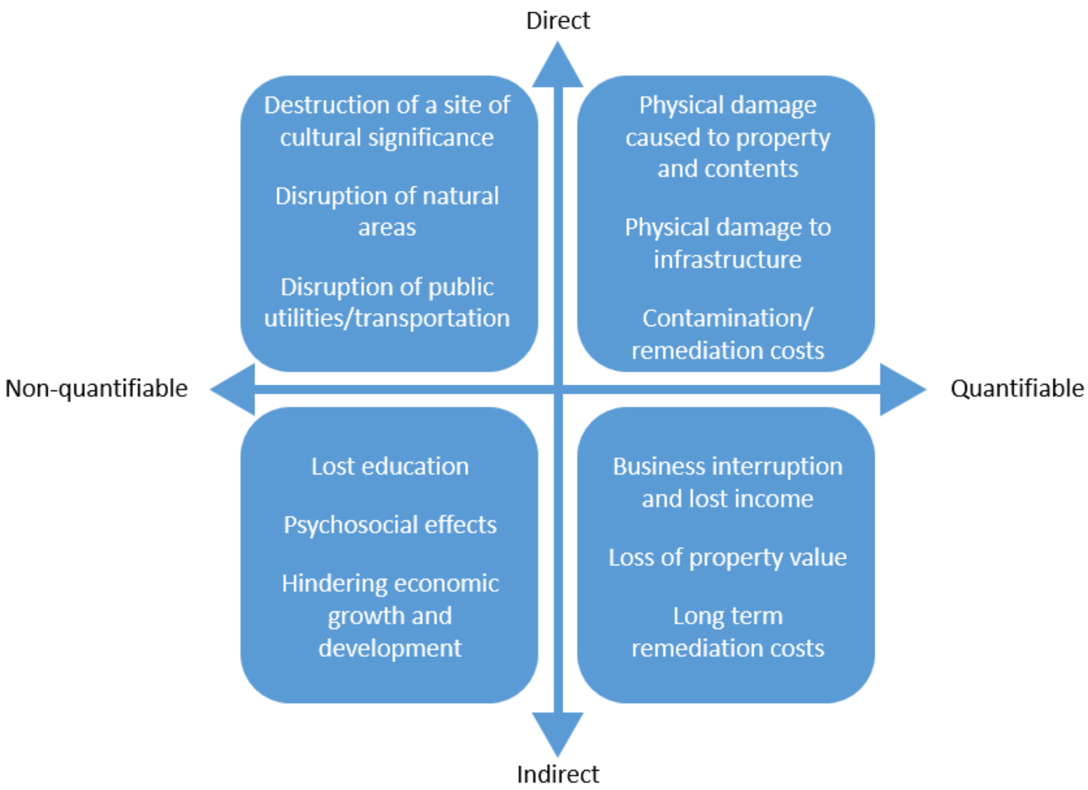


Figure 3-6. Adapted and expanded from Understanding Disaster Risk (UNDRR) available at https://preventionweb.net/ illustrates the potential consequences of flooding, distinguishing between direct and indirect impacts, as well as quantifiable and non-quantifiable aspects. Indirect consequences, such as the impact on economic growth and development of an area, are often more challenging to quantify and monitor/remediate compared to direct consequences like the loss in property value. These indirect consequences, although significant, may involve complex and multifaceted factors that make their assessment and mitigation more difficult.

To identify areas that can be more vulnerable than others if impacted by flooding this study is introducing the index of multiple deprivation, which is available for different countries in GB (Scottish Government, 2020b, English Government, 2019, Wales Government, 2019). Although the indexes are not directly comparable they were created with the same goal and can suggest, within each country, where people are experiencing disadvantages across different aspects of their lives. To give an idea of how the indexes were created the Scottish Index of Multiple Deprivation (SIMD), was estimated by splitting the population in small areas with similar population numbers (6,976 in 2020). Subsequently, 30 indicators of deprivation were identified and grouped into 7 domains (i.e., income, employment, education, etc.). The domains were finally combined into one index, ranking each data zone from 1 (most deprived) to 6,976 (least deprived). A similar process was performed for England and Wales.

For this Chapter, the Index of Multiple Deprivation quintile division was used to determine areas where the resident population is more vulnerable to flooding compared to others. The lowest quintile (no.1) represents the 20% of the population which is most vulnerable and therefore has the higher attribute value associated to it (5). The opposite applies to the quantile no.5 which represents the 20% of the population least deprived that would be best equipped to deal with and recover from and even of pollution (the value associated is 1).

(10) Rural/urban classification

The rural/urban classification in GB is provided by different countries and is not directly comparable, but for the scope of this study it can be used for determining if the disposal sites are located in urbanised or remote/rural areas within the country of residence. Datasets are openly available for Scotland, England and Wales (Scottish Government, 2020a, DEFRA, 2011, Welsh Government, 2011). In Scotland 77% of the country is defined as Remote Small Town or Remote Rural Areas, compared to Wales (78%) and England (64%). Significantly lower is the percentage of land classified as urbanised: Wales has the higher percentage (10%) of Urban Cities and Towns, followed by England with 6% and Scotland at 1% (Figure 3-7).

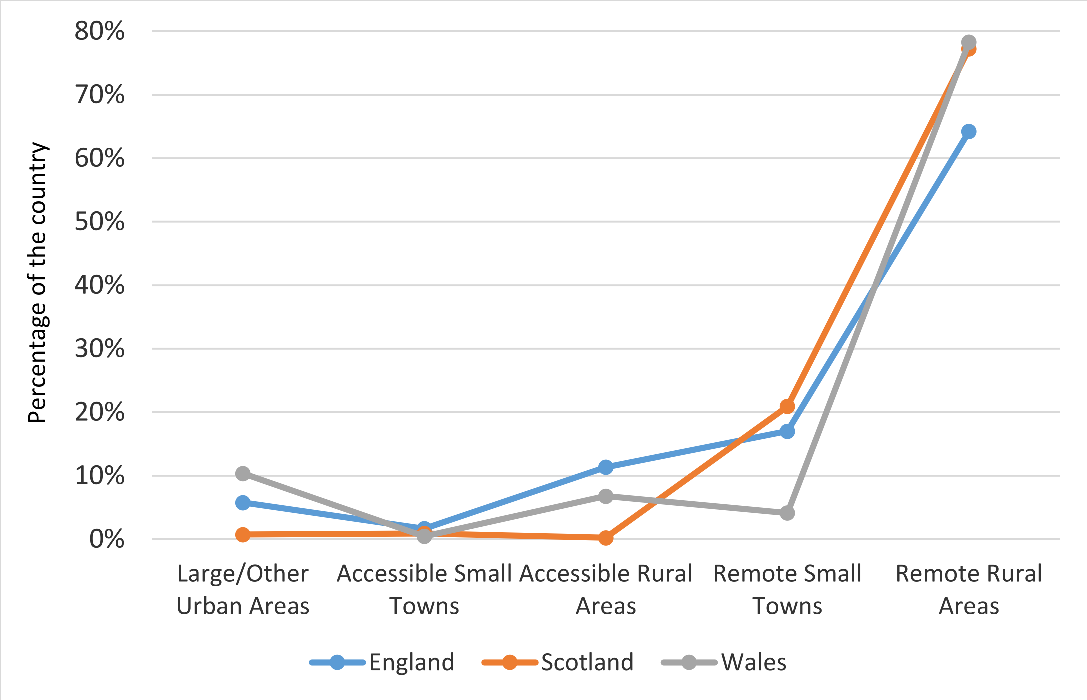


Figure 3-7. Urban/Rural classification for Scotland, England and Wales.

For the scope of this Chapter, the specific urban/rural classification for each country is organised from the most urban/major conurbation (least vulnerable) to the remote rural areas (most vulnerable in an event of pollution). Therefore, the values associated to the classification (from 1 to 5) start with the lowest (1) for to the least vulnerable areas and 5 to the most vulnerable ones.

3.5.2 Sensitivity Analysis

A sensitivity analysis is useful to acknowledge uncertainty, estimate the variability, and to identify the relative influence of each criterion on the overall results (Bartzas *et al.*, 2015). The single-parameter sensitivity analysis (Bartzas *et al.*, 2015, Babiker *et al.*, 2005, Arrighi *et al.*, 2018a) introduced in this study helps understanding the influence of different criteria on the overall vulnerability index. The calculation is performed by removing one criterion at the time from the vulnerability index for each waste facility as per Eq. 3-4. For simplification purposes, the weighting of criteria is not considered at this stage. Results are summarised based on the country (Scotland, England or Wales), and the influence of each criterion on the total score is showed expressed in percentage in the results Section 3.6.2.

$$S_{y,z} = \frac{|\ VI_y - VI_{y,z}\ |}{VI_y} \qquad Eq.\ 3\text{-}4$$

where (S) refers to the sensitivity value estimated for each waste facility (y) and criteria (z), (VI) represents the Vulnerability Index as per Eq. 3-2, finally ($VI_{y,z}$) indicates the Vulnerability Index for the waste site (y) with the removal of the index value associated to one criterion at the time (z).

3.6 Results and discussion

The index-based method introduced in Section 3.2 for the estimation of the level of risk posed by waste management facility when exposed to flooding was applied to disposal sites operative in 2019 in GB (EA, 2019a, Natural Resources Wales, 2019, EA, 2019d, SEPA, 2019c). The location of facilities was tested against the Fathom-UK flood maps for different likelihood and sources, and the vulnerability of receptors was analysed through weights and values associated to attributes and criteria (as per Appendix A). Outcomes were estimated per criterion and summarised per variable following Eq. 3-2, and finally variables were multiplied to establish a hazard index per disposal site (Eq. 3-3). Results are divided in the following Sections: 3.6.1 assesses the vulnerability results for all the waste facilities in GB independently from the risk of flooding; the same results are also analysed by the sensitivity analysis in 3.6.2; 3.6.3 reports the results from the interactions of waste facilities with the risk of flooding (hazard and the exposure); and finally 3.6.4 shows the overall risk index estimated for each waste facility and summarised per LA in GB.

3.6.1 The vulnerability of receptors in GB

The results for each criterion demonstrate the vulnerability of waste management facilities and potential receptors. It is important to note that these results are based on the entire dataset of waste facilities and do not take into account the specific risk of flooding. This approach allows for an independent analysis of vulnerability, separate from the uncertainties associated with flood risk predictions that are subject to change. Additionally, it is worth noting that cases of contaminant dispersion can also occur as a result of fire or explosions, where polluted firewater is used for firefighting.

Table 3-9 shows the distribution of waste facilities (as a percentage of the total 7,292 waste management facilities in GB), divided per attribute value and criteria.

Table 3-9. Results showing the summary of waste facilities (%) per attribute value per different criteria under the vulnerability variable.

	Criteria	Attribute values				
		1	2	3	4	5
Waste facility vulnerability	Water Risk Index (WRI)	19	51	30	-	-
	Compliance assessment scheme	91.6	5.1	1.5	1.4	0.4
	Waste facilities categories	-	2.9	-	72.5	24.6
Environmental vulnerability	Land cover	0.1	8.4	36.8	53.8	1.0
	Terrain slope	-	-	7.9	92	0.1
	Permeability	3.4	5	19.2	43.9	28.6
	Natural protected areas	50.1	-	-	-	49.9
	Aquatic classification	1.2	-	2.5	-	96.3
Social vulnerability	Index of Multiple Deprivation	11.1	22.8	25.1	19.7	21.3
Location accessibility	Urban/Rural classification	25.2	6.9	20.1	16.3	31.5

The results for the WRI criterion show the majority of waste facilities (51%) having a moderate WRI (9-13) associated with them. If considering the quantity of waste received annually, a moderate WRI corresponds to an interval between 8,000 and 440,000 tonnes of discarded material that was received/stored/treated on site annually. While 30% of disposal sites were found in the highest WRI interval (>13), which represents from 440,000 to 2M tonnes of waste received per annum by each facility. In addition because the WRI is based also on the hazardous property of waste, medium and high WRI represent sites where the proportion of recognised dangerous (versus non-dangerous) materials managed on site is higher than other facilities. The identification of such hotspots is important because the reception/management of high quantities of waste (including high volumes of hazardous discarded materials) may lead to: (1) more discarded materials temporary stored on site, potentially outdoor; (2) higher volume of waste that can degenerate and fragmentise due to mechanical and chemical treatment; (3) an increase in discarded materials quantities that can be mobilised during an event of flooding (or during a fire/explosion); and therefore (4) higher risk of hazardous particles dispersion in flood waters (or firewater).

For the land cover criterion, the majority of waste facilities (54%) resulted located in the proximity (1km) of at least one source of fresh or salty water, which can be highly impacted in an event of pollution. This is due partially to the mandatory location of some type of waste facilities close to sources of water (such as waste water treatment plants), but also by the conscious decision taken in the past of building disposal sites next to rivers and the sea when the danger posed to the environment was unknown (i.e., landfills). In addition, 37% of waste facilities resulted located close to at least one arable land where micro-components of discarded synthetic materials could migrate through flood waters. It is reassuring that only 1% of waste sites was found next to highly vulnerable wetland (bog), but among these 71 waste facilities, 10 are landfills where the monitor of potential contaminant dispersion during flooding events is the hardest to establish and control.

The outcomes from the permeability criterion show that 29% of waste facilities have at least one area with a “very high permeability” located within the footprint of the sites, the percentage grows to 44% for “high permeability” areas. This means that 73% of disposal sites are surrounded by at least one area classified with high or very high permeability, where contaminated flood waters could easily penetrate the soil and reach underground water if present. Approximately 50% of the total number of waste facilities in GB are located within a 1km proximity to natural protected areas. This comprises over 200 landfills, 600 metal recyclers, 1,000 treatment centres, and more than 1,500 transfer stations. This pattern can be partly attributed to the considerable expansion of protected areas on land in the UK, which has seen a 17% increase (11,265 km2) between 2000 and 2022 (JNCC, 2022).

Approximately 96% of waste facilities in GB were identified to be located within a 5km radius of at least one river sample or river (in the case of Wales) with an overall surface water body quality classified as poor/bad or non-compliant. The majority of these sites (~1,800) are situated in England, which is consistent with the unfavourable outcomes observed in the overall water quality samples conducted in 2020 (discussed in Section 3.5.1), the number of samples conducted in 2020 in Great Britain is presented, categorized by overall status classification outcomes. The majority of samples from Scotland showed high/very high quality, while 24% resulted in very low/low quality. In Wales, nearly 50% of the samples were classified as moderate. Notably, in England, a significant number of samples (over 400,000) were reported in 2020, indicating non-compliance with permits or general monitoring requirements. Waste facilities located within 5km from a water sample were given the attribute value of 1 for high/good/compliant samples, 3 for moderate, and 5 for poor/bad/non-compliant water quality samples. Pre-existing sings of poor water overall quality can affect the capacity of water flows to recover in case of an event of pollution and can worsen consequences in the medium/long term compared to rivers in good or moderate overall status.

Outputs for the Index of Multiple Deprivation are detailed in Table 3-10 per country. Results for Scotland and Wales show a uniform spread of facilities with the majority of them distributed in the middle range quintiles of deprivation (2, 3, and 4). A similar situation is observed for England, although the waste facilities that were found located in the proximity of at least one most deprived area are twice the number of sites having least deprived communities nearby.

Table 3-10. Based on the location of waste facilities considered in this study: distribution of areas from the least to the most deprived accordingly to national Index of Multiple Deprivation for Scotland, England and Wales.

Index of Multiple Deprivation				
	Attribute value	Scotland	England	Wales
least deprived	1	46	726	55
	2	176	1396	126
	3	220	1568	77
	4	160	1209	98
most deprived	5	86	1431	70
	Total values	688	6330	426

Finally, the analysis of the location accessibility variable reveals notable findings. In GB, a significant proportion (48%) of waste facilities are situated in areas classified as remote small towns/rural, while 20% are located in accessible towns/rural areas, and only 32% are found in urban areas where waste generation is more concentrated. This pattern is particularly evident in England, where the majority of disposal sites (~3,000) are situated in largely/mainly rural areas, followed by approximately 1,400 in accessible rural areas/small towns, and around 1,800 in large urban areas. Conversely, in Wales, waste facilities are predominantly concentrated in urban cities and towns (~260), with a smaller presence in rural towns and fringe (~60), and even fewer in rural village and dispersed areas (~100). In Scotland waste facilities are primarily located in the Central Belt and along the east coast, with 337 facilities in large/other urban areas, followed by 332 in remote/rural areas, and only 24 in accessible small/rural areas. These findings indicate a tendency to establish disposal sites in close proximity to major cities, which typically translates to rural or remote areas. This trend is further supported and visually represented in the results presented in Section 3.6.4, where the risk index is summarised per LA.

A visual representation of the vulnerability index values estimated for each waste facility, summarised for Local Authority and normalised per capita is available in Appendix B. To facilitate the comparison across different criteria, the results are divided into five classes using natural breaks. Darker red areas on the map indicate Local Authorities that are more vulnerable to pollution events compared to others, based on the specific criterion being represented.

3.6.2 *Sensitivity analysis*

The single-parameter sensitivity analysis was performed following the methodology outlined in Eq. 3-4 to investigate the relative influence of each input on the overall results. The findings were summarized by criterion and country to provide insights into their respective contributions. The results, presented in Table 3-11, are expressed as percentages indicating the mean and standard deviation. The mean values range from a minimum of 2.4% to a maximum of 12.1%. Among the criteria the category of waste facilities exhibited the highest influence on the overall results. This can be attributed to the predominance of waste treatment and transfer stations, accounting for 32.4% and 38.5% of the total number of waste facilities considered, respectively, and the high attribute value associated with them (4). The aquatic classification criterion also demonstrated notable percentages, particularly in England and Scotland. This could partially be attributed to the inherent limitations associated with the surface water quality datasets, as explained in 3.5.2. Additionally, the reports on surface water in 2020 indicated non-compliance rates of 65% in England and poor or bad overall quality in 24% of samples in Scotland. The slope criterion yielded high results across all countries, which can be attributed to the values associated with the attributes of the criterion (3, 4, and 5). The majority of facilities fell into the categories of low or medium-high slope degree, which corresponded to a value of 4. The presence of natural protected areas within a 1km proximity to waste facilities had a medium to high impact on the vulnerability index score, primarily due to the high attribute value associated with it (5). Additionally, the proximity of waste management facilities to protected areas was found particularly pronounced in England (2830 sites), followed by Scotland and Wales (511 and 389 sites, respectively). The compliance assessment scheme exhibited the lowest results (within 3% for all countries) due to the inclusion of waste facilities that were not subjected to review in 2019 and were therefore assigned a neutral value (1). Finally, the WRI criterion reported relatively low percentages, which can be attributed to the values associated with the attributes, ranging from 1 to a maximum of 3. This decision was made based on the limited information available on the management practices of waste facilities, such as the duration of waste storage on-site, the presence of underground or above-ground tanks for liquids, and other relevant factors.

Table 3-11. Summary of sensitivity analysis statistics (%) divided per vulnerability criteria and country.

	CRITERIA	ENGLAND		SCOTLAND		WALES	
		mean	st.dev	mean	st.dev	mean	st.dev
Waste facility vulnerability	Water Risk Index	5.2	1.6	3.8	1.5	4.8	1.4
	Compliance assessment scheme	2.7	1.2	3.0	1.8	2.4	0.3
	Waste facilities categories	11.9	2.1	12.0	1.5	12.1	1.6
Environmental vulnerability	Land cover	8.4	1.8	8.4	1.8	8.0	2.1
	Terrain slope	9.5	1.4	9.3	1.4	9.4	1.2
	Permeability	9.5	2.5	8.4	2.2	9.1	2.1
	Natural protected areas	6.2	4.0	9.1	3.8	11.1	2.7
	Aquatic classification	12.0	1.8	11.8	1.8	9.3	3.2
Social vulnerability	Index of Multiple Deprivation	7.8	3.6	7.4	3.0	7.3	3.3
Location accessibility	Urban/Rural classification	7.7	3.4	7.0	3.2	5.2	3.5

3.6.3 *Hazard and exposure*

The exposure of waste facilities to the risk of flooding was evaluated against the Fathom-UK flood maps for the selected returning periods (Section 3.3). 22% of the total amount of sites (7,292) were found with a river flooding probability of 10% in any given year, the number rises with the lower probabilities (0.5 and 0.1) to 28% and 30% of facilities respectively. The same trend was reported by the risk of pluvial flooding (surface water), impacting 19% of disposal sites with a 10% probability of being flooded in any given year. Results significantly increase by more than 3 times for the 1 in 200-year (0.5% probability) and 1 in 1,000-year (0.1% probability) return periods, with the 60% and 73% of waste facilities potentially affected by an event of flooding at any given year. The reasons behind the higher numbers obtained by pluvial versus fluvial flood can be partially explained by the simulation of extreme rainfall events when floods are created both from overflowing water bodies and independently from them. This results in a flood extent comprised in part by the fluvial flood map extent but additional areas also present as a result of the pluvial map. The resulting potential overestimation of disposal sites at risk of pluvial flooding is counterbalanced by the absence of flood risk projection for the years 2050 and/or 2080 that weren't available for GB at the time of writing. Future studies should aim to apply flooding projections to the analysis initiated in this Chapter to understand how the numbers of disposal sites at risk of flooding will change in future years. The vulnerability analysis revealed that among different types of waste facilities transfer stations and treatment centres were found to be the most exposed to flooding events for various flood return periods and sources. The number of sites potentially affected ranged from approximately 1,100 to 1,570 for high fluvial flood risk, and from 950 to 3,850 for low pluvial flood risk. Metal recyclers and landfills showed a maximum of 850 and 320 sites, respectively, that could be impacted by a low pluvial flood. These findings are significant, particularly considering that metal recyclers and landfills are known to be highly susceptible to the release of contaminants, as discussed in Section 3.3. The identification of their vulnerability to low-risk pluvial flooding highlights the importance of addressing the potential consequences and implementing appropriate mitigation measures in these facilities.

The hazard index was estimated by considering the depth of flooding and the debris factor for different return periods and sources. The results obtained by applying the flood water depth intervals to each waste facility in GB are shown in Figure 3-8. The outcomes for the water depth intervals associated to the pluvial risk of flooding are significantly higher compared to the river risk of floods, which is in line with the exposure outputs. Numbers increase exponentially for both the source of flooding when looking at the depth interval higher than 120cm. The rapid increase can be somewhat explained by the location of the disposal sites close to permanent water (predominantly fresh water), and by the use of buffers to represent the footprint of waste facilities. Because of the size of buffers being overgenerous, as detailed in Section 2.2.4, it's possible that the water depth intercepted by the footprint of facilities higher than 120cm is related to permanent water that is classified by Fathom-UK maps with the value of 9999. The hypothesis is reinforced by estimating the number of buffers that intersect permanent water for the 1 in 1,000-year return period within the total number of sites captured by the >120cm water depth. Results show 67% of sites intercepted by permanent water for the fluvial risk, and 33% for the pluvial risk.

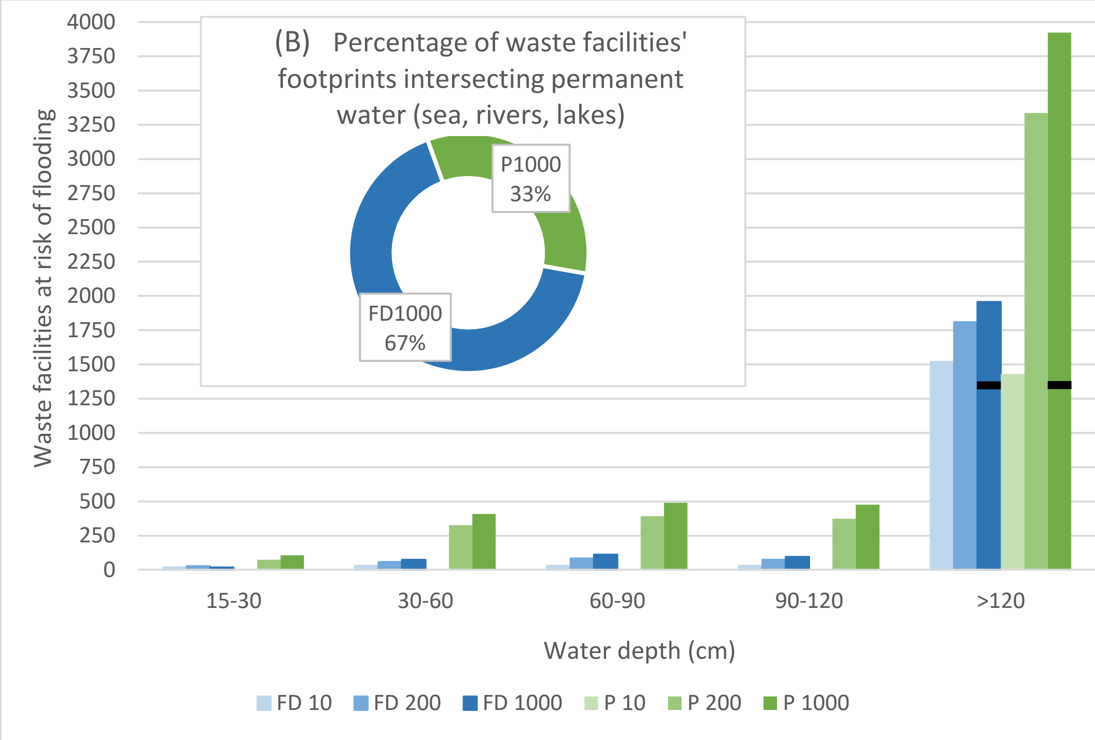
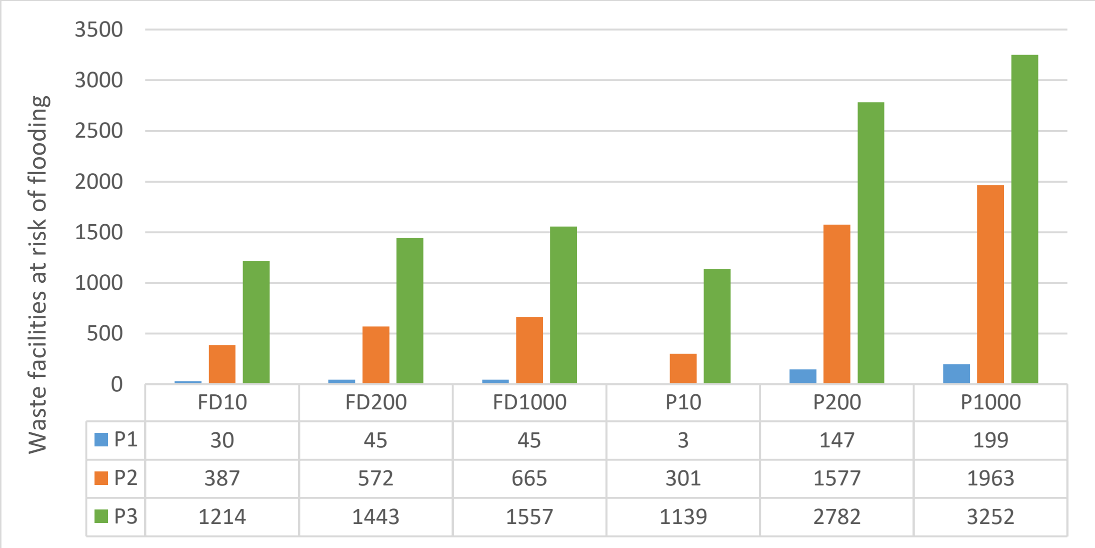


Figure 3-8. The water depth intervals applied to waste management facilities in GB (FD= fluvial defended, P=pluvial). The place markers on the chart show the number of facilities located adjacent to, or with buffers intersecting, permanent water.

Two significant outcomes arise from the estimation of the hazard index. Firstly, it is important to note that while the number of waste facilities potentially exposed to pluvial/fluvial floods above a water depth of 120cm may have been overestimated due to the presence of permanent water, it should be acknowledged that the analysis did not include the risk of flooding from the sea or future climate projections (such as those for 2050/2080). Therefore, the actual number of sites at risk could be even higher than estimated. Secondly, the proximity of waste facilities to water sources emphasises the criticality of this research in assessing the level of risk associated with pollution events and the subsequent need for mitigation, containment, and remediation measures. The presence of waste facilities in close proximity to water bodies underscores the potential impact and significance of the findings of this research in addressing and managing the risks posed by flooding events.

3.6.4 *Ranking sites for action prioritisation and risk mitigation*

Because the hazard index was generated per each flood likelihood and sources as described in Section 3.3, the overall level of risk calculated per each facility varies based on flooding return periods, and typology: pluvial or fluvial. Values range from 30 to 648. Equal intervals in between the risk index results were established to highlight waste facility with a low risk (between 30 and 217), medium risk (between 217 and 433), and finally high risk when values are higher than 433. Higher risk sites are more likely to release contaminant in flood waters, affect people and the environment, it's therefore important to identify them and rank them for action. The results show that 16% (~1200) of waste management facilities in GB (7,292 sites) were estimated with a high risk index and are located in areas likely to be impacted by fluvial flooding (with 10% probability any given year). The percentage grows to 19% and 21% when considering flood risk with lower probabilities (0.5% and 0.1% flood probability at any given year). Outcomes for the pluvial flood risk are again higher with 15% of sites with a high index of risk located in areas prone to flooding (10% probability), 37% and 44% when looking at flood risk with 0.5% and 0.1% probability at any given year. Two significant constants were discovered and presented in Figure 3-9 for the risk index of the facilities impacted by different flood likelihoods and sources: (1) the number of sites with high risk index (P3) is always the highest compared to medium (P2) and low risk (P1), and (2) the latter is particularly accentuated for the fluvial and pluvial high likelihood (FD10 and P10), where the number of high risk facilities is respectively 3.14 and 3.78 times the medium risk sites. The two observations suggest that when disposal sites are impacted by flood waters, because of the spatial context of waste facilities and the adapted proximity of risk methodology, the vulnerability of receptors is very often triggered at the full potential, especially in the case of high probability flood risk.



	FD10	FD200	FD1000	P10	P200	P1000
P1	30	45	45	3	147	199
P2	387	572	665	301	1577	1963
P3	1214	1443	1557	1139	2782	3252

Figure 3-9. Number of waste facilities at risk of flooding from different flood likelihood and sources (FD= fluvial defended, P=pluvial), divided in the estimated 3 index risk intervals (P1 = low, P2 = medium, and P3 = high).

In order to select sites to be prioritized for action, the waste facilities that scored the highest risk index (648) across the different flood likelihoods and sources were identified (5) and revealed. The facilities are located in England: two waste treatment sites, one metal recycler, one incineration facility and one landfill. Two of them are on the border between England and Wales in the counties of Shropshire and Gloucester, while the other three are located in the Norfolk County. The listed hotspots are facilities that should be subjected to further studies at the local scale to reduce the risk (i.e., flood protections, elevation of waste when stored, typology and quantity of waste received/treated, etc.), and to contain potential release of micro synthetic components in the environment (i.e., on site filtering/monitoring of surface runoff, second containment systems for tanks, etc.).

To analyse the distribution of waste management facilities and their estimated risk index, the resulting risk index values were summarised per LA and normalised by population for GB (Office for National Statistics, 2021, National Records of Scotland, 2022). Figure 3-10 shows the outputs obtained for the 1 in 10, 200, and 1,000-year return periods for pluvial and fluvial flood. Looking at the high flood likelihood (both fluvial and pluvial) in Scotland, the LAs with the highest risk index per capita are the Highlands and the Shetlands, while England shows higher concentration of risk in the North and Central-East area (from Carlisle to Peterborough, Cardiff and Newport for Wales), which are the LAs next to the main central cities (Leeds, Doncaster, Nottingham). In the 1 in 200-year fluvial flood the LAs of Perth and

Kinross stands out from the rest of Scotland, perhaps because of its position just north of Falkirk and Edinburgh and close to the medium-size cities on the East coast; although the LAs reporting the highest results in terms of risk index per capita are the LAs of Selby (located just outside Leeds and York), and Newark and Sherwood (in between Mansfield, Lincoln, and Nottingham cities). A similar situation resulted from the low flood likelihood, with the addition of the LA of Lancaster presenting the highest risk index score for the flood return period. Few areas south-east of London and the Newport LA are also emerging compared to the medium likelihood.

Figure 3-10. Risk index results summarised per Local Authority and normalised per capita for different flood return periods and source (pluvial and fluvial).

Table 3-12. List of the top ten Local Authorities with the highest risk index, ranked by return period and source.

Fluvial flood 10-year return period	Fluvial flood 200-year return period	Fluvial flood 1000-year return period	Pluvial flood 10-year return period	Pluvial flood 200-year return period	Pluvial flood 1000-year return period
City of London	Selby	Lancaster	City of London	Shetland Islands	Shetland Islands
Dover	City of London	Selby	Dover	Selby	Selby
Tonbridge and Malling	Newark and Sherwood	Newport	Tonbridge and Malling	North Lincolnshire	North Lincolnshire
Bassetlaw	Dover	City of London	Bassetlaw	Mid Suffolk	Ryedale
Shetland Islands	Tonbridge and Malling	Tonbridge and Malling	Swale	Barrow-in-Furness	Mid Suffolk
Swale	Bassetlaw	Barking and Dagenham	Shetland Islands	Richmondshire	East Cambridgeshire
Newark and Sherwood	Stockton-on-Tees	Newark and Sherwood	North Lincolnshire	Dover	Eden
North Lincolnshire	Lancaster	Dover	Newark and Sherwood	Newark and Sherwood	Barrow-in-Furness
Richmondshire	Swale	Neath Port Talbot	Richmondshire	East Cambridgeshire	Newark and Sherwood
Newport	Neath Port Talbot	Bassetlaw	Newport	Allerdale	Richmondshire

The flood likelihoods that really makes a difference on the impact on waste management facilities and the vulnerability of receptors are, once again, the 1 in 200 and 1 in 1,000-year pluvial return periods. Scotland displays a significant increase of risk especially in the Shetlands and the Outer Hebrides islands resulting from the vulnerability of the environment and the remoteness of the locations. England shows a further increment of risk localised mostly outside the main cities, with the highest results obtained for the areas of Richmondshire, Selby, North Lincolnshire, and Newark and Sherwood in the centre/north of England, and Mid Suffolk and Dover down west. Wales reports a generalised increased in risk levels outside the main cities for the first time. With the low probability pluvial flooding the tendency of the risk to concentrate outside main cities is clearly visible in the Central Belt in Scotland reporting lower risk levels (especially around Glasgow), and the major settlements in England. The Greater London area also strikes for the lower risk index per capita compared to mean risk values, especially in comparison with the LAs of Tandridge (just outside London) and Swale (just east of Gillingham). The observed tendency of high risk index values concentrating outside main urban settlements is the results of several aspects: (1) 48% of waste facilities in GB are located in areas classified as remote small towns/rural, vs 32% located in urban areas

(Section 3.6.1), (2) the vulnerability of environmental receptors, specifically, is higher, as expected, far from main urban conurbations with peaks in the north and north/east of Scotland, the counties of Durham in the north of England, Dorset and Wiltshire in the south, and finally Cornwall in the south-west, and (3) the number of facilities highly impacted by floods for different likelihoods and sources is located in the Highlands in Scotland, with peaks in the Northumberland, the Yorkshire and in the counties just north and west of London.

Finally to identify the LAs that need to be prioritised a selection was made by looking at the overall risk index across different flood likelihoods and sources. A total of nine LAs were identified as in need of further assessment at the local scale for risk monitoring and reduction. In Scotland, the Shetland Islands proved to be constantly very high in the risk index across different flood likelihoods, this is due to the extent of flooding risk in combination of high scores obtained by resident waste facilities in the attributes of land slope, aquatic classification, social vulnerability, and the location accessibility. In England, the LAs with the highest risk index scores across different flood likelihoods and sources were divided in three main areas: in the north the LA of Eden that obtained high results for the WRI and the overall aquatic classification, in the east side the LAs of Selby and Ryedale presented significantly high scores for the WRI, the compliance assessment scheme, the soil permeability, and the location accessibility. Finally, the south-west of England with South Hams and Teignbridge that reported particularly high scores for the attributes of overall aquatic classification and the WRI.

3.7 Conclusions

This Chapter introduced a novel approach that contextualises the risk posed by waste management facilities by considering relevant spatial factors. It also quantified the risk on a per-facility and per-Local Authority (LA) basis, enabling prioritisation of actions to reduce the threat. The methodology developed employs a multi-index-based assessment to identify waste management facilities with a higher likelihood of accidentally releasing contaminants in flood waters. By evaluating sub-variables, criteria, and attributes representing the components of risk (hazard, vulnerability, and exposure), a risk index value was estimated for each disposal site in GB. The identification of hotspots that require prioritisation for further actions by local governments and environmental agencies facilitates risk monitoring and reduction efforts. Notably, this method is applicable at a national scale and can be replicated in other countries as well.

The obtained results reveal that approximately 21% (1,600) of the total number of waste facilities in Great Britain are at risk of experiencing high flood likelihood, encompassing both pluvial and fluvial flooding. The numbers increase for medium and low likelihood, constituting approximately 60% (4,500) and 73% (5,400) of the total, respectively. Among the facilities identified as at risk across different likelihoods and sources of flooding, those categorised with a high-risk index consistently accounted for over 60% of the total, with peaks observed for high likelihood scenarios in both fluvial (60%) and pluvial flooding (74%). The latter suggests that the considered receptors are likely to be triggered at the full potential when facilities are impacted by flooding, independently from the level of impact (especially for the high probability of flood risk).

The hazard analysis conducted revealed that the majority of facilities at risk of flooding were found to intersect with flood depth intervals exceeding 120cm. This overlap was observed for a significant number of sites, ranging between 1,400 and 3,900, depending on the flood likelihood and source. The justification for this finding was attributed to the presence of waste site footprints that intersect with permanent water bodies such as seas, rivers, and lakes. This highlighted the tendency of disposal sites to be located in close proximity to water sources, thereby increasing their vulnerability to potential pollution events.

The study acknowledges the limitation arising from not including the coastal effects within the flood modelling, such as coastal erosion and coastal flooding. Future studies should consider incorporating coastal erosion (e.g., using Dynamic Coast tools), coastal flood risk assessment and projections of sea level rise to comprehensively address the risks associated with waste facility locations.

4 A framework to assess the impact of flooding on the release of microplastics from waste management facilities

4.1 Introduction

Floods are among the most frequent natural hazards to cause industrial accidents that may result in fires, explosions, and the release of hazardous materials (Piccinelli and Krausmann, 2013). These events are defined as 'Natech' accidents (i.e., natural hazards triggering technological accidents) because of the capacity of natural hazards to cause technological disaster. Natech accidents mainly refer to refineries, petrochemical complexes, and oil and gas pipelines that deal with dangerous substances as identified by the Seveso-III-Directive (2012/18/EU), which is the main EU legislation aiming at the prevention of technological disasters involving dangerous substances. However, the types of synthetic products that can cause harm are not limited to the list included in the Seveso-III-Directive. For example, plastic items are made from polymers mixed with a complex blend of additives, some of which belong to the list of emerging contaminants, which are released due to plastic degradation with potential risks to the environment and human health (Gunaalan *et al.*, 2020). In recent years, plastic pollution has become one of the major environmental concerns, with an exponential increase in plastic created (Geyer *et al.*, 2017, PlasticsEurope, 2021) and an expected 3-fold increase in plastic waste by 2030 (Borrelle *et al.*, 2020b). However, despite the growing interest developed by the scientific community, flood-induced plastic debris mobilisation from terrestrial sources has yet to be fully understood.

It is estimated that between 4 and 12 million metric tons (Mt) of plastic end up in the marine environment, globally, per year (Jambeck *et al.*, 2015, Geyer *et al.*, 2017, Boucher *et al.*, 2020). The issue is exacerbated by the persistence of plastic debris in the environment and the inevitable breakdown processes resulting in the fragmentation of the initial (microplastic) component into microplastics (MPs). The origin of ocean plastics has been increasingly attributed to terrestrial sources (Hurley *et al.*, 2018), and recent attention has been given to rivers, considered as a major pathway for plastic transport from inland areas to the ocean (van Emmerik *et al.*, 2019, He *et al.*, 2021, Wang *et al.*, 2021). The consequences of flooding on plastic loads in rivers was extensively studied by Hurley et al. (2018), which investigated 40 rivers across the northwest of England before and after a period of severe flooding in winter 2015/2016. The study confirmed firstly the presence of MPs in all of the studied river channel beds, and secondly the capacity of flooding to export approximately 70% to 100% of the total MPs load stored in the riverbeds.

The interaction between plastic transport and river flood events has been further investigated by Roebroek et al. (2021), with the introduction of global mismanaged plastic waste (MMPW) as a terrestrial diffuse source of plastic debris. By combining MMPW data with river flood extents for different return periods, the study was able to estimate the flood-driven plastic mobilisation per country with results showing a tenfold global increase in potential plastic transport during 10-year return period flood compared to non-flood conditions. Among the methodology's limitations listed by Roebroek et al. (2021), two aspects are particularly relevant for the purpose of this research. The first is data on MMPW are estimated on the waste generation rates per country that ignore the potential build-up over time. The second is that Roebroek et al. (2021) only focuses on mismanaged plastic, while plastic waste properly disposed could also be mobilised by flood waters. Data on waste characteristics and quantity received per facility (tonnes) are reported annually by Member States (European Parliament and Council of the European Union, 2008) and made publicly available. In the UK, in 2019, the total amount of waste received by waste facilities was equal to 257 million tonnes. Although it may overestimate the actual quantity of waste produced for the same year due to the complex movement of discarded items through the waste management network, the information on waste received provides a greater detail and richness compared to national statistics on waste produced annually because it includes data on the location, quantity, and characteristics of waste dealt with by each waste management facility rather than a gross value at a national scale (a list with type and description of waste management facilities selected for this study is available in Appendix C).

Waste management facilities become potential terrestrial sources for plastic mobilisation and therefore plastic pollution when the discarded materials are temporary stored, treated, or disposed within sites located in areas at risk of flooding. During recent decades, few works investigated the impacts of flooding on managed waste. Arrighi et al. (2018a) is one of the few studies available in literature that clearly defines wastewater treatment plants, waste handling facilities, and contaminated sites as environmental hotspots because of the risk posed by the presence of contaminants within sites located in flood-prone areas. Comparatively, the potential inundation of solid waste landfills has received more attention in the literature (Laner *et al.*, 2009, Neuhold and Nachtnebel, 2011, Neuhold, 2012) although many questions remain unanswered. For example, Nicholls et al. (2021) recently published a comprehensive European study on coastal landfills and the rising of sea levels, highlighting a lack of data and a general underestimation of the threat posed by the potential release of solid and liquid waste from coastal landfills.

The purpose of this chapter is to better understand quantity and characteristics of waste received and stored by waste management facilities, with the assumption that managed waste is potentially one of the biggest terrestrial sources of flood-induced MPs mobilisation. The developed novel methodology was applied to the UK by combining publicly available data on waste received by waste facility in 2019 together with flood likelihood map extents for different return periods and source (fluvial and pluvial). The research aims to (i) identify waste at risk of releasing MPs among the European Waste Catalogue (EWC) code in addition to the well-known plastic waste, (ii) estimate the quantity of waste at risk of flooding in the UK which could lead to MPs' mobilisation in flood waters, and (iii) identify spatial patterns where the level of risk requires further studies at the local scale.

4.2 Methods

This chapter introduces a framework to estimate the quantity of waste at risk of releasing MPs in flood waters by combining publicly available annual data on waste received by waste management facilities with flood extent maps for different sources (fluvial and pluvial) and likelihoods of flooding (5, 10, 20, 50, 100, 200, 500 and 1,000 years) at the national level. Due to the novelty of this assessment, two methodologies are presented to (1) identify the codes within the List of Waste (European Commission, 2000b) referring to solid waste that could deteriorate and fragment into MPs, and (2) determine the quantity of waste at risk of flooding based on the percentage of overlapping between the estimated footprints representing waste management facilities (Chapter 2) and flood map extents.

The methods were applied to the UK where datasets on annual quantity, type, and location of waste received by waste facilities, and INSPIRE Index Polygons (described in Section 2.2.1)) are publicly available (with the exception of Northern Ireland). The datasets were subsequently combined with flood map extents in a geographical information systems (ArcGIS and ArcGIS Pro) to determine national and local quantities of waste at risk of releasing MPs during inundation.

4.2.1 *Identification of waste at risk of deteriorating into synthetic micro components within the European Waste Catalogue (EWC): not just plastic waste*

The waste classification code, also referred to as LoW (List of Waste) or EWC (European Waste Catalogue) code was introduced in 2000 by the European Commission Decision 2000/532/EC (further revised in 2014 and 2017). Unlike more straightforward legislation on chemicals, because of the complexity and alterability of discarded substances, the LoW does not refer to a waste's chemical components for classification purposes but rather to alternative criteria such as (i) the waste source, (ii) the waste type, and (iii) the recognition of waste not otherwise specified because it is mixed or undifferentiated. The classification system was conceived to help operators to assign a standardised, accurate six-digit code to each entry of waste. The first two digits of the LoW refer to the waste source (e.g., 17 Construction and demolition wastes), the second series of digits assign the waste type (e.g., 17 02 Construction waste wood, glass and plastic), and the last two digits represent the final entry description (e.g., 17 02 03 Plastic construction waste). The LoW recognises three types of entry and marks with an asterisk (*) what is considered as hazardous waste. Absolute hazardous (AH*) entries display one or more of the fifteen hazardous property as indicated in Annex III to the Directive on waste 2008/98/EC (Waste Framework Directive or WFD), such as explosive, ecotoxic, mutagenic, infectious, etc.; absolute non-hazardous (ANH) entries identify waste lacking any hazardous component. Finally, mirror entries represent the case of mixed substances where further assessment needs to be undertaken to classify the waste as mirror hazardous (MH*) or mirror non-hazardous (MNH) (EA, 2014).

Table 4-1. Number of codes from the European Waste Catalogue (EWC) code per entry type (AH, MR, MNH, ANH) based on discarded materials' hazardous properties.

842 Entries in the List of Waste (LoW)			
408 Hazardous entries		434 Non-hazardous entries	
230 Absolute Hazardous	178 Mirror Hazardous	188 Mirror Non Hazardous	246 Absolute Non Hazardous

The absence of cross-categories (e.g., the substance state such as solid, liquid or gaseous) and the lack of a controlled vocabulary for the entries' description does not allow straightforward data inquiries and the same type of waste can be found throughout different LoW groupings. The European Commission published the Commission Notice On Technical

Guidance On The Classification Of Waste in 2018 (2018/C 1 24/01), which added a substance-oriented identifier approach, including a non-exhaustive list of plastic waste entries. Out of 842 LoW codes (Table 4-1), there are 29 plastic waste classification codes identified by the commission notice, divided into (6) absolute non-hazardous, (10) mirror hazardous, and (13) mirror non-hazardous categories. The word 'mirror' in the legislation indicates the case of entries presenting a mix of hazardous and non-hazardous materials. Surprisingly, no entries were selected among the absolute hazardous substances, even if within the LoW there are several absolute hazardous codes that identify mix waste, which are likely to have some plastic components: for example, code 18 01 10* amalgam waste from dental care. In addition, although MPs in waste is an increasingly explored issue in literature (e.g., MPs in wastewater treatment plants and in landfill leachate (He *et al.*, 2019, Sun *et al.*, 2021), in the commission notice there is no reference to the presence of MPs in waste. The description of the source of waste for entries related to plastic waste versus Microplastic Releasers is available in Appendix C.

The source of MPs can be direct: primary microplastic, specifically manufactured for commercial use such as cosmetics; and indirect: secondary microplastic, resulting from the deterioration and fragmentation of certain materials such as plastic items, synthetic fabrics and rubber, due to mechanical stress, photo-oxidation and weathering processes (Golwala *et al.*, 2021). Although the presence of primary MPs is well known within the waste industry, for example in landfill leachate or within waste-water treatment plants, this Chapter focuses on secondary MPs only, leaving the assessment of primary MPs to future work. The release of MPs from fabric is prominently documented, with studies reporting from 10 to 1700 mg of MPs per kg of washed fabric (Karkkainen and Sillanpaa, 2021); (Kapp and Miller, 2020) reported 35-70mg for 483g blanket (drying cycle) equivalent to 7-145mg MP per kg of fabric; De Falco et al. (2018) estimated 0.43-1.27g per 5kg wash, equivalent to 86-254mg MP per kg of fabric; and Napper et al. (2016) found on average 700,000 MP fibres per 6kg wash load, equivalent to 65-224mg MPs per kg of fabric. Another important source of MPs is road tyre wear emissions which was calculated ranging from 0.2 to 5.5 kg of global emissions of Tyre Wear Particles (TWPs) per capita (Baensch-Baltruschat *et al.*, 2020, Evangeliou *et al.*, 2020). Other potential emission sources include plastic manufacturers and industries where plastic is used (for example carpet, wallpaper and cosmetic/pharmaceutical manufacturers), waste management facilities, agricultural areas, road networks (beyond tyre and brake wear) and urban residential/commercial areas (Xu *et al.*, 2020, Allen *et al.*, 2022). It is noted that direct and diffuse source emission rates beyond fabric washing/drying and tyre wear have yet to be quantitatively characterised to date and is an important focus of future research.

For the purpose of this study, and in an effort to clearly identify waste that could degenerate into synthetic micro components, an extended description of the selection of plastic waste identified by the European Commission Notice (2018) has been adopted. The description of plastic waste by the European Commission Notice (2018) has been extended to include possible sources of secondary MPs in waste, defined as Microplastic Releasers (MPRs) (Figure 4-1) described as: (i) codes related to synthetic textile and rubber waste that could release microfibers and rubber polymers, and (ii) codes referring to mixed and undifferentiated materials (e.g., mixed household waste). These waste products are selected specifically as they are likely to contain larger plastic materials, synthetic clothes, and discarded items with rubber components and therefore act a source of MPs.

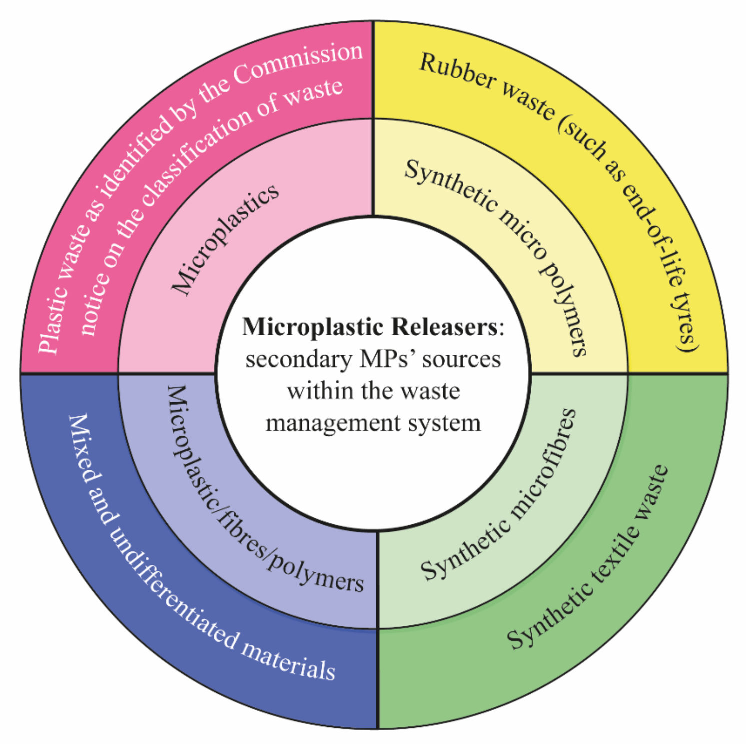


Figure 4-1. Four types of waste at risk of deteriorating and fragmentise into micro components: main type of waste on the outside of the circle and the related micro components on the inside. The study has named the selected waste as Microplastic Releasers (MPRs).

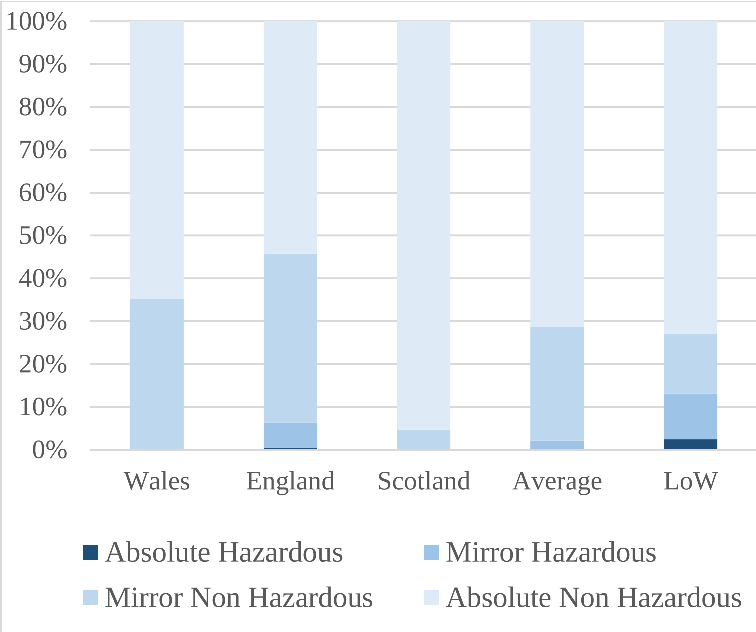


Figure 4-2. Microplastic Releasers distribution per List of Waste entry type. Each code within the European Waste Catalogue has an entry type associated to it based on the hazardous characteristics of the discarded items (Absolute Hazardous, Mirror Hazardous, Mirror Non Hazardous, and Absolute Non Hazardous). Figure 2 shows the distribution of the selected MPRs waste codes among the four entry types for each country in the 2019, with the exception of Northern Ireland, and allow the comparison with the average distribution for Great Britain and within the full List of Waste.

In addition to the physical stress due to transportation, treatment, and weathering phenomena (especially when waste is accumulated outside in containers and/or piled on hard surfaces), the deterioration and fragmentation of materials can be accelerated by flood water (as a mechanical force that may break particles) (Zhang *et al.*, 2021). In the case of flooding, discarded items could be subjected to the flow's rapidly changing conditions and the presence of suspended sediments, which could lead to turbulent mixing (mechanical abrasion and wear) and the collision with debris and other built infrastructures. Micro waste components could escape the perimeter of the facility within flood waters, with the flood water acting as the micro waste's transport vector and mixing or discharging into waterways, ecosystems, flooded areas and floodplains downstream. Therefore, it is important that waste facilities located in flood-prone areas are identified, alongside the information on the quantity and location of discarded items prone to release MPs in flood waters.

4.2.2 *Defining and mapping the different likelihoods for fluvial and pluvial flood risk map extents*

The Fathom-UK flood map extents, indicating the probability of flooding over space, were made available by SSBN UK Limited (SSBN UK Limited, 2021a). The method used to derive the fluvial maps refers to the global model detailed by Sampson et al. (2015a), further improved with higher quality data such as terrain, hydrography, stream gauge, rainfall, and flood defence data.

Fathom's hydraulic modelling is an implementation of the LISFLOOD-FP numerical scheme (Bates *et al.*, 2010), combined with Neal et al. (2018) approach to improve and optimise central and graphical processing units through parallelization to significantly reduce the model runtime. The hydraulic model is executed at 1 arc second (between 20 and 25m resolution) across the UK using a composite Digital Elevation Model (DEM) built using LiDAR elevation data from relevant national government agencies (which covers ~70% of UK land area), together with Ordnance Survey terrain data. Extreme flows on every river were predicted via statistical modelling based on a dense array of river gauges with long historical records available within the National River Flow Archive (NRFA). Channel locations were defined using Ordnance Survey channel location data, and used to construct a flow accumulation grid together with the DEM.

In terms of river bathymetry, the Global River Widths from Landsat (GRWL) database from Allen and Pavelsky (2018) was used to estimate river widths, while an estimate of channel beds elevations was produced by adopting an innovative channel solver (Neal *et al.*, 2021), and by combining data on an estimate of bank-full discharge (for return period of ~ 1 in 2 years), channel widths and slope from the DEM. The reason behind linking channel geometry to discharge return period is to ensure channels are appropriately sized for the simulated flow. For the pluvial model, Intensity-Duration-Frequency curves were calculated from CEH-GEAR1h, an hourly gridded rainfall dataset at 1 km spatial resolution, and 1-hour, 6-hour, and 12-hour intensity-frequency relationships were computed. The rainfall dataset input water directly onto the 2D base model LIDFLOOD-FP's staggered grid, with the addition of a 1D model solver for channels smaller than the grid size.

In terms of flood defences, data came primarily from the Environmental Agency and Natural Resource Wales. For location missing data, particularly in Scotland, a levee detection algorithm was adopted to fill the gaps (Wing *et al.*, 2019).

Fathom-UK hazard map extents were validated against Environmental Agency (England) and Natural Resources Wales flood maps, unfortunately at this stage no validation is available for Scotland and Northern Ireland. Results indicated that the two datasets are within proximity of each other, with the error potentially due to typical uncertainties in extreme flow estimation and terrain data accuracy. However, validation tests focussed only on extreme floods (<100-year return period), ongoing research at Fathom will further validate the methodology against both observed and lower return period flood events.

The Fathom-UK flood map extents adopted for the scope of the presented Chapter included several return periods (5, 10, 20, 50, 100, 200, 500 and 1,000 years), for fluvial (considering flood defences) and pluvial flooding. Data were given in flood modelling output (as a GeoTIFF raster) at 1/3 arc sec (~10 m) resolution for the entire UK, with cell values representing maximum inundation depths in centimetres from 0 to 9999 (9999 = permanent water). Unfortunately, no indication of the flood velocity was provided, nor was coastal flood likelihood extent mapping available. Therefore, a simplistic approach was adopted to differentiate flooded versus non-flooded pixels based on water depth: for fluvial, a binary map of wet and dry was established considering as flooded any depth higher than 0 cm; while for pluvial flooding, the same was applied to depths higher than 15 cm. The 15 cm threshold adopted was based on Environmental Agency's flood risk information (2019e) suggesting that at 15 cm flooding would likely exceed kerb height and damp-proof course. Raster maps were merged for the UK, reclassified with two intervals representing wet and dry, and converted in polygons in the GIS environment to allow further spatial analysis with the waste dataset.

4.3 Estimating the quantity of managed waste at risk of releasing microplastics in flood waters

The estimation of the quantity of waste at risk of being mobilised and releasing MPs in flood waters at any given day in 2019 was estimated based on the selection of Microplastic Releasers, from datasets on annual total waste received by facilities per country in the UK. Annual quantities were summarised per operator and divided by 365 to give approximate daily quantities. The simplification was necessary because of the lack of daily data on waste received and the average time spent by waste materials stored on site, which are both relevant considerations when simulating the impact of flooding on waste facilities. As a result of the limitations, the flooding event is considered as a single day duration only, which may underestimate the quantity of waste at risk of being affected. An additional build-up component was adopted only for landfills for their intrinsic accumulation properties. Therefore, for landfills operative in 2019, MPRs were selected among the annual quantity of total waste and summarised for every year from 2007 to 2018, and the relative quantities added on top of the daily tonnage from 2019.

4.3.1 Estimation of daily quantities (tonnes) of Microplastic Releasers received per facility in the UK in 2019

Data about annual quantities of waste received per facility for each typology (based on the LoW classification) are publicly available for all countries in the UK: Scottish waste sites and capacity tool (SEPA, 2019b), Waste Data Interrogator (EA, 2019d), Waste Permit Returns Data Interrogator (Natural Resources Wales, 2019), and Authorised waste sites (NIEA, 2019). From the public datasets, the operative facilities in 2019 were selected together with the category of waste facility, discarding sites with incomplete information (e.g., missing coordinates, waste codes, etc.). Subsequently, MPRs' codes were selected for each country and annual quantities (tonnes) were summarised per facility. The dataset for Northern Ireland doesn't specify quantities of waste for each LoW code, therefore the percentage of Microplastic Releasers was estimated by referring to the average obtained for Scotland, England and Wales (32% of the total amount of waste received). As indicated in Figure 4-2, the total annual amount of waste received by facilities in 2019 in the UK was 257 million tonnes, of which almost 30% (~74 million tonnes) were MPRs, consisting of mixed and undifferentiated materials (70%), plastic waste (29%), synthetic textile (7%), and rubber (1%). Although not all the operational waste management sites dealt with MPRs in 2019, results show approximately 66% of facilities for Scotland, 47% for England, 64% for Wales and 53% for Northern Ireland managed MPRs waste.

Table 4-2. Quantity (tonnes) of total waste and Microplastic Releasers received by waste facilities in 2019 in the UK.

	Active waste facilities in 2019	Annual total waste received (tonnes)	Annual Microplastic Releasers received (tonnes)	Percentage of Microplastic Releasers waste on the total
UK	7,676	257 x 10^6	73,8 x 10^6	29%
Scotland	697	16,7 x 10^6	5,8 x 10^6	35%
England	6,151	222,9 x 10^6	62,1 x 10^6	28%
Wales	444	9,9 x 10^6	3,5 x 10^6	35%

Northern Ireland	384	$7{,}5 \times 10^6$	$*2{,}4 \times 10^6$	32%

**Results for Northern Ireland were estimated based on the average found for the other countries in UK (32%)*

4.3.2 Microplastic Releasers accumulation in landfills

In the UK in 2018 50.7 million tonnes of waste were landfilled (44.1 in England) among 629 operative disposal sites (534 in England), with a remaining capacity of 129.3 million tonnes (converted from 415,069,000 m^3 by considering a waste density of 311.73 kg/m^3) (DEFRA and Government Statistical Service, 2021). On top of this, in England alone, around 20,000 historical landfills (where there is no environmental permit in force, including sites that existed before landfills were regulated) were mapped by the Environment Agency in 2022. Approximately 1,200 of those are located within flood zones of 1 in 200-year return period (Brand *et al.*, 2018), and ~3,400 are at risk for low likelihood but high intensity flooding (CCC, 2018). Awareness of the risk presented by landfills to the environment has been raised and has led to a significant increase of landfill-related publications in the last two decades: from 662 in 2000 to 2,335 in 2017 (Sabour *et al.*, 2020b). However, despite the increase in severity of landfill related regulations, Directive 1999/31/EC on the landfill of waste and the Amending Directive (EU) 2018/850 and encouraging governmental strategies in response of the Zero Waste movement, more measures need to be put in place to control and contain the potential leakage of waste from disposal areas to the environment.

In this Chapter, the publicly available data on waste received by landfills for different countries in the UK were used for two objectives: firstly, to advance the quantitative assessment of Microplastic Releasers received by landfills for the period 2007 – 2019 compared to the total amount of waste; and secondly, to select the quantities of MPRs accumulated by landfills classified as operative in 2019 to be added to the daily estimates of waste at risk of releasing MPs in flood waters. The latter is an attempt to approximate the build-up of MPRs that occurred through the years in landfills, which is relevant when simulating the impact of flood on disposal sites.

For the first part of the analysis, although historical landfills (intended as opposite to licensed/permitted sites) pose the highest risk due to their predominant location in low-lying estuarine and coastal areas, and the absence of leachate management systems (Brand *et al.*, 2018), inadequate records on the waste received and/or landfilled prevented them from being included in the current study. Instead, for landfills licensed/permitted in the UK, publicly available datasets were analysed to quantify the waste annually received per facility from 2007 to 2019 (EA, 2019d, Natural Resources Wales, 2019, NIEA, 2019, SEPA, 2019b). Subsequently, waste codes identified as potential Microplastic Releasers (MPRs) were selected for each country, summarised per year, and divided by the country's population to allow the comparison among different sizes of countries in the UK. Results shown in Figure 4-3 reveal a significant decrease of the total amount of waste received by disposal sites in Scotland, England and Wales from 2007 to 2014, with a slightly rise of approximately 3 million tonnes between 2014 and 2015. The percentage of MPRs compared to the total amount of waste also substantially reduced from 65% in 2007 to 38% in 2014, and 35% in 2019 (the available data for Northern Ireland are only for the period 2014-2019).

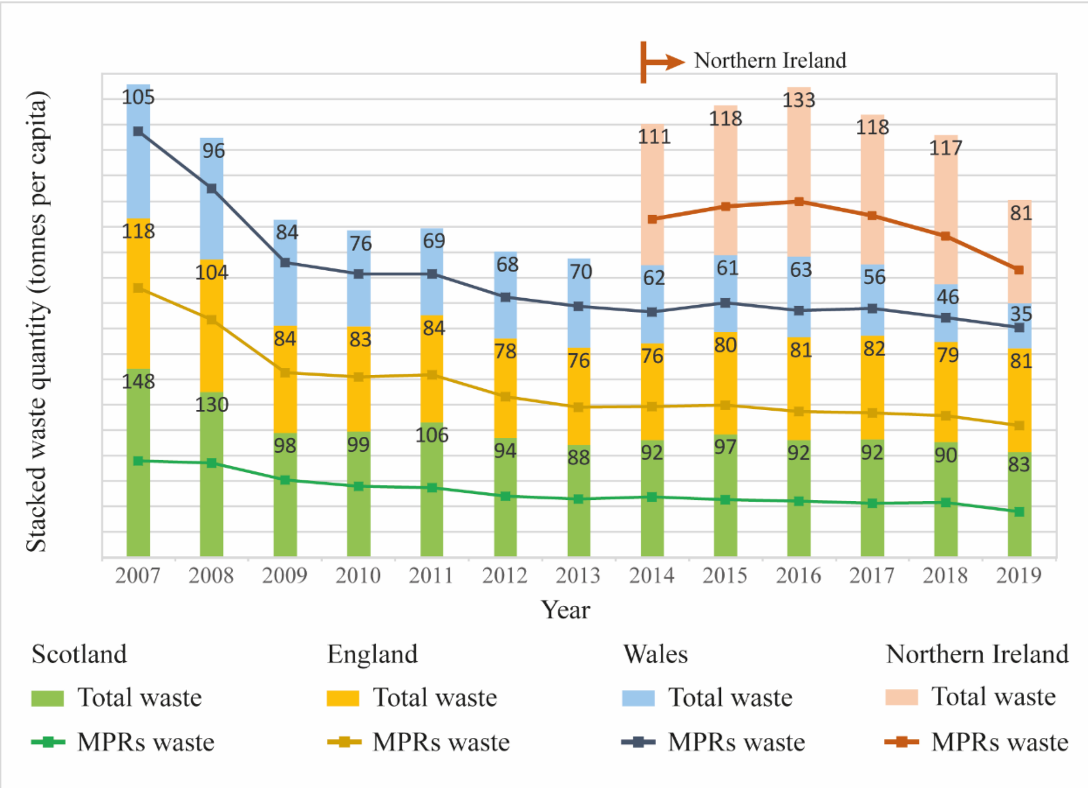


Figure 4-3. Per capita annual waste and Microplastic Releasers waste received by landfills per country and per year. MPRs waste was selected from annual quantity of total waste received by landfills from 2007. Both total waste and MPRs quantities were divided by population and organised in a stacked bar graph to highlight the differences between different years and countries in the UK (Scotland at the bottom, then England, Wales, and Northern Ireland). Each bar has a number on it referring to the quantity of total waste (tonnes per capita). Data for Northern Ireland are only available from 2014

and the quantity of MPRs was estimated based on the average for the other countries (32%).

The tendency of sending less waste to landfills is likely the result of the European Landfill Directive 99/31/EC, which aimed to reduce or prevent, as far as possible, negative impacts from landfill to the environment. Some of the Directive's consequences can be read through the UK achievement in terms of (i) biodegradable waste going to landfills that decreased from 10.3 million tonnes in 2012 to 7.2 in 2018; (ii) the rates of packaging waste recycled or recovered increased from 61,4% in 2012 to 62,1% in 2018; and (iii) the imposed rise in price for accepted discarded materials, to include the costs related to the closure and after-care of a site, translated in a reduction of more than 5 million tonnes between 2012 and 2014 in the total amount of commercial and industrial waste produced (Department for Environment Food & Rural Affairs and Government Statistical Service, 2021).

For the second objective of the analysis, an approximation regarding the quantity of MPRs accumulated in landfills was estimated with the intent of recreating the condition faced by flood waters during a hypothetical flood event occurring on any day in 2019. Unfortunately, disposal sites' data lacks consistency, especially regarding the location of landfills. For example, for England, sites' geographic coordinates are available from 2012, before that permit numbers were reported from 2010, and for the period 2007 - 2010 only site names or operators are available to identify a landfill's location. Therefore, the accumulation factor was performed only for landfills operative in 2019, and the total sum of MPRs for the period 2007-2018 was added to daily quantities for 2019. The aim of the methodology is to highlight the importance of considering what has been buried in landfills in previous years that could be mobilised in the case of a flood event. It does not take into consideration the MPs already contained in the disposal site's body and/or leachate, and does not investigate additional mechanisms such as waste decomposition, degradation, and landfill erosion which are left to future studies.

4.4 Results

A dataset with daily quantities of waste at risk of releasing MPs in flood waters was created for facilities operative in the UK in 2019, with the addition of accumulation estimates for landfills. A buffer area for each facility was created based on the methodology described and combined with Fathom-UK flood map extents for different return periods (from 1 in 5 to 1 in 1,000-year return periods) and flood type (fluvial and pluvial flooding). Both the number of waste facilities at risk of flooding, and the extent of the interaction (percentage of overlap) between buffer areas and flood map extents were estimated. The percentage of overlap was subsequently used to determine the quantity of waste at risk of releasing MPs in flood waters per site.

4.4.1 *Impact of floods on waste management facilities: quantity and location of waste at risk of releasing MPs*

The results across the UK are heavily influenced by the population spatial variation between England and the rest of the UK, however a general common trend was identified. As expected, Figure 4-4A shows a steady increase in the number of facilities affected by fluvial flooding from high to low likelihood scenarios, starting with approximately 450 sites for the 5-year return period (with 78% of the sites located in England, 10% in Scotland, 8% in Wales, and 3.5% in Northern Ireland), and consistently rising with an addition on average of 54 sites per return period. Pluvial flooding presents a significantly higher impact on waste management facilities compared to fluvial, with numbers almost doubling (from 737 to 1266) in between the 20 and 50-year return periods. This reaches the highest impact with the low likelihood scenario affecting 65% of the total number of facilities which dealt with MPRs in 2019. For the same return period, the landfills at risk of flooding are 135 for pluvial flooding, and 44 for fluvial flooding. This translates respectively to 10.8 and 1.2 million tonnes of MPRs at risk of flooding, predominantly landfill type waste facilities, in the period 2007-2018. A focus on the different categories of waste facilities affected by pluvial flood is available in Figure 4-4B, highlighting treatment facilities and transfer stations as the categories most affected. This is not surprising due to their higher number compared to other types of facilities.

Figure 4-4C represents the estimation of the quantity (million tonnes) of MPRs at risk of flood for different return periods and sources. The quantity of MPRs affected by fluvial flood is increasing consistently by ~30,000 tonnes per return period, starting at nearly 1 million tonnes for the high likelihood flooding (647 facilities affected) and reaching 1.2 million tonnes with the low likelihood flooding (1,024 facilities). Variations in quantities of waste at risk of flooding compared to the number of facilities affected depend on the quantity of MPR waste received by each facility in 2019, and by the presence of landfills where the waste accumulation was estimated for the period 2007-2018. The estimated numbers of MPRs affected by pluvial flooding are significantly higher, showing a 10-fold increase from 5 to 1,000-year return flood event, at the same time the number of facilities exposed drastically grows from 591 to 2,472 and the number of landfills increases from 39 to 135. In both cases, the percentage of sites located in England is approximately 80%. The biggest increment is between 20 and 50-year pluvial return period where the amount of MPRs at risk is 3 times higher (from ∼1,5 to ∼5 millions of tonnes) for the 50-year return period flood. The reasons behind the greater numbers obtained by pluvial versus fluvial flood can be partially explained by the simulation of extreme rainfall events when floods are created both from overflowing water bodies and independently from them. This results in a flood extend comprised in part by the fluvial flood map extent but will additional areas also present as a result of the pluvial map.

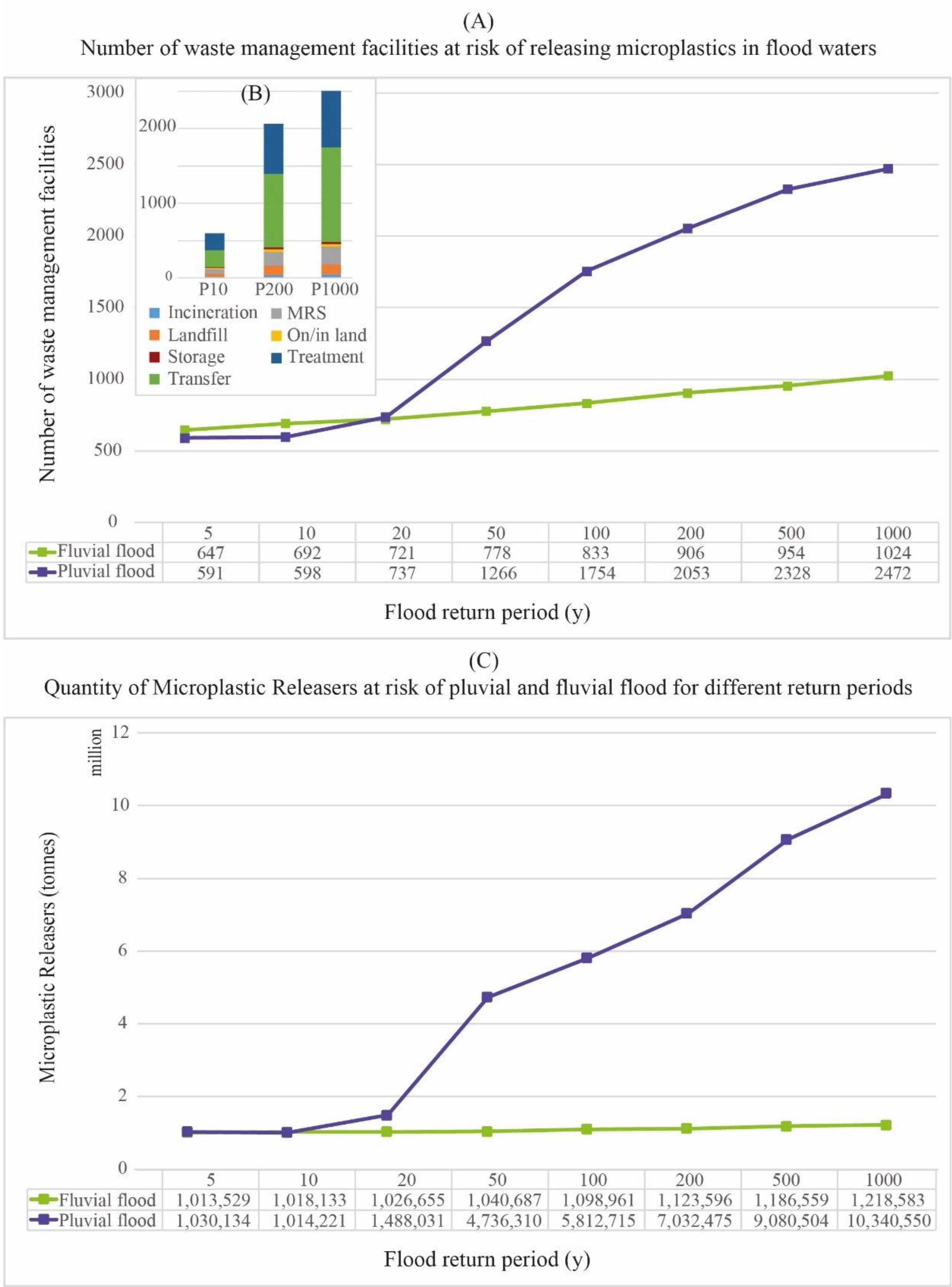


Figure 4-4. (A) Quantity of waste management facilities which received MPRs on site in 2019 at risk of flood for different return periods and peril (fluvial and pluvial flood). (B) Category of waste management facilities based on the Environment Agency in England Waste Data Interrogator at risk of flooding for each pluvial flood returning period organised on stack bar graph. (C) Estimated amount (million tonnes) of Microplastic

Releasers at risk of flooding for different return periods and source (fluvial defended and pluvial flood).

The present Chapter does not aim to advance the methodology for evaluating the mechanisms and rate of fragmentation of plastic debris. However, an estimation of the potential quantities of MPs originating from waste facilities could be made by applying the findings from fabric tearing in laundries as an early indicator of potential degradation rates, acknowledging that recycling and environmental specific studies are needed. Based on an analysis of various studies referenced in Section 4.2.1, the estimated range of microplastics released from fabric tearing during laundry processes is approximately 42-580mg per kg of fabric. When this average is applied to MPRs at risk in a 10-year fluvial flood return period (1.02E+09kg), the potential MPs at risk of inundation falls between 4.28 and 5.91 billion mg. For more precise predictions of MPs concentrations, future studies should consider collecting waste samples from disposal sites. This would allow for a more accurate assessment of the quantity, shape, type, and density of MPs.

4.4.2 Microplastic Releasers (MPRs) at risk of flood per local authority: emerging spatial patterns

Figure 4-5 spatially represents the MPRs distribution in the UK summarised by Local Authorities (LA) at risk of pluvial and fluvial flooding for high (1 in 10-year), medium (1 in 200-year) and low likelihoods (1 in 1,000-year) scenarios. Results were normalised based on local authorities' area (km²) in the GIS environment and graphically divided into same values intervals to allow a direct comparison among different likelihoods. This is used to identify hotspots where elevated concentrations of MPRs at risk of flood require further analysis at a local scale.

Figure 4-5. Amount of managed waste at risk of releasing microplastics in flood waters for different flood likelihoods and sources (fluvial defended and pluvial), summarised and normalised per Local Authority area (km2).

Regionally, in Scotland, the impact of the 10-year fluvial flood on waste facilities is particularly high in Fife (141 tonnes/km² of MPRs), followed by Moray (34 tonnes/km²), East Lothian, and North Ayrshire with respectively (14 and 10 tonnes/km²). Interestingly, the numbers estimated for both the cities of Glasgow and Edinburgh are significantly lower compared to areas in the immediate proximity, suggesting waste is mainly received and treated by facilities located in other districts (a tendency encountered also for other main cities in England, Wales and Northern Ireland). In England, MPRs are concentrated in the area north of London including the authorities from Peterborough through Cambridge and Harlow, with an average of 70 tonnes/km² and highest concentration between Peterborough and Cambridge (212 tonnes/km²). The impact on Wales is predominantly localised in Caerphilly, north of Newport, with a high number of 1,040 tonnes/km² for a total of ~300,000 tonnes of MPRs at risk of flooding in the district. In terms of medium and lower fluvial flood likelihoods, the local authority of Leeds stands out with an increase in the MPRs' concentration from 1 tonne (high likelihood) to 74 and 77 tonnes/km² respectively.

The overlap of pluvial flood map extent on MPRs for the 1 in 10-year is very much similar to the fluvial, but far more interesting is the impact estimated for the medium likelihood. For example, in the local authority of Moray, Scotland, the number of MPRs affected is 3 times higher: from 34 (high likelihood) to 97 tonnes/km² (medium likelihood), for a total of ~220,000 tonnes if considering the entire district's surface. A similar significant increase is evident for the Shetland Islands, rising from 1 to 101 tonnes/km² from high to medium likelihood. In England, when considering LAs with quantity of MPRs higher than 50 tonnes/km², several spatial patterns can be recognised (Figure 4-5). In particular, in addition to the districts already mentioned located north of London, another hotspot with an average of 230 tonnes/km² can be recognised between Blackburn, Liverpool, Manchester and Stoke-on-Trent. The spatial pattern with the highest concentration of MPRs at risk of medium pluvial flood likelihood is a corridor of 13 LAs in between England and Wales, from Telford south to Bristol and then splitting both west toward Swansea and south in direction of the South Somerset districts; data suggests an average of 700 tonnes per square kilometre. Finally, in the low likelihood but high intensity pluvial flooding, numbers are generally increasing compared to medium and high likelihood with peaks in West Lothian (1,437 tonnes/km²), Newport area (4,456 tonnes/km²), the zone in between Oxford and Luton (Aylesbury Vale with 3,416 tonnes/km²), and Belfast with the highest concentration in the UK: 4,789 tonnes/km².

4.5 Discussion

This research presents a novel methodology based on publicly available data on the characteristics and quantity of waste annually received by waste management facilities. A new terminology was introduced to refer to waste able to deteriorate and fragmentise into synthetic microplastic components: Microplastic Releasers (MPRs). MPRs are a selection from the European Waste Catalogue code, comprised of (1) plastic waste (as defined by the European Commission in 2018), (2) synthetic textile and rubber waste (well-known for releasing fibres and micro particles under mechanical stress), and (3) mix and undifferentiated materials which are likely to contain different sizes and types of plastic items, synthetic clothes, and rubber components. The new terminology was essential to enable a quantitative estimation of plastic waste occurring in waste facilities across the UK.

Quantities of MPRs at risk of fluvial and pluvial flood were estimated for the UK for different return periods. Coastal flood risk was not taken into consideration at this stage, but its inclusion is strongly recommended in future analyses, including considering global sea level rise projections. Results from the simulated impact of flood on MPRs were significantly higher for pluvial flooding compared to fluvial flooding, with a 10-fold increase from the high likelihood to the low likelihood (from approximately 1 to 11 million tonnes). The biggest increment was a 3-fold increase between 20 and 50-year return periods. Interestingly, a similar result was obtained by Roebroek *et al*. (2020) in their global assessment of flood induced mobilisation of plastic, where mismanaged plastic waste was generically estimated as a percentage of total waste generation per country relative to gross domestic profit, where they reported the biggest increment (4-fold increase) between 20 to 50-year return periods versus the 3.5-fold increase between 1 and 10-years. This represents the average at the global scale of approximately 151 tonnes of plastic mobilisation potential per administration unit were estimated (accounting for flood defences) for the 1 in 20-year return period, which became ~612 tonnes for the 1 in 50-year return period. Additionally, the quantity of MPs potentially at risk of inundation was estimated for the 10-year return period as between 4.28 and 5.91 billion mg.

The methodologies introduced can be applied elsewhere providing appropriate supporting assumptions and the limitation in data are considered. MPRs were selected from all the available origins (households, commercial and industrial activities, construction, and demolition and excavation), by considering only waste that is physically solid, therefore, sludge from waste-water treatment plants, leachate from landfills, used oils and derivate were not included. While some primary sources of MPs, such as wastewater treatment plants and landfill leachate, were excluded at this stage, they are recognised as significant and are recommended for consideration in future studies. In addition, due to the complexity and uncertainties related to the movement and storage of waste in a country's waste management system, the length of time required by plastic items to fragment into different sizes of microplastics was not included in the scope of the present work. Further studies could assess the feasibility of adding a modelling component to take into account (1) the eventual transport and fate of MPs already present in flood waters, (2) the waste storage periods within facilities, and (3) the extent of MPR degradation into MPs during those storage periods. By doing so future work will improve the introduced methodology to estimate the contribution of waste facilities to the total load of MPs in freshwaters due to flooding. The estimation of the footprints of the facilities was achieved through the creation of buffer areas based on the average size of the INSPIRE Index Polygons per facility type and country. Additional data about actual facility dimensions and boundaries description could be included in future mandatory reports on annual quantity of waste received. The latter would be extremely beneficial for more accurate predictions since the quantities of waste at risk of releasing MPs in flood waters were estimated based on the overlap between flood maps and the footprint of the facilities. Another useful piece of information would be data on flood velocity that could be used to estimate potential impact scenarios and significantly improve the assessment of waste at risk of flooding. At this stage, partially because of the adopted national scale, and partially due to the lack of available data, site-specific analysis on flood risk exposure could not be implemented. This includes the consideration of the design of the waste facilities, waste containment systems, and the existence of flood protection measures other than flood defences included within the Fathom flood risk map extents. In addition, available data on waste received for each waste management facility could be improved by including information on waste storage conditions and the build-up period (residency time) to allow the simulation of multiple days' flood scenarios. Finally, the adopted methodology to estimate the amount of MPRs buried in landfills for the period 2007-2018 was the first step in understanding the risk they pose when located in areas at risk of flood. No additional mechanisms of the landfill sites were considered, such as leachate formation, percolation through the body, decomposition stage for different type of waste, or erosion.

4.6 Conclusion

In this Chapter, a framework was developed to identify waste using the European Waste Catalogue (EWC) code that is at risk of releasing MPs. This goes beyond the traditionally recognised plastic waste, enabling the estimation of both the quantity and location of potential MP sources at the national scale, provided that waste datasets are publicly available. The methodology was applied to the UK to estimate the quantity of waste at risk of flooding which could lead to MPs' mobilisation in flood waters. Daily quantities of Microplastic Releasers ware estimated for all waste management facilities for the year 2019, only for the landfills operative in the same year, an accumulation factor has been considered by summarising annual quantities of MPRs received from 2007 to 2018 to approximate the real conditions potentially faced in case of inundation.

Results show the impact of pluvial flood being much higher compared to fluvial which can be partially explained by the simulation behind the flood map extents where floods were created both from overflowing water bodies and independently from them, but also it further proves the necessity of assessing the risk related to present and future extreme rainfall events. Results at the national scale were investigated further with the identification of spatial patterns at the local scale for pluvial and fluvial floods for the high, medium and low likelihood. Quantities of MPRs at risk of flooding were combined per Local Authority and normalised per area (km^2) identifying UK hotspots in need of future research in terms of risk management and mitigation measures at the local scale. Depending on the localities, stakeholders and policymakers could rethink the location of existing and new waste management facilities outside flood-prone areas, if the location cannot be changed, mitigation measures can be applied both to the flood origin and pathways with additional flood defences, and by intervening on site-specific containment systems able to limit the mobilisation of synthetic micro components during an event of flood.

5 Discussions and recommendations

5.1 Introduction

Worldwide, approximately 80% of ocean plastics comes from inland sources (Li *et al.*, 2016). Plastic debris (from larger items to micro and nanoplastics) can enter fluvial systems through mismanaged plastic waste (Lebreton and Andrady, 2019) and land-based activities including landfills, domestic waste water and industrial activities (Dris *et al.*, 2018, Lebreton and Andrady, 2019). The risk posed by waste management facilities at the national scale has been addressed in Chapter 3 with the identification of hotspots where the potential impact on receptors is particularly high, and in Chapter 4 by investigating the waste materials received by disposal sites that can deteriorate and fragment into synthetic micro components.

This Chapter has the goal to review a selected number of datasets and methodologies previously introduced to discuss limitations, constraints, and recommendations for further risk assessment on waste management facilities. Sections 5.2.1, 5.2.2, and 5.2.3 focus on limitations and constraints for national scale assessments, while Section 5.2.4 discusses the advantages of a local scale investigation. Recommendations for future work are reported from Section 5.3 onwards. The applicability of existing microplastics (MPs) fragmentation rate and transport models within the risk assessment methodology is introduced and discussed as key to advancing the research on contaminant release from disposal sites.

5.2 Limitations and constraints

5.2.1 The importance of harmonising datasets in the UK: a call to action

Dataset harmonisation is the process of aligning datasets collected from different sources or by different methods to ensure compatibility and comparability. In the context of European Union (EU) and UK legislation, dataset harmonisation is crucial to ensuring compliance with legal requirements, facilitating data sharing, and promoting transparency and accountability. In the UK several regulations, acts and initiatives were introduced over the past decade to promote the standardisation of datasets among countries. For example, the Open Data Institute (ODI) is an independent organisation established in 2012 that works with government and private sector organisations to promote accessible, usable, and understandable data. The Digital Economy Act 2017 and the Data Sharing and Governance Act 2019 are measures introduced to improve the sharing and use of public sector data, including the creation of a new code of practice for data-driven public service delivery and the provision of a legal framework for the sharing of data between public sector bodies to facilitate better decision-making and service delivery. The Joined-up Data Standards (JUDS) is a project led by NHS Digital to develop data standards and a common vocabulary for health and care organizations across the UK. These regulations and initiatives have been created to ensure that data is managed, shared and used for the benefit of people in the UK, enabling better collaboration and informed decision-making across sectors.

Throughout this research several openly available datasets for Scotland, England, Wales and Northern Ireland were used for risk assessment purposes at the national scale. The datasets have different information organisation and/or are based on different classification systems/methods, which makes the comparison and the identification of trends and patterns challenging. A selection of key datasets is reported below, each of which explores harmonisation issues for discussion.

The information reported by waste datasets (EA, 2019d, Natural Resources Wales, 2019, SEPA, 2019b, NIEA, 2019) is not consistent: Northern Ireland stores information on quantity and typology (European Waste Catalogue (EWC)) of waste but the two are not linked together, therefore the estimation of quantities of plastic waste received per facility cannot be estimated. Another inconsistency is the categories associated to waste management facilities: while England differentiates among site category (8 categories) and facility type (60 typologies), Scotland reports only the site category (12 categories), and often more than one is reported at the same time: such as metal recycler / transfer station, or landfill / civic amenity / transfer station / composting, resulting in (1) less information on the type of facility, and (2) ambiguity when different typology are reported for the same site.

Other data harmonisation issues were found when comparing the Index of Multiple Deprivation (IMD) (Scottish Government, 2020b, Wales Government, 2019, English Government, 2019): the size of IMD areas and the ranking system varies per country, which means that the same score can be associated to different deprivation areas relative to the country (e.g., most deprived areas in England may have the same score of medium deprived areas in Wales or Scotland but the true economic deprivation is not necessarily the same).

The same issue arises to the distinction between rural and urban classification (DEFRA, 2011, Scottish Government, 2020a, Scottish Government, 2020c) that is also non-comparable between countries since the indicators used to define them are different. This poses two problems: the number of classes considered in between rural and urban is different, and the sizes of the areas are therefore non comparable because of the different indices used to define the classes (i.e., England defines settlements with less than 10,000 resident population as rural, while for Scotland is less than 3,000 people).

Because the datasets in their original form exhibited issues related to harmonisation, some inaccuracies may result when estimating and comparing the vulnerability indices for different countries. This limitation is exacerbated when looking at waste facilities located at the border between different countries. If considering a buffer of 5km around waste management facilities in England and Wales, 87 buffer areas intersect both countries for example. This means that potential contaminant dispersed in flood waters from waste facilities could affect both countries but the severity of the impact based on environmental receptors is estimated only in relation of the country of residence. Thus, potential underestimation or uncertainty in estimation is likely to happen.

5.2.2 Improving the access and use of flood maps for risk assessment

Another example of key datasets in need of harmonisation, standardisation, and greater public accessibility is flood maps. Fathom-UK (SSBN UK Limited, 2021a) flood maps were adopted for this research because of the lack of accessibility of flood maps used by governmental environmental protection bodies in the different UK countries. Floods can vary spatially and detailed studies are needed to model flood hazards with high resolution. Flood hazard maps are a very important and powerful tool; they provide the basis for discussion around the vulnerability of people, the environment, and the assets at risk. Globally flood maps are not standardised, which makes direct comparison an impossible task (Dohmen *et al.*, 2016). Variations may be due to different data availability and quality, regulatory framework, risk management objectives, funding and resources, etc. Different initiatives have been created in recent years aiming to promote consistency and comparability of flood maps across different regions and countries (i.e., the Global Flood Partnership, the United Nations Office for Disaster Risk Reduction (UNDRR), and the World Meteorological Organization). In Europe, the European Directive on the Assessment and Management of Flood Risks (2007/60/EC) known as the Floods Directive, aims to establish a framework for assessing and managing flood risks across the European Union (EU). The Flood Directive requires EU member states to produce flood risk maps and management plans using a common methodology and classification system. This helps to ensure that flood maps are consistent and comparable across different member states.

By looking into how different countries in the UK responded to the Directive guidelines, two main aspects are particularly relevant to this Chapter: (1) flood data typology and availability is not consistent among different countries; and (2) although the European Directive on flood risk specifically refers to the importance of including information on potential sources of pollution that could increase the impact of flooding on receptors, this still represents a considerable gap for most of the countries in the UK.

5.2.2.1 Inconsistency among flood maps in the UK

Table 5-1 shows the inconsistency across available information (at the time of writing) on type and characteristics of flooding adopted at the country level. The comparison of the different hydrological and hydraulic modelling approaches behind the flood maps are outside the scope of this Chapter, but further harmonisation studies should also consider resolution, digital elevation model, validation, and so forth. Additionally, this study does not take into account information on coastal erosion and coastal flooding. Future work should address this by integrating coastal effects datasets (e.g., using Dynamic Coast tools). The Scottish *Flood Risk Management Maps* (SEPA, 2022b) are the most comprehensive (some have been downloadable since summer 2022), containing information on water depth and velocity for the low (once in every 1,000-years), medium (once in every 200-years), and high (once in every

10-years) likelihood from river, surface water and coastal floods. The maps are also the only ones in the UK to include the contributing factor posed by groundwater (for the low risk only), intended as a type of flooding generated by water rising up from underlying rocks or flowing from springs, which can influence the duration and extent of flooding from other sources. Future extent of floods by the 2080s is available separately in the *Flood Hazard And Flood Risk Information* map (SEPA, 2022a) for river and coastal medium likelihood (1 in 200-year return period). In England flood maps are divided between the flood risk map: *Development Planning And Flood Risk Assessments* (EA, 2021) that reports river and sea flooding data only. The flood type is divided into zones: zone 1 (low probability), zone 2 (medium probability), and zone 3 (high probability). Flood defences and water storage areas are also included. The second map available for England is the *Long Term Flood Risk Maps* (EA, 2019c) that includes the risk of flooding from surface water, the velocity of flooding (less of over 0.25 m/s) for the low (1 in 100 to 1 in 1,000-years), medium (1 in 30 to 1 in 100-years), and high (greater than 1 in 30-years) risk. The direction of the water flow and the extent of flooding from reservoirs are also displayed. No information is available for future risk, nor to the impact of flood on population, economic activities, or environmental areas. Flood maps in Wales are also divided between *Flood Maps for Planning* (Natural Resources Wales, 2020a), and *Flood Risks Assessment Wales Maps* (Natural Resources Wales, 2020b) and some are available for downloading through the DataMapWales catalogue (available at https://datamap.gov.wales/). The first one shows a similar division in flood zones observed for England, with the addition of recorded flood extents, flood risk from reservoirs and coastal erosion. While the *Flood Risks Assessment Wales Maps* expands the flood assessment by reporting the low, medium and high likelihood from rivers, surface water and the sea (Table 5-1). Interestingly, the map includes flood alerts and warnings that are considered separately in other countries. Wales flood maps do not presently consider potential pollution sources, groundwater flooding, and future projections. Flood maps for Northern Ireland are together in the *Flood Hazard & Flood Risk Maps for NI* (NI, 2023). The map reports detailed climate change projection for surface water, and floodplain for rivers and the sea. Historic flooding is also included while current flood hazards (rivers, sea and surface water) are viewable through PDFs accessible from the map.

Table 5-1. Type and characteristics of flooding considered by country-specific maps in the UK for a direct comparison.

	Scotland	England	Wales	Northern Ireland

Type of flooding	River, surface water, coastal	River, surface water, coastal	River, surface water, coastal	River, surface water, coastal
Likelihood (flood risk)	High (1:10), medium (1:200), low (1:1,000)	Zone 1 (less than 1:1,000), Zone 2 (1:100 - 1:1,000 of river flooding; or 1:200 – 1:1,000 of sea flooding), and Zone 3 (1:100 or greater of river flooding; or 1 in 200 or greater of sea flooding)	For rivers: high (up to 1:30), medium (1:30 – 1:100), and low (1:100 – 1:1,000). For the sea: high (up to 1:30), medium (1:30 – 1:200), and low (1:200 – 1:1,000)	High (1:10), medium (1:100), low (1:1,000)
Extent	Yes	Yes	Yes	Yes
Depth	Yes	Yes	No	No
Velocity	Yes	Yes	No	No
Flood defences	Yes	Yes	Yes	Yes
Areas benefitting from flood defences	No	No	Yes	Yes
Groundwater	Yes	No	No	No
Impact of flooding	Yes	No	No	Yes (available separately)
Natural flood management	Flood risk from reservoirs, water storage area, runoff reduction, sediment management, estuarine surge attenuation, and wave energy dissipation	Flood risk from reservoirs, water storage area	Flood risk from reservoirs, water storage area	No
Natural susceptibility to coastal erosion	Predominant direction of sediment movement	No	Shoreline management plan and coastal erosion	No
Historical floods	No	Yes	Yes	Yes
Flood alerts and warnings	Yes (available separately)	Yes (available separately)	Yes	No
Climate change, future projection	Yes	No	No	Yes
Download availability	Yes (some)	No	Yes (some)	No
Flood map name and link	SEPA Flood Maps https://map.sepa.org.uk/floodmap/map.htm	Development planning and flood risk assessments https://flood-map-for-	Flood Map for Planning https://naturalresources.wales/flooding/flood-map-for-	Flood Maps (NI) https://dfi-ni.maps.arcgis.com/apps/webappviewer/index.h

		planning.service.gov.uk/ Long term flood risk https://check-long-term-flood-risk.service.gov.uk/map	planning-development-advice-map/?lang=en Flood Risks Assessment Wales Maps https://naturalresources.wales/flooding/check-your-flood-risk-on-a-map-flood-risk-assessment-wales-map/?lang=en	tml?id=fd6c0a01b07840269a50a2f596b3daf6

5.2.2.2 The importance of including potential sources of pollution in flood maps

In 2018, Scotland introduced a consideration of what is at risk of flooding (impact on population, economic and community activities and environmental sites) within the National Flood Risk Assessment (NFRA) data explorer tool for the first time (SEPA, 2018). In 2022, the impact of flooding was also added to SEPA *Flood Risk Management Maps* (SEPA, 2022b). For the first time, the Integrated Pollution Prevention and Control (IPPC) installations are mentioned specifically (instead of the generic "industry" category), and the location of those is provided within the flood map. A step further, for the scope of this Chapter, was taken with the indication of the environmental sites potentially affected by IPPC installations for the Potentially Vulnerable Areas (PVAs), which is where significant flood risk exists now or is likely to occur in the future. While the map information may not explicitly indicate flooding as the primary cause of pollution from IPPC installations, the identification of environmental sites that could be impacted by industrial facilities does address the broader concern of environmental risks associated with IPPC installations and industrial activities in general.

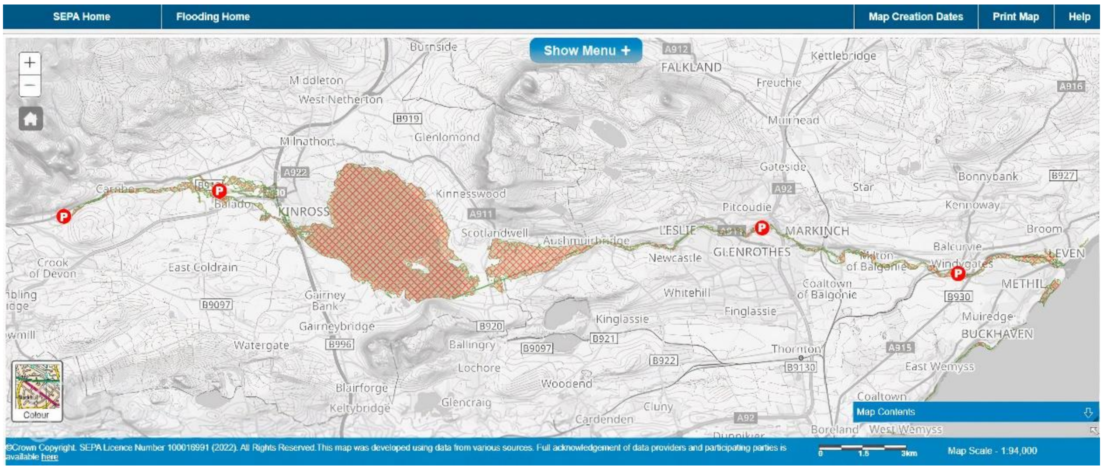


Figure 5-1. Extract from SEPA flood maps (SEPA, 2022b) with the location of Integrated Pollution Prevention and Control (IPPC) installations (red dots) and the environmental sites potentially affected by IPPC facilities between the cities of Kinross (Perth and Kinross) and Leven (Fife) now included in the flood map.

The consideration of potential point sources of pollution on flood maps represents a first step in recognising a link between the flood hazard and factors that can increase the impact of flooding by contaminating flood waters. The lack of information represents an existing gap within the majority of flood risk maps in the UK. Even the countries that display the position of industrial facilities (Scotland and Northern Ireland), they only refer to IPPC installations. Other industrial activities such as Control of Major Accident Hazards (COMAH) sites (European Parliament, 2012), and Waste Management Licences (WMLs) should be considered. The gap is a missed opportunity to assess the exposure of potential contaminant releasers to inundation and to promote further assessments of the risk at the national and local scale.

5.2.3 The challenges of hazardous waste classification for plastic components under the European Waste Catalogue

In Chapter 1 of this Thesis, the Water Risk Index (WRI) methodology was selected from the literature because of its capacity to identify substances dangerous for people and the environment when in contact with water. The method is particularly effective if applied at the national scale when only information on the typology and quantity of substances received on-site are available. Due to the different classification systems, the WRI was applied to the European Waste Catalogue (EWC) that classifies discarded materials as hazardous or non-hazardous based on the Hazard Property Code (HP Code), such as explosive, irritant, carcinogenic, mutagenic, etc. Although plastics can contain chemical hazardous additives to give colour/transparency and to improve resistance to degradation (Campanale *et al.*, 2020), the EWC classifies plastic waste as non-hazardous unless specifically contaminated by materials with hazardous properties (17 02 04 glass, plastic and wood containing or contaminated with hazardous substances). The legislation has not seen updates on the subject since its creation in 2000 (2000/532/EC). In particular, as investigated in Chapter 4, despite the publication in 2018 of the *Commission Notice On Technical Guidance On The Classification Of Waste* (2018/C 1 24/01), the hazardous properties of plastic and the recognition of waste materials that can disintegrate in synthetic micro-components has not been assessed leaving the waste classification system with several limitations:

- ❖ The waste classification that explicitly refers to plastic waste is reductive compared to the number of codes referring to mix/undifferentiated materials where plastic can likely be contained. The European Commission notice (2018/C 124/01) identified 29 codes (3.4% of the total) that are likely to contain plastic, when only 9 of those include the word "plastic" in the description. In comparison 12% of the total 843 codes refers to mix/undifferentiated materials, waste from composite materials, and waste not otherwise specified;
- ❖ The waste classification does not take into consideration the potential of certain materials to deteriorate into synthetic micro components (such as tyres, textiles, and others identified in Chapter 4 as potential Microplastics Releasers (MPRs)) and the hazardous properties of those micro components. In the United States, Europe, Australia and Japan, plastics are classified as non-hazardous solid waste (Rochman *et al.*, 2013), even though more than 50% of the ingredients commonly found in plastics (e.g., plasticizers, stabilizers and pigment) are labelled as hazardous by the United Nations' Globally Harmonised System of Classification and Labelling of Chemicals (Lithner *et al.*, 2011).

Degradation processes will fragment and disintegrate plastic materials forming a broad particle size and shape distribution (Lambert *et al.*, 2017). The degradation pathways depend on environmental conditions and their forecasting is an important aspect when considering waste management facilities as potential inland MPs sources. The same applies to the fragmentation rates for different substances which will help targeting the overall risk assessment. Unfortunately, missing national data about the length of time waste is stored within a waste facility, storing conditions, and the level of protection adopted against flooding, are impacting on eventual forecasting methodologies at the national scale. Investigating the fragmentation rate is material for a local scale exploration.

5.2.4 *Proximity of risk and vulnerability of receptors: the potentiality of a local scale risk assessment*

In Chapter 3, a multi-index risk assessment methodology was established to highlight waste facilities at risk of potentially causing accidents involving people and the environment if flooded. The exposure to flooding was assessed by considering the overlap between flood map extents for different return periods and the footprints of waste facilities. When the site footprint is inundated the receptors located within specific buffers (1 or 5km of diameter) from a waste site are marked as potentially affected by contaminated flood waters, despite their actual location in respect to the flood extent. The methodology is effective to highlight hotspots where higher levels of risk may lay at the national scale. However, at the local scale, the approach should be re-evaluated and cross-checked using on-the-ground observations. Priority should be given to the top ten Local Authorities listed as having the highest risk index (see Table 3-10). Specifically, the City of London is identified for the 1 in 10-year return period (covering both fluvial and pluvial risks), while the Shetland Islands stand out for the pluvial risks associated with 1 in 200 and 1 in 1000-year return periods. The identified Local Authorities should consider expanding upon this research by developing:

- ❖ Flood maps with higher resolution and complete flooding description (e.g., updated flood maps now available from SEPA including water depth, velocity, direction) from rivers, surface water and the sea including climate change projections;
- ❖ Collation of additional information including location, containment systems and length of stay on site for materials that can disintegrate into MPs to assess fragmentation mechanisms and rate for specific type of plastic waste;
- ❖ Assessment of MPs transport and fate in aquatic flows during inundation given by fate-transport models; and
- ❖ Collection of samples within waste facilities and in the proximity of disposal sites as data input and validation of the MPs fate-transport model.

5.3 Recommendations for future risk assessment

5.3.1 Enhancing existing datasets: towards harmonisation, standardization, and greater public accessibility

The harmonisation and/or standardisation and accessibility of datasets in the UK is essential to allow the direct comparison of results that is useful to identify regional variations, evaluating policies, and advancing research. For example, a direct comparison of the IMD across different countries would help highlight areas of high deprivation and target interventions and resources available at the national scale towards those areas. The direct comparison of the classification of urban and rural datasets would also impact the identification of areas with different demographic characteristics and inform policymaking and resource allocation. The latter can be particularly important for healthcare, where the needs of rural communities may differ from those of urban communities. An attempt to harmonise the IMD datasets in the UK was made by Abel *et al*. (2016) by developing adjusted indices of multiple deprivation that can be used to compare levels of deprivation across different geographic areas in the UK. The method applied indirect standardisation to mortality rates, which accounted for differences in age, sex, and cause-specific mortality rates between areas. The results showed that the adjusted IMDs enabled the comparison within and between different regions and countries of the UK. Despite some limitations in the methodology that relies on mortality data, which may not capture all aspects of deprivation, the adjusted IMDs could be useful for informing policy and targeting resources to areas with the highest levels of deprivation and mortality.

The harmonisation of flood maps across the UK would benefit from existing country-specific knowledge and resources, and improve the accuracy of flood risks assessment, resource allocation, and effective flood management strategies at the national scale. Standardised flood maps should be freely downloadable (to allow further research) and should report the extent of flooding from different sources (i.e., river, surface water and the sea), including groundwater, for high, low, and medium likelihoods. The latter should be defined by a specific selection of flood return periods for consistency (e.g., the low likelihood can be represented by the 1 in 100-year flood return period). The extent of flooding should include information on the flood depth, velocity, and direction. Maps should display information on flood defences and natural flood management such as floodplain storage, runoff reduction, etc. Historic flood extent, future projections, and flood alerts and warnings should also be incorporated in flood maps. Finally, the factors that can increase the impact of flooding (due to potential flood water contamination) should be reported by locating industrial facilities vulnerable to inundation such as IPPC installations, COMAH sites, and waste management facilities. The latter should also include an evaluation of the environmental areas that could be affected by pollution in case of release from the industrial installations mentioned above.

5.3.2 Tackling microplastics in municipal solid waste

In addition to the chemical components of plastic at the manufacturing stage (i.e., plasticisers, flame retardants, UV stabilisers, and pigments), plastic debris have greatly specific surface areas, suggesting that they possess significant adsorption capabilities for highly toxic pollutants in both aquatic and soil environments (Zhao *et al.*, 2022). With global annual production of thermoplastics expected to reach 445.25M metric tons in 2025 and a further increase of more than 30% between 2025 and 2050 (IEA, 2020), we recommend policymakers tackle the current limitations in the classification of plastic (reported in Section 5.2.3) with the following:

- Recognise the presence of environmental hazardous chemicals into plastic (e.g., resins can release Bisphenol A at high temperature) and consequently initiate a scrutiny process with the goal of associating hazardous properties to different types of plastic (such as PET, HDPE, PVC, LDPE, PP, PS, and other). Declaring certain plastics as hazardous will help both with the recording and therefore control of the movement of plastics through the waste system and it would rectify the storing techniques for hazardous plastics to avoid being accumulated outdoors with no containment systems in place;
- Initiate a discussion around other materials that can release synthetic micro components such as fibres, spheres, pellets, lines, sheets, flakes, and foam (Kooi et al., 2018), to investigate and revise the current storing techniques to minimise the dispersion of MPs during transportation, temporary storage, treatment and final disposal; and
- Standardise and decrease the number of codes referring to mixed and undifferentiated materials to boost the separation and recycling rate of materials. In England in 2019, the amount of waste received by waste facilities classified as mixed and undifferentiated was approximately 850,000 tonnes (EA, 2019d).

5.3.3 Flood characterisation, MPs fragmentation rate and transport at the local scale

The local scale risk assessment should be estimated by giving priority to Local Authorities (LAs) with higher level of risk were identified using the national scale risk assessment (Section 3.6.4). For each local authority country/regional level, flood maps can be used (if reporting all the information needed is available), otherwise watershed level simulations of flooding using hydrologic and hydrodynamic models is recommended. Flood maps with higher resolution are more accurate and provide more detailed information about the water flow, physical geography (e.g., landforms, vegetation, topography, etc.), and morphological information (vegetation, buildings, roads) that can affect flood risk. This increased accuracy can reduce uncertainty when flood extent is overlapped with the footprint of waste facilities and potential receptors. Flood maps should report information on the spatial extent, water depth, velocity, direction (and length of the event ideally) for the low, medium, high flood likelihood including climate change projection. Flood characterisation can be used to establish the potential damage that could occur during a flooding event to above-ground tanks, pipelines, and industrial sites through the assessment of depth-damage probability curves (Antonioni *et al.*, 2009, Landucci *et al.*, 2012, Cozzani *et al.*, 2014, Landucci *et al.*, 2014, Antonioni *et al.*, 2015). The latter can replace the water depth intervals introduced in Section 3.3 for national scale risk assessment with limited information on flooding, to increase accuracy in the assessment of contaminant mobilisation due to inundation.

5.3.3.1 Fragmentation rate and mechanisms of plastic waste

To reduce the uncertainty and initiate a first assessment on the fragmentation rate of plastic items into micro components within waste facilities, additional information should be collected at the local scale about (1) residence times for type of plastic waste temporary stored; (2) storage conditions (inside/outside, with/without containment systems); (3) mechanical treatments for plastic waste alongside details of the mix and undifferentiated materials; and (4) the level of protection adopted against flooding and the containment/control of runoff or flood water. The fragmentation of bigger items into micro components is due to a combination of physical, chemical, and biological processes. Environmental stresses involve mechanical treatments (tearing and compression due to direct physical treatment or accidentally during transport and storage), thermal degradation, and direct exposure to rain, wind, and sunlight. Photo-oxidation and hydrolysis are reported as the most significant mechanisms of environmental degradation (Gerritse *et al.*, 2020, Masry *et al.*, 2021). The exposure to sunlight (very frequent when waste is stored outside on pavements or inside containment tanks without ceiling) weakens and embrittles the plastic affecting the fragmentation rate, including during mechanical stresses *(Andrady et al., 2022).* In addition, during flood events the mobilisation of waste through turbulent mixing, collision and swelling-deswelling can further fragment weakened plastics *(Andrady et al., 2022).* Observations of the process of weakening, embrittlement, and fragmentation due to exposure to rain, sunlight, and mechanical stresses have been made both on beaches (Corcoran *et al.*, 2009) and in surface water environments (Garvey *et al.*, 2020, Alimi *et al.*, 2022) and provide an insight into degradation rates that could be incorporated into future risk assessments.

5.3.3.2 MP fate-transport model in freshwater systems

The key properties of MPs that can influence the transport behaviour and fate in the aquatic environment are the buoyancy/non-buoyancy, size (macroplastics are larger than 5mm, microplastics range from 5nm to 100nm, and nanoplastics are smaller than 100nm (Kooi *et al.*, 2018)), density (related to the production volumes of the different polymers), and shape (i.e., fragments, fibres, spheres, pellets, lines, sheets, flakes, and foam). By taking in account the MPs unique properties, because MPs exhibit behaviours that are in part similar to sediment particles, Kooi *et al.* (2018) suggests that sediment transport models can be utilised to simulate their fate and transport in aquatic systems. Ockelford *et al*. (2020) proved the concept by investigating the concentration of MPs in the sediment before and after flooding events, finding a decrease in concentration of MP after floods indicating that fluvial processes can effectively remove MPs contamination from river channel beds in a short amount of time. Another model that could be considered for this assessment that incorporates a sediment transport (and surface runoff) module is the INCAcontaminants (Nizzetto *et al.*, 2016). INCAcontaminants also considers emissions from sewage sludge, surface runoff, effluents from wastewater treatment plants, advection, settling, resuspension, and store depletion, which makes it particularly suitable for assessing the potential dispersion of MPs from waste facilities. However, among the gaps identified by Kooi *et al.* (2018) in existing plastic debris transportation models (including INCAcontaminants), the lack of detail about the MPs actual size, shape and polymer density distributions is crucial to address to advance the research. The characterisation of plastic in environmental samples from within waste facilities and in the proximity would address the gap by providing information on plastic debris size, distribution, and shape classification.

5.4 Summary of key issues

In this Chapter, three critical issues related to environmental management in the UK were discussed and recommendations introduced. Firstly, the need to enhance existing datasets through harmonisation, standardisation and greater public accessibility was highlighted to enable direct comparisons between regions and countries. The study on adjusted Indices of Multiple Deprivation made in Abel *at al.* (2016) was described as an example of the advantages of a direct comparison of levels of deprivation across different geographic areas in the UK. The lack of availability and the inconsistency of information among country-specific flood maps in the UK was also investigated and a standardised approach suggested.

Secondly, the current limitations and inconsistency in the classification of plastic waste were discussed and recommendations were listed for policymakers to address the potential harmful consequences of microplastics originated in municipal solid waste. The key recommendations were to recognise the presence of hazardous chemicals in different types of plastic and initiate a discussion around materials that release synthetic micro components. Additionally, recommendations were made to standardise and decrease the number of classification codes for mixed and undifferentiated materials to improve the separation and recycling rate of materials.

Finally, additional recommendations for local scale risk assessments were introduced. The importance of a detailed flood characterisation is emphasised to improve spatial analysis and to reduce the uncertainty around the impact of flooding on waste facilities with the use of depth-damage probability curves. Methods and models on microplastics fragmentation rate and transport were discussed to highlight the risk posed by deteriorating waste located within disposal sites at risk of inundation. Based on the findings of fabric tearing in laundries studies, the 1,02M tonnes of Microplastic Releasers (i.e., plastic, textile, rubber, and mix waste) estimated at risk of inundation (high likelihood) (Ponti *et al.*, 2022), could release between 4.28 and 5.91 billion mg of MPs in flood waters. To fill the current gap in understanding MPs fragmentation rate, transport and fate, and improve the risk assessment methodology, it is highly recommended that future studies characterise environmental samples within and in the proximity of waste management facilities.

6 Conclusions

6.1 Summary of key findings

Through this Thesis, notable findings have been made, offering valuable insights into various dimensions of risks associated with waste management facilities and their susceptibility to flood-related impacts. The research contributes to a deeper understanding of the complex relationship between waste facilities and flooding, shedding light on critical aspects such as facility footprints, risk assessment methodologies, potential contamination risks, and the need for improved data harmonisation and standardisation. These findings have implications not only for waste management practices but also for broader discussions on environmental management, risk assessment, and the resilience of industries in the face of natural hazards. The research questions were formulated by identifying gaps based on the initial findings highlighted in the introduction. The subsequent sections present a comprehensive account of the answers to these research questions, addressing each one sequentially.

6.1.1 RQ1

RQ1: Considering the necessity of assessing the exposure of sites to flood risk, what can be used to represent the footprint of waste management facilities?

Chapter 2 investigated the lack in public waste datasets of specific georeferenced boundaries indicating waste activities, impeding effective spatial GIS analysis with flood maps. The INSPIRE Index Polygons spatial dataset (European directive 2007/2/EC), which serves as a European open-source dataset containing georeferenced polygons representing registered property information, was tested against the footprint of waste facilities. However, the direct use of these polygons was deemed inadequate for the research because (1) the limited coverage provided, which did not encompass a sufficiently comprehensive range of waste activities; and (2) the observed tendency for the polygon areas to extend beyond the actual footprint of waste activities. To address the limitations, two buffer methodologies were investigated. In GIS-based studies, buffers are commonly employed for numerous purposes. These include assessing landslide risks (Saha *et al.*, 2005), determining land-use effects on river water purity (Sliva and Williams, 2001), and general geographic data tasks (Liu *et al.*, 2015). Buffers have also been utilised in inspecting facilities from the EPA's 1994 Toxic Release Inventory to understand industrial pollution impact on the environment (Sheppard *et al.*, 1999, Chakraborty and Armstrong, 2013). However, specific buffer metrics for medium to small industrial activities, such as waste management facilities, remain absent. This Chapter concentrates on formulating a strategy to determine the best buffer sizes for recreating waste site boundaries. The first approach involved establishing reliable buffers based on the annual waste intake of a limited population sample of facilities, but the results were non-statistically significant. The second approach expanded the sample population to include 1,049 waste facilities in GB. Buffers were created based on the averaging of polygon areas per waste facility category and country. These buffers and polygons were then tested against different flood likelihood scenarios: high, medium, and low. The key results are reported below.

- As flood likelihood decreases, the number of affected sites and the severity of impact increases. Specifically, the high flood likelihood scenario impacted 228 (22%) of the 1,049 sites, while the medium and low flood likelihood scenarios impacted 289 (28%) and 335 (32%) of the sites, respectively. Furthermore, for the low flood likelihood scenario, 111 (33%) of the polygons and 117 (35%) of the buffers reported an exposure to flooding (overlap) higher than 40%. In comparison, for the high flood likelihood scenario, only 20 (9%) of the polygons and 23 (10%) of the buffers were impacted in the same manner.
- Buffers are more susceptible to flooding and exhibit a greater extent of inundation compared to polygons, particularly in situations with lower flood likelihoods. When considering scenarios where either buffers or polygons are overlapped by flood extents (with an overlap greater than 0% on buffers and 0% on polygons, or vice versa), the exposure of buffers is approximately 2.3 to 2.4 times higher than that of polygons in the low likelihood scenario. In contrast, in the high likelihood scenario, the exposure of buffers is approximately 1.5 times higher than that of polygons.
- The overall difference in flood inundation between polygons and buffers is low, indicating a largely alignment in terms of the areas affected by flooding. Among the sites impacted by various flood likelihoods, approximately 80% have a maximum overlap difference of 22%. When considering sites with at least 40% overlap, the maximum difference decreases even further to just 4% of the sites.

To explore the disparity between polygons and the actual footprint of waste facilities, an additional investigation was conducted comparing polygons with a larger set of manually verified footprints. Polygons were observed to be 3.5-times bigger than footprints, indicating a potential constraint within the methodology. To address this disparity in future studies, a scaling factor was implemented as a corrective measure.

6.1.2 RQ2 and RQ3

RQ2: How can the methodology of the Water Risk Index be expanded to consider the spatial context of waste facilities when estimating the risk posed to receptors? How can this be expressed simply to enable the comparison between sites?

RQ3: How is the risk distributed? Which sites and Local Authorities require further investigation at the local scale for risk mitigation and management?

Initial findings from the application of the Water Risk Index (WRI) methodology to waste management activities in Scotland highlighted potential environmental concerns and the need for further analysis. In Chapter 3 a novel approach is introduced to incorporate spatial factors and contextualise the risk associated with waste management facilities. The approach employs a multi-index-based assessment that represents the three components of risk: (1) hazard, (2) vulnerability, and (3) exposure. Various methods have been explored in the literature to define the three components using specific weighted indicators and parameters. Three key studies were selected for their relevance. The DRASTIC model by Linda et al. (1987) focused on groundwater vulnerability and scrutinised a selection of factors affecting aquifer contamination. Similarly, the FIGUSED method by Nerantzis et al. (2015) highlighted areas prone to flooding, taking into account geological structures and water accumulation. Finally, Arrighi et al. (2018a) adopted a multi-index approach to analyse waste management sites in flood-vulnerable regions. Because these studies targeted river basin or regional scale risk assessments, a gap was identified in national-level risk assessment, particularly regarding flood impacts on disposal locations. The methodology developed in this Chapter enables the estimation of risk on a per-facility and per-Local Authority (LA) basis, facilitating the prioritisation of actions to mitigate potential threats. Key findings are reported below divided by risk components.

6.1.2.1 Hazard

Results exhibit a consistent rise in the severity of impact as the likelihood of flooding diminishes. An exponential growth is notable when flood depth surpasses the 120cm interval. Within this range, the count of affected facilities varies from 1,400 to 3,900 sites, depending on flood likelihood and source. Comparatively, the number of disposal sites falling within the water depth range of 90-120cm is significantly lower, spanning from 10 to 500 sites. The elevated figures can be partially attributed to two factors: the overestimation of buffers in representing waste facility footprints and the tendency for waste facilities to be located near permanent water bodies such as seas, rivers, and lakes. Considering the lack of available information on coastal flooding and climate projections at the time of writing, the overestimation of buffers serves as a compensating factor. However, it is recommended that future studies incorporate these aspects to enhance the accuracy and comprehensiveness of the analysis. The close proximity of waste disposal sites to water sources amplifies their vulnerability to pollution incidents during flooding, which in turn increases the potential consequences in the event of a contaminant release.

6.1.2.2 Vulnerability

The vulnerability of receptors was assessed based on their proximity to waste management activities and by a selection of geophysical criteria. The latter were chosen to evaluate the severity of the impact and the potential consequences in case of a contaminant release. Key findings reveal the following:

- 54% of waste facilities are located within 1km of a fresh or salty water source, increasing the number of pathways in case of contaminant dispersion;
- 73% of disposal sites are surrounded by areas with high or very high permeability, making it easier for contaminated floodwaters to penetrate the soil and reach underground water if present;
- approximately 50% of waste facilities in GB are located within a 1km radius of natural protected areas. These include over 200 landfills, 600 metal recyclers, 1,000 treatment centres, and more than 1,500 transfer stations;
- about 96% of disposal sites are situated within a 5km radius of a river sample or river (in the case of Wales) with poor/bad or non-compliant surface water quality. The presence of pre-existing signs indicating poor overall water quality can affect the ability of water flows to recover in the event of pollution, thereby exacerbating the consequences in the medium to long term;
- a significant portion of waste management activities, accounting for 48%, is located in remote small towns and rural areas. Additionally, 20% of these activities are found in accessible towns and rural areas, while only 32% are situated in urban areas where waste generation is typically higher. The consequences of this distribution can have several impacts, including increased travel distances for

waste, resulting in a larger carbon footprint. Moreover, outside urban areas there is a higher likelihood of encountering permeable terrains, water sources, and protected natural areas that can increase the potential for environmental impacts; and

- results from the single-parameter sensitivity analysis indicate that the category criterion has the highest influence due to the prevalence of waste treatment and transfer stations, accounting for 32.4% and 38.5% of the total waste facilities considered. The aquatic classification criterion also exhibited significant percentages, particularly in England and Scotland, likely due to high non-compliance rates (65%) in England and poor/bad overall quality in 24% of samples in Scotland. Proximity of waste facilities to natural protected areas within a 1km radius significantly affected the vulnerability index score. England had the most facilities in this category (2,830), followed by Scotland (511) and Wales (389).

6.1.2.3 Exposure

The assessment of the exposure of disposal sites to the risk of flooding (regardless the severity of impact) indicates that pluvial flooding (surface water) has a more significant impact compared to fluvial flooding (rivers). This disparity is particularly noticeable in the medium and low pluvial flood likelihood scenarios, where approximately 60% (~4,500 sites) and 73% (~5,400 sites) of waste activities are at risk, respectively. Transfer stations and treatment centres exhibited the highest susceptibility to flooding across different flood return periods and sources, ranging from approximately 1,100 to 1,570 for high fluvial flood risk and from 950 to 3,850 for low pluvial flood risk. Metal recyclers and landfills showed a maximum of 850 and 320 sites, respectively, that could be impacted by a low pluvial flood. These findings carry significant implications, especially when considering the potential release of contaminants in metal recyclers and landfills.

Once the hazard, vulnerability, and exposure factors were combined to derive an overall level of risk for each waste facility, the resulting risk values were classified into three categories: low, medium, and high. Findings indicate that approximately 16% (around 1,200) of waste management facilities out of a total of 7,292 sites in Great Britain were estimated to have a high risk index and are situated in areas susceptible to fluvial flooding with a 10% probability in any given year. When considering flood risk with lower probabilities, the percentage increases to 19% and 21% for 0.5% and 0.1% flood probabilities, respectively. In terms of pluvial flood risk, the outcomes are even higher. Approximately 15% of sites with a high risk index are located in areas prone to flooding with a 10% probability in any given year. This percentage rises to 37% and 44% when considering flood risk with 0.5% and 0.1% probabilities, respectively. The analysis revealed two key findings in the risk assessment of facilities impacted by different flood likelihoods and sources. Firstly, the number of facilities with a high risk index consistently surpassed those with medium and low risk. Secondly, this difference is particularly pronounced in high likelihood flood scenarios, whether fluvial or pluvial, where the number of high-risk facilities was significantly higher compared to the medium-risk sites. These findings suggest that when disposal sites are affected by floodwaters, particularly in situations of high probability flood risk, the vulnerability of receptors is triggered to its full potential.

The index-based risk assessment identified the waste facilities with the highest risk index (648) across various flood likelihoods and sources, aiming to prioritise sites for necessary actions. Further local-scale studies are recommended for these identified hotspots to reduce the risk through measures such as flood protections, elevation of stored waste, and careful management of waste typology and quantity. Additionally, the distribution of waste management facilities and their risk index was summarised per Local Authority (LA) and normalised by population to further highlight the distribution of risk.

6.1.3 RQ4, RQ5 and RQ6

RQ4: apart from well-known plastic waste, which other type of waste classified within the EWC possess the potential to degrade and release microplastics (MPs)?

RQ5: what quantity of waste received by disposal sites in the UK is at risk of flooding, potentially leading to the mobilisation of MPs in floodwaters?

RQ6: which sites and Local Authorities warrant further localised studies to evaluate the level of risk associated with the release of MPs from waste management facilities?

Chapter 4 presented a framework to identify waste at risk of releasing microplastics (MPs) among the European Waste Catalogue (EWC) code. The methodology allowed for the estimation of the quantity and location of potential MP sources at the national scale using publicly available waste datasets. The fragmented substances (MPs) can then be mobilised and dispersed in case of flooding. Although recent studies have increasingly identified terrestrial sources as primary contributors to ocean plastics (Hurley *et al.*, 2018), flood-induced plastic debris mobilisation has yet to be fully understood. Rivers, in particular, have been recognised as significant conduits for the transit of plastics from inland regions to the oceans (van Emmerik *et al.*, 2019, He *et al.*, 2021, Wang *et al.*, 2021). Hurley et al. (2018) examined 40 rivers in northwest England both before and after the severe floods in winter 2015/2016. The study observed the presence of microplastics (MPs) in every riverbed analysed, and determined that floods could displace and transport about 70% to 100% of the total MPs previously residing in these riverbeds.

In recent years, the effects of flooding on managed waste have been sparingly examined. Arrighi et al. (2018a) identified wastewater treatment plants, waste handling facilities, and contaminated locations as environmental concerns due to the risks associated with contaminants in flood-vulnerable zones. Comparatively, the possible flooding of solid waste landfills has gained more academic interest (Laner *et al.*, 2009, Neuhold, 2012, Neuhold and Nachtnebel, 2011, Nicholls *et al.*, 2021), but many aspects remain unexplored. This chapter seeks to enhance the understanding of terrestrial sources of MPs by examining the likelihood of waste management facilities releasing microplastics during flood events.

The Chapter presented a novel methodology based on publicly available data on the characteristics and quantity of waste annually received by waste management facilities. A new terminology was introduced to refer to waste able to deteriorate and fragmentise into synthetic microplastic components: Microplastic Releasers (MPRs). MPRs are a selection from the European Waste Catalogue code, comprised of (1) plastic waste (as defined by the European Commission in 2018), (2) synthetic textile and rubber waste (well-known for releasing fibres and micro particles under mechanical stress), and (3) mix and undifferentiated materials which are likely to contain different sizes and types of plastic items, synthetic clothes, and rubber components. The new terminology was essential to enable a quantitative estimation of plastic waste handled by waste facilities across the UK.

Quantities of MPRs at risk of fluvial and pluvial flooding were estimated for the UK across different return periods. The amount of MPRs at risk of flooding consistently increases by approximately 30,000 tonnes per return period. Initially, for high likelihood flooding, nearly 1 million tonnes (affecting 647 facilities) are at risk, while for low likelihood flooding, this figure reaches 1.2 million tonnes (with 1,024 facilities affected). The estimated numbers of MPRs affected by pluvial flooding are significantly higher, with a 10-fold increase observed from a 5 to 1,000-year return flood event. Additionally, the number of facilities exposed to pluvial flooding shows a drastic growth from 591 to 2,472. The most significant increment occurs between the 20 and 50-year pluvial return periods, where the amount of MPRs at risk is three times higher, ranging from approximately 1.5 million to 5 million tonnes for the 50-year return period flood. It is important to note that coastal flood risk was not considered in this stage, but we strongly recommend its inclusion in future analyses, including the consideration of global sea level rise projections.

Lastly, in order to address RQ6, significant sites and Local Authorities with a notably high concentration of MPRs were identified. The impact of low likelihood pluvial flooding was found to be the most severe. The areas with the highest concentration of MPRs were West Lothian, reaching 1,437 tonnes per km^2, the Newport area with 4,456 tonnes per km^2, the region between Oxford and Luton (Aylesbury Vale) with 3,416 tonnes per km^2, and Belfast having the highest MPR concentration in the entire UK, with 4,789 tonnes per km^2.

6.2 Future applications & impact of research

Based on the key issues in UK environmental management identified and discussed in Chapter 5, future applications of the methodologies introduced and tested in the previous Chapters should consider the following improvements:

- Enhancing the resilience of industries to natural hazards: the findings of the research have implications for improving the resilience of industries in the face of natural hazards. By integrating the spatial extent of additional natural hazards such as landslides, heatwaves, wildfires, etc., into the presented methodologies, similar studies can be conducted to further assess the susceptibility of disposal sites. This information can inform the development of strategies to enhance resilience, including site selection, infrastructure improvements, and emergency response planning;
- Raising public awareness and engagement: the research emphasises the need for greater public accessibility to harmonised and standardised environmental datasets and information. Future applications can focus on improving user-friendly platforms and tools that enable the public to access environmental data and understand the factors that can increase the adverse impact of flooding. This approach can promote awareness, engagement, and participation in environmental management and decision-making processes;

- Further enhancing risk assessment methodologies: future applications should aim to enhance risk assessment methodologies such as the Water Risk Index and the multi-index based assessment of spatial factors. These methodologies can be used to contextualise risks associated with waste management facilities. Similar approaches can be applied in other industries and sectors to assess and prioritise risks based on hazard, vulnerability, and exposure factors. This can help identify areas and assets at high risk and guide the implementation of risk management measures.

In conclusion, this Thesis has contributed to a deeper understanding of the complex relationship between waste facilities and flooding, shedding light on critical aspects such as facility footprints, risk assessment methodologies, potential contamination risks including microplastics release, and the importance of improved data harmonisation and standardisation. These findings have significant implications not only for waste management practices but also for broader discussions on environmental management, risk assessment, and the resilience of industries in the face of natural hazards.

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Appendix A

Table A-1. A compilation of selected weighted criteria (e.g., WATER RISK INDEX with weight = 2), attributes (e.g., low), and corresponding values (e.g., 1) employed to define the vulnerability variable in a multi-index-based risk assessment for waste management activities.

MULTI INDEX-BASED RISK ASSESSMENT (part 1/2)							
VULNERABILITY							
WASTE FACILITY VULNERABILITY		ENVIRONMENTAL VULNERABILITY		PEOPLE VULNERABILITY		LOCATION ACCESSIBILITY	
WATER RISK INDEX per site (weight = 2)		LAND USE (weight = 1)		INDEX OF MULTIPLE DEPRIVATION (weight = 1)		URBAN RURAL CLASSIFICATION (weight = 2)	
low (<9)	1	built-up areas and gardens	1	quintile n. 1	5	**Scotland**	
moderate (9-13)	2	natural areas (mountain, heath, woodland, grassland)	2	quintile n. 2	4	Large Urban Areas	1
high (13-max)	3	arable	3	quintile n. 3	3	Other Urban Areas	2
COMPLIANCE ASSESSMENT SCHEME (weight = 1)		coastal (including saltwater)	4	quintile n. 4	2	Accessible Small Towns	3
Scotland		freshwater	4	quintile n. 5	1	Remote Small Towns	4
Excellent	1	wetland (bog)	5			Accessible Rural Areas	5
Good	1	TERRAIN SLOPE (%) (Weight = 1)				Remote Rural Areas	5
Broadly Compliant	2	< 1 (very low)	5			**England**	
At Risk	3	1-2 (low)	4			Urban with Major Conurbation	1
Poor	4	2-4 (moderate-low)	3			Urban with Minor Conurbation	2
Very Poor	5	4-10 (moderate-high)	4			Urban with City and Town	3
England		> 10 (high)	5			Urban with Significant Rural	4
Band A	1	PERMEABILITY				Largely Rural	5

		(weight = 1)					
Band B	1	very low	1			Mainly Rural	5
Band C	2	low	2			**Wales**	
Band D	3	moderate	3			Urban city and town	1
Band E	4	high	4			Urban city and town in a sparse setting	2
Band F	5	very high	5			Rural town and fringe	3
WASTE FACILITIES CATEGORY (weight = 1)		NATURAL PROTECTED AREAS (weight = 2)				Rural town and fringe in a sparse setting	4
On/In Land	2	no	1			Rural village and dispersed	5
Incineration plant	4	yes	5			Rural village and dispersed in a sparse setting	5
Waste treatment	4	AQUATIC CLASSIFICATION surface waters (weight = 1)					
Waste transfer	4	**Scotland**					
Landfill	5	High	1				
Waste storage and selection	5	Good	1				
Electronic waste treatment	5	Moderate	3				
		Poor	5				
		Bad	5				
		England					
		compliant	1				
		non compliant	5				
		Wales					
		Good	1				
		Moderate	3				
		Poor	5				

Table A-2. A compilation of selected weighted criteria (e.g., FLOOD DEPTH with weight = 2), attributes (e.g., 1-15), and corresponding values (e.g., 1) employed to define the hazard and exposure variables in a multi-index-based risk assessment for waste management activities.

MULTI INDEX-BASED RISK ASSESSMENT (part 2/2)			
HAZARD		EXPOSURE	
FLOOD DEPTH (cm) (weight = 2)		EXPOSURE (pluvial and fluvial flooding for different likelihoods) (weight = 1)	
1-15	1	water depth < 15cm	0
15-30	1	water depth > 15cm	1
30-60	2		
60-90	3		
90-120	4		
>120	5		
DEBRIS FACTOR - flood water depth (cm) (weight = 1)			
1 - 25	1		
25 - 75	2		
> 75	2		

Table A-3. Additional information on the datasets used to populate the vulnerability criteria for GB.

Criteria	Data	Year	Description	Format	Source	Last visited
WATER RISK INDEX and WASTE HANDLING FACILITY CATEGORY	Waste from all sources Discover Data tool	2019	Summary data on Scottish waste generated and waste managed in Scotland for the period 2011 onwards	csv	SEPA	25/07/2022
	2019 Waste Data Interrogator	2019	All operators of regulated waste management facilities have to provide us with details of the quantities and types of waste they deal with i.e., waste received into site and waste sent on from site to other facilities or processes	csv	Environmental Agency	25/07/2022
	Waste Permit Returns Data Interrogator 2019	2019	as above	csv	Welsh Government	25/07/2022
COMPLIANCE ASSESSMENT SCHEME	Compliance Assessment Scheme (CAS) for Scotland	2019	To rate an operator's environmental performance against its licence conditions	csv	SEPA	15/08/2022
	2019 Compliance Classification System - Waste operations and installations	2019	Condition breaches on Environmental Permitting Regulations waste operations and installation permits for 2019	csv	environment.data.gov.uk	15/08/2022
	Compliance Rating Dataset for England	2019	Evaluating operator's environmental performance against its licence conditions and condition breaches	csv	EA	15/08/2022

LAND USE	Land Cover Map 2020 (1:250 000)	2020	Edina Digimap Land Cover (vector) The LCM2020 product consists of six datasets, three for GB and Northern Ireland. For each, a 10m dataset of classified pixels, a dataset of classified land parcels, and a 25m pixel dataset or rasterised land parcels	GeoPackage	Edina Digimap	25/07/2022
SLOPE	European Digital Elevation Model (EU-DEM), version 1.1	2016	EU-DEM v1.1 is available in Geotiff 32 bits format. It is a contiguous dataset divided into 1,000 x 1,000 km tiles, at 25m resolution with vertical accuracy: +/- 7 meters RMSE	geotiff	European Environment Agency (EEA)	25/07/2022
	Slope	2016	Terrain slope	geotiff	calculated from DEM	-
PERMEABILITY	Permeability index (1:50 000)	2020	Edina Digimap permeability (vector) based on the bedrock minimum permeability	vector	Edina Digimap	25/07/2022
NATURAL PROTECTED AREAS	Sites of Special Scientific Interest (SSSIs)	2022	Areas of land and water that considered to best represent the natural heritage	shp	Natural England, Scottish Government, Welsh Government	02/09/2022
	Special Areas of Conservation	2019	Areas designated by Scottish Ministers under the EC Habitats Directive	shp	JNCC	02/09/2022
	Ancient Woodland Inventory	2021	Woodland	shp	Natural England, Scottish Government, Welsh Government	02/09/2022
	Special Protection Areas	2021	These are areas of the most important habitat for rare (listed	shp	JNCC	02/09/2022

			on Annex I to the Directive) and regularly occurring migratory birds within the European Union			
	World Heritage Site	2021	GIS spatial data for World Heritage Sites and their Buffer Zones, where existing, as inscribed by the World Heritage Committee of UNESCO. World Heritage Sites and their Buffer Zones are defined by a polygon defining the extent of the protected area	shp	Scottish Government, Historic England	02/09/2022
	Wetlands of international importance (Ramsar)	2019	These sites comprise of globally important wetland areas and may extend into the marine environment up to a depth of 2m	shp	JNCC	02/09/2022
	National Nature Reserves	2018	NNRs contain examples of some of the most important natural and semi-natural terrestrial and coastal eco-systems in GB	shp	Natural England, NatureScot (Scotland), Welsh Government	02/09/2022
AQUATIC CLASSIFICATION	Water quality archive datasets	2020	England water quality archive dataset	csv	Department for Environment Food & Rural Affairs	02/09/2022
	Water classification hub	2020	Scotland water quality dataset	csv	SEPA	02/09/2022
	Wales 2018 C2 Interim Classification	2018	Overall water body status for rivers and canals	csv	Natural Resources Wales	02/09/2022
	Main Rivers	2022	Main rivers in Wales	shp	Natural Resources Wales	
INDEX OF MULTIPLE DEPRIVATION	SIMD, Scottish Index of Multiple Deprivation 2016	2020	A tool for identifying areas with relatively high levels of deprivation	csv	Scottish Government	02/09/2022
	English indices of deprivation 2019	2019		csv	English Government	02/09/2022

	Welsh Index of Multiple Deprivation: 2019	2019		csv	Welsh Government	02/09/2022
URBAN RURAL CLASSIFICATION	Scottish Government Urban Rural Classification 2020	2020	Based upon two main criteria: (i) population as defined by National Records of Scotland (NRS), and (ii) accessibility based on drive time analysis to differentiate between accessible and remote areas in Scotland	shp	Scottish Government	02/09/2022
	2011 Local Authority Rural Urban Classification	2011	2011 Rural Urban Classification of Local Authority Districts and other higher-level geographies	csv	English Government	02/09/2022
	Rural-Urban Classification for LSOAs	2011	Same as above but for Wales	shp	Welsh Government	02/09/2022

Appendix B

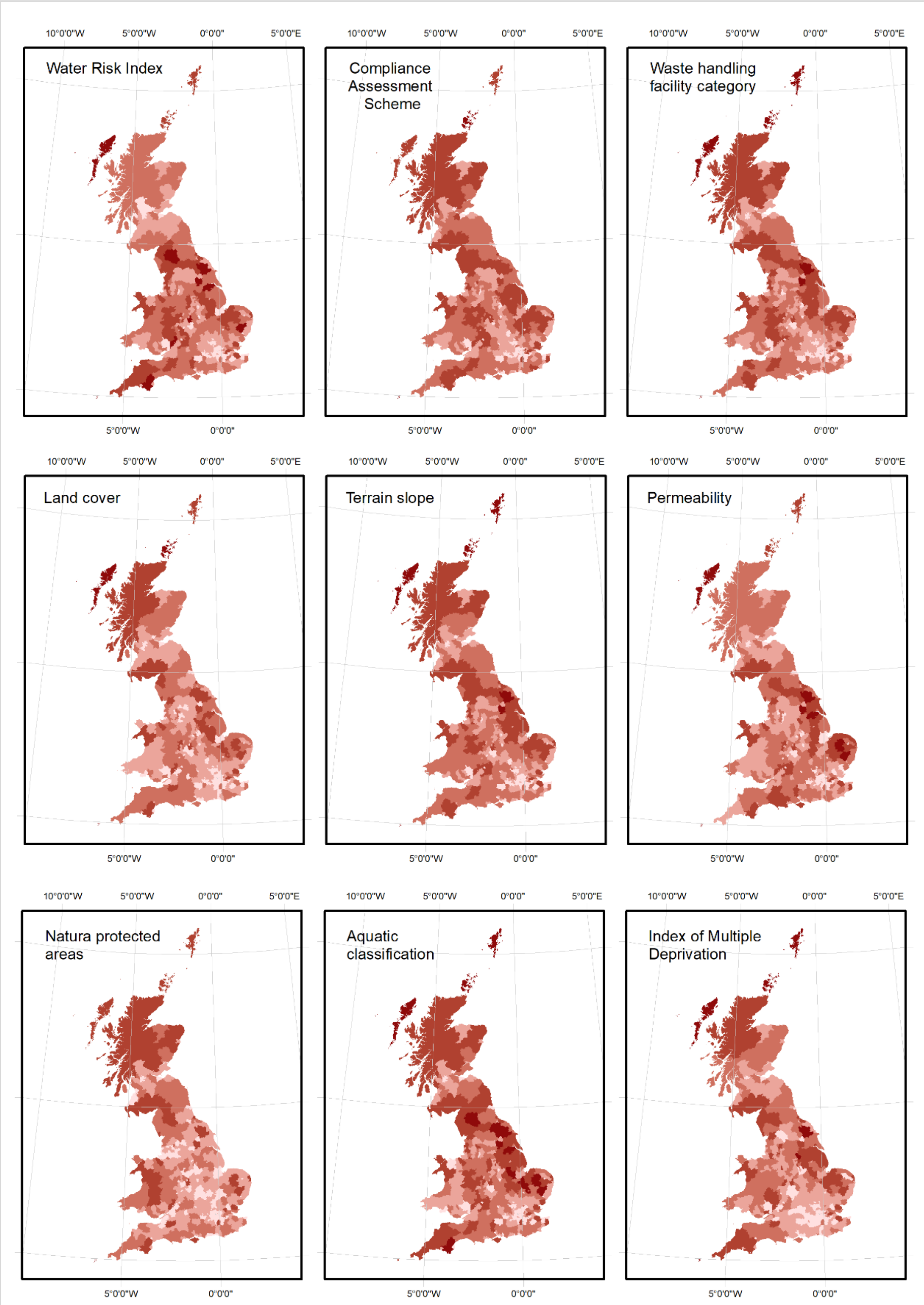

Figure B-1. The vulnerability index values are estimated for waste facilities based on various criteria such as WRI, compliance assessment scheme, waste facility category, land cover, terrain slope, permeability, natural protected areas, aquatic classification, and IMD. These values are then summarised for each Local Authority and normalised *per capita. To facilitate comparison across different criteria, the results are divided into five classes using natural breaks. Darker red areas on the map indicate Local Authorities that are more vulnerable to pollution events compared to others, based on the specific criterion being represented.*

Appendix C

The appendix reports the supplementary materials related to the published paper "A framework to assess the impact of flooding on the release of microplastics from waste management facilities" Ponti *et al.* (2022).

Table C-1. Categories' name and description for the waste management facilities considered in the study based on the classification included in the dataset "Waste Permit Returns Data Interrogator 2019" for England and Wales available at https://data.gov.uk/dataset/d409b2ba-796c-4436-82c7-eb1831a9ef25/2019-waste-data-interrogator (accessed: 25 April 2022).

Category of waste facility	Waste activity in detail
Incineration	Animal By-Products Incinerator Biomass Clinical Waste Incinerator Co-Incinerator Co-Incinerator (Haz) EFW Incinerator Hazardous Waste Incinerator Municipal Waste Incinerator Pet Crematorium
Landfill	Hazardous Merchant LF Hazardous Restricted LF Inert LF Non Haz (SNRHW) LF Non Hazardous LF Restricted LF
Transfer	CA site Clinical waste transfer Haz waste transfer Inert waste transfer Non-haz waste transfer
Treatment	Anaerobic digestion Biological treatment Chemical treatment Clinical waste treatment Composting Haz waste treatment Material recycling facility Mechanical biological treatment Non-haz waste treatment Physical treatment Physical-chemical treatment Recovery of waste WEEE treatment facility Animal and food waste Non-Ferrous metal re-processing Paper and pulp re-processing Paper recycling
MRS (metal treatment)	Car Breaker Metal Recycling

	Vehicle depollution facility Ferrous metal re-processing Metal re-processing
On/In Land	Deposit of waste to land (recovery) Lagoon Mining waste management (non-hazardous)
Storage	In-House storage Storage - A/D Storage - Dredging Storage - Incinerator Storage - Metal Reprocessing Storage - oils Temporary storage installation

Table C-2. Description of the source of waste for (A) entries from the LoW referring to plastic waste as identified by the Commission notice on technical guidance on the classification of waste (2018/C 124/01) and (B) entries selected from the LoW as Microplastic Releasers (plastic waste included) as defined in the section 2.1. of the method. The table shows how MPRs can be found predominantly as the result of previous waste treatments and from thermal, organic and inorganic chemical processes.

Source of waste description	(A) Plastic waste entries	(B) Microplastic Releasers entries
Wastes From Waste Management Facilities	7	24
Wastes From Thermal Processes	-	13
Wastes From Inorganic Chemical Processes	-	11
Wastes From Organic Chemical Processes	3	10
Municipal Wastes	1	9
Construction And Demolition Wastes	7	8
Wastes From Agriculture, Horticulture, Aquaculture, Forestry, Hunting And Fishing, Food Preparation And Processing	1	8
Wastes Not Otherwise Specified In The List	3	8
Waste Packaging, Absorbents, Wiping Cloths, Filter Materials And Protective Clothing	4	5
Wastes From The Leather, Fur And Textile Industries	-	5
Wastes From Shaping And Physical And Mechanical Surface Treatment Of Metals And Plastics	3	4
Wastes From Coatings (Paints, Varnishes And Vitreous Enamels), Adhesives, Sealants And Printing Inks	-	4
Wastes From Chemical Surface Treatment	-	3
Wastes From Petroleum Refining, Natural Gas Purification And Pyrolytic Treatment Of Coal	-	3
Wastes Resulting From Exploration, Mining, Quarrying, And Physical And Chemical Treatment Of Minerals	-	3

Wastes From Wood Processing And The Production Of Panels And Furniture, Pulp, Paper And Cardboard	-	2
Wastes From Human Or Animal Health Care	-	1
Wastes From The Photographic Industry	-	1
TOTAL NUMBER OF ENTRIES	29	122

Table C-3. The table reports the list of MPRs codes selected from the European Waste Catalogue (Chapter 4) described both by the EWC-Stat Rev 4 definition, and the classification of waste according to the EWC categories.

Chapter description from EWC-Stat Rev 4 (European Waste Classification for Statistics, version 4) available at: https://ec.europa.eu/eurostat/ramon/other_documents/ewc_stat_4/index.cfm?TargetUrl=DSP_EWC_STAT_4	**Classification of waste according to EWC-Stat categories** available at: https://ec.europa.eu/eurostat/documents/342366/351806/Guidance-on-EWCStat-categories-2010.pdf/0e7cd3fc-c05c-47a7-818f-1c2421e55604	**Description**
02 Chemical preparation wastes		
02.1 Off-specification chemical wastes		
02.14 Other chemical preparation wastes		
	07_02_16	wastes containing dangerous silicones
	07_02_17	waste containing silicones other than those mentioned in 07 02 16
02.3 Mixed chemical wastes		
02.33 Packaging polluted by hazardous substances		
	15_01_10	packaging containing residues of or contaminated by dangerous substances
06 Metallic wastes		
06.2 Metal wastes, non-ferrous		
06.26 Other metal wastes		
	17_04_11	cables other than those mentioned in 17 04 10
07 Non-metallic wastes		
07.3 Rubber wastes		
07.31 Used tyres		
	16_01_03	end-of-life tyres
07.4 Plastic wastes		
07.41 Plastic packaging wastes		
	15_01_02	plastic packaging

07.42 Other plastic waste		
	02_01_04	waste plastics (except packaging)
	07_02_13	waste plastic
	12_01_05	plastics shavings and turnings
	16_01_19	plastic
	17_02_03	plastic
	19_12_04	plastic and rubber
	20_01_39	plastics
07.6 Textile wastes		
07.61 Worn clothing		
	20_01_10	clothes
07.62 Miscellaneous textile wastes		
	04_02_09	wastes from composite materials (impregnated textile, elastomer, plastomer)
	04_02_21	wastes from unprocessed textile fibres
	04_02_22	wastes from processed textile fibres
	15_01_09	textile packaging
	19_12_08	textiles
	20_01_11	textiles
08 Discarded equipment		
08.4 Discarded machines and equipment components		
08.43 Other discarded machines and equipment components		
	16_02_15	hazardous components removed from discarded equipment
	16_02_16	components removed from discarded equipment other than those mentioned in 16 02 15
10 Mixed wastes		
10.1 Household and similar waste		
10.11 Household wastes		
	20_03_01	mixed municipal waste
	20_03_02	waste from markets
	20_03_07	bulky waste
	20_03_99	municipal wastes not otherwise specified
10.12 Street cleaning residue		
	20_03_03	street-cleaning residues

10.2 Mixed and undifferentiated materials		
10.21 Mixed packaging		
	15_01_05	composite packaging
	15_01_06	mixed packaging
10.22 Other mixed and undifferentiated materials		
	01_03_99	wastes not otherwise specified
	01_04_99	wastes not otherwise specified
	01_05_99	wastes not otherwise specified
	02_01_99	wastes not otherwise specified
	02_02_99	wastes not otherwise specified
	02_03_99	wastes not otherwise specified
	02_04_99	wastes not otherwise specified
	02_05_99	wastes not otherwise specified
	02_06_99	wastes not otherwise specified
	02_07_99	wastes not otherwise specified
	03_01_99	wastes not otherwise specified
	03_03_99	wastes not otherwise specified
	04_01_99	wastes not otherwise specified
	04_02_99	wastes not otherwise specified
	05_01_99	wastes not otherwise specified
	05_06_99	wastes not otherwise specified
	05_07_99	wastes not otherwise specified
	06_01_99	wastes not otherwise specified
	06_02_99	wastes not otherwise specified
	06_03_99	wastes not otherwise specified
	06_04_99	wastes not otherwise specified
	06_06_99	wastes not otherwise specified
	06_07_99	wastes not otherwise specified
	06_08_99	wastes not otherwise specified
	06_09_99	wastes not otherwise specified
	06_10_99	wastes not otherwise specified
	06_11_99	wastes not otherwise specified
	06_13_99	wastes not otherwise specified
	07_01_99	wastes not otherwise specified
	07_02_99	wastes not otherwise specified
	07_03_99	wastes not otherwise specified
	07_04_99	wastes not otherwise specified
	07_05_99	wastes not otherwise specified
	07_06_99	wastes not otherwise specified
	07_07_99	wastes not otherwise specified
	08_01_99	wastes not otherwise specified
	08_02_99	wastes not otherwise specified

	08_03_99	wastes not otherwise specified
	08_04_99	wastes not otherwise specified
	09_01_99	wastes not otherwise specified
	10_01_99	wastes not otherwise specified
	10_02_99	wastes not otherwise specified
	10_03_99	wastes not otherwise specified
	10_04_99	wastes not otherwise specified
	10_05_99	wastes not otherwise specified
	10_06_99	wastes not otherwise specified
	10_07_99	wastes not otherwise specified
	10_08_99	wastes not otherwise specified
	10_09_99	wastes not otherwise specified
	10_10_99	wastes not otherwise specified
	10_11_99	wastes not otherwise specified
	10_12_99	wastes not otherwise specified
	10_13_99	wastes not otherwise specified
	11_01_99	wastes not otherwise specified
	11_02_99	wastes not otherwise specified
	11_05_99	wastes not otherwise specified
	12_01_99	wastes not otherwise specified
	16_01_99	wastes not otherwise specified
	16_03_04	inorganic wastes other than those mentioned in 16 03 03
	16_07_99	wastes not otherwise specified
	19_01_99	wastes not otherwise specified
	19_02_99	wastes not otherwise specified
	19_05_99	wastes not otherwise specified
	19_06_99	wastes not otherwise specified
	19_08_99	wastes not otherwise specified
	19_09_99	wastes not otherwise specified
	19_11_99	wastes not otherwise specified
	20_01_99	other fractions not otherwise specified
	16_03_03	inorganic wastes containing dangerous substances
	17_04_10	cables containing oil, coal tar and other dangerous substances
	18_01_10	amalgam waste from dental care
10.3 Sorting residues		
10.32 Other sorting residues		
	19_02_03	premixed wastes composed only of non-hazardous wastes

	19_02_10	combustible wastes other than those mentioned in 19 02 08 and 19 02 09
	19_05_01	non-composted fraction of municipal and similar wastes
	19_05_02	non-composted fraction of animal and vegetable waste
	19_05_03	off-specification compost
	19_10_04	fluff-light fraction and dust other than those mentioned in 19 10 03
	19_10_06	other fractions other than those mentioned in 19 10 05
	19_12_10	combustible waste (refuse derived fuel)
	19_12_12	other wastes (including mixtures of materials) from mechanical treatment of wastes other than those mentioned in 19 12 11
	19_02_04	premixed wastes composed of at least one hazardous waste
	19_02_09	solid combustible wastes containing dangerous substances
	19_04_03	non-vitrified solid phase
	19_10_03	fluff-light fraction and dust containing dangerous substances
	19_10_05	other fractions containing dangerous substances
	19_12_11	other wastes (including mixtures of materials) from mechanical treatment of waste containing dangerous substances
12 Mineral wastes		
12.1 Construction and demolition wastes		
12.13 Mixed construction wastes		
	17_06_04	insulation materials other than those mentioned in 17 06 01 and 17 06 03
	17_09_04	mixed construction and demolition wastes other than those mentioned in 17 09 01, 17 09 02 and 17 09 03
	17_06_03	other insulation materials consisting of or containing dangerous substances

	17_02_04	glass, plastic and wood containing or contaminated with dangerous substances
	17_09_03	other construction and demolition wastes (including mixed wastes) containing dangerous substances
12.5 Various mineral wastes		
12.51 Artificial mineral wastes		
	12_01_16	waste blasting material containing dangerous substances
	12_01_17	waste blasting material other than those mentioned in 12 01 16

Appendix D

The scientific article "A framework to assess the impact of flooding on the release of microplastics from waste management facilities." was published by the Journal of Hazardous Materials Advances.

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A framework to assess the impact of flooding on the release of microplastics from waste management facilities

Abstract

The impact of flood on waste management facilities can induce the release of micro pollutants to freshwater systems with concerning impacts on the marine environment, agricultural ecosystems, and human health. Almost 30% of the total waste managed in the UK in 2019 was characterised by Microplastic Releasers (MPRs): plastic waste, synthetic textile, rubber waste, and mix/undifferentiated materials that are able to or contain items that can deteriorate and fragment into micro components. In recent years, the management of solid waste and its contribution to flood-driven microplastic pollution has been limited with a focus on plastic waste mismanagement specifically, and the assessment of the risk is long overdue. We present a new methodology combining publicly available data on waste with pluvial and fluvial flood extent maps. The methodology was applied to the UK where the impact of pluvial flood on waste management facilities shows a 3-fold increment between 20 and 50-year return period resulting in almost 5 million tonnes of waste per day at risk of releasing microplastics during inundation. We conclude that further studies at the local scale are necessary to establish site-specific mitigation measures and containment systems able to decrease the flood-induced microplastic mobilisation from waste management facilities.

Keywords: microplastic pollution, emerging contaminants, floods, waste management facilities, municipal solid waste

Introduction

Floods are among the most frequent natural hazards to cause industrial accidents that may result in fires, explosions, and the release of hazardous materials (Piccinelli and Krausmann, 2013). These events are defined as 'Natech' accidents (i.e., natural hazards triggering technological accidents) because of the capacity of natural hazards to cause technological disaster. Natech accidents mainly refer to refineries, petrochemical complexes, and oil and gas pipelines that deal with dangerous substances as identified by the Seveso-III-Directive (2012/18/EU), which is the main EU legislation aiming at the prevention of technological disasters involving dangerous substances. However, the types of synthetic products that can cause harm are not limited to the list included in the Seveso-III-Directive.

For example, plastic items are made from polymers mixed with a complex blend of additives, some of which belong to the list of emerging contaminants, which are released due to plastic degradation with potential risks to the environment and human health (Gunaalan *et al.*, 2020). In recent years, plastic pollution has become one of the major environmental concerns, with an exponential increase in plastic created (Geyer *et al.*, 2017, PlasticsEurope, 2021) and an expected 3-fold increase in plastic waste by 2030 (Borrelle *et al.*, 2020a). However, despite the growing interest developed by the scientific community, flood-induced plastic debris mobilisation from terrestrial sources has yet to be fully understood.

It is estimated that between 4 and 12 million metric tons (Mt) of plastic end up in the marine environment, globally, per year (Jambeck *et al.*, 2015, Geyer *et al.*, 2017, Boucher *et al.*, 2020). The issue is exacerbated by the persistence of plastic debris in the environment and the inevitable breakdown processes resulting in the fragmentation of the initial (microplastic) component into microplastics (MPs). The origin of ocean plastics has been increasingly attributed to terrestrial sources (Hurley *et al.*, 2018), and recent attention has been given to rivers, considered as a major pathway for plastic transport from inland areas to the ocean (van Emmerik *et al.*, 2019, He *et al.*, 2021, Wang *et al.*, 2021). The consequences of flooding on plastic loads in rivers was extensively studied by Hurley et al. (2018), which investigated 40 rivers across the northwest of England before and after a period of severe flooding in winter 2015/2016. The study confirmed firstly the presence of MPs in all of the studied river channel beds, and secondly the capacity of flooding to export approximately 70% to 100% of the total MPs load stored in the riverbeds.

The interaction between plastic transport and river flood events has been further investigated by Roebroek et al. (2021), with the introduction of global mismanaged plastic waste (MMPW) as a terrestrial diffuse source of plastic debris. By combining MMPW data with river flood extents for different return periods, the study was able to estimate the flood-driven plastic mobilisation per country with results showing a tenfold global increase in potential plastic transport during 10-year return period flood compared to non-flood conditions. Among the methodology's limitations listed by Roebroek et al. (2021), two aspects are particularly relevant for the purpose of this research. The first is data on MMPW are estimated on the waste generation rates per country that ignore the potential build-up over time. The second is that Roebroek et al. (2021) only focuses on mismanaged plastic, while plastic waste properly disposed could also be mobilised by flood waters. Data on waste characteristics and quantity received per facility (tonnes) are reported annually by Member States (European Parliament and Council of the European Union, 2008) and made publicly available. In the UK, in 2019, the total amount of waste received by waste facilities was equal to 257 million tonnes. Although it may overestimate the actual quantity of waste produced for the same year due to the complex movement of discarded items through the waste management network, the information on waste received provides a greater detail and richness compared to national statistics on waste produced annually because it includes data on the location, quantity, and characteristics of waste dealt with by each waste management facility rather than a gross value at a national scale (a list with type and description of waste management facilities selected for this study is available in the supplementary materials, Table S1).

Waste management facilities become potential terrestrial sources for plastic mobilisation and therefore plastic pollution when the discarded materials are temporary stored, treated, or disposed within sites located in areas at risk of flooding. During recent decades, few works investigated the impacts of flooding on managed waste. Arrighi et al. (2018a) is one of the few studies available in literature that clearly defines wastewater treatment plants, waste handling facilities, and contaminated sites as environmental hotspots because of the risk posed by the presence of contaminants within sites located in flood-prone areas. Comparatively, the potential inundation of solid waste landfills has received more attention in the literature (Laner *et al.*, 2009, Neuhold and Nachtnebel, 2011, Neuhold, 2012) although many questions remain unanswered. For example, Nicholls et al. (2021) recently published a comprehensive European study on coastal landfills and the rising of sea levels, highlighting a lack of data and a general underestimation of the threat posed by the potential release of solid and liquid waste from coastal landfills.

The purpose of this study is to better understand quantity and characteristics of waste received and stored by waste management facilities, with the assumption that managed waste is potentially one of the biggest terrestrial sources of flood-induced MPs mobilisation. The novel methodology developed within this study was applied to the UK by combining publicly available data on waste received by waste facility in 2019 together with flood likelihood map extents for different return periods and source (fluvial and pluvial). The research aims to (i) identify waste at risk of releasing MPs among the European Waste Catalogue (EWC) code in addition to the well-known plastic waste, (ii) estimate the quantity of waste at risk of flooding in the UK which could lead to MPs' mobilisation in flood waters, and (iii) identify spatial patterns where the level of risk requires further studies at the local scale.

Methods

The study introduces a framework to estimate the quantity of waste at risk of releasing MPs in flood waters by combining publicly available annual data on waste received by waste management facilities with flood extent maps for different sources (fluvial and pluvial) and likelihoods of flooding (5, 10, 20, 50, 100, 200, 500 and 1000 years) at the national level. Due to the novelty of this assessment, three methodologies are presented to (1) identify the codes within the List of Waste (European Commission, 2000a) referring to solid waste that could deteriorate and fragment into MPs, (2) estimate the waste facilities' footprint from a point coordinate based on the INSPIRE Cadastral Parcel dataset, and (3) determine the quantity of waste at risk of flooding based on the percentage of overlapping between the estimated facilities' footprint and flood map extents.

The methods were applied to the UK where datasets on annual quantity, type, and location of waste received by waste facilities, and Cadastral Parcel polygons (which represent land ownership) are publicly available (with the exception of the Cadastral Parcel dataset for Northern Ireland which is not publicly available). The datasets were subsequently combined with flood map extents in a geographical information systems (ArcGIS and ArcGIS Pro) to determine national and local quantities of waste at risk of releasing MPs during inundation.

Identification of waste at risk of deteriorating into synthetic micro components within the European Waste Catalogue (EWC) code: not just plastic waste

The waste classification code, also referred to as LoW (List of Waste) or EWC (European Waste Catalogue) code was introduced in 2000 by the European Commission Decision 2000/532/EC (further revised in 2014 and 2017). Unlike more straightforward legislation on chemicals, because of the complexity and alterability of discarded substances, the LoW does not refer to a waste's chemical components for classification purposes but rather to alternative criteria such as (i) the waste source, (ii) the waste type, and (iii) the recognition of waste not otherwise specified because it is mixed or undifferentiated. The classification system was conceived to help operators to assign a standardised, accurate six-digit code to each entry of waste. The first two digits of the LoW refer to the waste source (e.g. 17 Construction and demolition wastes), the second series of digits assign the waste type (e.g. 17 02 Construction waste wood, glass and plastic), and the last two digits represent the final entry description (e.g. 17 02 03 Plastic construction waste). The LoW recognises three types of entry and marks with an asterisk (*) what is considered as hazardous waste. Absolute hazardous (AH*) entries display one or more of the fifteen hazardous property as indicated in Annex III to the Directive on waste 2008/98/EC (Waste Framework Directive or WFD), such as explosive, ecotoxic, mutagenic, infectious, etc.; absolute non-hazardous (ANH) entries identify waste lacking any hazardous component. Finally, mirror entries represent the case of mixed substances where further assessment needs to be undertaken to classify the waste as mirror hazardous (MH*) or mirror non-hazardous (MNH) (EA, 2014).

Table 1. Number of codes from the European Waste Catalogue (EWC) code per entry type (AH, MR, MNH, ANH) based on discarded materials' hazardous properties.

842 Entries in the List of Waste (LoW)			
408 Hazardous entries		434 Non-hazardous entries	
230 Absolute Hazardous	178 Mirror Hazardous	188 Mirror Non Hazardous	246 Absolute Non Hazardous

The absence of cross-categories (e.g. the substance state such as solid, liquid or gaseous) and the lack of a controlled vocabulary for the entries' description does not allow straightforward data inquiries and the same type of waste can be found throughout different LoW groupings. The European Commission published the Commission Notice On Technical Guidance On The Classification Of Waste in 2018 (2018/C 1 24/01), which added a substance-oriented identifier approach, including a non-exhaustive list of plastic waste entries. Out of 842 LoW codes (Table 1), there are 29 plastic waste classification codes identified by the commission notice, divided into (6) absolute non-hazardous, (10) mirror hazardous, and (13) mirror non-hazardous categories. The word 'mirror' in the legislation indicates the case of entries presenting a mix of hazardous and non-hazardous materials. Surprisingly, no entries were selected among the absolute hazardous substances, even if within the LoW there are several absolute hazardous codes that identify mix waste, which are likely to have some plastic components: for example code 18 01 10* amalgam waste from dental care. In addition, although MPs in waste is an increasingly explored issue in literature (e.g. MPs in wastewater treatment plants and in landfill leachate (He *et al.*, 2019, Sun *et al.*, 2021)), in the commission notice there is no reference to the presence of MPs in waste. The description of the source of waste for entries related to plastic waste versus Microplastic Releasers is available in the supplementary materials, Table S2.

The source of MPs can be direct: primary microplastic, specifically manufactured for commercial use such as cosmetics; and indirect: secondary microplastic, resulting from the deterioration and fragmentation of certain materials such as plastic items, synthetic fabrics and rubber, due to mechanical stress, photo-oxidation and weathering processes (Golwala *et al.*, 2021). Although the presence of primary MPs is well known within the waste industry, for example in landfill leachate or within waste-water treatment plants, for this study we focused on secondary MPs only, leaving the assessment of primary MPs to future work. The release of MPs from fabric is prominently documented, with studies reporting from 10 to 1700 mg of MPs per kg of washed fabric (Karkkainen and Sillanpaa, 2021). Another important source of MPs is road tyre wear emissions which was calculated ranging from 0.2 to 5.5 kg of global emissions of Tyre Wear Particles (TWPs) per capita (Baensch-Baltruschat *et al.*, 2020, Evangeliou *et al.*, 2020). Other potential emission sources include plastic manufacturers and industries where plastic is used (for example carpet, wallpaper and cosmetic/pharmaceutical manufacturers), waste management facilities, agricultural areas, road networks (beyond tyre and brake wear) and urban residential/commercial areas (Xu *et al.*, 2020, Allen *et al.*, 2022). It is noted that direct and diffuse source emission rates beyond fabric washing/drying and tyre wear have yet to be quantitatively characterised to date and is an important focus of future research.

For the purpose of this study, and in an effort to clearly identify waste that could degenerate into synthetic micro components, an extended description of the selection of plastic waste identified by the European Commission Notice (2018) has been adopted. The description of plastic waste by the European Commission Notice (2018) has been extended to include possible sources of secondary MPs in waste, defined as Microplastic Releasers (MPRs) (Figure) described as: (i) codes related to synthetic textile and rubber waste that could release microfibers and rubber polymers, and (ii) codes referring to mixed and undifferentiated materials (e.g. mixed household waste). These waste products are selected specifically as they are likely to contain larger plastic materials, synthetic clothes, and discarded items with rubber components and therefore act a source of MPs.

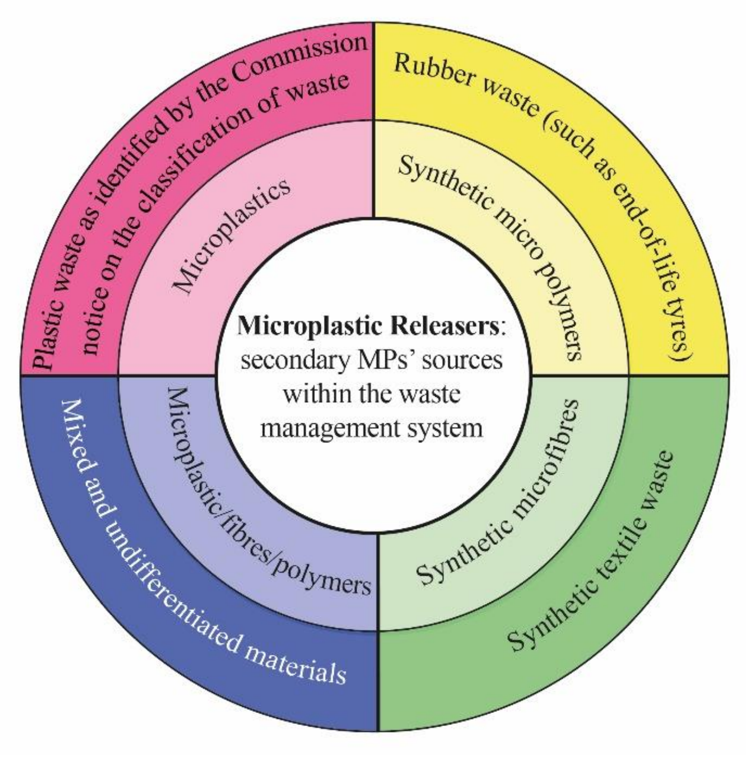

Figure 1. Four types of waste at risk of deteriorating and fragmentise into micro components: main type of waste on the outside of the circle and the related micro components on the inside. The study has named the selected waste as Microplastic Releasers (MPRs).

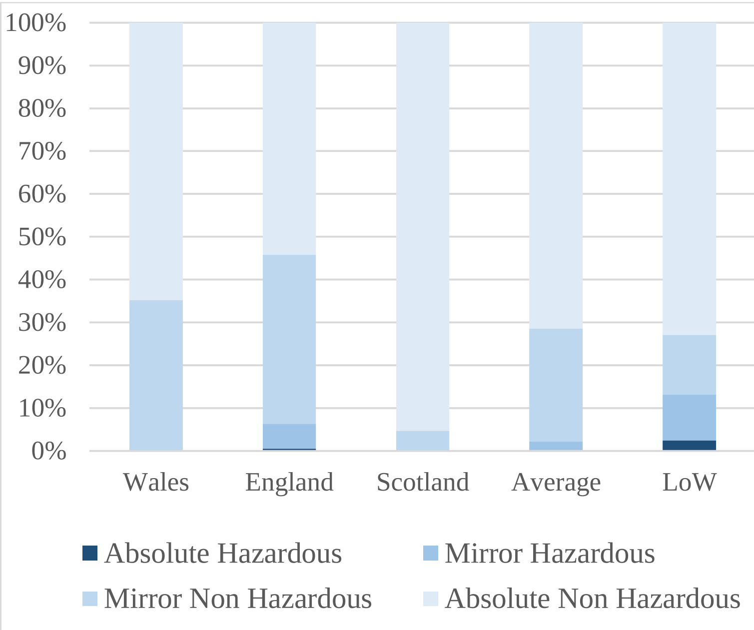

Figure 2. Microplastic Releasers distribution per List of Waste entry type. Each code within the European Waste Catalogue has an entry type associated to it based on the hazardous characteristics of the discarded items (Absolute Hazardous, Mirror

Hazardous, Mirror Non Hazardous, and Absolute Non Hazardous). Figure 2 shows the distribution of the selected MPRs waste codes among the four entry types for each country in the 2019, with the exception of Northern Ireland, and allow the comparison with the average distribution for Great Britain and within the full List of Waste.

In addition to the physical stress due to transportation, treatment, and weathering phenomena (especially when waste is accumulated outside in containers and/or piled on hard surfaces), the deterioration and fragmentation of materials can be accelerated by flood water (as a mechanical force that may break particles) (Zhang *et al.*, 2021). In the case of flooding, discarded items could be subjected to the flow's rapidly changing conditions and the presence of suspended sediments, which could lead to turbulent mixing (mechanical abrasion and wear) and the collision with debris and other built infrastructures. Micro waste components could escape the perimeter of the facility within flood waters, with the flood water acting as the micro waste's transport vector and mixing or discharging into waterways, ecosystems, flooded areas and floodplains downstream. Therefore, it is important that waste facilities located in flood-prone areas are identified, alongside the information on the quantity and location of discarded items prone to release MPs in flood waters.

Defining the waste management facilities' footprint estimation

The publicly available UK waste facility datasets (SEPA, 2019b, EA, 2019d, Natural Resources Wales, 2019, NIEA, 2019) report the postcodes and/or geographic coordinates for each facility, but no surface area or boundaries information are associated with each entry. To enable the assessment of both data describing waste at the facilities' location and the flood map extents, an averaged footprint area for each type of facility in GB was estimated by regulatory region (per country) using the publicly available INSPIRE Cadastral Parcels spatial dataset (HM Land Registry, 2021, Registers of Scotland, 2021), which is available for Scotland, England, and Wales, and applied to all licensed waste facilities in the UK. Unfortunately, the cadastral parcel dataset is not currently publicly available for Northern Ireland. The INSPIRE cadastral parcels dataset complies to the European directive 2007/2/EC of the European Parliament and of the Council of 14 March 2007 establishing an Infrastructure for Spatial Information in the European Community (INSPIRE), and contains the ownership polygon at ground level for each registered property. Unfortunately, the dataset has several limitations: (i) the spatial extent of the database is limited, especially outside main cities where big portions of land are unmapped; (ii) the land ownership polygon may not match the actual footprint of a site but includes green areas, residential properties, and parking areas; (iii) polygons may not have been kept up-to-date and therefore they may not represent the sites' current extent. To overcome the limitations two methodologies were designed and tested to understand how to best represent the footprint of a waste facility when the only information available are: 1) the geographic coordinates indicating a point randomly located within or nearby the facility boundaries; 2) the category of waste facility (e.g. landfill, transfer station, etc.); and 3) the quantity of annual waste intake.

Methodology no.1 investigated the potential correlation between waste sites' footprint (manually checked and corrected when necessary from INSPIRE polygons for 154 sites in Scotland, 22% of the total Scotland registered waste sites), and the annual waste intake (tonnes of waste received on site per year). Unfortunately, no significant correlation was discovered ($r = 0.377$, p-value= 0.516), especially for waste transfer stations which are the prevalent category (75%) among the 154 sites' dataset but the most unpredictable in terms of surface area. Therefore, Methodology no.2 was developed. All the waste facilities in the UK having a cadastral polygon were selected (1,052 from the more than 7,000 waste sites recorded), and circular buffer areas (the creation of polygons around input features to a specified distance) were created based on the cadastral polygon average size per category of waste facility and per country (Scotland, England and Wales). Consequently, three flood risk map extents (fluvial high likelihood (1 in 10-year return period), medium likelihood (1 in 200-year return period) and low likelihood (1 in 1000-year return period)) were selected by looking at the most common approach adopted in terms of flood risk perception by different countries in the UK (EA, 2018, NIFRA, 2018, SEPA, 2020b, Natural Resources Wales, 2021) although these will need to be reconsidered in the future due to climate change. To compare the performance of buffer areas versus cadastral parcels, both datasets were combined with flood likelihood maps to estimate the number of facilities affected by flood and the extension of the interactions (percentage of overlap) in the Geographical Information System (GIS) environment Figure 3. The differences were analysed to establish the applicability of the methodology for this study and to other (non-microplastic specific) purposes. The INSPIRE dataset is not only available for the UK but it is a European dataset publicly available for most Member States, which opens up the possibility of applying the methodology for waste management analysis for other countries in Europe, providing a possible comparable future risk assessment of microplastic release from waste facilities across Europe due to flooding.

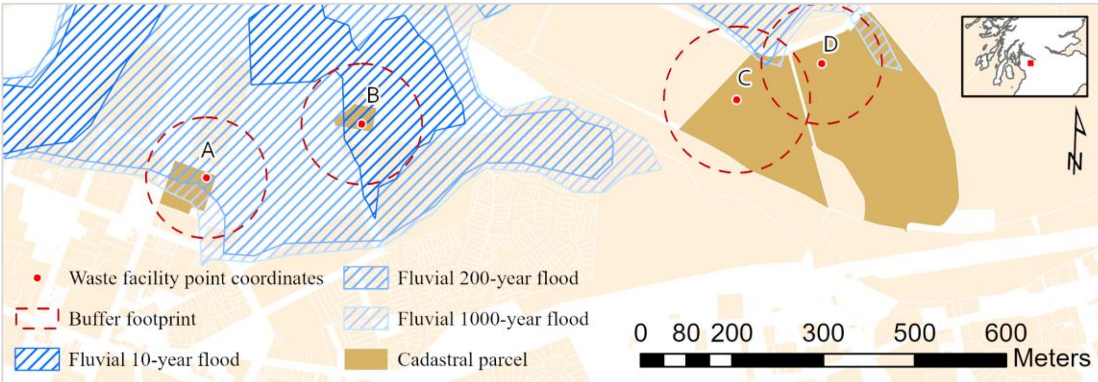

Figure 3. An example of overlap between flood map extents for different return periods and buffer areas (dashed circles) originated by the average size of the INSPIRE Cadastral Parcels polygons (Registers of Scotland, 2021) for site category and country. The figure shows the difference in size of buffer areas compared to cadastral parcels and their relationships with flood maps. Site (A) is overlapped by 72.5% by 1 in 200-year fluvial flood, and 85% by 1 in 1000-year return period; Site (B) is completely overlapped by 200 and 1000-year flood maps and partially overlapped (70%) by the 10-year fluvial return period. Sites (C) and (D) present opposite results both for the size of the cadastral parcels compared to the buffer areas and for the minimal overlap reported with the medium and low flood likelihoods.

The correlation coefficient values between buffer areas and cadastral parcels obtained for the tested flood risk return periods showed a constant improvement from the higher flood likelihood to the medium and lower flood likelihood: respectively r = 0.846, 0.897, and 0.900. The medium likelihood presents values closer to the best fit line, and the highest significance level (0.07) in terms of extreme results probability value (P-value). Low and high likelihood have less statistically significant results presenting respectively p-value=0.16, and p-value=0.28. In terms of the *coefficient* of determination (R-squared) values, the medium likelihood (1 in 200-year return period) has the best result with 0.888, followed by the low likelihood with 0.810 and high likelihood 0.717. Despite the limitations, methodology n.2 was considered fit for the purpose of recreating waste facilities' footprint at the national level where cadastral parcel datasets are not available but a facility point coordinate is known. Further studies should be undertaken toward advancing this methodology for future studies.

Defining and mapping the different likelihoods for fluvial and pluvial flood risk map extents

The Fathom-UK flood map extents, indicating the probability of flooding over space, were made available by SSBN UK Limited (SSBN UK Limited, 2021b). The method used to derive the fluvial maps refers to the global model detailed by Sampson et al. (2015b), further improved with higher quality data such as terrain, hydrography, stream gauge, rainfall, and flood defence data.

Fathom's hydraulic modelling is an implementation of the LISFLOOD-FP numerical scheme (Bates *et al.*, 2010), combined with Neal et al. (2018) approach to improve and optimise central and graphical processing units through parallelization to significantly reduce the model runtime. The hydraulic model is executed at 1 arc second (between 20 and 25m resolution) across the UK using a composite Digital Elevation Model (DEM) built using LiDAR elevation data from relevant national government agencies (which covers ~70% of UK land area), together with Ordnance Survey terrain data. Extreme flows on every river were predicted via statistical modelling based on a dense array of river gauges with long historical records available within the National River Flow Archive (NRFA). Channel locations were defined using Ordnance Survey channel location data, and used to construct a flow accumulation grid together with the DEM.

In terms of river bathymetry, the Global River Widths from Landsat (GRWL) database from Allen and Pavelsky (2018) was used to estimate river widths, while an estimate of channel beds elevations was produced by adopting an innovative channel solver (Neal *et al.*, 2021), and by combining data on an estimate of bank-full discharge (for return period of ~ 1 in 2 years), channel widths and slope from the DEM. The reason behind linking channel geometry to discharge return period is to ensure channels are appropriately sized for the simulated flow. For the pluvial model, Intensity-Duration-Frequency curves were calculated from CEH-GEAR1h, an hourly gridded rainfall dataset at 1 km spatial resolution, and 1-hour, 6-hour, and 12-hour intensity-frequency relationships were computed. The rainfall dataset input water directly onto the 2D base model LIDFLOOD-FP's staggered grid, with the addition of a 1D model solver for channels smaller than the grid size.

In terms of flood defences, data came primarily from the Environmental Agency and Natural Resource Wales. For location missing data, particularly in Scotland, a levee detection algorithm was adopted to fill the gaps (Wing *et al.*, 2019).

Fathom-UK hazard map extents were validated against Environmental Agency (England) and Natural Resources Wales flood maps, unfortunately at this stage no validation is available for Scotland and Northern Ireland. Results indicated that the two datasets are within proximity of each other, with the error potentially due to typical uncertainties in extreme flow estimation and terrain data accuracy. However, validation tests focussed only on extreme floods (<100-year return period), ongoing research at Fathom will further validate the methodology against both observed and lower return period flood events.

The Fathom-UK flood map extents adopted for the scope of the presented study included several return periods (5, 10, 20, 50, 100, 200, 500 and 1000 years), for fluvial (considering flood defences) and pluvial flooding. Data were given in flood modelling output (as a GeoTIFF raster) at 1/3 arc sec (~10 m) resolution for the entire UK, with cell values representing maximum inundation depths in centimetres from 0 to 9999 (9999 = permanent water). Unfortunately, no indication of the flood velocity was provided, nor was coastal flood likelihood extent mapping available. Therefore, a simplistic approach was adopted to differentiate flooded versus non-flooded pixels based on water depth: for fluvial, a binary map of wet and dry was established considering as flooded any depth higher than 0 cm; while for pluvial flooding, the same was applied to depths higher than 15 cm. The 15 cm threshold adopted was based on Environmental Agency's flood risk information (2019e) suggesting that at 15 cm flooding would likely exceed kerb height and damp-proof course. Raster maps were merged for the UK, reclassified with two intervals representing wet and dry, and converted in polygons in the GIS environment to allow further spatial analysis with the waste dataset.

Estimating the quantity of managed waste at risk of releasing microplastics in flood waters

The estimation of the quantity of waste at risk of being mobilised and releasing MPs in flood waters at any given day in 2019 was estimated based on the selection of Microplastic Releasers, from datasets on annual total waste received by facilities per country in the UK. Annual quantities were summarized per operator and divided by 365 to give approximate daily quantities. The simplification was necessary because of the lack of daily data on waste received and the average time spent by waste materials stored on site, which are both relevant considerations when simulating the impact of flooding on waste facilities. As a result of the limitations, the flooding event is considered as a single day duration only, which may underestimate the quantity of waste at risk of being affected. An additional build-up component was adopted only for landfills for their intrinsic accumulation properties. Therefore, for landfills operative in 2019, MPRs were selected among the annual quantity of total waste and summarised for every year from 2007 to 2018, and the relative quantities added on top of the daily tonnage from 2019.

Estimation of daily quantities (tonnes) of Microplastic Releasers received per facility in the UK in 2019

Data about annual quantities of waste received per facility for each typology (based on the LoW classification) are publicly available for all countries in the UK: Scottish waste sites and capacity tool (SEPA, 2019b), Waste Data Interrogator (EA, 2019d), Waste Permit Returns Data Interrogator (Natural Resources Wales, 2019), and Authorised waste sites (NIEA, 2019). From the public datasets, the operative facilities in 2019 were selected together with the category of waste facility, discarding sites with incomplete information (e.g., missing coordinates, waste codes, etc.). Subsequently, MPRs' codes were selected for each country and annual quantities (tonnes) were summarised per facility. The dataset for Northern Ireland doesn't specify quantities of waste for each LoW code, therefore the percentage of Microplastic Releasers was estimated by referring to the average obtained for Scotland, England and Wales (32% of the total amount of waste received). As indicated in Table 2, the total annual amount of waste received by facilities in 2019 in the UK was 257 million tonnes, of which almost 30% (~74 million tonnes) were MPRs, consisting of mixed and undifferentiated materials (70%), plastic waste (29%), synthetic textile (7%), and rubber (1%). Although not all the operational waste management sites dealt with MPRs in 2019, results show approximately 66% of facilities for Scotland, 47% for England, 64% for Wales and 53% for Northern Ireland managed MPRs waste.

Table 2. Quantity (tonnes) of total waste and Microplastic Releasers received by waste facilities in 2019 in the UK.

	Active waste facilities in 2019	Annual total waste received (tonnes)	Annual Microplastic Releasers received (tonnes)	Percentage of Microplastic Releasers waste on the total
UK	7,676	257×10^6	$73,8 \times 10^6$	29%
Scotland	697	$16,7 \times 10^6$	$5,8 \times 10^6$	35%
England	6,151	$222,9 \times 10^6$	$62,1 \times 10^6$	28%
Wales	444	$9,9 \times 10^6$	$3,5 \times 10^6$	35%
Northern Ireland	384	$7,5 \times 10^6$	*$2,4 \times 10^6$	32%

**Results for Northern Ireland were estimated based on the average found for the other countries in UK (32%)*

Microplastic Releasers accumulation in landfills

In the UK in 2018 50.7 million tonnes of waste were landfilled (44.1 in England) among 629 operative disposal sites (534 in England), with a remaining capacity of 129.3 million tonnes (converted from 415,069,000 m^3 by considering a waste density of 311.73 kg/m^3) (DEFRA and Government Statistical Service, 2021). On top of this, in England alone, around 20,000 historical landfills (where there is no environmental permit in force, including sites that existed before landfills were regulated) were mapped by the Environment Agency in 2022. Approximately 1,200 of those are located within flood zones of 1 in 200-year return period (Brand *et al.*, 2018), and ~3,400 are at risk for low likelihood but high intensity flooding (CCC, 2018). Awareness of the risk presented by landfills to the environment has been raised and has led to a significant increase of landfill-related publications in the last two decades: from 662 in 2000 to 2,335 in 2017 (Sabour *et al.*, 2020a). However, despite the increase in severity of landfill related regulations, Directive 1999/31/EC on the landfill of waste and the Amending Directive (EU) 2018/850 and encouraging governmental strategies in response of the Zero Waste movement, more measures need to be put in place to control and contain the potential leakage of waste from disposal areas to the environment.

In this study, the publicly available data on waste received by landfills for different countries in the UK were used for two objectives: firstly, to advance the quantitative assessment of Microplastic Releasers received by landfills for the period 2007 – 2019 compared to the total amount of waste; and secondly, to select the quantities of MPRs accumulated by landfills classified as operative in 2019 to be added to the daily estimates of waste at risk of releasing MPs in flood waters. The latter is an attempt to approximate the build-up of MPRs that occurred through the years in landfills, which is relevant when simulating the impact of flood on disposal sites.

For the first part of the analysis, although historical landfills (intended as opposite to licensed/permitted sites) pose the highest risk due to their predominant location in low-lying estuarine and coastal areas, and the absence of leachate management systems (Brand *et al.*, 2018), inadequate records on the waste received and/or landfilled prevented them from being included in the current study. Instead, for landfills licensed/permitted in the UK, publicly available datasets were analysed to quantify the waste annually received per facility from 2007 to 2019 (EA, 2019d, Natural Resources Wales, 2019, NIEA, 2019, SEPA, 2019b). Subsequently, waste codes identified as potential Microplastic Releasers (MPRs) were selected for each country, summarised per year, and divided by the country's population to allow the comparison among different sizes of countries in the UK. Results shown in Figure 4-3 reveal a significant decrease of the total amount of waste received by disposal sites in Scotland, England and Wales from 2007 to 2014, with a slightly rise of approximately 3 million tonnes between 2014 and 2015. The percentage of MPRs compared to the total amount of waste also substantially reduced from 65% in 2007 to 38% in 2014, and 35% in 2019 (the available data for Northern Ireland are only for the period 2014-2019).

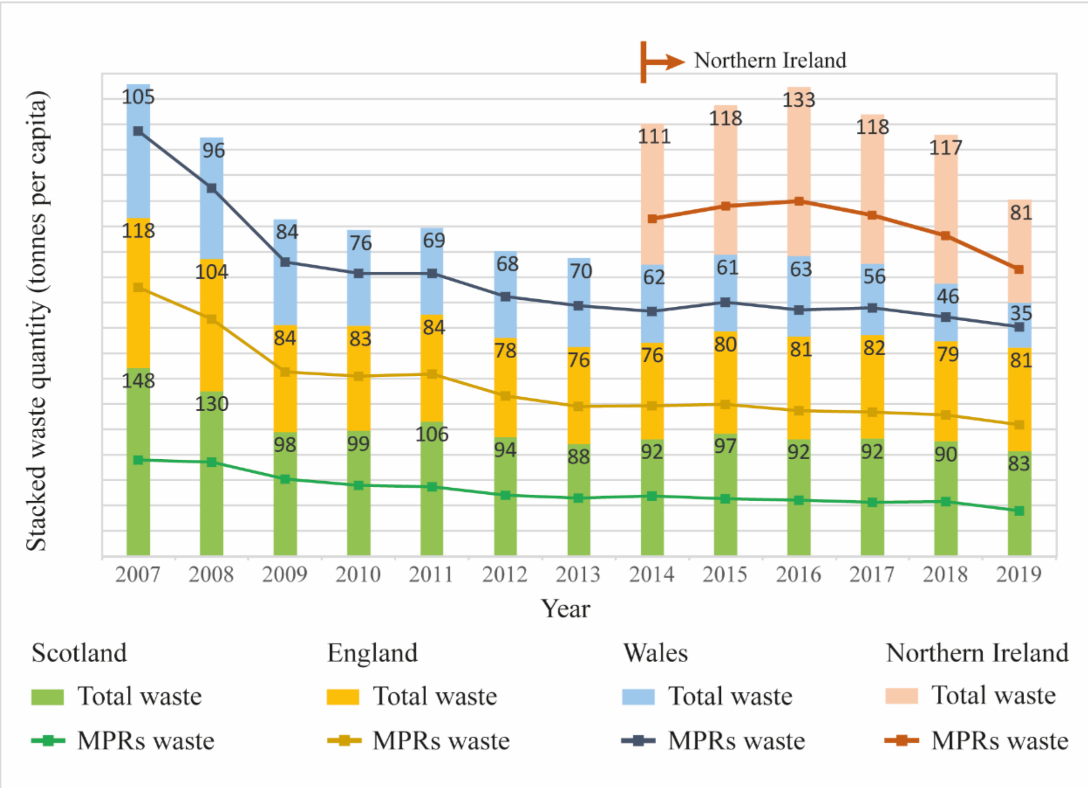

Figure 4. Per capita annual waste and Microplastic Releasers waste received by landfills per country and per year. MPRs waste was selected from annual quantity of total waste received by landfills from 2007. Both total waste and MPRs quantities were divided by population and organised in a stacked bar graph to highlight the differences between different years and countries in the UK (Scotland at the bottom, then England, Wales, and Northern Ireland). Each bar has a number on it referring to the quantity of total waste (tonnes per capita). Data for Northern Ireland are only available from 2014 and the quantity of MPRs was estimated based on the average for the other countries (32%).

The tendency of sending less waste to landfills is likely the result of the European Landfill Directive 99/31/EC, which aimed to reduce or prevent, as far as possible, negative impacts from landfill to the environment. Some of the Directive's consequences can be read through the UK achievement in terms of (i) biodegradable waste going to landfills that decreased from 10.3 million tonnes in 2012 to 7.2 in 2018; (ii) the rates of packaging waste recycled or recovered increased from 61,4% in 2012 to 62,1% in 2018; and (iii) the imposed rise in price for accepted discarded materials, to include the costs related to the closure and after-care of a site, translated in a reduction of more than 5 million tonnes between 2012 and 2014 in the total amount of commercial and industrial waste produced (Department for Environment Food & Rural Affairs and Government Statistical Service, 2021).

For the second objective of the analysis, an approximation regarding the quantity of MPRs accumulated in landfills was estimated with the intent of recreating the condition faced by flood waters during a hypothetical flood event occurring on any day in 2019. Unfortunately, disposal sites' data lacks consistency, especially regarding the location of landfills. For example, for England, sites' geographic coordinates are available from 2012, before that permit numbers were reported from 2010, and for the period 2007 - 2010 only site names or operators are available to identify a landfill's location. Therefore, the accumulation factor was performed only for landfills operative in 2019, and the total sum of MPRs for the period 2007-2018 was added to daily quantities for 2019. The aim of the methodology is to highlight the importance of considering what has been buried in landfills in previous years that could be mobilised in the case of a flood event. It does not take into consideration the MPs already contained in the disposal site's body and/or leachate, and does not investigate additional mechanisms such as waste decomposition, degradation, and landfill erosion which are left to future studies.

Results

A dataset with daily quantities of waste at risk of releasing MPs in flood waters was created for facilities operative in the UK in 2019, with the addition of accumulation estimates for landfills. A buffer area for each facility was created based on the methodology described and combined with Fathom-UK flood map extents for different return periods (from 1 in 5 to 1 in 1000-year return periods) and flood type (fluvial and pluvial flooding). Both the number of waste facilities at risk of flooding, and the extent of the interaction (percentage of overlap) between buffer areas and flood map extents were estimated. The percentage of overlap was subsequently used to determine the quantity of waste at risk of releasing MPs in flood waters per site. For example, in Figure 3 the waste management facility (A) is a transfer station which dealt with 3,204 tonnes of MPRs waste in 2019 for an approximate daily quantity of 9 tonnes. The medium (200-year) fluvial flood likelihood for the waste facility in Figure 3 is overlapping the facility's buffer area by 72.5% which means that 6.4 daily tonnes of waste are considered as at risk of releasing MPs in 200-year flood waters.

Impact of floods on waste management facilities: quantity and location of waste at risk of releasing MPs

The results across the UK are heavily influenced by the population spatial variation between England and the rest of the UK, however a general common trend was identified. As expected, Figure 5 (A) shows a steady increase in the number of facilities affected by fluvial flooding from high to low likelihood scenarios, starting with approximately 450 sites for the 5-year return period (with 78% of the sites located in England, 10% in Scotland, 8% in Wales, and 3.5% in Northern Ireland), and consistently rising with an addition on average of 54 sites per return period. Pluvial flooding presents a significantly higher impact on waste management facilities compared to fluvial, with numbers almost doubling (from 737 to 1266) in between the 20 and 50-year return periods. This reaches the highest impact with the low likelihood scenario affecting 65% of the total number of facilities which dealt with MPRs in 2019. For the same return period, the landfills at risk of flooding are 135 for pluvial flooding, and 44 for fluvial flooding. This translates respectively to 10.8 and 1.2 million tonnes of MPRs at risk of flooding, predominantly landfill type waste facilities, in the period 2007-2018. A focus on the different categories of waste facilities affected by pluvial flood is available in Figure 5 (B), highlighting treatment facilities and transfer stations as the categories most affected. This is not surprising due to their higher number compared to other types of facilities.

Figure 5 (C) represents the estimation of the quantity (million tonnes) of MPRs at risk of flood for different return periods and sources. The quantity of MPRs affected by fluvial flood is increasing consistently by ~30,000 tonnes per return period, starting at nearly 1 million tonnes for the high likelihood flooding (647 facilities affected) and reaching 1.2 million tonnes with the low likelihood flooding (1,024 facilities). Variations in quantities of waste at risk of flooding compared to the number of facilities affected depend on the quantity of MPR waste received by each facility in 2019, and by the presence of landfills where the waste accumulation was estimated for the period 2007-2018. The estimated numbers of MPRs affected by pluvial flooding are significantly higher, showing a 10-fold increase from 5 to 1000-

year return flood event, at the same time the number of facilities exposed drastically grows from 591 to 2,472 and the number of landfills increases from 39 to 135. In both cases, the percentage of sites located in England is approximately 80%. The biggest increment is between 20 and 50-year pluvial return period where the amount of MPRs at risk is 3 times higher (from ∼1,5 to ∼5 millions of tonnes) for the 50-year return period flood. The reasons behind the greater numbers obtained by pluvial versus fluvial flood can be partially explained by the simulation of extreme rainfall events when floods are created both from overflowing water bodies and independently from them. This results in a flood extend comprised in part by the fluvial flood map extent but will additional areas also present as a result of the pluvial map.

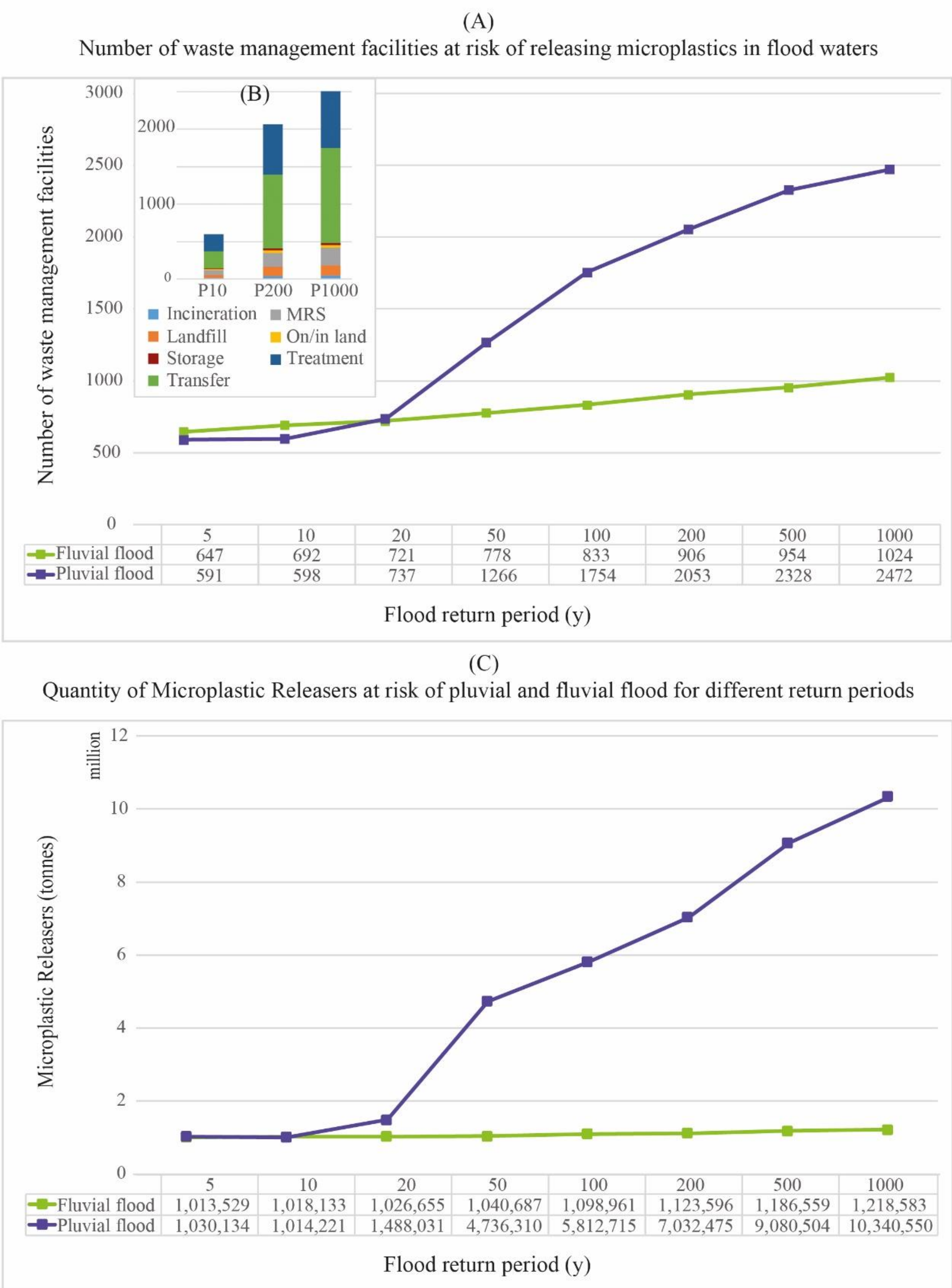

Figure 6-1. (A) Quantity of waste management facilities which received MPRs on site in 2019 at risk of flood for different return periods and peril (fluvial and pluvial flood). (B) Category of waste management facilities based on the Environment Agency in England Waste Data Interrogator at risk of flooding for each pluvial flood returning period organised on stack bar graph. (C) Estimated amount (million tonnes) of Microplastic

Releasers at risk of flooding for different return periods and source (fluvial defended and pluvial flood).

Microplastic Releasers (MPRs) at risk of flood per local authority: emerging spatial patterns

Figure 6 spatially represents the MPRs distribution in the UK summarised by Local Authorities (LA) at risk of pluvial and fluvial flooding for high (1 in 10-year), medium (1 in 200-year) and low likelihoods (1 in 1000-year) scenarios. Results were normalised based on local authorities' area (km^2) in the GIS environment and graphically divided into same values intervals to allow a direct comparison among different likelihoods. This is used to identify hotspots where elevated concentrations of MPRs at risk of flood require further analysis at a local scale.

Figure 6. Amount of managed waste at risk of releasing microplastics in flood waters for different flood likelihoods and sources (fluvial defended and pluvial), summarised and normalised per Local Authority area (km2).

Regionally, in Scotland, the impact of the 10-year fluvial flood on waste facilities is particularly high in Fife (141 tonnes/km² of MPRs), followed by Moray (34 tonnes/km²), East Lothian, and North Ayrshire with respectively (14 and 10 tonnes/km²). Interestingly, the numbers estimated for both the cities of Glasgow and Edinburgh are significantly lower compared to areas in the immediate proximity, suggesting waste is mainly received and treated by facilities located in other districts (a tendency encountered also for other main cities in England, Wales and Northern Ireland). In England, MPRs are concentrated in the area north of London including the authorities from Peterborough through Cambridge and Harlow, with an average of 70 tonnes/km² and highest concentration between Peterborough and Cambridge (212 tonnes/km²). The impact on Wales is predominantly localised in Caerphilly, north of Newport, with a high number of 1,040 tonnes/km² for a total of ~300,000 tonnes of MPRs at risk of flooding in the district. In terms of medium and lower fluvial flood likelihoods, the local authority of Leeds stands out with an increase in the MPRs' concentration from 1 tonne (high likelihood) to 74 and 77 tonnes/km² respectively.

The overlap of pluvial flood map extent on MPRs for the 1 in 10-year is very much similar to the fluvial, but far more interesting is the impact estimated for the medium likelihood. For example, in the local authority of Moray, Scotland, the number of MPRs affected is 3 times higher: from 34 (high likelihood) to 97 tonnes/km² (medium likelihood), for a total of ~220,000 tonnes if considering the entire district's surface. A similar significant increase is evident for the Shetland Islands, rising from 1 to 101 tonnes/km² from high to medium likelihood. In England, when considering LAs with quantity of MPRs higher than 50 tonnes/km², several spatial patterns can be recognised (Figure 6). In particular, in addition to the districts already mentioned located north of London, another hotspot with an average of 230 tonnes/km² can be recognised between Blackburn, Liverpool, Manchester and Stoke-on-Trent. The spatial pattern with the highest concentration of MPRs at risk of medium pluvial flood likelihood is a corridor of 13 LAs in between England and Wales, from Telford south to Bristol and then splitting both west toward Swansea and south in direction of the South Somerset districts; data suggests an average of 700 tonnes per square kilometre. Finally, in the low likelihood but high intensity pluvial flooding, numbers are generally increasing compared to medium and high likelihood with peaks in West Lothian (1,437 tonnes/km²), Newport area (4,456 tonnes/km²), the zone in between Oxford and Luton (Aylesbury Vale with 3,416 tonnes/km²), and Belfast with the highest concentration in the UK: 4,789 tonnes/km².

Discussion

This research presents a novel methodology based on publicly available data on the characteristics and quantity of waste annually received by waste management facilities. A new terminology was introduced to refer to waste able to deteriorate and fragmentise into synthetic microplastic components: Microplastic Releasers (MPRs). MPRs are a selection from the European Waste Catalogue code, comprised of (1) plastic waste (as defined by the European Commission in 2018), (2) synthetic textile and rubber waste (well-known for releasing fibres and micro particles under mechanical stress), and (3) mix and undifferentiated materials which are likely to contain different sizes and types of plastic items, synthetic clothes, and rubber components. The new terminology was essential to enable a quantitative estimation of plastic waste occurring in waste facilities across the UK.

Quantities of MPRs at risk of fluvial and pluvial flood were estimated for the UK for different return periods. Coastal flood risk was not taken into consideration at this stage, but we strongly recommended its inclusion in future analyses, including considering global sea level rise projections. Results from the simulated impact of flood on MPRs were significantly higher for pluvial flooding compared to fluvial flooding, with a 10-fold increase from the high likelihood to the low likelihood (from approximately 1 to 11 million tonnes). The biggest increment was a 3-fold increase between 20 and 50-year return periods. Interestingly, a similar result was obtained by Roebroek et al. (2020) in their global assessment of flood induced mobilisation of plastic, where mismanaged plastic waste was generically estimated as a percentage of total waste generation per country relative to gross domestic profit, where they reported the biggest increment (4-fold increase) between 20 to 50-year return periods versus the 3.5-fold increase between 1 and 10-years. This represents the average at the global scale of approximately 151 tonnes of plastic mobilisation potential per administration unit were estimated (accounting for flood defences) for the 1 in 20-year return period, which became ~612 tonnes for the 1 in 50-year return period.

The methodologies introduced can be applied elsewhere providing appropriate supporting assumptions and the limitation in data are considered. MPRs were selected from all the available origins (households, commercial and industrial activities, construction, and demolition and excavation), by considering only waste that is physically solid, therefore, sludge from waste-water treatment plants, leachate from landfills, used oils and derivate were not included. By not considering waste-water treatment plants and landfill leachate, we excluded at this stage some of the main well-known sources of primary MPs, which we strongly recommend to be added in the estimation of the total MP load in future studies. In addition, due to the complexity and uncertainties related to the movement and storage of waste in a country's waste management system, the length of time required by plastic items to fragment into different sizes of microplastics was not included in the scope of the present work. Further study could assess the feasibility of adding a modelling component to take into account (1) the eventual transport and fate of MPs already present in flood waters, (2) the waste storage periods within facilities, and (3) the extent of MPR degradation into MPs during those storage periods. By doing so future work will improve the introduced methodology to estimate the contribution of waste facilities to the total load of MPs in freshwaters due to flooding. The estimation of the footprints of the facilities was achieved through the creation of buffer areas based on the average size of the cadastral parcels per facility type and country.

Additional data about actual facility dimensions and boundaries description could be included in future mandatory reports on annual quantity of waste received. The latter would be extremely beneficial for more accurate predictions since the quantities of waste at risk of releasing MPs in flood waters were estimated based on the overlap between flood maps and the footprint of the facilities. Another useful piece of information would be data on flood velocity that could be used to estimate potential impact scenarios and significantly improve the assessment of waste at risk of flooding. At this stage, partially because of the adopted national scale, and partially due to the lack of available data, site-specific analysis on flood risk exposure could not be implemented. This includes the consideration of the design of the waste facilities, waste containment systems, and the existence of flood protection measures other than flood defences included within the Fathom flood risk map extents. In addition, available data on waste received for each waste management facility could be improved by including information on waste storage conditions and the build-up period (residency time) to allow the simulation of multiple days' flood scenarios. Finally, the adopted methodology to estimate the amount of MPRs buried in landfills for the period 2007-2018 was the first step in understanding the risk they pose when located in areas at risk of flood. No additional mechanisms of the landfill sites were considered, such as leachate formation, percolation through the body, decomposition stage for different type of waste, or erosion.

Conclusion

Within our study we developed a framework to identify waste at risk of releasing MPs among the European Waste Catalogue (EWC) code in addition to the well-known plastic waste, which allows the estimation of the quantity and location of potential MPs' sources at the national scale as long as datasets about waste are publicly available. In order to overlap the information about waste type and location with flood extent maps, the footprint of waste management facilities has been estimated with the use of buffer areas based on the publicly available European INSPIRE Cadastral Parcel dataset. The methodology was applied to the UK to estimate the quantity of waste at risk of flooding which could lead to MPs' mobilisation in flood waters. Daily quantities of Microplastic Releasers ware estimated for all waste management facilities for the year 2019, only for the landfills operative in the same year, an accumulation factor has been considered by summarising annual quantities of MPRs received from 2007 to 2018 to approximate the real conditions potentially faced in case of inundation.

Results show the impact of pluvial flood being much higher compared to fluvial which can be partially explained by the simulation behind the flood map extents where floods were created both from overflowing water bodies and independently from them, but also it further proves the necessity of assessing the risk related to present and future extreme rainfall events. Results at the national scale were investigated further with the identification of spatial patterns at the local scale for pluvial and fluvial floods for the high, medium and low likelihood. Quantities of MPRs at risk of flooding were combined per Local Authority and normalised per area (km^2) identifying UK hotspots in need of future research in terms of risk management and mitigation measures at the local scale. Depending on the localities, stakeholders and policymakers could rethink the location of existing and new waste management facilities outside flood-prone areas, if the location cannot be changed, mitigation measures can be applied both to the flood origin and pathways with additional flood defences, and by intervening on site-specific containment systems able to limit the mobilisation of synthetic micro components during an event of flood.

Declaration of Competing Interest

The authors declare no conflict of interest.

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